Efectos sobre la producción y el medio ambiente de la fertilización con lodos de depuradora urbana en sistemas silvopastorales establecidos con *Fraxinus excelsior* L., *Pinus radiata* D. Don y *Quercus rubra* L.

Tesis Doctoral

Autora: Nuria Ferreiro Domínguez

Directores: Dra. Mª Rosa Mosquera Losada
Dr. Antonio Rigueiro Rodríguez

UNIVERSIDAD DE SANTIAGO DE COMPOSTELA

ESCUELA POLITÉCNICA SUPERIOR DE LUGO

DEPARTAMENTO DE PRODUCCIÓN VEGETAL

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Realizada en el Departamento de Producción Vegetal de la Universidad de Santiago de Compostela

Autorizamos:

La presentación de la citada Tesis Doctoral, realizada por Nuria Ferreiro Domínguez dado que consideramos que reúne las condiciones necesarias para su defensa.

Febrero de 2011

Fdo. Mª Rosa Mosquera Losada   Fdo. Antonio Rigueiro Rodríguez

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Nuria Ferreiro Domínguez

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RESUMEN

RESUMEN

En Galicia, la productividad de los sistemas silvopastorales (SS) con arbolado joven o poco denso se ve limitada por la baja fertilidad del suelo que puede modificarse mediante diferentes técnicas de manejo, como la fertilización. La fertilidad edáfica puede mejorarse mediante el uso de los lodos de depuradora urbana, ya que este residuo es rico en materia orgánica y macronutrientes, especialmente en N y P. La mejora de la fertilidad del suelo en SS debe tener en cuenta el tipo de suelo de partida (agrícola o forestal), la especie forestal, el tipo de fertilizante y la dosis. Los lodos de depuradora urbana han de ser sometidos a diferentes procesos de estabilización antes de ser empleados en la agricultura lo que les confiere diferente capacidad fertilizante. La UE promueve el uso de los lodos digeridos anaeróbicamente o compostados, pero ambos poseen elevados contenidos en agua, por lo que es interesante evaluar otro tipo de productos como los estabilizados mediante procesos térmicos (lodos peletizados). El objetivo de esta tesis fue el de evaluar el efecto de la aplicación de diferentes dosis de lodos de depuradora urbana en suelo agrícola y forestal estabilizados mediante digestión anaeróbica, compostaje y peletización sobre los cambios más relevantes que se producen en las propiedades químicas del suelo, el crecimiento del arbolado, la producción, biodiversidad (de especies vegetales vasculares) y calidad del pasto y el posible riesgo ambiental provocado por el lavado de nitratos en comparación con los tratamientos de control (no fertilización y fertilización mineral) en sistemas silvopastorales establecidos con *Pinus radiata* D. Don (pino radiata), *Fraxinus excelsior* L.(fresno) y *Quercus rubra* L (roble americano). Los sistemas silvopastorales con fresno y roble fueron establecidos en la parroquia de Baltar (A Pastoriza, Lugo) después de una siembra con dactilo, raigrás y trébol antes de la plantación. En el ensayo con fresno se evaluó el efecto del lodo (320 kg N total ha^{-1}) sometido a diferentes tratamientos de estabilización (lodo anaeróbico, compostado y peletizado). En el caso de la experiencia con roble americano se estudió el efecto de tres dosis diferentes de lodo anaeróbico (100, 200 y 400 kg N total ha^{-1}), el más abundante en las grandes ciudades gallegas. En cuanto al sistema establecido con pino radiata, se instalaron 27 macetas de 2 m^3, con sus respectivos lisímetros, en la parroquia de Piugos (Lugo). En quince macetas se empleó suelo agrícola y en doce suelo forestal. Las macetas con suelo agrícola fueron todas sembradas con dactilo, raigrás y trébol. Parte de estas macetas se dejaron sin fertilizar, otras se fertilizaron con lodo de depuradora anaeróbico (320 kg N total ha^{-1}) y el resto con abono mineral (500 kg ha^{-1}). Además, en algunas de las macetas fertilizadas se plantó pino radiata. Las macetas con suelo forestal fueron todas plantadas con pino radiata, siendo la mitad de las macetas fertilizadas con lodo de depuradora anaeróbico (320 kg N total ha^{-1}) y el resto con mineral (500 kg ha^{-1}) y sembrando con una mezcla pratense de dactilo, raigrás y trébol la mitad de las macetas fertilizadas. Los resultados obtenidos nos permiten concluir que la fertilización con lodos de depuradora permite el reciclaje de este residuo a la vez que incrementa la producción de pasto y el crecimiento del arbolado. Las dosis de 200 y 400 kg de N total ha^{-1} incrementaron de forma similar la producción de pasto y el crecimiento del arbolado por lo que desde un punto de vista económico y medioambiental sería suficiente con aplicar dosis de lodo que implicaran insumos de 200 kg de N total ha^{-1}. Además, debería promoverse el uso del lodo peletizado ya que mejora la productividad y, al tener un menor contenido en agua su aplicación al suelo es más sencilla y su transporte y almacenamiento son más baratos.

ABSTRACT

ABSTRACT

In Galicia, the silvopastoral systems (SS) productivity with young trees or in low density stands is limited by the low soil fertility which could be modified by different management techniques, such as the fertilization. An interesting option that could increase soil fertility is the use of sewage sludge as fertilizer in SS, since this residue is rich in organic matter and macronutrients, especially N and P. The improvement of the soil fertility in the SS should take into account the initial soil type (agronomic or forest), the forest species, the type of fertilizer and the dose. Sewage sludge should be stabilised before using as fertilizer. The stabilisation process could cause differences in the mineralisation rates and therefore in the fertilizer efficiency. The most important types of sludge stabilization are anaerobic digestion and composting, both of which are promoted by the EU. However, both types of waste contain high proportions of water so it is interesting to evaluate other sewage sludge stabilization techniques such as thermic processes (pelletized sewage sludge). The aim of this thesis was to evaluate the effects of the application of different dose of sewage sludge stabilized by anaerobic digestion, composting and pelletization in agronomic and forest soil on changes in soil chemical properties, tree growth, pasture production, biodiversity in terms of sward botanical composition, quality of pasture and the possible environmental risk caused by the nitrate leaching as compared to the control treatments (no fertilisation and mineral) in silvopastoral systems under *Pinus radiata* D. Don, *Fraxinus excelsior* L. and *Quercus rubra* L. The silvopastoral systems development under *Fraxinus excelsior* L. and *Quercus rubra* L. were established in Baltar (A Pastoriza, Lugo). In both experiments pasture was sown with a mixture of *Dactylis glomerata* L., *Trifolium repens* L. and *Lolium perenne* L. and naked-root *Fraxinus excelsior* L. and *Quercus rubra* L. plants were then planted in each experiment. In the study with *Fraxinus excelsior* L., it was evaluated the effect of sewage sludge (320 kg total N ha^{-1}) stabilized by anaerobic digestion, composting and pelletization. In the experiment with *Quercus rubra* L., it was studied the effect of three different doses of anaerobic sewage sludge (100, 200 and 400 kg total N ha^{-1}), being this type of sludge the most abundant in the Galician large cities. Finally, in the study with *Pinus radiata* D. Don, 27 pots of 2 m^3 with their lysimeters were installed in Piugos (Lugo). Fifteen pots were filled with agronomic soil and twelve pots with forest soil. All pots with agronomic soils were sown with *Dactylis glomerata* L., *Trifolium repens* L. and *Lolium perenne* L. Part of these pots were no fertilized, other pots were fertilized with anaerobic sewage sludge (320 kg total N ha^{-1}) and the rest of the pots with mineral (500 kg ha^{-1}). Moreover, in some fertilized pots a *Pinus radiata* D. Don. Tree was planted. On the other hand, all pots filled with forest soil were planted with *Pinus radiata* D. Don being half of these pots fertilized with anaerobic sewage sludge (320 kg total N ha^{-1}) and the rest of the pots with mineral (500 kg ha^{-1}). Half of the pots with forest soil and fertilized with sewage sludge or with mineral were also sowed with *Dactylis glomerata* L., *Trifolium repens* L. and *Lolium perenne* L. The results obtained showed that in the silvopastoral systems, the fertilization with sewage sludge allows nutrient recycling of this residue while increases the pasture production and the tree growth. The doses of 200 and 400 kg total N ha^{-1} increased similarly pasture production and tree growth, therefore, from an economic and environmental point of view would be sufficient to apply 200 kg total N ha^{-1}. The pelletized sludge should be promoted because this type of sludge enhances productivity and presents lower proportion of water than anaerobic sludge and composted sludge which reduces application and storage costs.

PARTE I

JUSTIFICACIÓN Y OBJETIVO

1. JUSTIFICACIÓN Y OBJETIVO

En Galicia, en los últimos años se está produciendo un abandono paulatino del medio rural (baja natalidad y busca de empleo en las ciudades), por lo que muchas explotaciones ganaderas están cesando su actividad, quedando así muchos terrenos agrícolas abandonados (Xunta de Galicia, 2009).

El gobierno autonómico de Galicia, dada la situación y viendo la gran demanda de madera existente en Europa, impulsó la forestación de tierras agrícolas abandonadas y zonas de matorral ofertando diferentes subvenciones (con cofinanciación estatal y de la Unión Europea) (UE, 2005) lo que provocó en los últimos años un importante incremento de la superficie arbolada (III IFN, 1998). Este incremento de la superficie forestal podría suponer un aumento del riesgo de incendios forestales, que es uno de los principales problemas medioambientales que existe en las masas forestales gallegas (Rigueiro-Rodríguez et al., 2009). Una opción alternativa al abandono de la actividad ganadera intensiva y que disminuiría el riesgo de los incendios forestales sería el empleo de sistemas silvopastorales en los que se compaginan los sistemas forestales y ganaderos, combinando la producción de productos ganaderos y madera de calidad, a la vez que se obtiene un efecto de control del sotobosque por parte de los animales (Rigueiro-Rodríguez et al., 2005a, 2009) y por tanto una disminución del riesgo de incendios. En España, el establecimiento de sistemas agroforestales se promueve a través de la Estrategia Forestal Española (MMA, 1999) y de la Directiva Europea CE 1698/2005 (UE, 2005) relativa a las ayudas al desarrollo rural a través del Fondo Europeo Agrícola de Desarrollo Rural que contempla ayudas directas para el establecimiento de sistemas agroforestales.

Ente las especies arbóreas interesantes para establecer sistemas silvopastorales en la zona atlántica española podemos citar *Pinus radiata* D. Don, ya que es la especie más empleada en repoblaciones de la provincia de Lugo en las últimas décadas (Álvarez et al., 2001) y es ampliamente empleada en sistemas agroforestales en otros países (Benavides et al., 2009), *Fraxinus excelsior* L., que ha presentado buenos resultados en la implantación de sistemas agroforestales en la zona atlántica europea (McAdam y Hoppe, 1996; McAdam y Sibbald, 2000) y *Quercus rubra* L. que es una frondosa cuyo uso se está extendiendo en los últimos años en Galicia (Molina-Rodríguez et al., 2004).

Por otra parte, los suelos gallegos suelen presentar baja fertilidad, lo que limita la productividad de los componentes herbáceo y arbóreo de los sistemas silvopastorales (Zas y Alonso, 2002). Una opción interesante para incrementar la fertilidad del suelo sería el uso de los lodos procedentes de depuradoras urbanas como fertilizante orgánico, ya que son ricos en materia orgánica, N y P (MMA, 2006). El uso de los lodos en la agricultura permitiría emplear este tipo de residuo de forma sostenible y resolver el problema de eliminación del mismo, ya que debido a la implementación de la Directiva 91/271/CEE (UE, 1991) que obliga a depurar las aguas residuales en poblaciones de más de 2.000 habitantes su producción se ha visto notablemente incrementada en los últimos años. En España, el R.D. 1310/1990 (BOE, 1990) y en Europa la Directiva 86/278/CEE (UE, 1986) recogen toda la normativa que regula la utilización agrícola de los lodos de depuradora, estableciendo unos valores límite máximos de concentración de metales pesados (Cu, Zn, Ni, Cd, Pb, Hg y Cr) en el suelo y en el lodo, para que éste pueda ser empleado como fertilizante en la agricultura. Las dosis de lodo de depuradora que se pueden aplicar al suelo van a depender de la cantidad de metales pesados y nitrógeno en el lodo y de la proporción de nitrógeno que es fácilmente mineralizable durante el primer año tras la aplicación del residuo (Barry et al., 1986; EPA, 1994; Smith, 1996). De este modo, la EPA (1994) nos indica que si las dosis de lodo de depuradora son muy superiores a las necesidades del cultivo existe riesgo de que se produzca un lavado de nitratos a través del perfil del suelo provocando una contaminación de las aguas.

Para que los lodos de depuradora urbana se puedan aplicar al suelo es necesario que sean previamente sometidos a tratamientos de estabilización (RD 1310/1990) (BOE, 1990) (digestión aeróbica, anaeróbica, compostaje y peletización son los más frecuentes). Las características de los lodos, su contenido en nutrientes y la tasa de incorporación al suelo varían en función de los procesos de estabilización a los que son sometidos en las estaciones de depuración de las aguas residuales (EDAR). La Unión Europea promueve el empleo como fertilizantes de los lodos estabilizados mediante digestión anaeróbica y compostaje (EEA, 2000); sin embargo, en casos contienen elevadas proporciones de agua, que se reducen en los lodos sometidos a secado térmico y posterior peletizado, lo que abarata el transporte y almacenaje y facilita su aplicación al suelo.

Por lo tanto, los aspectos más importantes a evaluar, desde los puntos de vista productivo y medioambiental, en relación con el impacto de la fertilización en sistemas agroforestales, están relacionados con el tipo de suelo, la especie forestal, el pasto, el tipo de fertilizante y la dosis de lodo de depuradora aplicada.

El objetivo de esta tesis fue evaluar el efecto de la aplicación de lodos de depuradora urbana en suelo agrícola y forestal sobre las propiedades químicas del suelo, el crecimiento del arbolado, la producción, biodiversidad (de especies vegetales vasculares) y calidad del pasto y el posible riesgo ambiental provocado por el lavado de nitratos, en comparación con los tratamientos de control (no fertilización y fertilización mineral) en sistemas silvopastorales establecidos con *Pinus radiata* D. Don, *Fraxinus excelsior* L. y *Quercus rubra* L.

Más concretamente los objetivos fueron:

- ✓ Evaluar el efecto del lodo de depuradora urbana que ha sido estabilizado mediante digestión anaeróbica, compostaje y peletización sobre las propiedades químicas del suelo (pH, capacidad de intercambio catiónico efectiva (CIC), porcentaje de saturación del Al, Cu y Zn total y Ca, Cu y Zn extraídos por el método Mehlich), el crecimiento del arbolado (altura y diámetro), la producción de pasto del sotobosque, la biodiversidad en términos de la composición botánica de la pradera y la concentración de Cu y Zn en el pasto en comparación con los tratamientos de control (no fertilización y fertilización mineral) en un sistema silvopastoral establecido con *Fraxinus excelsior* L.
- ✓ Evaluar el efecto de la plantación de *Pinus radiata* D. Don, de la siembra de pasto y de la fertilización con lodo de depuradora anaeróbico y con fertilizante mineral sobre distintas variables del suelo (pH, capacidad de intercambio catiónico efectiva (CIC), materia orgánica, N y P total, P extraído por el método Mehlich y lavado de nitratos), del arbolado (altura y diámetro), y del pasto (producción, composición botánica de la pradera y cantidad de proteína y P del pasto) en sistemas forestales arbolados, praderas y en sistemas silvopastorales establecidos en suelo agrícola y en suelo forestal.
- ✓ Evaluar el efecto de diferentes dosis de lodo de depuradora anaeróbico (100 kg N total ha^{-1}, 200 kg N total ha^{-1} y 400 kg N total ha^{-1}) sobre diferentes variables del suelo (pH, capacidad de intercambio catiónico efectiva (CIC) y porcentaje de saturación de Al, K, Ca, Mg y Na), del arbolado (altura y diámetro) y del pasto (producción y biodiversidad de plantas vasculares), en comparación con el tratamiento de control de no fertilización, en un sistema silvopastoral establecido con *Quercus rubra* L.

PARTE II

INTRODUCCIÓN

2. INTRODUCCIÓN

2.1. LOS SISTEMAS AGROFORESTALES (SAF)

Los sistemas agroforestales son técnicas de manejo sostenible de la tierra que emplean árboles, cultivos y/o animales sobre la misma unidad de terreno en cualquier forma de ordenación espacial o temporal, registrándose entre ellos interacciones ecológicas y económicas (Nair, 1993; Rigueiro-Rodríguez et al., 2008a). Este tipo de sistemas tienen siempre una mayor complejidad que cualquier sistema forestal o agrícola por separado, debido a los diferentes tipos de las relaciones que se establecen entre los componentes (Silva-Pando y Rozados-Lorenzo, 2002).

Los sistemas agroforestales son formas de uso del territorio vinculadas al sedentarismo de la especie humana y empleadas desde antiguo, ya que algunas formas de sistemas agroforestales se empleaban 9.000 años a.C. en África o 2.500 años a.C. en España (Eichhorn et al., 2006). Sin embargo, es a partir de los años ochenta del pasado siglo cuando los sistemas agroforestales empiezan a ser reconocidos y se incorporan a los programas nacionales de investigación agrícola y forestal de muchos países desarrollados, debido principalmente a problemas de gestión de la tierra como la deforestación tropical, la escasez de leña, la degradación del suelo y la pérdida de biodiversidad (Nair et al., 2008). En 1992, en la Conferencia de las Naciones Unidas para el Desarrollo y el Medio Ambiente (UNCED) celebrada en Río de Janeiro, se desarrolló el plan de acción denominado “Agenda 21” (UN, 1992) en el cual se proponía el mantenimiento y la sostenibilidad de las zonas arboladas, especificando la adecuación de los sistemas agroforestales como método de manejo sostenible de la tierra. Además, en España, la Estrategia Forestal Española también promueve el establecimiento de los sistemas agroforestales a través del uso múltiple de los terrenos forestales (MMA, 1999). En la actualidad este tipo de sistemas son promovidos por la UE (Reglamento1698/2005, UE, 2005) y son vistos por parte de los agricultores europeos como formas de gestión del territorio susceptibles de ser empleados, como demuestran las encuestas realizada a 214 agricultores de 14 regiones diferentes de Europa, en las que más de la mitad indicaron que les gustaría establecer sistemas agroforestales en sus granjas (Graves et al., 2008).

Para poder clasificar los sistemas agroforestales se pueden considerar diferentes criterios. En la Tabla 2.1 se presenta un resumen de los criterios empleados por distintos autores (Wiersum, 1981; Nair, 1985, 1990; Silva-Pando y Rozados-Lorenzo, 2002), entre los cuales la naturaleza de los componentes ha sido el criterio más utilizado, dando lugar a la Agroselvicultura, Silvopascicultura o a la Agrosilvopascicultura entre otras denominaciones. La aplicación de criterios como la ordenación espacial y temporal de los componentes, distribución biogeográfica, nivel de aportes tecnológicos, relaciones coste beneficio o tipo de función darán lugar a una mayor diferenciación y complejidad.

Clasificación de los SAF basada en su estructura y función			Tipos de SAF de acuerdo a su extensión y gestión	
Estructura (Naturaleza y ordenación de los componentes especialmente los leñosos)		Función (papel y/o producciones de los componentes, especialmente los leñosos)	Adaptabilidad agroecológica y medioambiental	Nivel socioeconómico y de gestión
Naturaleza de los componentes	Ordenación de los componentes			
Agroselvicola (cosechas agrícolas y árboles, incluyendo arbustos/árboles) ***Silvopastoral*** (pastos/animales y árboles) ***Agrosilvopastoral*** (cosechas agrícolas, pasto/animales y árboles) ***Otros*** (grupos de árboles multiproducto, apicultura con árboles, acuicultura con árboles, etc.)	***En el espacio*** (espacial) * Mezclas densas (p.e.: casa-jardín) * Mezclas laxas (p.e.: la mayoría de los sistemas de árboles sobre pastizales) * Por fajas (el ancho de la faja debe ser mayor que una fila de árboles) * Límites (árboles en los límites de grupos/campos) ***En el tiempo*** (temporal) Coincidente Concomitante Superpuesto Secuencial (separado) Interpolado	***Función productiva*** Alimento Ramón Leña Otras maderas Otros productos ***Función protectora*** Cortavientos Conservación de suelo Conservación de la humedad Sombra (para cosechas animales y el hombre) Conservación de la biodiversidad Prevención de incendios	***Sistemas en/por*** Tierras bajas tropicales húmedas Tierras altas tropicales húmedas (p.e.: Alturas mayores de los 1.200 m.s.n.m., Malasia) Tierras bajas tropicales subhúmedas (p.e.: zona de sabana de África, cerrado de Suramérica) Zonas templadas (p.e.: Europa, Norteamérica, China)	***Basada en el nivel del aporte tecnológico*** Bajos aportes (marginal) Aportes medios Altos aportes ***Basado en la relación coste/beneficio*** Comercial Intermedio Subsistencia

Tabla 2.1: Clasificación de los sistemas agroforestales (Fuente: Nair, 1985; Silva-Pando y Rozados-Lorenzo, 2002).

Según la "Association for Temperate Agroforestry" (AFTA, 1999) los sistemas agroforestales se caracterizan por su intencionalidad, intensividad, interactividad y por ser sistemas integrados. El término intencionalidad implica que el sistema es diseñado y manejado de forma intencionada como una unidad y la intensividad indica que los sistemas son manejados para obtener beneficios tanto monetarios como de protección. Las interacciones físicas y biológicas entre los diferentes componentes (arbolado, cultivo/pasto, suelo y animales) se encuentran implícitas en el término interactividad y se dice que son sistemas integrados ya que entre sus componentes se registran interacciones estructurales y funcionales, dando lugar a una unidad de manejo.

Por otro lado, cabe indicar que en todo el mundo existen numerosos sistemas tradicionales en los que se reconocen las características mencionadas anteriormente. Dichos sistemas son específicos de cada zona, ya que se describen bajo unas condiciones locales, surgiendo así un amplio número de sistemas agroforestales, aunque se puede decir que sus componentes presentan un determinado patrón de distribución espacio-tiempo, dando lugar a lo que Nair y Nair (2002) definen como prácticas agroforestales.

Dentro de las zonas templadas se identifican las prácticas agroforestales que se recogen en la Tabla 2.2 (Nair y Nair, 2002; Alavalapati et al., 2004; Mosquera-Losada et al., 2008a).

Prácticas agroforestales	Definición
Cultivo en callejones	Distribución del arbolado en hileras simples o agrupadas con cultivos herbáceos intercalados
Cultivo en el bosque	Cultivo en zonas arboladas de plantas medicinales, ornamentales, culinarias, etc., y/o aprovechamiento de plantas silvestres útiles que crecen en los bosques
Bosques de ribera	Zonas de vegetación perenne (árboles, arbustos o herbáceas) entre los cultivos/pastos y los cursos de agua, y/o cultivos entre el arbolado en franjas de ribera
Silvopastoreo	Combinación de la producción forestal con la forrajera (pasto o heno) y la ganadera
Zonas cortaviento	Plantaciones lineales de árboles alrededor de granjas y campos de cultivo, cuya finalidad es la protección de los animales, el cultivo y el suelo frente a los efectos del viento

Tabla 2.2: Prácticas agroforestales (Fuente: Nair y Nair, 2002; Alavalapati et al., 2004; Mosquera-Losada et al., 2008a).

En esta tesis nos centraremos en los sistemas silvopastorales, los que más potencial de uso tienen en la comunidad autónoma gallega, la cual posee una importante ganadería bovina y una amplia superficie destinada a pastos y forrajes (25% del territorio) y a terrenos forestales (más del 60%). Por otra parte, los sistemas silvopastorales son de los más antiguos tipos de sistemas agroforestales desarrollados en las zonas templadas (Nair, 1991) y son una de las prácticas agroforestales más extendidas hoy en día en toda Europa (Agroforestry Forum, 2007).

2.2. LOS SISTEMAS SILVOPASTORALES

Los sistemas silvopastorales son un tipo de sistema agroforestal en el cual el arbolado y el pasto se manejan buscando una integración entre la producción maderera y animal (Rigueiro-Rodríguez et al., 2005a) y se pueden considerar como una de las prácticas agroforestales más empleadas en el pasado y en la actualidad en toda Europa (Mosquera-Losada et al., 2008a).

La especie arbórea que se utiliza para establecer el sistema silvopastoral desempeña una serie de funciones, entre las que destacan la producción maderera y proporcionar alimento al ganado. Con el objeto de que el arbolado permita la producción del pasto bajo sus copas, es conveniente que reúna una serie de características como son una buena dominancia apical, buena poda natural o tolerar podas intensas (Beaton y Hislop, 2000). La relación diámetro de copa/diámetro del tronco del árbol debe ser baja y la copa tiene que ser clara para que deje pasar la luz al suelo y no intercepte mucha lluvia, además, la descomposición de sus restos no debe producir efectos alelopáticos sobre las especies herbáceas del sotobosque. Asimismo, deben ser eficaces bombas de nutrientes y el sistema radical del árbol tiene que ser capaz de explorar horizontes profundos del suelo para disminuir la competencia con los estratos arbustivo y herbáceo, obteniéndose así una mayor productividad de los componentes arbóreos y forrajeros. La especie arbórea establecida en el sistema silvopastoral también debe ser compatible con el tipo de ganado introducido en el sistema (Rigueiro-Rodríguez et al., 2005b).

El ganado empleado será a su vez capaz de alimentarse de la vegetación que se desarrolla en el sotobosque (Silva-Pando, 1988; Rigueiro-Rodríguez et al., 1997; Rigueiro-Rodríguez, 2000) y compatible con el arbolado. Las cabras y los caballos serán adecuados para controlar el combustible vegetal leñoso vivo del sotobosque disminuyendo el riesgo de incendios (Rigueiro-Rodríguez et al., 2005b), mientras que las ovejas y las vacas consumen bien el pasto herbáceo, introduciéndose en el sistema cuando el sotobosque se encespeda debido al pastoreo previo de las cabras y los caballos o cuando establecemos una pradera artificial bajo repoblación (Rigueiro-Rodríguez, 1992).

2.3. CARACTERÍSTICAS DE LOS SISTEMAS SILVOPASTORALES

En los sistemas silvopastorales se debe presentar una estabilidad entre sus diferentes componentes (árboles, pasto y animales) que permita que el sistema sea viable tanto en el tiempo como en el espacio (Silva-Pando y Rozados-Lorenzo, 2002). La integración de los recursos agropecuarios y forestales que se registra en este tipo de sistemas favorece su productividad y el aporte de sus beneficios ambientales y sociales (Rigueiro-Rodríguez et al., 2008a).

2.3.1. Productividad de los sistemas silvopastorales

La productividad de los sistemas silvopastorales deriva de la multiplicidad de productos obtenidos. Así, a partir de la misma unidad territorial podemos obtener simultáneamente madera, alimentos, forraje, leña, plantas medicinales, etc. (Rigueiro-Rodríguez et al., 2008a). Los sistemas silvopastorales presentan una mayor estabilidad económica que los exclusivamente forestales, debido a los rápidos retornos monetarios de los productos agrícolas y/o ganaderos que se compatibilizan y complementan con los más tardíos del arbolado, como la madera. Estudios realizados por Sibbald (1996) señalan hasta un 15% de aumento del beneficio económico al combinar el ganado con especies arbóreas como *Fraxinus excelsior* L., en comparación con el pastoreo en zonas no arboladas en áreas no montañosas del Reino Unido. Estos beneficios proporcionan a los propietarios una mayor continuidad de ingresos a lo largo de la vida de la explotación, en comparación con el manejo forestal tradicional (Anderson y Sinclair, 1993; Sharrow, 1999).

La productividad de un sistema silvopastoral depende del tipo de especie arbórea elegida y de la densidad del arbolado y se ve positivamente influenciada cuando la asociación árbol-plantas forrajeras es adecuada en tiempo y espacio. Los árboles pueden beneficiar a los cultivos que se desarrollan en su entorno, promoviendo la producción de productos de origen animal a bajo coste y de alta calidad (Rigueiro-Rodríguez et al., 2008a). En estudios realizados por Fernández-Núñez et al (2007) en los que se comparaba la rentabilidad de sistemas silvopastorales establecidos con *Pinus radiata* D.

Don a diferente densidad de plantación (2500 y 833 árboles ha^{-1}) con la rentabilidad de sistemas exclusivamente forestales y ganaderos, se observó que en los sistemas silvopastorales con menor densidad de arbolado la producción aumentaba un 17% y un 53% con respeto a los sistemas ganaderos o forestales, respectivamente. Por lo tanto, el establecimiento de sistemas silvopastorales en tierras con uso exclusivamente forestal o ganadero podría considerarse como una buena alternativa económica siempre y cuando la especie arbórea y su densidad, el pasto y el ganado y carga ganadera sean adecuados.

2.3.2. Beneficios ambientales de los sistemas silvopastorales

Los sistemas silvopastorales presentan grandes ventajas de tipo ambiental, debido a que aportan externalidades de gran interés: mejoran el microclima y la calidad paisajística, previenen la erosión eólica e hídrica y los incendios forestales, aumentan la biodiversidad y reducen la contaminación de las aguas por lavado de nitratos y fósforo (Hislop y Sinclair; 2000; McAdam, 2000; Rigueiro-Rodríguez et al., 2008a), al mismo tiempo que fijan carbono (Nair et al., 2008; Fernández-Núñez et al., 2010a; Howlett et al., 2010). Nos centraremos en los efectos sobre la biodiversidad y la calidad de las aguas por ser estos aspectos evaluados en esta tesis.

a) Biodiversidad

La Unión Europea se ha planteado el ambicioso objetivo de detener la pérdida de la biodiversidad antes del año 2010 (UE, 2002), ya que la conservación de la biodiversidad es necesaria por razones económicas, sociales y ecológicas. En los bosques de la Unión Europea y en las tierras agrícolas de todo el mundo, la intensificación de la agricultura, la silvicultura monoespecífica y la propagación de especies alóctonas invasoras son algunos de los factores clave responsables de los cambios del estado de los nutrientes del suelo, de la productividad del sotobosque y de la pérdida de diversidad vegetal (Augusto et al., 2002; Mosquera-Losada y Rigueiro-Rodríguez, 2005; CBD, 2010).

Los sistemas silvopastorales podrían ser considerados una herramienta para fomentar la conservación de la biodiversidad, ya que estos sistemas generan diferentes condiciones de humedad, luz y fertilidad del suelo que fomentan el desarrollo de diferentes especies microbianas, de artrópodos, arácnidos y vegetales, en comparación con los ecosistemas exclusivamente forestales o agrícolas, potenciando así el crecimiento y desarrollo de especies adaptadas a esos microclimas (Mosquera-Losada et al., 2006; Rois-Díaz et al., 2006; McAdam et al., 2007; McAdam y McEvoy, 2008). Además, los sistemas silvopastorales también contribuyen a incrementar la biodiversidad paisajística, y amplían el terreno forestal y agrícola a nivel territorial, creando biotopos apropiados para distintas especies de aves y mamíferos y formando corredores que comunican ecosistemas reduciendo la fragmentación de hábitats (Rois-Díaz et al., 2006).

En los sistemas silvopastorales, la biodiversidad del pasto se va a modificar con el paso del tiempo en función de la especie arbórea introducida y de su crecimiento y de la estructura, anatomía y disposición de las hojas de los árboles que van a permitir que llegue al pasto diferente intensidad y cantidad de luz (Rigueiro-Rodríguez et al., 2008a). En los primeros años del establecimiento del sistema silvopastoral, la biodiversidad de las especies del pasto será muy similar a la encontrada en áreas exclusivamente pascícolas y sin presencia de arbolado. Sin embargo, con el paso del tiempo el efecto del arbolado sobre la biodiversidad del pasto será diferente en función de la especie arbórea introducida. De este modo, en sistemas silvopastorales establecidos con *Pinus radiata* D. Don se observó, siete años después del establecimiento del estudio, una disminución de la biodiversidad del pasto debido a las condiciones de sombra, como consecuencia de la elevada densidad de plantación (Mosquera-Losada et al., 2006), sin embargo, en un sistema silvopastoral establecido con *Betula alba* L., también siete años después del inicio del ensayo, no se observó un efecto significativo del arbolado sobre la biodiversidad del pasto (Rigueiro-Rodríguez et al., 2005c). Por el contrario, las dehesas son consideradas en toda Europa como el paisaje de mayor biodiversidad creado por el hombre, en el cual la combinación de árboles y pasto sirve de hábitat para una gran variedad de insectos, aves y otro tipo de fauna y flora (Moreno-Marcos y Pulido, 2008).

En cuanto a biodiversidad de la fauna en los sistemas silvopastorales, experiencias descritas por autores como Cuthbertson y McAdam (1996) demuestran que los sistemas silvopastorales incrementan la población de la fauna invertebrada, debido a que ésta se desarrolla mejor bajo las condiciones heterogéneas de los sistemas agroforestales. Un ejemplo de esto es la mayor presencia de carábidos y otros artrópodos (Dennis et al., 1996; McAdam et al., 2007) y el aumento del número de pájaros y de otros animales (Burgess, 1999; McAdam et al., 2007) en los sistemas agroforestales en comparación con sistemas exclusivamente agrícolas.

b) Contaminación de las aguas

En los sistemas agrícolas tradicionales menos de la mitad del N o del P de los fertilizantes aplicados es usado por los cultivos, lo que implica que el exceso de fertilizante, sobre todo el nitrógeno,, es lavado a través del perfil del suelo contaminando los acuíferos, reduciendo la calidad de las aguas y provocando fenómenos de eutrofización (Cassman, 1999). Se considera que la contaminación de las aguas que causa el exceso de N aplicado al suelo es mayor que la que produce el exceso de P, debido al ciclo más complejo del N (Whitehead, 1995) y al mayor número de vías de eliminación del N del sistema (lavado y volatilización) y a que el P es retenido por el Al y el Fe en los suelos ácidos, como es el caso de los gallegos.

El establecimiento de sistemas agroforestales podría reducir el lavado de nitratos y por lo tanto mejorar la calidad de las aguas, ya que las raíces de los árboles exploran capas más profundas del suelo que las raíces del pasto y pueden absorber el N no aprovechado por las especies pascícolas tal como se muestra en la Figura 2.1. Estudios realizados en zonas tropicales (Van Noordwijk et al., 1996) y en zonas templadas (Nair y Graetz, 2002; Allen et al., 2004; Nair y Graetz, 2004; Nair et al., 2007) muestran el importante papel importante del arbolado en la reducción de la contaminación de las aguas por lavado de nitratos y fósforo. Autores como Allen et al. (2004), en sistemas agroforestales establecidos en el noroeste de Florida con *Carya illinoinensis* (Wangenh.) K. Koch y algodón mostraron que el lavado de nitratos se reducía en un 72% en comparación con los monocultivos de algodón. Nair et al. (2007) y Mosquera-Losada et al. (2010a) también obtuvieron una mayor retención de nutrientes en sistemas silvopastorales establecidos en Florida con *Pinus elliottii* Engelm. y en Galicia con *Pinus radiata* D. Don, respectivamente, que en sistemas agrícolas sin arbolado. Por lo tanto, los resultados nos muestran que los sistemas agroforestales y en particular los sistemas silvopastorales podrían desempeñar un papel importante en la mitigación del daño que la fertilización puede causar a la calidad de las aguas en las prácticas de agricultura intensiva.

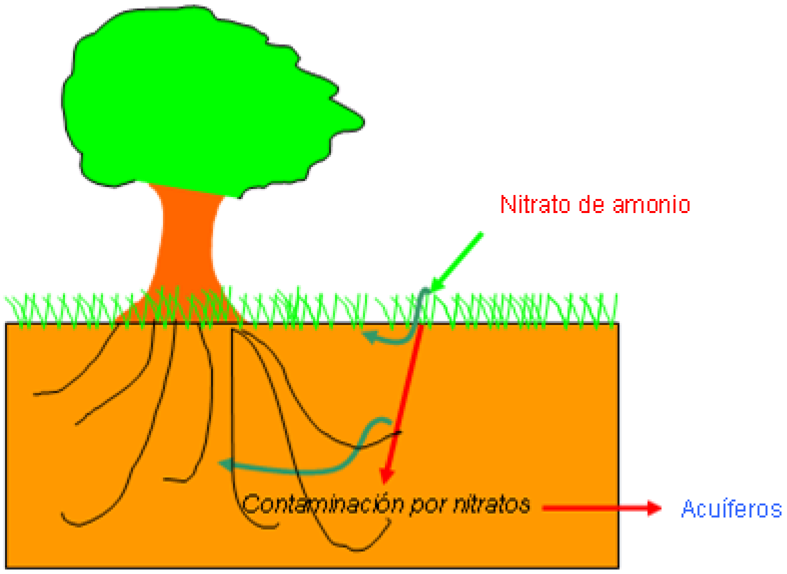

Figura 2.1: Uso de los nitratos por las raíces del pasto y del arbolado (líneas verdes) y lavado de los nitratos (líneas rojas) (Fuente: Rigueiro-Rodríguez et al., 2008a)

2.3.3. Beneficios sociales de los sistemas silvopastorales

Desde el punto de vista social, los sistemas silvopastorales contribuyen a mejorar las condiciones socioeconómicas de las zonas rurales mediante la creación de empleo y el aumento de ingresos (Nair, 1991). Se trata de sistemas de gestión de la tierra que permiten un amplio disfrute de las áreas rurales para el público en general, porque incrementan, mejoran y ayudan a preservar prácticas tradicionales y culturales. A su vez pueden ayudar a mejorar la productividad en la agricultura y, si se asocian con turismo rural, puede vincularse con la obtención de productos ecológicos de alta calidad con denominación de origen que incrementan la rentabilidad de estos sistemas, contribuyendo así a la estabilización de la población rural (Pardini, 2008; Rigueiro-Rodríguez et al., 2008a).

2.4. LA SUPERFICIE FORESTAL Y AGRÍCOLA EN GALICIA

Según los datos del III Inventario Forestal Nacional (III IFN, 1998) la superficie forestal de Galicia representa el 69% de la superficie de la Comunidad Autónoma, siendo arbolado más del 50% del terreno forestal. Galicia es la Comunidad Autónoma española con mayor proporción de superficie arbolada de España, ya que el monte arbolado gallego representa el 8% de las masas arbóreas españolas y es una de las regiones con mayor importancia forestal de toda Europa. Por otro lado, el 28% de la superficie gallega se corresponde con terreno agrícola y el 3% con otros usos del territorio (III IFN, 1998) (Figura 2.2).

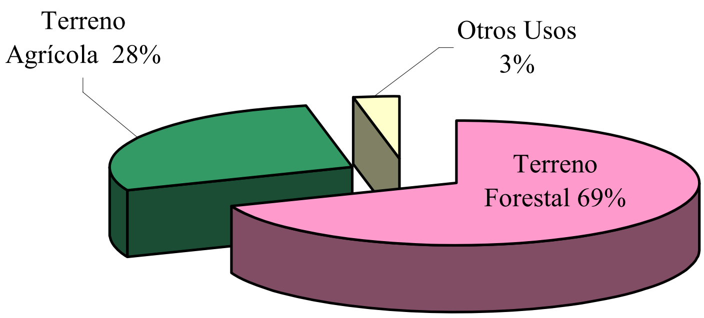

Figura 2.2: Usos del territorio en Galicia (Fuente: III IFN (1998))

Según la misma fuente en Galicia existe un equilibrio entre las masas de coníferas (506.026 ha) y las masas de frondosas incluyendo los eucaliptos (562.417 ha) y hay una importante superficie ocupada por la mezcla de coníferas y frondosas (359.267 ha). Tal y como se muestra en la Figura 2.3, dentro del grupo de las coníferas, la especie dominante es *Pinus pinaster* Aiton (383.631,78 ha), seguido en menor presencia por *Pinus sylvestris* L. (63.195,6 ha) y *Pinus radiata* D. Don (59.198,27 ha). Dentro del grupo de las frondosas autóctonas destacan las quercíneas, con *Quercus robur* L. (187.788,97 ha) como principal especie, seguida de *Quercus pyrenaica* Willd. (100.503,78). Entre los eucaliptos la especie principal es *Eucalyptus globulus* Labill. (174.210,40 ha), mientras que en la mezcla de coníferas y frondosas predomina la combinación de *Pinus pinaster* Aiton con *Eucalyptus globulus* Labill. (159.413,93 ha).

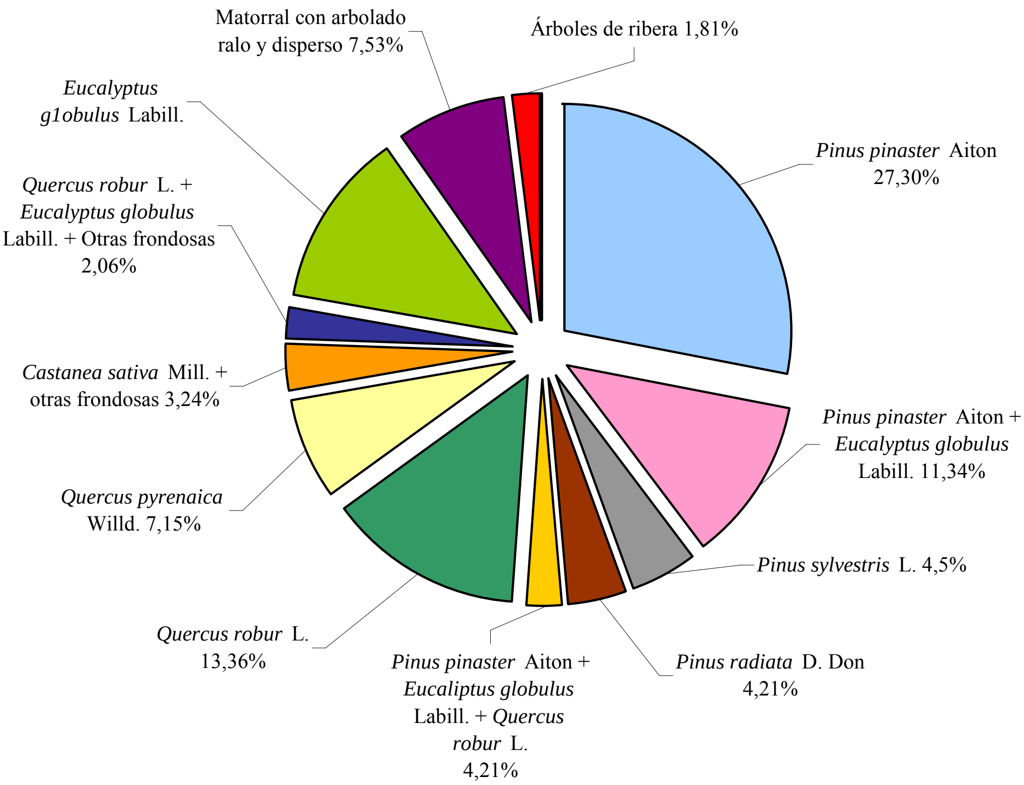

Figura 2.3: Porcentaje de la superficie arbolada en Galicia según la especie (Fuente: III IFN (1998))

En los últimos años, hay que destacar el incremento de la superficie arbolada que se está produciendo debido al abandono paulatino de muchas explotaciones agropecuarias, lo que conlleva un cambio del uso de la tierra de agrícola a forestal. Según los datos del II Inventario Forestal Nacional (II IFN, 1987) y III Inventario Forestal Nacional (III IFN, 1998) la superficie arbolada creció entre esos años un 27,1% en el caso de montes particulares y un 64,8% en el caso de los montes públicos y vecinales. Este cambio de uso de la tierra es apoyado por la política de la Unión Europea con líneas de ayudas para el abandono voluntario de explotaciones agrarias y para la reforestación de tierras agrarias (UE, 2005). Los datos del II Inventario Forestal Nacional (II IFN, 1987) y III Inventario Forestal Nacional (III IFN, 1998) indican que entre ambos inventarios se ha incrementado en 400.000 ha la superficie arbolada de Galicia. Este aumento tan importante de la superficie arbolada supone problemas de gestión, debido a que el mantenimiento de las masas, con las necesarias labores de desbroces, claras y podas, es costoso. La implantación de sistemas silvopastorales disminuiría estos costes porque permitiría reducir el combustible vegetal (desbroce biológico por el ganado) a la par que se facilitaría la transitabilidad por el monte, favoreciendo la realización de trabajos selvícolas, como podas y claras, que incrementarían la producción de madera de calidad en turnos más cortos y la producción de pasto, disminuyendo a su vez el riesgo de incendios de forma notable (Rigueiro-Rodríguez et al., 2005a, 2009). Los sistemas silvopastorales también podrían ser una alternativa al abandono de la actividad ganadera intensiva, dado que permiten compaginar la producción ganadera y la forestal y obtener beneficios de ambas. Además, la Directiva Europea CE 1698/2005 (UE, 2005) relativa a las ayudas al desarrollo rural a través del Fondo Europeo Agrícola de Desarrollo Rural contempla ayudas directas para el establecimiento de sistemas agroforestales. Según el informe realizado en el año 2009 sobre la aplicación de las medidas forestales en el marco del Reglamento 1698/2005 de desarrollo rural (UE, 2005) se espera que entre los años 2007-2013 sean implantados sistemas agroforestales en 60.000 ha de superficie agrícola, beneficiándose así a 3.000 propietarios de tierras y correspondiéndole a cada beneficiario 18 ha (UE, 2009).

En esta tesis se desarrollan estudios relativos al *Pinus radiata* D. Don, la especie forestal más empleada en repoblaciones de la provincia de Lugo en las últimas décadas (Álvarez et al., 2001), *Fraxinus excelsior* L., especie que ha presentado buenos resultados en la implantación de sistemas agroforestales en la zona atlántica europea (McAdam y Hoppe, 1996; McAdam y Sibbald, 2000) y *Quercus rubra* L., frondosa cuyo uso se está extendiendo en los últimos años en Galicia (Molina-Rodríguez et al., 2004).

2.4.1. *Pinus radiata* D. Don en los sistemas silvopastorales

Pinus radiata D. Don (pino insigne o pino de Monterrey) es una especie forestal originaria de la costa oriental norteamericana, siendo su área de distribución natural muy reducida (costa de California, sur de San Francisco e islas de Santa Rosa, Santa Cruz y Guadalupe) (Dans et al., 1999). Sin embargo, la importante tasa de crecimiento del *Pinus radiata* D. Don, junto con la precocidad con la que alcanza su máxima producción en volumen y la calidad aceptable de su madera para diferentes usos, ha fomentado el empleo de esta especie en repoblaciones en gran parte del mundo, siendo introducida en muchos países de la zona templada mundial, como Chile, Nueva Zelanda, Australia, Sudáfrica y varios países europeos.

En Europa, el *Pinus radiata* D. Don se introdujo sobre todo en el sur (Italia, Francia, España, Portugal), pero actualmente es España el único país en el que las masas de esta especie ocupan importantes extensiones, sobre todo en Galicia y la cornisa Cantábrica (Figura 2.4), y en el que se ha desarrollado un tejido industrial basado en la transformación de su madera (Dans et al., 1999). En Galicia, en el año 1950, el pino insigne fue introducido en las provincias de Lugo y A Coruña a través de los trabajos del Patrimonio Forestal del Estado, que con posterioridad fueron continuados por el ICONA. Hoy en día, el *Pinus radiata* D. Don es una de las especies forestales de mayor importancia en Galicia, ya que en el programa de forestación de tierras agrarias, que comenzó en Galicia en Noviembre de 1993 a través de la aplicación de los Reglamentos (CEE) nº 2080/1992 (UE, 1992) y nº 1257/1999 (UE, 1999), muchos propietarios forestales se sintieron inclinados a elegir esta especie para reforestar sus tierras agrarias marginales y otros terrenos de carácter forestal, debido al crecimiento moderadamente rápido de esta especie. En concreto, entre el año 1993 y el año 2000 se repoblaron un total de 62.885 ha, principalmente con *Eucalyptus* spp. (*Eucalyptus globulus* Labill. en la zona costera y *Eucalyptus nitens* H. Deane & Maiden en la zona de interior) y con pinos (*Pinus pinaster* Aiton y *Pinus radiata* D. Don, principalmente) (Pérez-Cruzado et al., 2006).

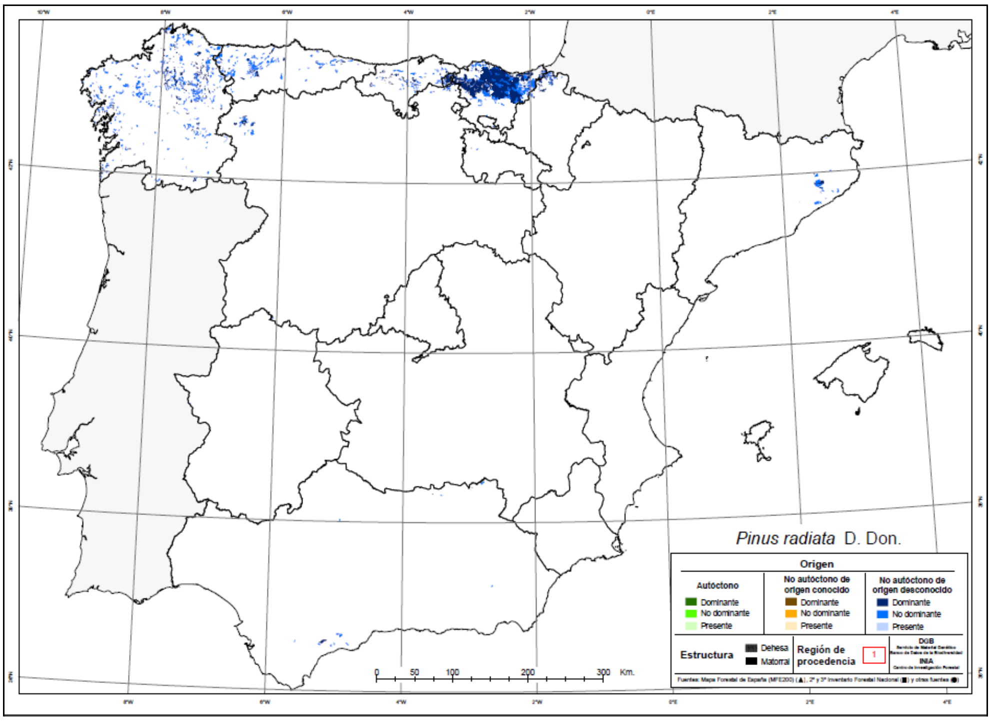

Figura 2.4: Área de distribución del *Pinus radiata* D. Don en España (Fuente: www.inia.es)

Por otra parte, el *Pinus radiata* D. Don es una especie arbórea muy empleada actualmente en el establecimiento de sistemas silvopastorales en zonas templadas como Australia, Nueva Zelanda y Chile, obteniéndose en estas zonas incrementos de beneficios económicos de hasta el 12% al realizar plantaciones con esta especie sobre pastos ya establecidos, en comparación con los beneficios económicos de los terrenos con uso de pradera (Hawke, 1991; Knowles, 1991; Benavides et al., 2009). En el norte y noroeste de España también se han establecido sistemas silvopastorales con *Pinus radiata* D. Don con el fin de disminuir la incidencia de los incendios forestales al remplazar a los arbustos altamente inflamables asociados al sotobosque del pinar por un tapiz herbáceo menos inflamable (Ibarra et al., 2000; López-Díaz et al., 2009; Rigueiro-Rodríguez et al., 2010a). La buena aceptación de esta especie para establecer sistemas silvopastorales se debe fundamentalmente a su rápido crecimiento a marcos de plantación amplios, a que tolera podas artificiales, a su dominancia apical y a que actúa como una bomba de extracción de nutrientes (Silva-Pando, 1998; Rigueiro-Rodríguez, 2000).

2.4.2. *Fraxinus excelsior* L. en los sistemas silvopastorales

Fraxinus excelsior L. (fresno) es una especie arbórea de amplia distribución en Europa y en gran parte de Galicia (Figura 2.5) que se integra perfectamente en los sistemas silvopastorales establecidos en la región biogeográfica atlántica de Europa (McAdam y Hoppe, 1996; McAdam y Sibbald, 2000).

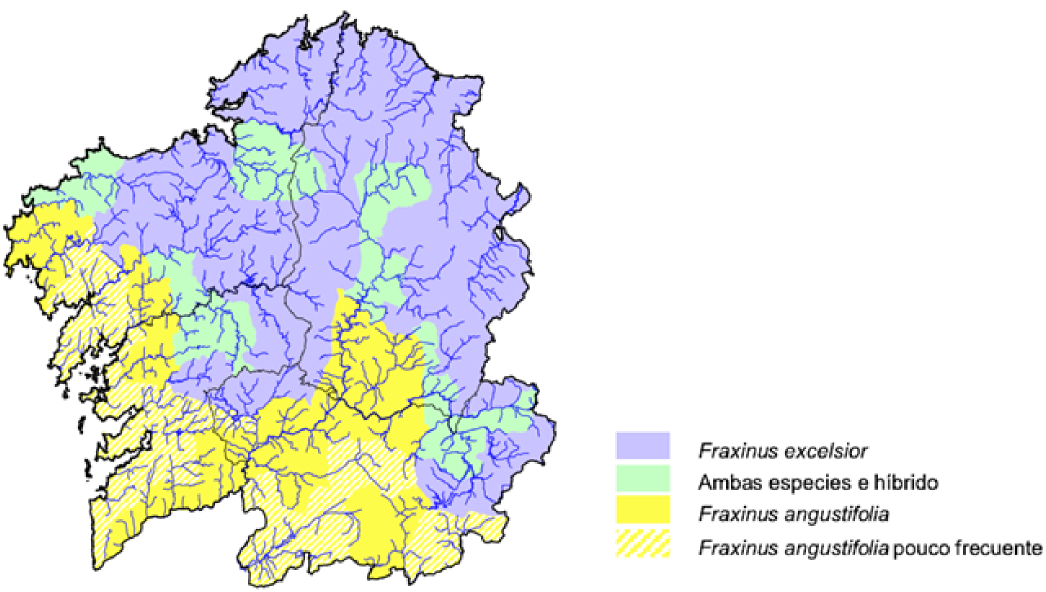

Figura 2.5: Área de distribución en Galicia del *Fraxinus excelsior* L. y otros fresnos (Fuente: Rigueiro-Rodríguez y Silva-Pando, 1990, modificado)

Los fresnos, se caracterizan por su dominancia apical y por sus raíces que alcanzan capas más profundas del suelo que las del pasto, con las que no compiten, permitiendo además un mejor reciclaje de nutrientes en el sistema en comparación con terrenos cubiertos exclusivamente por especies pratenses. Por otra parte,, como se trata de una especie de hoja caduca, durante el otoño y la primavera la cantidad de radiación interceptada es menor que en el caso de los sistemas silvopastorales establecidos bajo coníferas siempre verdes, y durante el verano proporciona sombra, reduciendo la evapotranspiración, aumentando por lo tanto la producción de pasto en comparación con el establecido bajo árboles de hojas persistentes o en zonas abiertas en áreas que presentan sequía estival, prolongando así la estación de crecimiento del pasto y la disponibilidad de forraje fresco y de buena calidad para el ganado. La copa clara de los fresnos también permite que llegue mayor cantidad de agua de lluvia al pasto, favoreciendo así su producción (McEvoy, 2004).

En el Reino Unido se realizaron ensayos para comprobar la compatibilidad del pasto con distintas especies arbóreas en sistemas silvopastorales, obteniéndose buenos resultados con *Fraxinus excelsior* L. (McAdam y Hoppe, 1996; McAdam y Sibbald, 2000). Esta especie también forma parte, junto con *Acer pseudoplatanus* L., de una red de experiencias silvopastorales establecidas en siete zonas diferentes del Reino Unido (Agroforestry Forum, 2009).

2.4.3. *Quercus rubra* L. en los sistemas silvopastorales

Quercus rubra L. (roble americano) es una especie procedente de la costa atlántica de Norteamérica, donde habita en un área amplia desde Canadá hasta Lousiana (Figura 2.6). Tiene por lo tanto una gran variedad de procedencias, lo cual explica su plasticidad (Álvarez-Álvarez et al., 2000).

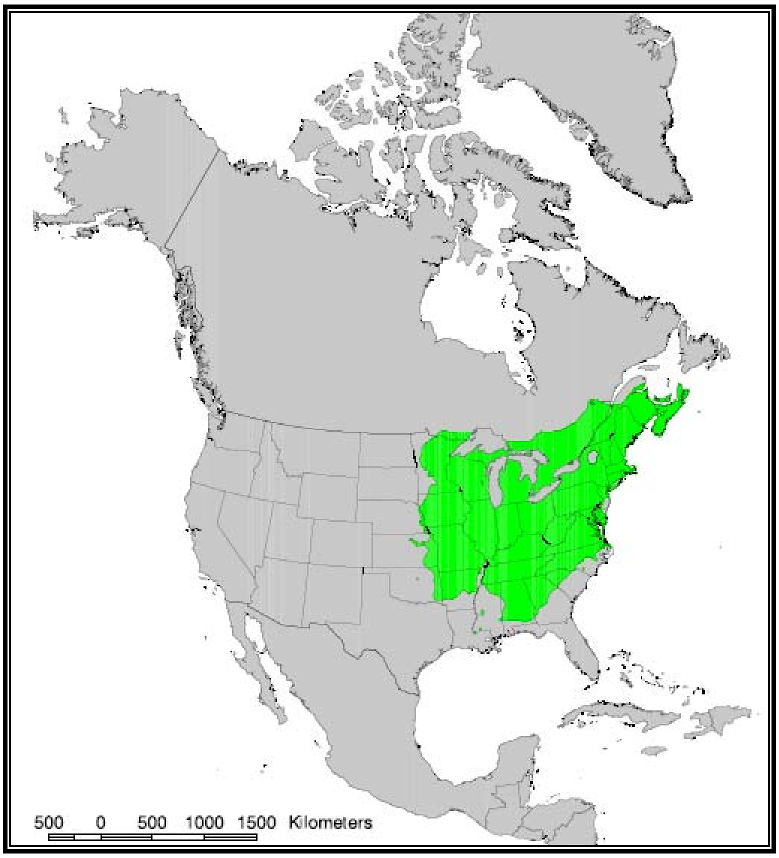

Figura 2.6: Área de distribución natural del *Quercus rubra* L. (Fuente: Sander, 1990)

En la Península Ibérica *Quercus rubra* L. está presente sobre todo en el norte y noroeste, encontrándose las repoblaciones de mayor edad y extensión en el País Vasco. En Galicia fue introducido hace más de 60 años y tuvo una buena aceptación por parte de los silvicultores dentro del programa de forestación de tierras agrarias, debido fundamentalmente a que su crecimiento es más rápido que el de la especies autóctonas de robles y, por lo tanto, permite a los silvicultores obtener beneficios económicos procedentes de la explotación forestal de forma más temprana (Renou-Wilson et al., 2008). Los resultados del III Inventario Forestal Nacional (III IFN, 1998) señalan una presencia destacada de esta especie en Galicia, con una tendencia a incrementarse.

Quercus rubra L. es una especie arbórea frecuentemente usada en el establecimiento de sistemas silvopastorales (Nuzzo, 1986; Balandier y Dupraz, 1999; Lehmkuhler et al., 2003; Rozados-Lorenzo et al., 2007) debido a su copa clara que permite el paso de la luz al pasto. Igual que *Fraxinus excelsior* L. es una especie caducifolia, que intercepta menor cantidad de luz que las coníferas siempre verdes en primavera y otoño, y durante el verano proporciona sombra a los animales y reduce la evapotranspiración incrementándose la producción de pasto. Además, dada la rapidez de su crecimiento y la calidad de su madera siempre interesará potenciar a esta especie desde un punto de vista económico (Álvarez-Álvarez et al., 2000).

2.4.4. Los pastos en los sistemas silvopastorales

La producción de pasto en una región depende de forma importante de las condiciones climáticas. En la Figura 2.7 se puede observar la distribución estacional media de la producción de pasto en Galicia, observándose que ésta no es estable en el tiempo, presentando períodos de escasez de pasto fresco para alimentar al ganado debido a la sequía en verano y al frío en invierno (Mosquera-Losada et al., 1999). Estas curvas de crecimiento se pueden ver modificadas por el efecto del arbolado, que varía con la evolución de la masa arbórea con el tiempo. En general, si no se regula adecuadamente la competencia, la producción de pasto puede verse mermada de forma importante debido al aumento de la cobertura del arbolado (Chang et al., 2002).

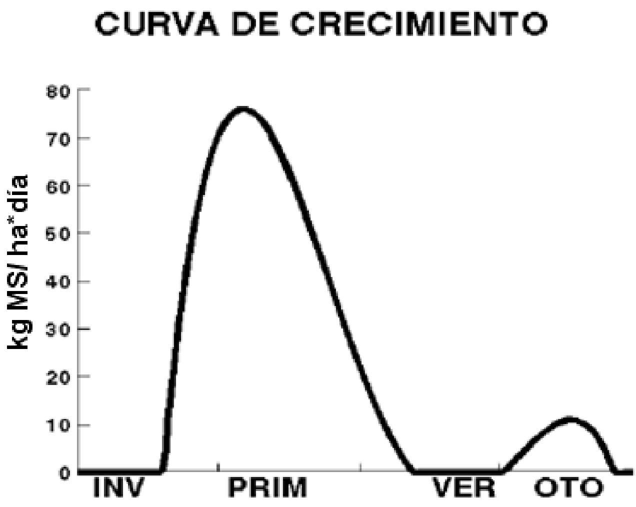

Figura 2.7: Curva de crecimiento de pasto por estación. INV: invierno; PRIM: primavera; VER: verano y OTO: otoño (Fuente: Mosquera-Losada et al., 1999).

En países como Italia, Grecia, España y Portugal la mayor parte de los sistemas silvopastorales incluyen gramíneas o leguminosas en el sotobosque para proveer de forraje a los animales (Alonso y Bento, 2005; Pardini, 2008). Cuando se van a establecer o renovar praderas en los sistemas silvopastorales es importante elegir especies pratenses que se puedan desarrollar bien en condiciones de sombra, ya que la producción de pasto en el sotobosque puede verse limitada si la densidad del arbolado es elevada, debido al efecto de la sombra del componente arbóreo. La densidad del arbolado y sobre todo su cobertura condicionan la producción potencial del pasto. Por lo tanto, debería buscarse un equilibrio entre la producción de pasto y la maderera a través de una densidad adecuada del arbolado. *Dactylis glomerata* L. es una especie pratense adecuada para establecer un tapiz herbáceo artificial en los sistemas silvopastorales, ya que se adapta bien a condiciones de sombra (Mosquera-Losada et al., 2001, 2006). *Trifolium repens* L. es una leguminosa que también podría establecerse en la pradera del sistema silvopastoral porque mejoraría la calidad del pasto, la producción y el crecimiento del arbolado debido a su capacidad de fijar nitrógeno (Whitehead, 1995; López-Díaz et al., 2009).

En la Figura 2.8 se puede apreciar la influencia de la cobertura del arbolado sobre la productividad del sotobosque. Si la cobertura de los árboles es demasiado elevada la producción y la calidad de pasto no serán suficientes para que el crecimiento y producción del ganado sean adecuados (McAdam y McEvoy, 2008). Sin embargo, la presencia de árboles en zonas de pastoreo tiene las ventajas de que proporcionan sombra en verano a los animales y actúan como barreras cortavientos en el invierno (Pardini, 2008).

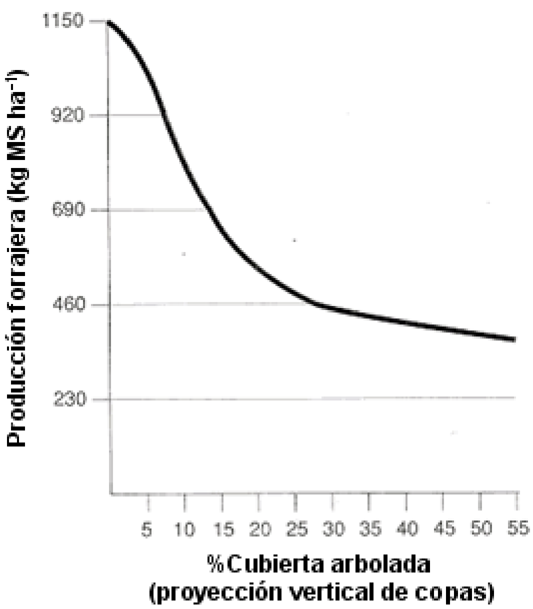

Figura 2.8: Variación de la producción de pasto natural en función de la densidad arbórea (Fuente: Rigueiro-Rodríguez et al., 1997)

2.5. LA FERTILIZACIÓN EN LOS SISTEMAS SILVOPASTORALES

En Galicia, la producción de pasto en los sistemas silvopastorales se ve limitada por la baja fertilidad de los suelos debido a su acidez natural (Zas y Alonso, 2002), que se asocia a un alto porcentaje de saturación de Al en el complejo de cambio y baja disponibilidad de cationes y fósforo (Prasad y Power, 1997; Rigueiro-Rodríguez et al., 2007).

El uso de fertilizantes orgánicos e inorgánicos es una herramienta de gestión que permite el incremento de la producción de pasto en los sistemas silvopastorales (Mosquera-Losada et al., 2006). No obstante, en las primeras etapas del desarrollo de los sistema silvopastorales establecidos mediante repoblación sobre pradera artificial la competencia entre los árboles y el pasto puede ser alta (Nair y Graetz, 2004; Rigueiro-Rodríguez et al., 2008a), por lo que deben potenciarse estrategias de fertilización que favorezcan el crecimiento conjunto del pasto y el arbolado. Sin embargo, a medida que los árboles crecen y la sombra del arbolado va limitando el crecimiento de la vegetación herbácea, el efecto de la fertilización será menor. En este momento, la reducida respuesta a la fertilización del pasto bajo arbolado indicará que la luz, más que los nutrientes, es el factor que restringe el crecimiento (McAdam y Sibbald, 2000).

El uso de los fertilizantes nitrogenados inorgánicos en la Unión Europea se está viendo reducido por el incremento que han experimentado últimamente los precios y por los efectos negativos de tipo ambiental de la aplicación de altas dosis de nitrógeno mineral al suelo (EFMA, 2009). Una buena opción sería el uso de los lodos de depuradora urbana como fertilizante orgánico de los sistemas silvopastorales, ya que son ricos en materia orgánica y macronutrientes, especialmente en nitrógeno y fósforo (MMA, 2006) y son en la actualidad una fuente de nitrógeno interesante desde un punto de vista económico. Por otra parte, su utilización como abono está promovida por la Unión Europea con el objeto de valorizar su uso y dar una salida adecuada a este residuo (UE, 1986).

2.6. LOS LODOS DE DEPURADORA URBANA

En este estudio se han utilizado como fertilizantes orgánicos los lodos procedentes de la depuración de aguas residuales urbanas. Comentaremos el proceso de formación de los mismos, desde la recogida de las aguas residuales y su tratamiento, así como su uso agronómico.

2.6.1. Las aguas residuales urbanas

Según la Directiva 91/271/CEE (UE, 1991) "se entiende por aguas residuales urbanas las aguas residuales domésticas producidas básicamente por el metabolismo humano y las actividades domésticas (aguas residuales domésticas) o bien la mezcla de las mismas con aguas residuales procedentes de locales comerciales o industriales (aguas residuales industriales) y/o aguas de escorrentía pluvial".

La Unión Europea a través de la Directiva 91/271/CEE (UE, 1991) obligaba a que los núcleos de población con más de 15.000 habitantes dispusieran de sistemas colectores de aguas residuales urbanas desde el 31 de diciembre del 2000, mientras que en el caso de núcleos urbanos de entre 2.000 y 15.000 habitantes se establecía una prórroga hasta el 31 de diciembre del 2005. Al mismo tiempo se exigía que estas aguas se sometieran a tratamientos secundarios antes de su vertido en núcleos con más de 15.000 habitantes, y en núcleos de más de 10.000 habitantes a partir del 31 de diciembre del 2005 (a partir del 2000 para los vertidos en aguas dulces o estuarios). Todo ello ha dado lugar al aumento del número de plantas depuradoras de aguas residuales y, por tanto, de la producción de lodos residuales (Figura 2.9), que son los residuos que se obtienen una vez que se tratan las aguas residuales urbanas.

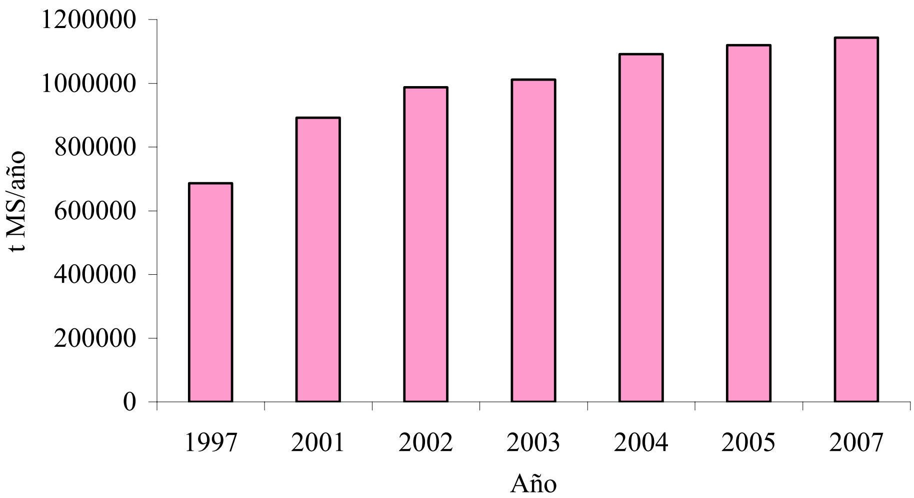

Figura 2.9: Evolución de la generación anual de lodo en España (t MS/año) (Fuente: MARM, 2008).

Como se observa en la gráfica anterior, entre los años 1997 y 2007 la producción de lodo en España aumentó en un 40% (MARM, 2008). Los datos de producción de lodo por Comunidades Autónomas en el año 2007 (Tabla 2.3), extraídos del Anuario de Estadística del año 2008, sitúan a Galicia en el sexto puesto a nivel nacional en la producción de lodo, que se corresponde con más de un 4% de la producción total a nivel nacional.

Debido al notable aumento de la producción de lodos en los últimos años, el hecho de eliminar, transformar o depositar los lodos de depuradora se ha convertido en un problema de gran dimensión. Actualmente, la gestión de lodos de depuradora está regulada por la directiva 86/278/CEE (UE, 1986), que establece las condiciones en las que podrán ser aplicados los lodos como fertilizante.

Comunidad Autónoma	Producción de lodo (t MS)
Cataluña	309.960
Madrid	238.229
Comunidad Valenciana	165.447
Andalucía	102.889
Baleares	66.811
Galicia	49.375
Castilla y León	45.361
Castilla La Mancha	35.746
Canarias	35.185
Aragón	31.986
País Vasco	24.399
Navarra	9.886
Extremadura	9.430
Cantabria	7.733
La Rioja	4.686
R. de Murcia	2.731
P. de Asturias	2.229
Ceuta y Melilla	1.460
TOTAL	1.143.553

Tabla 2.3: Producción de lodo por Comunidad Autónoma en el año 2007 (Fuente: MARM, 2008).

2.6.2. Tratamiento de las aguas residuales urbanas

Las plantas de depuración de aguas residuales urbanas funcionan mediante diferentes procesos de tipo mecánico, biológico, químico o combinaciones de los mismos. En su mayoría los procesos de depuración incluyen varias etapas:

✓ Pretratamiento de las aguas: se eliminan principalmente los sólidos de mayor tamaño. Mediante unas rejas colocadas en un canal de entrada a la estación depuradora quedan retenidos los cuerpos más voluminosos, y en unas instalaciones de desarenado y desengrasado se eliminan las arenas por decantación y las grasas por desemulsión y recogida superficial.

✓ Tratamiento primario: mediante una decantación las partículas suspendidas más pesadas se separan sedimentándose en el fondo de unos tanques denominados decantadores primarios, que disponen de unos mecanismos de arrastre y extracción de los fangos que se van depositando en el fondo.

✓ Tratamiento secundario: su finalidad es la reducción de la materia orgánica presente en las aguas residuales una vez superadas las fases de pretratamiento y tratamiento primario. El tratamiento secundario más comúnmente utilizado es un proceso biológico aerobio seguido por una decantación, denominada secundaria. El proceso biológico puede llevarse a cabo por distintos procedimientos. Los más usuales son el proceso denominado fangos activos y el denominado de lechos bacterianos o percoladores.

✓ Tratamiento terciario: constituye un complemento de la depuración de las aguas residuales para adaptar la calidad de las mismas a su destino final o uso posterior.

2.6.3. Tratamiento de estabilización de los lodos obtenidos

En el tratamiento de las aguas residuales se obtienen grandes volúmenes de lodo con un reducido contenido en sólidos altamente biodegradables. Para el uso de estos lodos como fertilizante en el suelo, en España el Real Decreto 1310/1990 (BOE, 1990) obliga a someter los lodos a procesos de estabilización que se conocen como post-tratamientos, los cuales básicamente buscan disminuir el contenido de agua del lodo, eliminar su potencial contaminante mediante la reducción de nitratos, malos olores y carga bacteriana y facilitar su manejo. La Unión Europea promueve el empleo como fertilizantes de los lodos estabilizados mediante digestión anaeróbica y compostaje, sin embargo, ambos tipos tienen elevadas proporciones de agua, que se reducen en los lodos sometidos a secado térmico y posterior peletizado (EEA, 2000), tratamientos que abaratan el transporte y facilitan el almacenaje y su aplicación al suelo. Estos procesos de estabilización se pueden resumir de la siguiente forma:

✓ Digestión anaeróbica: las bacterias degradan la materia orgánica en ausencia de oxígeno hasta formar compuestos inertes más o menos estables. El proceso tiene lugar en unos depósitos cerrados en cuyo fondo se depositan los fangos digeridos. Durante el proceso de digestión se produce un gas biológico, con un contenido de metano del 65-70%, que puede convertirse en un subproducto valioso a través del cual se genera una gran parte de la energía que la planta depuradora necesita para su funcionamiento. Este proceso es caro, pero el volumen tratado es mayor en comparación con el aeróbico. La digestión anaeróbica es el tratamiento más empleado en grandes ciudades como Madrid, Barcelona y Vigo, y también se emplea en Lugo.

✓ Compostaje: el lodo compostado se obtiene a partir de un proceso biológico aeróbico, mediante el cual los microorganismos actúan sobre la materia rápidamente biodegradable permitiendo obtener "compost". En el proceso de compostaje los biosólidos son mezclados con restos de biomasa triturada (restos de podas, serrín u otros restos vegetales), la cual actúa como agente estructurante y permite obtener un producto de calidad y valor agronómico. El compostaje puede ser una vía importante de estabilización de los sólidos de los lodos y además garantiza los siguientes criterios de calidad: materia seca mínima del 40%, práctica ausencia de patógenos, fermentación aerobia completa (3 días a más de 55ºC) y características físicas uniformes.

✓ Peletizado: el lodo peletizado se obtiene mediante un secado térmico que reduce el volumen del lodo por la evaporación del agua superficial y el agua de los capilares, transformando el lodo en un producto seco, en forma de pellets o bolas de entre 1 y 3 mm de diámetro.

Las características de los lodos, su contenido en nutrientes (Mosquera-Losada et al., 2010b) y la tasa de incorporación al suelo (EPA, 1994) varían en función de los procesos de estabilización a los que deben ser sometidos en las plantas de depuración. La incorporación al suelo y su tasa de mineralización depende también del clima local.

La EPA (Agencia Medio Ambiental Estadounidense) señalaba en 1994 la importancia del conocimiento de la concentración de nitrógeno en los distintos tipos de lodo y de la tasa de mineralización anual, ya que estas variables influirán de forma importante en las recomendaciones de manejo de estos residuos cuando se pretende su uso como fertilizante. En este sentido la EPA (1994) pone de manifiesto que existen claras diferencias entre los lodos estabilizados mediante compostaje y digestión aeróbica o anaeróbica en relación a esos parámetros, señalando que la digestión anaeróbica produce lodos con una proporción de nitrógeno total disponible en el primer, segundo y tercer año tras la aplicación del 20, 10 y 5%, resultando estas cifras del 10, 5 y 2,5% si la estabilización del lodo es mediante compostaje. La EPA (1994) no cifra la tasa de mineralización anual para el lodo peletizado, pero, si es lodo de digestión anaerobia sometido a un proceso de secado térmico, se le supone la tasa de mineralización anual de los lodos digeridos anaeróbicamente.

2.6.4. Normativa del empleo de los lodos en la agricultura

Los lodos de depuradora urbana se han eliminado tradicionalmente por diferentes vías, entre las más utilizadas se encuentran la descarga a vertederos (cuya tasa de llenado se acelera), la incineración (proceso contaminante y de coste elevado), el vertido al mar (prohibido desde 1999) y la aplicación en la agricultura como fertilizante.

La UE promueve el uso del lodo de depuradora urbana como fertilizante del suelo debido a su alto contenido en materia orgánica y macronutrientes, principalmente nitrógeno y fósforo (MMA, 2006). No obstante, debe hacerse un estudio pormenorizado de estos residuos, ya que, debido a su procedencia y a los procesos físico-químicos que intervienen en el tratamiento de las aguas residuales, el lodo tiende a concentrar trazas de metales pesados y compuestos orgánicos (productos químicos de uso doméstico, productos fitosanitarios, etc.) poco biodegradables que pueden provocar efectos nocivos sobre el suelo, la vegetación, los animales y la salud humana.

En España el Real Decreto 1310/1990 (BOE, 1990) recoge toda la normativa que regula la utilización agrícola de los lodos de depuradora en este país. Establece unos valores límite de concentración de metales pesados en el suelo y en el lodo, similares a los establecidos por la Directiva Europea 86/278/CEE (UE, 1986). De este Real Decreto, es especialmente importante, desde nuestro punto de vista, el artículo 3º que señala entre otros aspectos:

✓ Los suelos sobre los que podrán aplicarse los lodos tratados deberán presentar una concentración de metales pesados inferior a la establecida en la Tabla 2.4.

	Valores límite (mg kg^{-1})	
Parámetros	Suelos con pH<7	Suelos con pH>7
Cadmio	1	3
Cobre	50	210
Cromo	100	150
Níquel	30	112
Mercurio	1	1,5
Plomo	50	300
Zinc	150	450

Tabla 2.4: Concentración máxima de metales pesados en suelos susceptibles de ser fertilizados con lodos. (Fuente: BOE, 1990).

✓ Los lodos tratados a utilizar en los suelos no excederán, en cuanto al contenido en metales pesados, de los valores límite recogidos en la Tabla 2.5.

	Valores límite (mg kg^{-1})	
Parámetros	Suelos con pH<7	Suelos con pH>7
Cadmio	20	40
Cobre	1000	1750
Cromo	1000	1500
Níquel	300	400
Mercurio	16	25
Plomo	750	1200
Zinc	2500	4000

Tabla 2.5: Concentración máxima de metales pesados en lodos destinados a uso agrícola o forestal (Fuente: BOE, 1990).

✓ Las cantidades máximas de lodos que podrán aportarse al suelo por hectárea y año serán las que, de acuerdo con el contenido en metales pesados de los suelos y lodos a aplicar, no rebasen los valores límite de incorporación de los metales pesados establecidos en la Tabla 2.6.

Parámetros	Valores límite (kg $ha^{-1}año^{-1}$)
Cadmio	0,15
Cobre	12
Cromo	3
Níquel	3
Mercurio	0,1
Plomo	15
Zinc	30

Tabla 2.6: Valores límite de incorporación de metales pesados en suelos basándose en una media de 10 años (Fuente: BOE, 1990).

El Real Decreto 1310/1990 (BOE, 1990) también establece algunas prohibiciones como: “Aplicar lodos tratados en praderas, pastizales y demás aprovechamientos a utilizar en pastoreo directo por el ganado, con una antelación menor de tres semanas respecto a la fecha de comienzo del citado aprovechamiento directo”.

2.6.5. Las dosis de lodo de depuradora

La fertilización orgánica difiere de la inorgánica en que, por lo general, la disponibilidad de los nutrientes es menor en la primera en comparación con la segunda. Esto se debe a que el fertilizante orgánico debe mineralizarse y este proceso es lento y depende de las condiciones existentes en el suelo, que intervienen en el desarrollo de la población microbiana encargada de esta mineralización, como son la temperatura, el contenido en humedad o el pH. De este modo, a la hora de planificar la fertilización orgánica con lodos de depuradora urbana hemos de tener en cuenta el ciclo de los nutrientes que contiene el lodo y las necesidades del cultivo, al igual que en el caso de la fertilización inorgánica, pero a mayores hemos de considerar la proporción de nitrógeno que es fácilmente mineralizable durante el primer año tras la aplicación del lodo (Barry et al., 1986; EPA, 1994; Smith, 1996). La EPA (1994) nos indica que si las dosis de lodo de depuradora son muy superiores a las necesidades del cultivo existe riesgo de que se produzca un lavado de nitratos a través del perfil del suelo, provocando una contaminación de las aguas, como citan para el País Vasco Egiarte et al. (2009). Por ello se deben aplicar dosis adecuadas de lodo en el momento en el que la absorción por parte de los cultivos sea máxima. Por lo tanto, al establecer las dosis de lodo que se van a aplicar hay que tener en cuenta, además de los aspectos económicos, los medioambientales, ya que la eficiencia de los fertilizantes nitrogenados disminuye a partir de una determinada dosis, a partir de la cual no se produce respuesta alguna en el crecimiento del pasto y del arbolado, e incluso, si se sigue aumentando, la respuesta en producción puede ser negativa (Whitehead, 1995), originando así problemas ambientales y pérdidas económicas.

En los sistemas silvopastorales, normalmente el componente arbóreo tiene una menor demanda anual de nutrientes que la vegetación herbácea, debido a que su crecimiento es mucho más lento. Por ello, en las primeras edades del arbolado, la dosis de lodo de depuradora a aplicar será aquella que garantice un crecimiento adecuado del pasto, siempre que no perjudique al arbolado. En algunos casos las necesidades de fertilización pueden ser menores que en un sistema agrícola tradicional, debido a la mejora en el ciclo de nutrientes que se produce en los sistemas agroforestales (López-Díaz, 2004).

A pesar de que hay normas generales para un uso adecuado de lodos de depuradora urbana en relación a la dosis a aportar, su potencial contaminante depende, entre otros aspectos, de las condiciones climáticas del año concreto de aplicación, así como del suelo. En estudios realizados en el País Vasco con *Pinus radiata* D. Don por Egiarte et al. (2009) se ha observado que hasta una dosis 2,4 t ha año^{-1} de lodo depuradora no había riesgo de contaminación de las aguas subterráneas, pero si la dosis se incrementaba hasta 60 t ha año^{-1} se producía contaminación de los acuíferos subterráneos por exceso de NO_3^--N y NH_4^+-N, con picos de 104,2 y 48,7 mg l^{-1}, respectivamente.

2.6.6. Efecto del aporte de lodo sobre el suelo

La aplicación de lodo afecta a las propiedades físicas y químicas del suelo. En el caso de las propiedades físicas, este tipo de residuos mejora la estructura y la estabilidad de los agregados, gracias al aporte de materia orgánica que se realiza, lo que da lugar a un incremento en la permeabilidad y en la retención hídrica (Navarro-Pedreño, 1995). En cuanto a las propiedades químicas podemos destacar:

a) Efecto del aporte de lodo sobre el pH del suelo

El efecto del aporte del lodo de depuradora urbana sobre el pH del suelo depende del tipo de lodo que se emplee y del suelo en el que se aplique. Estudios desarrollados en suelos muy ácidos muestran un incremento del pH debido al aporte de cationes que se realiza, en especial Ca (López-Díaz et al., 2007). Sin embargo, en suelos de menor acidez o próximos a la neutralidad,, el incremento del pH del suelo no resulta ser relevante (Mosquera-Losada et al., 2006), debido a que el aporte de lodo incrementa en mayor medida la producción de pasto y el crecimiento del arbolado, lo cual supone una importante extracción de cationes del suelo.

b) Efecto del aporte de lodo sobre el contenido de materia orgánica del suelo

La fertilización con lodos a corto plazo no suele producir modificaciones en los contenidos de materia orgánica del suelo (López-Mosquera et al., 2002), debido a que las dosis aplicadas no son lo suficientemente elevadas como para que se note un efecto en un primer momento en los suelos de Galicia y al incremento en el ritmo de mineralización (López-Díaz et al., 1999). Sin embargo, con las sucesivas aplicaciones de lodo al suelo el incremento en el contenido de materia orgánica se va haciendo más evidente (Andrade-Couce et al., 1985; Tsadilas et al., 1995; Krebs et al., 1998) y aumenta al hacerlo las dosis aplicadas (Canet et al., 1996). En sistemas silvopastorales establecidos en Galicia con *Pinus radiata* D. Don se observó un incremento de la materia orgánica del suelo debido a la fertilización con lodo y a los insumos de materia orgánica procedentes de los restos del arbolado y del pasto (Fernández-Núñez et al., 2010a).

c) Efecto del aporte de lodo sobre la CIC del suelo

La capacidad de intercambio catiónico de un suelo puede verse mejorada tras el aporte de lodo, sobre todo en suelos arenosos y si se incrementa el contenido de materia orgánica (Piccolo et al., 1992). Sin embargo, también existen suelos en los que esto no ocurre, debido a la rápida mineralización del lodo o a que el suelo al que se aporta posee una elevada proporción de materia orgánica en comparación a la aplicada con el lodo (Gigliotti et al., 2001). Así, el aporte de lodo ocasionó un incremento de la fertilidad del suelo en terreno de monte al ser el Al reemplazado por el Ca en el complejo de cambio (López-Díaz et al., 2007) y este efecto no fue observado en terreno agrícola (Mosquera-Losada *et al.*, 2006).

d) Efecto del aporte de lodo sobre los niveles de macronutrientes del suelo

El efecto sobre los niveles de macronutrientes en el suelo varía en función del tipo de estabilización que haya sufrido el lodo antes de ser empleado (Mosquera-Losada et al., 2010b), de la dosis aplicada, de la tasa de mineralización y del tipo de suelo en el que se aporta. Los lodos procedentes de digestión anaeróbica y aeróbica poseen una menor proporción de cationes que los compostados, que son a su vez menos ricos en nitrógeno (Mosquera-Losada et al., 2010b).

En general, la aplicación de lodo de depuradora urbana al suelo produce un incremento de los niveles de nitrógeno en el mismo (Rodríguez-Barreira 2007), lo que se relaciona con el aumento de materia orgánica. Por otra parte, el incremento del pH debido a la aplicación del lodo de depuradora sobre suelos ácidos deriva de la reducción de la presencia de aluminio en el complejo de cambio, debido a los aportes de calcio que se producen con este residuo, tal y como encontraron Andrade-Couce et al., (1985), Vivekanandan et al., (1991) y López-Díaz (2004). Otros estudios demuestran un aumento de la presencia de fósforo total, principalmente cuando existe una carencia del mismo (Mosquera-Losada et al., 2008b), del nitrógeno disponible y del fósforo intercambiable (Cucci et al., 2008). En relación al magnesio, la aplicación de lodo aumenta los niveles de este elemento en el suelo e incrementa su disponibilidad al aumentar el pH del suelo (Vivekanandan et al., 1991; López-Mosquera et al., 2002;

López-Díaz, 2004). Finalmente, la cantidad de sodio también suele incrementarse debido a la descomposición de estos residuos (López-Mosquera et al., 2002), aunque este efecto se reduce con el tiempo debido a la alta solubilidad de este elemento. Hay que señalar, que en algunos estudios (Giddens et al., 1997; Mosquera-Losada et al., 2006) se observó un incremento de la concentración del sodio en el suelo que no se atribuyó a la fertilización, ya que la cantidad de sodio aumentaba debido a que el arbolado interceptaba en el aire sal procedente del mar y este se transfería al suelo.

e) Efecto del aporte de lodo sobre los niveles de micronutrientes y metales pesados del suelo

Hay que tener en cuenta que debido a su naturaleza, los lodos de depuradora pueden contener elementos como los metales pesados que, a determinada concentración, pueden encontrarse disponibles para las plantas, con la posibilidad de producirse su entrada en la cadena trófica y provocar problemas de toxicidad. Por lo tanto, uno de los principales problemas del uso de lodos de depuradora urbana en la agricultura está relacionado con el mayor nivel de metales pesados que poseen en comparación con el suelo (Smith, 1996; Mosquera-Losada et al., 2010b). Si bien los suelos tienen cierta capacidad para retener contaminantes, cuando esta capacidad se excede se pueden producir daños en los recursos naturales, tales como aguas superficiales y subterráneas, plantas y animales, incluido el hombre (Calmano et al., 2001), ya que los metales pesados son carcinogénicos y pueden provocar importantes alteraciones en la fisiología y funcionamiento de plantas y animales. Por otra parte, y según McGrath et al., (1995), los metales pesados contenidos en el lodo pueden actuar como contaminantes en el suelo reduciendo la riqueza de microorganismos y causando problemas en la fertilidad edáfica. Por ello, el uso de los lodos de depuradora urbana como fertilizante está regulado tanto a nivel europeo, Directiva Europea 86/278/CEE (UE, 1986), como nacional, RD 1310/90 (BOE, 1990), siendo mucho más restrictivas estas regulaciones en el caso de los suelos ácidos, debido al incremento de la disponibilidad de los metales en los mismos (Figura 2.10), ya que la disponibilidad de algunos elementos como el hierro, manganeso, zinc y cobre aumenta cuando el pH del suelo pasa de básico a ácido (Porta et al., 2003).

En este sentido, para el empleo de lodo como fertilizante interesan los lodos procedentes de efluentes urbanos con una preponderancia doméstica igual o superior al 70% del volumen de agua tratado, así como los que se obtengan de efluentes industriales con un origen exclusivamente agroalimentario, ya que esto limitaría considerablemente la cantidad de metales pesados. Las aguas residuales con un componente industrial superior al 30% y las que proceden de operaciones industriales no agroalimentarias se considera que generan lodos con un mayor contenido en metales pesados, lo que limitaría de forma considerable su empleo agrícola.

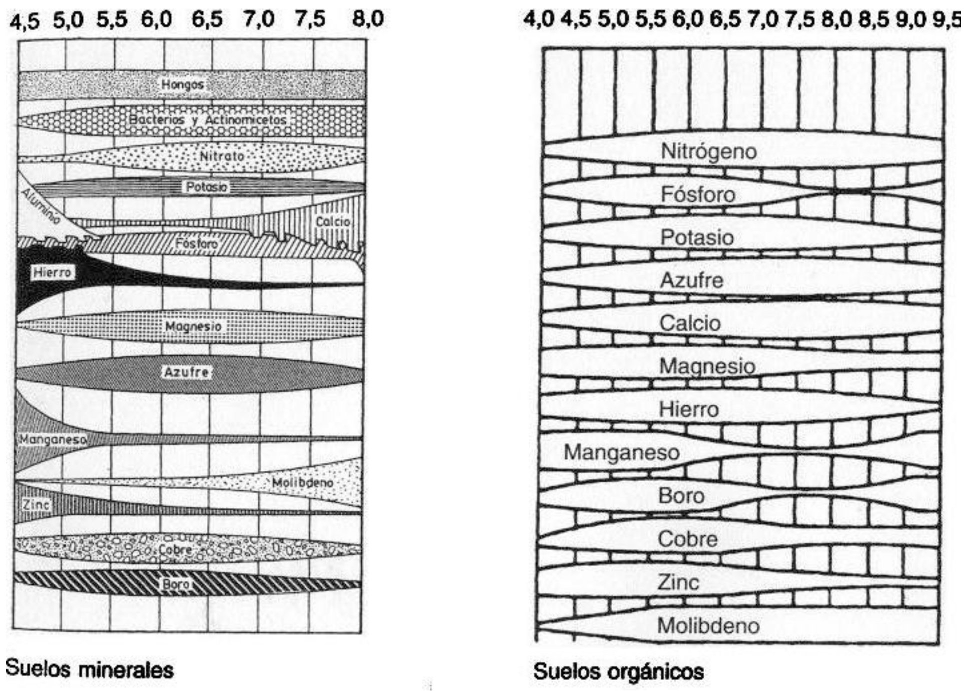

Figura 2.10: Disponibilidad de los nutrientes en relación con el pH según el tipo de suelo (Fuente: Porta et al., 2003).

El efecto del lodo de depuradora sobre los niveles de metales pesados en el suelo va a depender de la concentración de metales pesados del fango. Así, el lodo suele tener elevados niveles de hierro y, entre los metales pesados regulados por la legislación española (cromo, zinc, cobre, mercurio, cadmio, níquel y plomo), es más rico en zinc y, por este orden, en cobre y cromo (Smith, 1996; Mosquera-Losada et al., 2010b). Por lo tanto, los principales riesgos de toxicidad son debidos al zinc, que es fácilmente absorbido, así como al cobre, que junto al zinc y al níquel pueden existir en gran cantidad en las aguas residuales contaminadas por las industrias. El cadmio es el elemento que más probabilidades tiene de ser tóxico para las personas que consumen productos procedentes de suelos tratados con lodos residuales, por ser fácilmente absorbido por las plantas. Afortunadamente, el plomo y el cromo, que son los principales contaminantes de ciertas zonas, no son absorbidos con tanta facilidad por las plantas como el cadmio. Estos elementos son acumulables en el suelo y pueden pasar bastantes años hasta que desaparezcan sus efectos tóxicos (Simpson, 1986; Pomares y Canet, 2001).

En general el aporte de lodo durante períodos de tiempo prolongados provoca aumentos en el suelo del contenido de hierro (López-Díaz, 2004), manganeso (Andrade-Couce et al., 1985; López-Díaz, 2004), cromo (Kabata-Pendías y Pendías, 2001), zinc, cobre y plomo (Andrade-Couce et al., 1985; Mosquera-Losada et al., 2010b, 2009b), cadmio (Canet et al., 1998) y níquel (Canet et al., 1996; López-Díaz, 2004; Rodríguez-Barreira, 2007), siendo la disponibilidad del plomo reducida debido a que este elemento se liga a la materia orgánica del suelo (Canet et al., 1998).

2.6.7. Efecto del aporte de lodo sobre el crecimiento del arbolado en sistemas silvopastorales

El aporte de lodos de depuradora como fertilizante suele mejorar el desarrollo del arbolado de especies de crecimiento rápido (Wolstenholme, 1992) en comparación con los abonos minerales de liberación más rápida, ya que el lodo es un abono que libera lentamente los nutrientes, lo que es sumamente importante para que el árbol, con menor tasa de crecimiento que el pasto, sea capaz de aprovechar mejor los nutrientes liberados, en comparación con los abonos minerales de liberación más rápida.

El efecto del aporte del lodo sobre el crecimiento del arbolado va a depender de la relación de competencia pasto-arbolado. Así, en terrenos agrícolas, cuando se realizan aportes de fertilizante inorgánico en un sistema silvopastoral desarrollado con *Pinus radiata* D. Don, se observa que aumenta la producción de pasto, lo que merma inicialmente el desarrollo del arbolado; sin embargo, cuando no se fertiliza se produce una merma en la producción de pasto, lo que incrementa el crecimiento del arbolado (Rigueiro-Rodríguez et al., 2000). El aporte de lodos mejora la producción de pasto,, al liberar nutrientes y mejorar la capacidad de retención de humedad del suelo, favoreciendo también el desarrollo del arbolado (Rigueiro-Rodríguez et al., 2000). Sin embargo, en terrenos de monte, cuando las dosis de lodo aplicadas son bajas, el crecimiento del arbolado es menor (López-Díaz et al., 2007), al aumentar el pH del suelo y liberarse nutrientes del lodo en mayor medida, lo que favorece la producción de pasto, que ejerce una fuerte competencia con el arbolado, reduciendo su crecimiento.

2.6.8. Efecto del aporte de lodo sobre el pasto en sistemas silvopastorales

a) Efecto del aporte de lodo sobre la producción de pasto

Existen diversas investigaciones que estudian el efecto de la fertilización con lodo sobre la producción de pasto en Galicia (Mosquera-Losada et al., 2006; López-Díaz et al., 2007; Rigueiro-Rodríguez et al., 2008b) y en otras partes del mundo (Sibbald et al., 2001; Etienne, 2005; Pontes et al., 2007).

El efecto de la fertilización con lodo sobre la producción de pasto en sistemas agroforestales depende de la capacidad de desarrollo del pasto, que se ve afectada por la cantidad de luz que le llega y, por tanto, por la cobertura forestal. En sistemas silvopastorales muy densos o con coberturas elevadas la respuesta del pasto a la fertilización está muy mermada debido a la falta de luz; sin embargo, en plantaciones jóvenes o con coberturas reducidas, la producción de pasto en sistemas exclusivamente pascícolas o en sistemas silvopastorales se ve favorecida tal y como ocurre en masas con *Pinus radiata* D. Don, tanto en suelos forestales (Mosquera-Losada et al., 2001; López-Díaz et al., 2007; López-Díaz et al., 2009) como en suelos agrícolas (Mosquera-Losada et al., 2006), en masas de *Betula alba* L. (Mosquera-Losada et al., 2006), de *Populus* x *canadensis* Moench (Mosquera-Losada et al., 2010c), *Ecalyptus globulus* Labill. y *Pinus pinaster* Aiton. (Mosquera-Losada y Rigueiro-Rodríguez, 2007).

b) Efecto del aporte de lodo sobre la composición botánica

En general, el empleo de lodos de depuradora incrementa la presencia de las especies gramíneas sembradas en el pasto (Mosquera et al., 2001; López-Díaz, 2004), que son más exigentes en fertilidad edáfica, en comparación con especies del género agrostis, con lo que se mejora la calidad y productividad del pasto (Mosquera et al., 1999). La respuesta a la fertilización orgánica es mayor en el caso de las gramíneas que en el de las leguminosas, ya que las primeras se benefician de los aportes de nitrógeno, perjudicándoles a las segundas dosis superiores a 120 kg N ha^{-1} (González, 1992; López-Díaz et al., 1999). En estudios realizados por Mosquera-Losada et al., (2009a) en sistemas silvopastorales establecidos con *Pinus radiata* D. Don, se encontró un aumento del número de especies vegetales vasculares después de seis años de experimento debido a los aportes de lodo de depuradora que se hicieron al suelo.

c) Efecto del aporte de lodo sobre la calidad del pasto

El efecto de la aplicación de lodo de depuradora sobre la calidad de pasto va a depender de las dosis de lodo aplicadas y de la presencia de especies sembradas en el pasto (López-Díaz et al., 2007). En diversos estudios llevados a cabo en suelos ácidos se ha observado que la aplicación de lodo de depuradora incrementaba el pH del suelo y por lo tanto reducía el porcentaje de saturación del Al en el complejo de cambio, provocando así un aumento de las especies sembradas (*Lolium perenne* L., *Dactylis glomerata* L. y *Trifolium repens* L.), mejorando la calidad del pasto (López-Díaz et al., 2007; Rigueiro-Rodríguez et al., 2007). El efecto contrario fue encontrado por Rigueiro-Rodríguez et al. (2005c) en un estudio en el que la proporción de especies no sembradas era mucho mayor que el de las especies sembradas. En general, se puede decir que el uso del lodo de depuradora urbana como fertilizante incrementa las concentraciones en el pasto de proteína (Whitehead, 1995; López-Díaz, 2004), de fósforo (Vivekanandan et al., 1991; López-Díaz, 2004; Rodríguez-Barreira, 2007), de potasio (Tiffany et al., 2000a; López-Díaz, 2004), de calcio (Mosquera et al., 2001; López-Díaz, 2004), de sodio (Tiffany et al., 2000a) y de magnesio (Vivekanan et al., 1991; Tiffany et al., 2000a; López-Díaz, 2004).

Por otro lado, los metales pesados contenidos en los lodos de depuradora pueden ser absorbidos por las plantas, Y si éstas no se extraen vuelven a restituir nuevamente estos elementos al suelo en forma relativamente asimilable, lo que provocará con el tiempo una acumulación de metales pesados con posibilidad de un creciente efecto nocivo. Si por el contrario la cosecha se destina a la alimentación animal, el riesgo podrá ser más grave o por lo menos más inmediato (Costa-Yagüe et al., 1987). De todas formas, hay que tener en consideración, que uno de los factores principales para la posibilidad de acumulación de los metales pesados es el tipo de planta que se desarrolla en el suelo fertilizado con lodos de depuradora, ya que las especies vegetales tienen una tolerancia muy variable con respeto a los metales tóxicos, existiendo también diferencias de tolerancia para una misma especie. Así, por ejemplo, *Lolium perenne* L. puede acumular concentraciones importantes de níquel y plomo cuando se desarrolla en suelos con altos contenidos de estos elementos (Sanders et al., 1986; Alloway, 1995), al igual que sucede con las leguminosas y las especies del género *Agrostis* con el níquel (Virgel-Mensaka, 2002), y con *Agrostis capillaris* L. en sustratos que presentan elevadas concentraciones de cadmio (Fergusson, 1990). En cambio, la concentración de cromo del raigrás inglés suele encontrarse por debajo del límite de detección (Smith, 1996).

En general, en diferentes estudios se ha visto que el uso del lodo de depuradora urbana como fertilizante incrementa las concentraciones en el pasto de cobre (Mosquera et al., 2001; Tiffany et al., 2000b; Mosquera-Losada et al., 2009b), de zinc (Loué, 1988; Mosquera-Losada et al., 2009b), de Mn (Williams et al., 1997) y de cadmio (Tsadillas et al., 1995). Sin embargo, no se ha encontrado que la aplicación del lodo de depuradora al suelo provoque modificaciones en la concentración de hierro (Tiffany et al., 2000b), cromo (Hamon et al., 1999; López-Díaz, 2004) y plomo (Canet et al., 1998; López-Díaz, 2004) en el pasto. En cuanto al contenido de níquel en planta, éste depende de la disponibilidad de níquel en el suelo; así, Sanders et al. (1986) observó un incremento de este elemento en la planta al emplear este tipo de residuos en un suelo básico en el que se produjo una reducción de pH, mientras que en Galicia López-Díaz (2004) observó un incremento de níquel en planta proporcional a la dosis de fertilizante orgánico empleado cuando el pasto crecía sobre un suelo ácido.

PARTE III

The effects of fertilization with anaerobic, composted and pelletized sewage sludge on soil, tree growth, pasture production and biodiversity in a silvopastoral system under ash (*Fraxinus excelsior* L.)

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3.1. ABSTRACT

In silvopastoral systems, tree growth and the composition and productivity of pasture can be modified by management practices such as initial fertilization when tree seedlings are more sensitive to understorey competition. The aim of this study was to compare the effects of fertilization with different types of sewage sludge (anaerobic sludge, composted sludge and pelletized sludge), using different rates of incorporation and mineralization with traditional treatments (with and without mineral fertilizers) on the growth of newly established ash (*Fraxinus excelsior* L.) and on pasture development, to obtain sustainable management practices that enhance the growth of both components. Soil characteristics, tree growth, sward composition and pasture development were modified differently according to the type of sewage sludge used, and for similar total N inputs. Anaerobic sludge had a higher initial effect on both tree and pasture productivity. Pelletized sludge sustained better tree and pasture production. Composted sludge was found to be the most appropriate treatment for improving soil characteristics over the long term on sandy soils. It was concluded that pelletized sludge should be promoted because it enhances productivity, allows for better nutrient recovery and is less costly to store and apply compared with anaerobic sludge and composted sludge. No toxic concentrations of Zn or Cu were found in plants or in the soil despite higher concentrations being present in the applied sludge than in soil.

Keywords: agroforestry; Zn; waste; afforestation; land use change; Spain

3.2. INTRODUCTION

Silvopastoral systems are a type of agroforestry systemin which trees and grazing are combined, resulting in benefits for wood production (with long-term economic returns) and livestock (with short-term economic returns) (Rigueiro-Rodríguez et al., 2005a). Agroforestry systems are sustainable land management techniques that are promoted by the EU (Council Regulation 1698 ⁄ 2005 (UE, 2005)). Agroforestry systems have also received favourable evaluations from farmers in Europe; for example, a sample of 214 farmers interviewed in fourteen areas of Europe, half indicated they would attempt silvo-arable agroforestry on their farm (Graves et al., 2008).

In the early stages of the development of silvopastoral systems established through afforestation, competition between trees and pasture can be high (Nair and

Graetz, 2004; Rigueiro-Rodríguez et al., 2008a). In newly established systems, adequate management should aim to optimize silvopastoral outputs through the selection of trees and pasture species as well as through fertilization inputs. Shrub development should be avoided through frequent clearing in order to avoid shrub–tree competition and fire risk (Rigueiro-Rodríguez et al., 2009). Moreover, in early stages of establishment pasture–tree competition should be avoided, either through mulching or by providing forage for grazing animals (Wagner et al., 2006).

The European Ash (*Fraxinus excelsior* L.) is a widely distributed tree species that integrates well into silvopastoral systems in the Atlantic biogeographic region of Europe (McAdam and Hoppé, 1996; McAdam and Sibbald, 2000). European ash trees possess apical dominance and deep roots that avoid root competition between trees and pasture; these roots enhance nutrient recovery from the deep soil layer up into the system, making them compatible with silvopastoral systems. Moreover, as a deciduous species, it allows better light penetration than conifers during the autumn and early spring, and provides shading during the summer, thereby reducing evapotranspiration and thus enhancing pasture production when compared with pasture under conifers or on open pasture sites. The open crowns of ash trees also allow light to reach the pasture surface, and they do not intercept much rain (McEvoy, 2004). The most appropriate pasture species for silvopastoral system implementation are those that are well adapted to shading, such as *Dactylis glomerata* L. (Mosquera-Losada et al., 2001, 2006). However, legume species also enhance pasture quality and production as well as tree growth (Whitehead, 1995; López-Díaz et al., 2009)

In Galicia, the natural soils have low fertility due to their acidity (Zas and Alonso, 2002). This acidity implies a high concentration of saturated aluminium in the exchange complex and low cation and phosphorus availability (Prasad and Power, 1997; Rigueiro-Rodríguez et al., 2007). The EU promotes the use of sewage sludge as a fertilizer because of its specific organic matter and content of macronutrients, particularly N (MMA, 2006). However, a higher concentration of heavy metals (mainly Zn and Cu) in sewage sludge than is normally found in soil, as well as long-term sludge loadings, limits the use of sewage sludge according to the Spanish (R.D. 1310 ⁄ 1990; BOE, 1990) and European Directives (86 ⁄ 278 ⁄ CEE; UE, 1986) in order to prevent harmful effects on soil and vegetation, and on animal and human health.

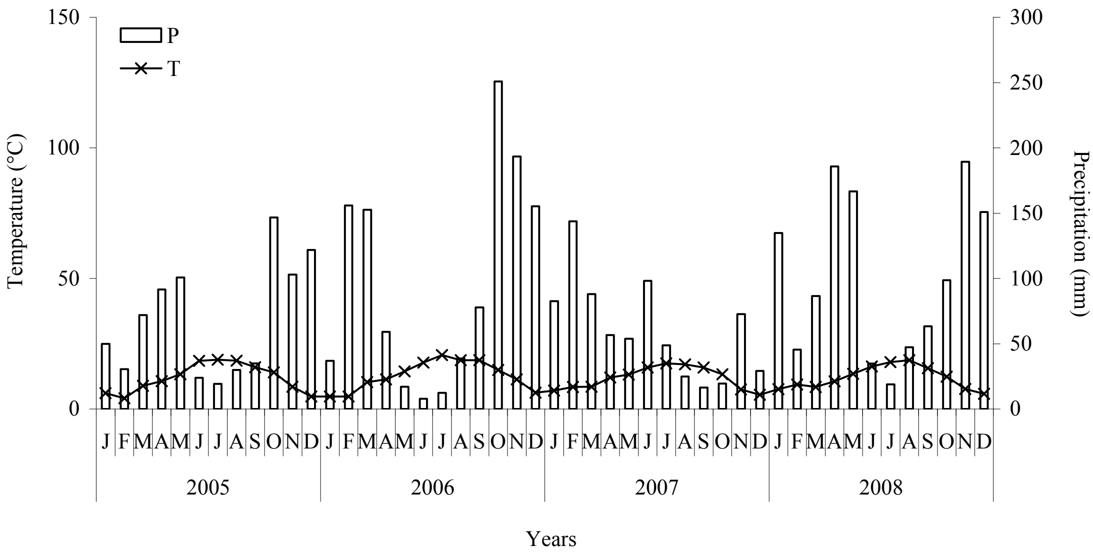
Anaerobic digestion and composting are two sewage sludge stabilization processes which are promoted by the EU (EEA, 2000) before the sludge is used as a fertilizer in agriculture. However, sewage sludge stabilized by these processes contains a high proportion of water. Pelletized sewage sludge is derived from the thermic treatment of anaerobic digested sewage sludge in order to reduce water content to 2%, which consequently reduces storage, transport and spreading costs compared with anaerobic or composted sludge (COM) (Mosquera-Losada et al., 2010b). Each type of sewage sludge has different characteristics, nutrient contents (Mosquera-Losada et al., 2010b) and rates of incorporation into the soil according to the treatment stabilization (EPA, 1994) and the specific local climate.

The aim of this study was to evaluate the effects of municipal sewage sludge that has been stabilized by either anaerobic digestion, composting or pelletization, on changes in soil chemical properties, tree growth, understorey production, biodiversity in terms of sward botanical composition and quality of pasture compared with treatments (receiving either mineral fertilizers or no fertilization) in a silvopastoral system under *F. excelsior* L. during a 4-year period.

3.3. MATERIALS AND METHODS

3.3.1. Characteristics of the study site

The experiment was conducted in A Pastoriza (Lugo, Galicia, NW Spain, European Atlantic Biogeographic Region; t 43' 14 N, 7º 21' W; 550 m a.s.l.). Figure 3.1 shows the mean monthly precipitation and temperatures for 2005, 2006, 2007 and 2008 and the previous 30-year mean. Total annual rainfall was 824.3, 1157.5, 734.4 and 1222.3 mm in 2005, 2006, 2007 and 2008 respectively. Very low precipitation was observed in 2005 and 2007 compared with the 30-year mean. There were periods of drought from April to July 2006 and from June to October 2008, which would have been unfavourable for tree growth and pasture production during these periods. The annual mean temperature was mild (12ºC).



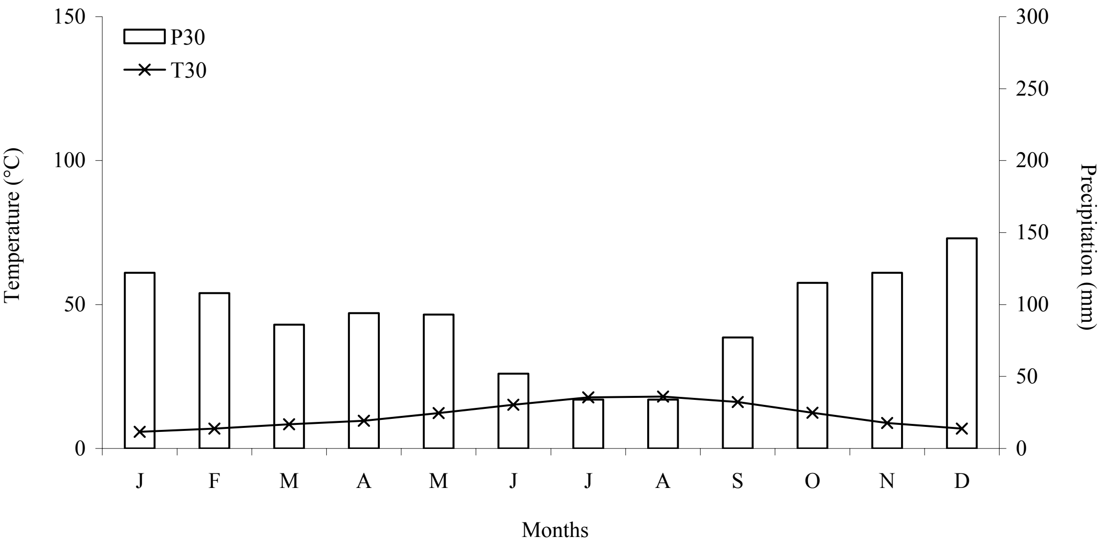


Figure 3.1: Monthly precipitation and mean temperatures for the study area in 2005, 2006, 2007 and 2008, and mean data for the last 30 years. T: mean monthly temperature (ºC); T30: 30-year mean temperature; P, mean monthly precipitation (mm); and P30: 30-year mean precipitation.

The experiment was carried out on abandoned agricultural land. The soil texture at the start of the experiment was sandy (91.81 sand, 4.92 silt and 3.27% clay) and pH (water) was moderately acidic at 5.6. Although the initial soil Mehlich 3-P concentration (35.1 mg kg^{-1}) can be considered high (Sawyer et al., 2008), no risk of P leaching has been found in the area because of the high soil acidity, which leads to a soil P storage capacity (Mosquera-Losada et al., 2008b). All heavy metal concentrations in the soil (Table 3.1) were below the maximum threshold for using sewage sludge as fertilizer as specified by the European Union Directive 86/278/CEE (UE, 1986) and Spanish legislation under R.D. 1310/1990 (BOE, 1990).

	Heavy metal concentrations (mg kg^{-1})					
Soil	Cd	Cu	Cr	Ni	Pb	Zn
Initial soil concentration	-	5.8	4.1	2.1	-	20.6
Spanish law limits	1-3	50-210	100-150	30-112	50-300	150-450

Table 3.1: Heavy metal concentrations in the soil at the beginning of the experiment and legal limits established by European Directive 86/278 and Spain R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH<7, maximum: soil pH>7). Concentrations of Cd and Pb were below detection limit of the technique used for determination.

3.3.2. Experimental design

At the beginning of the experiment, the soil was double ploughed to a depth of 50 cm, which is traditional practice in the area, and the pasture was sown with a mixture of *D. glomerata* L. var. Artabro (12.5 kg ha^{-1}), *Lolium perenne* L. var. Brigantia (12.5 kg ha^{-1}) and *Trifolium repens* L. var. Huia (4 kg ha^{-1}) in autumn 2004. Bare-rooted 1-year old plants of *F. excelsior* L. (typical height of 25 cm) were planted at a density of 952 trees ha^{-1}, with a distance between rows of 3.0 x 3.5 m. The experimental design was a randomized block with three replicates and five treatments distributed in experimental units of 168 m^2 with twenty-five trees arranged in a frame of 5 x 5 trees. The treatments consisted of (i) no fertilization (NF); (ii) mineral fertilization (MIN) with 500 kg ha^{-1} 8:24:16 compound fertilizer ($N:P_2O_5:K_2O$) at the beginning of the growing season and 40 kg N ha^{-1} after first harvest (MIN); (iii) fertilization with anaerobically digested sludge with an input of 320 kg total N ha^{-1} before pasture sowing (ANA); (iv) fertilization with composted sewage sludge with an input of 320 kg total N ha^{-1} before pasture sowing (COM) and (e) application of pelletized sewage sludge, which involved a total input of 320 kg total N ha^{-1} split into 134 kg total N ha^{-1} just after pasture sowing in 2004, and 93 kg N ha^{-1} at the end of 2005 and 2006, which correspond to similar total inputs of 320 kg total N ha^{-1} (PEL). Based on previous experiments in the area and EPA (1994) recommendations, it was assumed that approximately 0.25 of the total N would be mineralized in the first year if compost or anaerobic sludge was added, and therefore approximately 80 kg N ha^{-1} year^{-1} was applied. There was no available information about the pellet fertilizer, but a total similar input of 320 kg total N ha^{-1} during the experiment was used.

3.3.3. Sewage sludge

Anaerobically digested sewage sludge, COM and pelletized sludge (PEL) were taken from municipal waste treatment plants at Lugo, Valladolid and Madrid respectively.

The calculation of the required amounts of sludge was conducted according to the percentage of the total N and dry-matter contents EPA (1994), taking into account the European Union Directive 86 ⁄ 278 ⁄ CEE (UE, 1986) and Spanish regulation R.D.1310 ⁄ 1990 (BOE, 1990) regarding heavy metal concentrations for the application of sewage sludge on to soil. The composition of the sewage sludge is summarized in Table 3.2.

The sludge used in the present experiment had a similar composition to the mean composition of the sludge described for plants all over Spain (Mosquera-Losada et al., 2010b). The proportion of N in the sludge was higher than the P and K concentrations in the sludge. As the calculations of the required amounts of sludge were based on N and pasture N, and as P needs are similar, contamination by phosphorus is not likely to occur. Furthermore, the acidity of the soil would prevent P from leaching from the soil (Mosquera-Losada et al., 2008b).

Parameters	Anaerobic sludge	Composted sludge	Pelletized sludge	Spanish legal limits
Dry matter (%)	29.47	65.19	95.4	
pH	7.25	7.28	7.25	
N (g kg^{-1})	26.2	8.8	35.5	
P (g kg^{-1})	21.4	3.9	10.7	
K (g kg^{-1})	1.9	2.7	1.8	
Ca (g kg^{-1})	6.0	49.8	60.6	
Mg (g kg^{-1})	4.5	14.7	12.4	
Na (g kg^{-1})	1.0	0.4	0.7	
Fe (g kg^{-1})	19.6	12.8	141.5	
Cr (mg kg^{-1})	92.3	3.9	16.6	1000-1500
Cu (mg kg^{-1})	238.5	121.2	136.1	1000-1750
Ni (mg kg^{-1})	69.5	95.3	91.9	300-400
Zn (mg kg^{-1})	1752.3	753.1	1130.4	2500-4000
Cd (mg kg^{-1})	14.4	<0.01	<0.01	20-40
Pb (mg kg^{-1})	281.1	104	58.5	750-1200
Mn (mg kg^{-1})	248.3	90.5	108.8	
	Total heavy metal inputs per treatment			
Parameters	Anaerobic sludge	Composted sludge	Pelletized sludge	Spanish legal limits
Cr (kg ha^{-1})	1.39	0.14	0.21	3
Cu (kg ha^{-1})	3.59	4.47	1.76	12
Ni (kg ha^{-1})	1.04	3.51	1.18	3
Zn (kg ha^{-1})	26.43	27.76	14.63	30
Cd (kg ha^{-1})	0.21	<0.001	<0.0001	0.15
Pb (kg ha^{-1})	3.74	3.33	1.40	15

Table 3.2: Chemical properties of the sewage sludge applied, total loadings supplied with the inputs of different types of sludge in this experiment and legal limits established by Spanish directive R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH<7, maximum: soil pH>7). Lower section shows total loadings supplied with the inputs of different types of sludge in this experiment.

3.3.4. Field samplings and laboratory determinations

Soil samples were collected to a depth of 25 cm, as described in the RD 1310/1990 (BOE, 1990) in February 2006, January 2007 and January 2008. In the laboratory, soil pH was determined in water (1:2.5) (Guitián and Carballás, 1976). The aluminium concentration in the exchange complex and the exchangeable cations were determined by extraction with 0.3 M $BaCl_2$. The K, Ca, Mg and Na exchangeable concentrations were measured with a Varian 220FS Spectrophotometer (Varian, Walnut Creek, CA, USA) using the atomic emissions for K and Na and the absorptions for Ca and Mg. Aluminium concentrations were analysed after valoration with 0.01 N NaOH, using phenolphthalein (1%) in an alcohol-based solution as an indicator (Mosquera and Mombiela, 1986). The effective exchange capacity (EEC) was determined by taking the sum of K+ Ca+ Mg+ Na+ Al and the aluminium percentage saturation using the quotient Al ⁄ EEC. The total soil Ca, Cu and Zn concentrations were determined after microwave digestion (CEM, 1994), and the available Ca, Cu and Zn were measured after extraction with Mehlich (1985) with the Varian 220 FS Spectrophotometer using atomic absorption.

Base tree height and diameter were measured with a graduated ruler and a calliper, respectively, at the beginning of 2005, 2006, 2007 and 2008.

Pasture production was determined randomly by taking four samples of pasture at a height of 2.5 cm per plot (0.3 m x 0.3 m) using an electric hand clipper in August and December 2005; June and December 2006; April, June and December 2007 and May and December 2008 before all of the plots were grazed by mature sheep (Galician breed) at a stocking rate of fifty sheep over the whole experimental area (2520 m^2) 1 week after sampling. Two pasture samples were dried for 48 h at 60ºC and weighed to estimate pasture production. The other two samples were separated by hand to determine the proportions of the different plant species and the senescent material, and then dried (60ºC for 72 h) to determine the botanical composition on a dry weight basis. Annual abundance diagrams (Magurran, 1988) were made which excluded senescent material. The total Zn in the harvested pasture herbage was determined by microwave digestion with nitric acid (CEM, 1994).

3.3.5. Statistical analysis

The data were analysed using ANOVA, and the differences between the averages were determined using the LSD test (at the alpha level of 0.05) using the SAS statistical package (SAS, 2001). All soil variables (soil water pH, EEC, Al saturation percentage, total (Cu and Zn) and Mehlich (Ca, Cu and Zn) soil concentrations) were analysed using the ANOVA model ($Y_{ij} = \mu + B_i + T_j + Y_k + BT_{ij} + TY_{jk} + BY_{ik} + \varepsilon_{ijk}$), where μ is the mean; Bi is block (two freedom degrees); T_j is treatment (four freedom degrees); Y_k is year (three freedom degrees), BT_{ij} (block treatment interaction (eight freedom degrees), TY_{jk} is treatment year interaction (twelve freedom degrees); BY_{ik} is block year interaction (six freedom degrees) and ε_{ijk} is the error term. All tree and pasture variables were analysed using the ANOVA model ($Yij = \mu + B_i + T_j + \varepsilon_{ijk}$), where μ is the mean; B_i is block (two freedom degrees); T_j is treatment (four freedom degrees); and ε_{ijk} is the error term. ANOVA type III errors were taken into account to determine significances.

3.4. RESULTS

3.4.1. Soil chemical properties

3.4.1.1. *pH in water, effective exchange capacity (EEC), aluminium saturation percentage and Ca extracted by Mehlich*

Composted sludge (COM) increased the mean soil pH ($P < 0.01$) and the amount of mean Ca extracted by Mehlich ($P < 0.05$), but the effect on ECC was not significant ($P > 0.05$). Because of these modifications (Figure 3.2), the mean percentage of aluminium saturation decreased with this type of sludge application ($P < 0.01$). There was a significant effect of year ($P < 0.001$) on soil pH (2006: 5.6^{a}; 2007: 5.5^{b} and 2008: 5.27^{c}); on EEC (2006: 8.6^{b}; 2007: 10.31^{a} and 2008: 9.99^{a} expressed as cmol (+) 100 g^{-1} soil); on aluminium saturation percentage (2006: 17.24^{b}; 2007: 24.15^{a} and 2008: 11.66^{c}) and amount of Ca extracted by Mehlich (2006: 0.69^{b}; 2007: 0.25^{c} and 2008: 1.43^{a} expressed as mg kg^{-1}) (in all cases different superscript letters indicate significant differences between years). The soil pH and aluminium saturation percentage were lower at the end of the study (2008) than at the beginning of the study (2006), whereas the EEC and the amount of Ca extracted by Mehlich were higher in 2008 than in 2006.

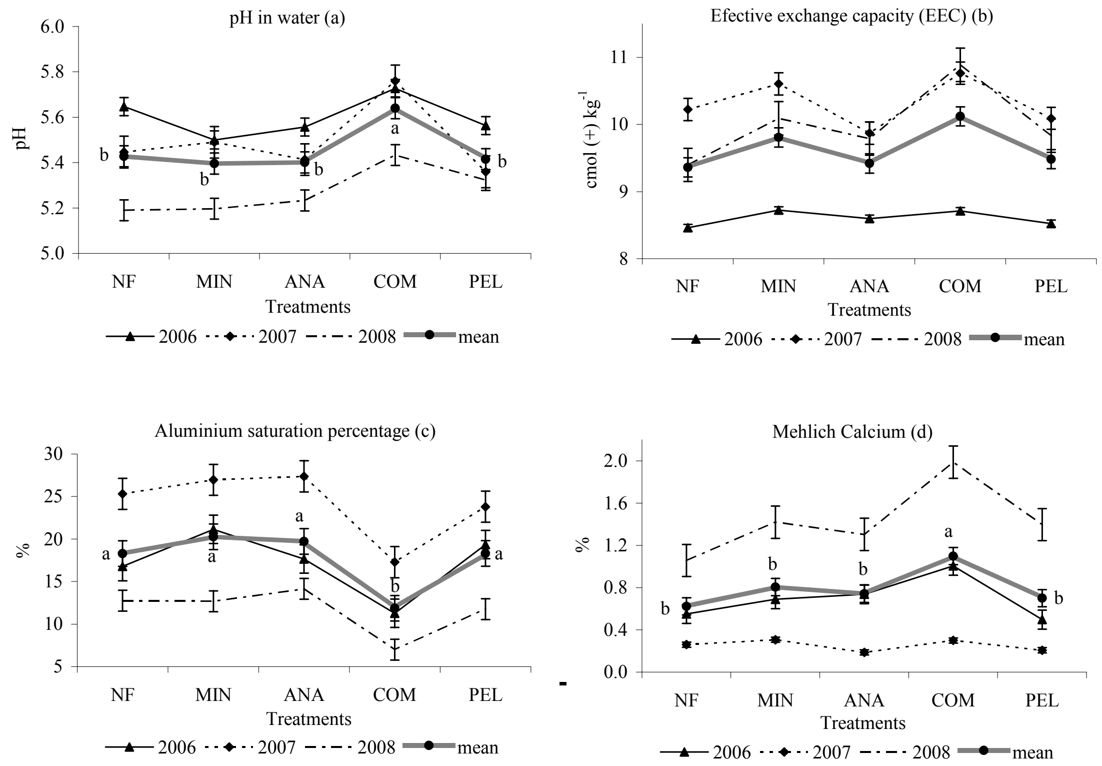


Figure 3.2: Soil pH in water (a), exchange capacity (cmol (+) kg^{-1}) (b), aluminium saturation percentage in soil exchange complex (%) (c) and amount of Ca extracted by Mehlich (%) (d) in each treatment in the years 2006, 2007 and 2008. NF, no fertilization; MIN, mineral; ANA, anaerobic sludge; COM, composted sludge and PEL, pelletized sludge. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

3.4.1.2. *Total and Mehlich extracted Cu and Zn*

The total and Mehlich soil levels of Cu and Zn in 2006, 2007 and 2008 are presented in Figure 3.3. The mean total Cu and Zn were significantly affected by the treatments ($P < 0.001$ and $P < 0.05$ respectively). There was a significant effect of year on total Cu ($P < 0.001$), total Zn ($P < 0.001$) and on the Zn extracted by Mehlich ($P < 0.001$). In the case of the Cu extracted by Mehlich, the interaction of treatment x year was significant ($P < 0.01$). All of the variables were increased by COM, and total Zn was increased by anaerobic sludge (ANA) compared with the other treatments. The total soil Cu and Zn found in the experiment were below the maximums set by Spanish regulations for the use of sewage sludge in agriculture for acid soils (Cu: 50 mg kg^{-1} and Zn: 150 mg kg^{-1}) (R.D. 1310 ⁄ 1990, (BOE, 1990)). The concentrations of total Cu and Zn and Zn extracted by Mehlich were lower in 2008 than in 2006.

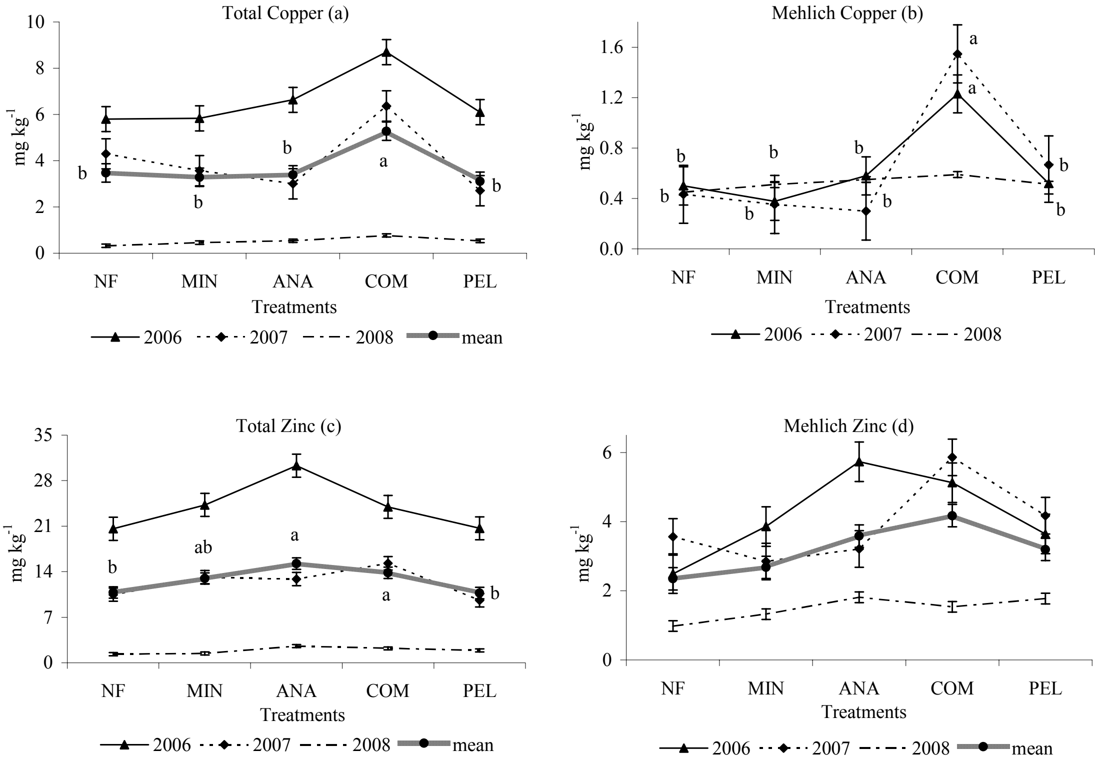


Figure 3.3: Total Cu (mg kg^{-1}) (a) and total Zn concentration (mg kg^{-1}) (c) in soil and amount of Cu (mg kg^{-1}) (b) and amount of Zn (mg kg^{-1}) (d) extracted by Mehlich in each treatment in the years 2006, 2007 and 2008. NF, no fertilization, MIN, mineral; ANA, anaerobic sludge; COM, composted sludge and PEL, pelletized sludge. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

3.4.2. Tree height and diameter

The average tree heights for each treatment in 2005, 2006, 2007 and 2008 are shown in Table 3.3. In all 2 years of the study, tree height was significantly modified by fertilization treatment (Table 3.4). Initially, the tree height was higher in all treatments that received organic fertilizer (ANA, COM and PEL) than the MIN or NF treatments. Tree height in 2005 and 2006 was increased by anaerobic (ANA) and PEL compared with the mineral fertilization (MIN) and no fertilization (NF) treatments, but this positive effect was only found in those plots receiving PEL in 2007 and 2008. Trees growing in NF and MIN treatments had the lowest height during the 4 years of the experiment.

Tree diameter (Table 3.3) was significantly modified by fertilization treatment in all 4 years of the study, with the exception of 2007, (Table 3.4). In 2005, 2006 and 2007, tree diameter was increased by ANA compared with the other treatments (NF, MIN, ANA, COM and PEL). In 2008, the COM and PEL treatments resulted in larger tree diameters than MIN treatment.

	Year	Treatments					SE
		NF	MIN	ANA	COM	PEL	
Tree height (cm)	2005	32.88	32.33	38	34.85	38.05	0.69
	2006	33.65	33.08	39.33	35.8	38.1	0.66
	2007	41.59b	42.58b	48.89ab	47.65ab	53.55a	1.05
	2008	114.36ab	107.56b	103.63b	121.84ab	135a	3.51
Tree diameter (mm)	2005	5b	4.83b	5.89a	4.7b	5b	0.14
	2006	5.12b	5.42b	6.56a	4.8b	5.15b	0.14
	2007	6.29	5.92	7.54	6.45	6.09	0.14
	2008	12.55ab	10.81b	12.29ab	13.68a	14.04a	0.37
Pasture Production t ha^{-1}	2005	1.96b	2.98a	3.17a	2.81ab	3.36a	0.14
	2006	4.91	3.98	5.15	4.82	5.24	0.18
	2007	5.66ab	5.88ab	4.81b	5.53ab	6.55a	0.18
	2008	3.96	4.25	3.37	3.94	4.51	0.14

Table 3.3: Mean values for tree height (cm), tree diameter (mm) and pasture production (t ha^{-1}) under the different fertilization treatments in the years 2005-2008. SE: Mean Standard Error, NF: no fertilization, MIN: mineral fertilizer; ANA: anaerobic sludge; COM: composted sludge, and PEL: pelletized sludge. Different letters indicate differences between treatments within the same year that were significant at $P<0.05$.

	Year	Height	Diameter	Pasture production
Treatment effect	2005	ns	*	*
	2006	ns	**	ns
	2007	**	ns	*
	2008	*	*	ns

Table 3.4: ANOVAs for tree height, diameter and pasture production in 2005, 2006, 2007 and 2008. ns: not-significant, * $P<0.05$, ** $P<0.01$, *** $P<0.001$.

3.4.3. Pasture understorey

3.4.3.1. *Pasture production*

The annual pasture production for the different fertilization treatments in 2005, 2006, 2007 and 2008 is summarized in Table 3.3. Significant differences were detected between the treatments in all years except 2006 and 2008 (Table 3.4). The highest levels of pasture production were found in 2007 (4.8–6.5 t ha^{-1}), whereas the lowest values were recorded in 2005 (1.9–3.3 t ha^{-1}). In 2005, there was a positive response to organic fertilization (ANA, COM and PEL) and inorganic fertilization (MIN) compared with no fertilization (NF), but this effect was only maintained in the PEL treatment until the end of the experiment. In the final years of the study (2007–2008), the annual pasture production tended to be lower than that of the PEL when the ANA was applied to the soil.

3.4.3.2. *Pasture abundance diagrams*

Figure 3.4 shows the abundance diagrams for the different fertilization treatments in 2005, 2006, 2007 and 2008. *Agrostis capillaris* L., *D. glomerata* L., *Holcus lanatus* L. and *L. perenne* L. were present in the sward in all treatments and in all years. The treatments with a high number of species were associated with a higher proportion of dicotyledonous species. *Agrostis capillaris* L. and *D. glomerata* L. were the most dominant species throughout the study. The proportion of *A. capillaris* L. was always over75%in the NF treatment and approximately50%in the other treatments and years, with the exception of mineral (MIN) and ANA fertilization in the final year. However, the mineral (MIN) and ANA treatments had higher proportions of *D. glomerata* L. in the first years of the study. In the PEL treatment, codominance between *D. glomerata* L. and *A. capillaris* L. was evident throughout the experiment. In terms of species richness, the ANA treatment had the lowest number of species at the start of the experiment, but the PEL treatment had the lowest number from 2006 onwards. The COM treatment in the second year of the experiment was an exception to this trend, exhibiting the lowest number of species. The PEL treatment was dominated by monocotyledonous species in the final year than the remaining treatments (NF, MIN, ANA and COM).

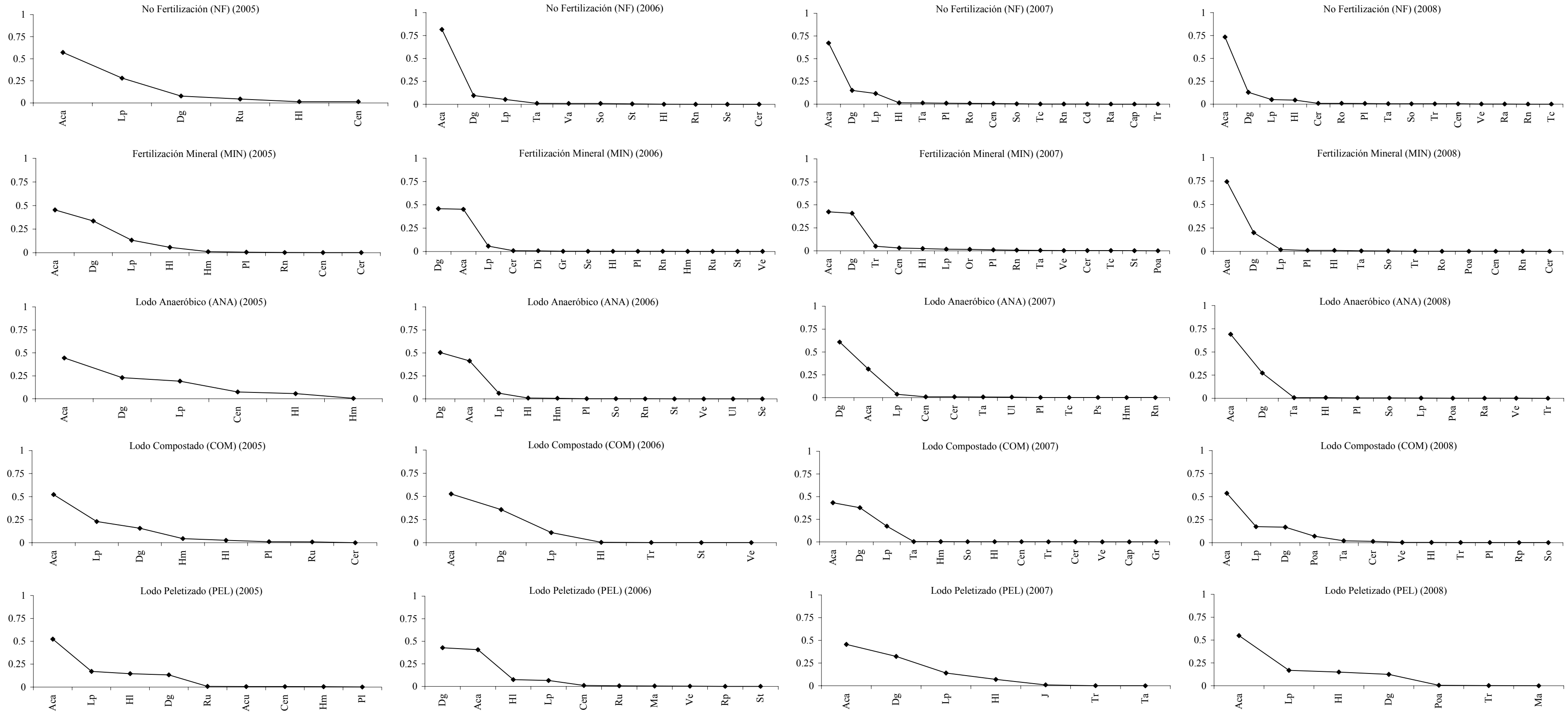


Figure 3.4 Abundance diagrams for the treatments applied in the years 2005, 2006, 2007 and 2008. Aca: *Agrostis capillaris* L.; Acu: *Agrostis curtisii* Kerguélen; Cd: *Carduus* sp; Cap: *Capsella bursa-pastoris* L.; Cen: *Centaurea limbata* Hoffmanns. / Link; Cer: *Cerastium glomeratum* Thuill; Di: *Digitalis purpurea* L.*;* Dg: *Dactylis glomerata* L.*;* Gr: *Geranium rotundifolium* L.; Hl: *Holcus lanatus* L.; Hm: *Holcus mollis* L.; J: *Juncus effusus* L.; Lp: *Lolium perenne* L.; Ma: *Matricaria* sp; Or: *Ornithopus compressus* L.; Pl: *Plantago lanceolata* L.; Poa: *Poa pratensis* L.*;* Ps: *Pseudarrhenatherum longifolium* (Thore) Rouy; Ra: *Rumex acetosa* L.; Rn: *Ranunculus repens* L.; Ro: *Rumex obtusifolius* L.; Rp: *Raphanus raphanistrum* L.; Ru: *Rubus* sp; Se: *Senecio jacobaea* L.; So: *Sonchus oleraceus* L.; St: *Stellaria media* L. (Vill); Ta: *Taraxacum officinale* Weber; Tc: *Trifolium campestre* Schreber; Tr: *Trifolium repens* L.; Ul: *Ulex europaeus* L.; Ve: *Veronica agrestis* L.

3.4.3.3. *Zn and Cu concentrations in pasture*

The concentration of Zn in the pasture was significantly affected by treatments in April 2007 ($P < 0.01$), and May 2008 ($P < 0.05$) (Figure 3.5). However, Cu levels in the pasture were not affected by any treatments (data not shown). In April 2007, the concentration of Zn in the pasture was higher when the ANA and PEL treatments were applied. In May 2008 the concentration of Zn in the pasture increased in the ANA treatment compared with the COM treatment. There were no responses to the treatments in terms of the pasture concentrations of Zn after harvests made in 2005, 2006, December 2007 and December 2008.

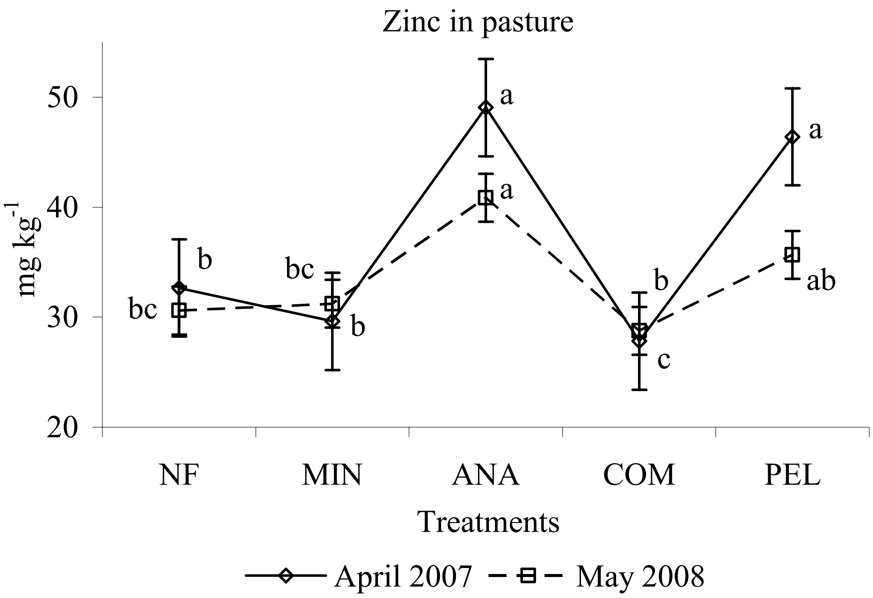


Figure 3.5: Concentrations of Zn in pasture (mg kg^{-1}) under the different fertilizer treatments in the harvests of April 2007 and May 2008. NF, no fertilization; MIN, mineral; ANA, anaerobic sludge; COM, composted sludge and PEL, pelletized sludge. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

3.5. DISCUSSION

This experiment demonstrates that there are very few negative effects of sewage sludge application, either from toxic metal loading or in terms of productivity in the specific edaphoclimatic conditions of this study. The growth of trees and pasture production can be limited by low soil pH, EEC and a high aluminium saturation percentage; however, they can be improved by applications of high pH organic waste and by fertilization (Mosquera-Losada et al., 2010b). The soil pH in this experiment ranged from acidic (5.1–5.5) to moderately acidic (5.6–6.0) (Slattery et al., 1999);

conditions which usually indicate deficiencies in the availability of cations and, therefore, limit pasture production (Whitehead, 2000). Moreover, the soil EEC was low and usually below 10 cmol (+) kg^{-1} soil, which could be explained by the high sand proportion in the soil of this experimental site (Brady and Weil, 2008). However, the aluminium saturation percentage was below 25%, which indicates the optimum characteristics for pasture and tree growth in Galicia (Mombiela and Mateo, 1984). Aluminium saturation was also found to have a diachronic variation, which usually improved from the start to the end of the experiment. The EEC increase could be explained by the tilling that occurred at the start of the experiment. The soil aggregate structure was probably destroyed by the tilling process (Dexter, 1988), but the structure is likely to have been improved with the addition of fertilizer and with the establishment of pasture and trees, and the potential for increased organic matter input into the soil. The soil pH, however, was lower at the end of the experiment, despite the increase in available Ca and the reduction in available Al caused by sludge incorporation into the soil (Smith, 1996); this could be explained by the increase in the proportion of H^+ related to the higher EEC. The soil acidity increase derived from the H^+ can be explained by the cation extraction of the crops, the mineralization process (NH_4^+ is transformed in NO_3^- and H^+ is released in the nitrification process) and the high mean rainfall of the area, which promotes cation leaching (Whitehead, 1995).

Composted sludge (COM) increased soil fertility compared with other treatments. The pH and mean amount of available Ca were improved by the COM treatment, which was expected and is explained by the different composition of the sludge and the rate of mineralization (Mosquera-Losada et al., 2010b). Applied doses of sewage sludge with compost were higher in order to meet the N required by EPA (Environmental Protection Agency) (1994) recommendations because the COM composition revealed a lower concentration of N than the anaerobic and PEL. The COM also had a low mineralization rate, as indicated by the EPA (Environmental Protection Agency) (1994) (the mineralization rate is approximately 20% for anaerobic compared with 10% for COM in the first year). Moreover, the pH and available Ca were increased in the COM treatment because of the higher concentrations of Ca, K and Mg compared with anaerobic and PEL (excepting Ca). The COM treatment applied approximately 1835.8 kg Ca ha^{-1}, 99.53 kg K ha^{-1} and 541.9 kg Mg ha^{-1}; meanwhile, only 90.51 kg Ca ha^{-1}, 28.66 kg K ha^{-1} and 67.88 kg Mg ha^{-1} were added with the ANA treatment, and 776.85 kg Ca ha^{-1}, 23.07 kg K ha^{-1} and 158.96 kg Mg ha^{-1} were the soil inputs with the

PEL treatment. The higher rate of Ca and Mg with the COM also explains the reduction in the aluminium saturation percentage for this treatment (Smith, 1996; Prasad and Power, 1997; Speir et al., 2004). In previous studies, the anaerobic sewage sludge effect on the soil pH and EEC was found to depend on the previous initial soil pH. When the initial soil pH was high, anaerobic sludge inputs increased acidity as extraction was promoted (Mosquera-Losada et al., 2006), but when the soil pH was very low (4.5), as described by López-Díaz et al. (2007), a positive effect of sewage sludge application on soil pH was found. In this study, although the soil fertility was enhanced by COM treatment, tree growth and pasture production were not promoted. The lack of enhancement in tree growth and pasture production was probably due to the lower mineralization rate and the N availability of COM treatment compared with that of the anaerobic (ANA) or PEL as described by the EPA (1994). Warman and Termeer (2005) found better crop production in response to ANA than to COM sludge. The mineralization rate depends on the local climate and on the mineralization rates (EPA, 1994), and in our case, the COM treatment seems to have an initial lower mineralization rate compared with the ANA treatment. As a result, the effect of COM on pasture production and tree growth could become apparent over a longer period, as was seen with tree diameter. The soil fertility is differently affected by sewage sludge application, and the effect is dependent on the initial soil pH and on the type of the sewage sludge applied.

At the end of the 4-year study, the heights of ash trees varied from 104 to 135 cm and stem diameters from 10.81 to 14.04 mm. The tree heights were within the range (23–275 cm) reported in a study carried out in UK after 3 years of experimentation (Mwase et al., 2008). The initial tree diameters in our study were also similar to those found by the same authors (15.3 mm). A positive effect was found for the ANA treatment in the two first years of the experiment; this trend has also been described for *Populus* x euroamericana (Rigueiro-Rodríguez et al., 2008b) and *Pinus radiata* D. Don, under soil conditions that were initially either very acid (López-Díaz et al., 2007) or neutral (Mosquera-Losada et al., 2006). In other work, Muys et al. (2004) and Weber-Blaschke and Rehfuess (2002) demonstrated that soil fertility improvements led to higher *F. excelsior* L. growth, relative to the control, on sites with sandy loam and loam soils respectively. In both the present experiment and in these previous studies, the positive effect of the ANA treatment could be attributed to the soil fertility and water retention improvements caused by anaerobic fertilizer applied at the establishment of the plantation (Wolstenholme et al., 1992). However, at the end of the experiment, only the PEL treatment showed a significant increase in tree height diameter and the COM treatment in tree diameter compared with the MIN treatment, which could be explained by the enhancement of pasture growth and increased competition between trees and pasture caused by the MIN treatment as compared with the NF treatment.

Annual pasture production was below the common levels in this region due to the droughts in 2005, 2006, 2007 and 2008 (mean monthly precipitation of these years were lower than mean precipitation over the previous 30 years) and the low temperatures at the beginning of the year. Pasture production was significantly increased by organic or inorganic fertilization as found by Mosquera-Losada et al. (2006) and by López-Díaz et al. (2009) in agrarian soils in Galicia afforested with *P. radiata* D. Don with a pH close to neutral (pH 6.8) and a slightly acidic pH (pH 6.3) respectively. At the end of this study, only the PEL treatment was found to have positively affected annual pasture production by the PEL treatment because of the annual application; however, no residual effect was found as a result of the COM or ANA treatments (Rigueiro-Rodríguez et al., 2000; López-Díaz et al., 2007).

The lower proportion of *A. capillaris* L. and the higher proportion of *D. glomerata* L. could be associated with low and high soil fertility, as demonstrated in soils with a pH below 4.97 (Mosquera-Losada et al., 2001). Modifications to soil fertility also caused variations in the dominance of these two species, with a high proportion of *D. glomerata* L. initially associated with the MIN and ANA treatments followed by a switch in the botanical composition to over 75% *A. capillaris* L. by the end of the experimental period.

Fertilizer treatment effects on soil fertility explain tree and pasture production, and affect biodiversity. The ANA treatment initially increased soil fertility, as demonstrated by higher tree growth, greater pasture production, a lower number of species and a higher proportion of *D. glomerata* L. (Mosquera-Losada et al., 2001) compared with the other treatments (NF, MIN, COM and PEL). These results are of particular interest in areas in which the initial tree development is important to guarantee tree survival. However, there is a reduction in tree and pasture growth when soil fertility is depleted, probably because the nutrient requirements of trees are greater than in the other treatments and because the capacity for nutrient retention in the ANA treatment is lower than in the other treatments because nutrient leaching occurs. On the contrary, the PEL treatments sustained better tree and pasture production over the long term as the nutrient inputs were supplied as split doses through time. The higher fertility of soils fertilized with PEL probably explains the dominance of monocotyledonous species and the low number of species throughout the experiment.

It is important to be aware of the effects of sewage sludge on the concentrations of Cu and Zn in the soil and in plants, because Cu and Zn are commonly present in higher proportions in the municipal sewage sludge (Smith, 1996; Mosquera-Losada et al., 2010b) relative to soil concentrations. As seen with the macronutrients, the COM treatment resulted in higher input rates of Cu and Zn (4.46 kg Cu ha^{-1} and 27.76 kg Zn ha^{-1}) into the soil than the ANA (3.59 kg Cu ha^{-1} and 26.43 kg Zn ha^{-1}) or PEL treatment (1.74 kg Cu ha^{-1} and 14.47 kg Zn ha^{-1}). The COM treatment increased the soil pH compared with the other treatments, which usually implies a reduction of Cu and Zn solubility because of the EEC increment (Prasad and Power, 1997). In China, Miao-Miao et al. (2007) also found that the use of COM as fertilizer increased the concentrations of Zn and Cu in the soil, but in that study the COM contained higher concentrations of Cu and Zn due to the effects of local industry (Cu: 2316 mg kg^{-1} and Zn: 2971 mg kg^{-1}) compared with the sludges used in our experiment. ANA also increases the total soil Zn because this type of sludge has a higher concentration of Zn than the COM or the PEL (anaerobic sludge: 1752.3 mg Zn kg^{-1} soil; COM: 478.7 mg Zn kg^{-1} soil and PEL: 753.1 mg Zn kg^{-1}). An increment of Cu and Zn in soil as a result of sewage sludge applications was also found by Yuan (2009). In contrast, in this study the concentrations of Zn and Cu in the soil were lower in 2008 than at the beginning of the experiment. This could be explained by leaching, but is more likely due to pasture and tree .extractions.

The range of Zn concentrations in the pasture in this experiment (18.63–49.31 mg kg^{-1}) was at the low end of the concentrations commonly found in pastures (27–150 mg kg^{-1}) and below the levels of 100 and 400 mg kg^{-1} considered excessive or toxic for plants (Kabata-Pendías and Pendías, 2001; Smith, 1996). ANA tended to increase the soil and pasture Zn concentrations in the acidic soils of the Galician region, as described by Mosquera-Losada et al. (2001). In spite of the higher Zn inputs with the COM treatment, which increased the total and the available Zn in the soil, no effect was detected in the pastures because of the increased soil pH compared with other treatments, which could in turn reduce the availability of absorbable Zn in the soil (Prasad and Power, 1997). A Zn concentration of 500 mg kg^{-1} is considered toxic for cattle, sheep and horses (Smith, 1996), but this value was not reached or exceeded in the pasture in this experiment.

3.6. CONCLUSION

Soil characteristics, tree growth, pasture species biodiversity and pasture development are modified by the type of sewage sludge used when similar N inputs are applied. The ANA treatment has a higher initial effect on tree and pasture productivity, but PEL treatment sustains better production as it is applied in several times and the COM treatment improved soil characteristics over the long term in sandy soils. The PEL treatment should be promoted because this treatment enhances productivity allows for better nutrient recovery and is less costly to apply than the other two treatments. No toxic Zn or Cu concentrations were found in the plants or in the soil in spite of the higher concentrations in sewage sludge than in the soil.

3.7. ACKNOWLEDGMENTS

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PARTE IV

Fertilization in pastoral and *Pinus radiata* D. Don silvopastoral systems developed in forest and agronomic soils of Northwest Spain

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4.1. ABSTRACT

The effects of fertilization, pasture sowing and tree plantation on soil fertility and tree and pasture production can vary depending on the soil type. Tree plantation is recognized as a way to reduce nutrient leaching and increase land profitability in agronomic and forest soils, meanwhile pasture fertilization and sowing is usually associated to better pasture productivity and quality. Fertilization can be performed with mineral fertilizers, which have become expensive in recent times, or with organic fertilizers like sludge, which is being promoted worldwide. This study aims to evaluate the effects of sludge fertilization, tree planting and pasture sowing on different variables of soil (KCl-pH, cation exchange capacity, total N, total and Mehlich P, nitrate and soil organic matter) and pasture (production, botanical composition, crude protein and P concentration) in treeless and agroforestry systems established in forest and agronomic soils. The experimental design was a randomized block following an incomplete factorial design with three replicates and nine treatments including two types of soils (forestry and agronomic), two types of vegetation (natural and sown), two types of fertilization (sludge fertilization and mineral fertilization, with a no fertilizer control) in afforested and treeless pastures. Pasture production and quality was better under agronomic soils, which also had higher levels of KCl-pH, cation exchange capacity, nitrate, total N and P than forest soils. Tree establishment did not modify nitrate or P leaching, probably due to the youth of the trees when most of nitrate was leached at the beginning of the experiment, but reduction of soil KCl pH and pasture crude protein was found in forest soils, when trees and pasture were together established, probably due to the high extractions of these systems compared with unsown forests. Moreover, the sludge inputs increased pasture production better than the mineral fertilizer in the forest soils, probably due to the greater amount of nutrients applied by the former. Sowing enhanced the presence of sown grasses in the forest understory, but their presence reduced pasture quality, and they disappeared within a short period of time. Therefore, the use of the sludge as fertilizer allows nutrient recycling of this residue in soils of low fertility and increases productivity and preserves fertility compared with mineral fertilizer at short (forest soils) and medium (agronomic soils) term.

Key Words: agroforestry, biodiversity, nitrate leaching, organic matter, P, sludge

4.2. INTRODUCTION

Agroforestry systems are sustainable land management techniques that are promoted by the EU (European Union, Council Regulation 1698/2005 (UE, 2005)) and are considered a good management tool that can be implemented by farmers in the different countries of Europe (Graves et al., 2008).

Monterey pine (*Pinus radiata* (D. Don)) is a tree species that is currently used in silvopastoral systems in temperate areas like Australia, New Zealand, and Chile (Hawke, 1991; Knowles, 1991; Benavides et al., 2009) due to its fast growth. The species is widely used in the Atlantic biogeographic region of Europe (mostly in the North of Spain and West of France) in both forestry and farm grassland soils. Adequate fertilization practices in Monterey pine silvopastoral systems should be implemented to increase tree and pasture growth simultaneously at the same time that nutrient leaching risk is reduced. Recent increases in inorganic fertilizer prices along with environmental concerns have reduced the use of inorganic N fertilizers in the EU (EFMA, 2009), which are currently being replaced by organic fertilizers like sewage sludge as a cheaper N resource.

In EU countries, sewage sludge production has increased since the early nineties due to the implementation of European Directive 91/271/EEC (UE, 1991), which was enacted to enhance continental water quality. Therefore, it is necessary to find an adequate means of disposal for these residues in compliance with the environmental policies of the EU. One alternative that has been adopted in various countries around the world is the application of sewage sludge to soils as fertilizer (EPA, 1994), which is regulated in Europe by the directive 86/278/EEC (UE, 1986). The use of sewage sludge as fertilizer is being promoted because it eliminates waste and reduces environmental pollution while imparting organic matter and macronutrients, particularly N and P, to the soil (Loehr et al., 1979; Rosswall, 1982; Beltrán et al., 2002; Mosquera-Losada et al., 2010b). The study of the proportion of N that is readily mineralizable is important in determining the dose of sewage sludge that should be applied to the soil (Barry et al., 1986) in order to enhance both understory and overstory production in silvopastoral systems and to evaluate nitrate leaching risks (Simon and Le Corre, 1992, EPA, 1994). Sludge mineralization depends on soil types, pH and microbial soil activity (Smith, 1996) and this process is usually faster in agronomic than in more acid forest soils in North Western Spain. Moreover, the impact of sludge mineralization also depends on land use, being nitrate leaching risk usually higher in exclusive agronomic use (grasslands) than in silvopastoral systems due to the presence of the tree that may use the nitrate not employed by the pasture (Nair et al., 2008; Rigueiro et al., 2008a).

Monterey pine and pasture growth response to sewage sludge inputs can be modified by the type of soil in which it is applied due to the different N and P availability in forestry and agronomic soils, which also affects nitrate leaching. Northwestern Spain forest soils usually have high organic matter content, which can act as a source of nutrients for crops. However, P availability is usually lower in forest soils due to high Al and Fe levels (Nair and Graetz, 2002). P and N availability can modify tree and pasture development as well as nitrate and P losses and pasture botanical composition (Campbell et al., 1993; Kellas et al., 1995; Nair et al., 2008). The impact of sewage sludge inputs in different soil types on nitrate and P cycling in agroforestry systems compared with treeless systems has not been evaluated in Western Europe.

The aim of this study is to evaluate the soil, productivity (tree and pasture) and environmental (nitrate and, P leaching) response to mineral or municipal sewage sludge inputs in grasslands and silvopastoral systems developed under Monterey pine established in agronomic and forest soils.

4.3. MATERIALS AND METHODS

4.3.1. Characteristics of the study site

The experiment was initiated in December 2006 through the use of 27 cilindric pots of about 2 m^3 (144 cm height x 134.5 cm width) that were installed in the town of Piugos (Lugo, Galicia, NW Spain, European Atlantic Biogeographic Region) at an altitude of 470 m above sea level and filled with soils. Soils are gleyic umbrisols (FAO classification) and Umbrept Inceptisols (USDA system). Figure 4.1 shows the monthly mean precipitation and temperature values for 2007, 2008, and 2009 and the normal mean precipitation and temperature values of the study area. The total annual rainfall was 658.1, 1000, and 872.7 mm in 2007, 2008, and 2009, respectively. Meanwhile, the rainfall registered in the spring of 2007, 2008, and 2009 was 439.8, 604.5, and 368.4 mm, respectively. In general, these years were drier than the mean year (998.3 mm) for the study area. However, the mean monthly precipitation in 2007 was higher than the mean normal precipitation from June to August, which reduced the drought period found in 2008 and 2009 that limited pasture growth. The annual mean temperature was mild (12 ºC).

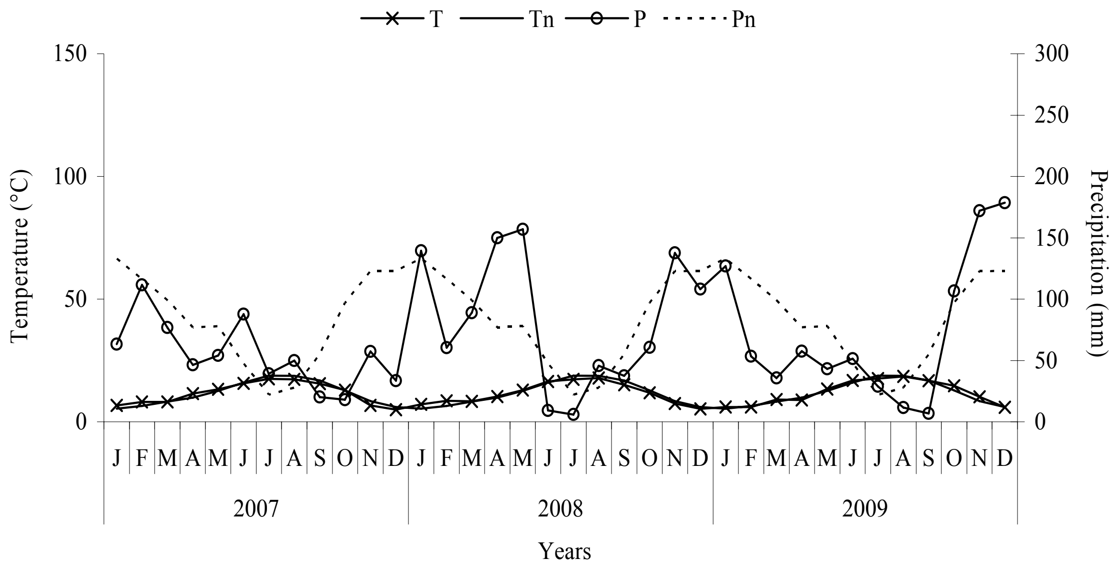

Figure 4.1: Mean monthly precipitation and mean monthly temperature in 2007, 2008, and 2009 and mean normal data for the study area. T: mean monthly temperature (ºC), Tn: mean normal temperature, P: mean monthly precipitation (mm), and Pn: mean normal precipitation.

Fifteen pots were filled with agronomic soil from Sarria (Lugo, Galicia, NW Spain) and the other with forest soil (12 pots) from Bascuas (Condesmo, Lugo, Galicia, NW Spain). In each pot, a lysimeter was installed at a depth of 135 cm to study the leaching of nutrients. The lysimiter was a PVC pipe of 2 cm of diameter introduced after making a hole in the pots. The tube is completely adjusted to the pot and no water is leached outside with the exception of the hole of the PVC pipe.

Initial agronomic soil analyses showed a highly acid KCl pH (4.46) (Faithfull, 2002), low soil organic matter (SOM: 36.3 g kg^{-1}) (Kowalenko, 2001), and a total N (1.9 g kg^{-1}) and P (0.3 g kg^{-1}) (Castro et al., 1990). Meanwhile, the forest soil analyses also had a highly acid KCl pH (4.27) (Faithfull, 2002), higher SOM (72 g kg^{-1}) (Kowalenko, 2001) and total P (0.8 g kg^{-1}) contents and a lower concentration of total soil N (1.8 g kg^{-1}) (Castro et al., 1990) than the agronomic soils. All heavy metal concentrations in both the agronomic and forest soils (Table 4.1) were below the maximum thresholds for the use of sewage sludge as fertilizer, as specified by the EU Directive 86/278/CEE (UE, 1986) and Spanish legislation under R.D. 1310/1990 (BOE, 1990).

Soil	Heavy metal concentrations (mg kg^{-1}) Cd	Cu	Cr	Ni	Pb	Zn
Initial agronomic soil	0.1	1	0.9	-	17.7	28.8
Initial forest soil	0.9	7.8	2	-	-	32.5
Spanish law limits	1-3	50-210	100-150	30-112	50-300	150-450

Table 4.1: Heavy metal concentrations in the agronomic soil and in the forest soil at the beginning of the experiment and legal limits established by European Directive 86/278 and Spain R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH<7, maximum: soil pH>7). -: element concentration below detection limit of the technique used in its determination.

4.3.2. Experimental design

The experimental design was a randomized block with three replicates and nine treatments. Treatments followed a design that consisted of a fractional factorial design of a 2p fully factorial, with "p" = 4 factors (2 levels per factor).The treatments established were chosen because they are the most traditional practices in the area of study (agronomic soil without tree, forest soil without pasture, and silvopastoral systems) in forest and agronomic lands. The treatments consisted of the following: (1) Agronomic soil + pasture sowing (Agronomic + P); (2) Agronomic soil + pasture sowing + sewage sludge (Agronomic + PS); (3) Agronomic soil + pasture sowing + mineral (Agronomic + PM); (4) Agronomic soil + pasture sowing + sewage sludge + tree (Agronomic + PST); (5) Agronomic soil + pasture sowing + mineral + tree (Agronomic + PMT); (6) Forest soil + sewage sludge + tree (Forest + ST); (7) Forest soil + mineral + tree (Forest + MT); (8) Forest soil + pasture sowing + sewage sludge + tree (Forest + PST); and (9) Forest soil + pasture sowing + mineral + tree (Forest + PMT). The following physical parameters were used:

- Pasture sowing (P): the pasture was sown with a mixture of cocksfoot (*Dactylis glomerata* (L.)) var. Artabro (12.5 kg ha^{-1}) (Dg), ryegrass (*Lolium perenne* (L.)) var. Brigantia (12.5 kg ha^{-1}) (Lp), and white clover (*Trifolium repens* (L.)) var. Huia (4 kg ha^{-1}) (Tr) in December 2006.
- Tree (T): a one-year-old Monterey pine tree was planted in January 2007.
- Sewage sludge (S): an anaerobically digested sludge with an input of 320 kg total N ha^{-1} applied in December 2006.

- Mineral (M): in the Agronomic + PM, Agronomic + PMT, Forest + MT, and Forest + PMT treatments, 500 kg ha^{-1} of 8 (% N):24 (% P_2O_5):16 (% K_2O) were applied at the beginning of the years 2007, 2008, and 2009, and 40 kg of N ha^{-1} as calcium ammonium nitrate (26% of N) was applied after each harvest.

4.3.3. Sewage sludge

Anaerobically digested sludge was taken from the municipal waste treatment plant of Lugo. A calculation of the required amount of sludge was conducted according to its percentage of total N and dry matter content (EPA, 1994) and taking into account that around 25% of the total N from anaerobically digested sewage sludge is available in the first year after application. EU Directive 86/278/CEE (1986) and Spanish regulation R.D. 1310/1990 (BOE, 1990) regarding heavy metal concentrations in the application of sewage sludge to soil were also considered. The composition of the sewage sludge applied is summarized in Table 4.2.

Parameters	Values	
	Anaerobic sludge	Spanish law limits
Dry matter, %	20.47	
pH	7.47	
N, g kg^{-1}	35	
P, g kg^{-1}	17.8	
K, g kg^{-1}	3.5	
Ca, g kg^{-1}	27.1	
Mg, g kg^{-1}	8.4	
Na, g kg^{-1}	1.5	
Fe, g kg^{-1}	17.9	
Cr, mg kg^{-1}	39.4	1000-1500
Cu, mg kg^{-1}	142.7	1000-1750
Ni, mg kg^{-1}	29.4	300-400
Zn, mg kg^{-1}	1248.56	2500-4000
Cd, mg kg^{-1}	0.7	20-40
Pb, mg kg^{-1}	84.4	750-1200
Mn, mg kg^{-1}	6.1	

Table 4.2: Chemical properties of the sewage sludge applied and legal limits established by European Directive 86/278 and Spain R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH<7, maximum: soil pH>7).

4.3.4. Field samplings and laboratory determinations

Soil samples were collected at a depth of 25 cm, as described in R.D. 1310/1990 (BOE, 1990) in March 2008 and in January 2009. In the laboratory, the pH of the KCl was determined as 1:2.5 soil: 0.1 M KCl (Faithfull, 2002). The total C content in the soil was determined by oxidation of the total organic matter with potassium dicromate and sulphuric acid. The excess of dicromate was valorated with Mohr salt (Kowalenko, 2001). The percentage of organic matter was calculated by multiplying the total C content of the soil by the de Van Bemmelen factor (1.724). Cation Exchange Capacity (CEC) was calculated as the sum of the concentrations of Ca, K, Mg, Na and Al expressed as cmol(+) kg^{-1} of soil, after extraction with 0.6 N $BaCl_2$ (Mosquera and Mombiela 1986). The total soil N and total soil P concentrations were determined after micro-kjeldahl digestion with a TRAACS-800+ autoanalyzer, as described by Castro et al. (1990) (US-786-86 A method, for N and US-787-86 A method, for P (Bran+Luebbe, 1979)). The available P was measured after extraction with Mehlich 3 (Mehlich, 1985) with the TRAACS-800+ autoanalyzer (US-787-86 A method (Bran+Luebbe, 1979) and water volume measured with a volumetric ware. Total nitrate leached was estimated by multiplying nitrate concentration and water leached in each sampling. Nitrate leached for a period in each sampling was summed to obtain the global period nitrate leached.

Water was extracted each week from the lysimeters unless drought caused a lack of water. Nitrate was determined according to Bremner (1965) using a continuous-flow analytical system (TRAACS-800+).

Tree height and diameter were measured with graduated ruler and calliper, respectively, in September 2009.

To estimate pasture production, botanical composition, and crude protein (CP) and P content of the pasture, two samples of pasture were randomly taken with an electric hand clipper at a height of 2.5 cm per pot (0.3 x 0.3 m^2) in May, June, and August 2007; in May and July 2008; and in May and June 2009 (autumn data were not used in this study). Later, the samples were labelled and transported to the laboratory, where the samples were weighed and separated by hand according to the different plant species and the senescent material. They were then dried at 60 °C for 72 hours to determine the harvest pasture production and the botanical composition weight. The CP and P content of the pasture were determined after micro-kjeldahl digestion with a TRAACS-800+ autoanalyzer, as described by Castro et al. (1990).

4.3.5. Statistical analysis

The data obtained from soil, tree and pasture variables were analyzed with three 2- way ANOVAs (proc glm procedure) following the model $Y_{ij} = \mu + F_i + T_j + \varepsilon_{ij}$. The first ANOVA was performed to discern the effects of soil type (agronomic vs forest) with mineral and sludge fertilization in Pasture + Tree (silvopastoral systems) with two levels of fertilization (F: sludge and mineral) x two types of soil (T: Forest and agronomic) and their interactions (treatments Agronomic + PMT, Agronomic + PST, Forest + PMT and Forest + PST); where Y_{ij} is the studied variable; μ is the variable mean; F_i is the fertilizer factor i; T_j is the soil type factor j; and ε_{ij} is the error. The second 2-way ANOVA was performed to discern the effects of two levels of pasture vegetation (S: sown and unsown pasture) with two levels of fertilization (F: sludge and mineral) and their interactions on forest soil (treatments Forest + PMT, Forest + PST, Forest + ST and Forest MT) where Y_{ij} is the studied variable; μ is the variable mean; F_i is the fertilizer factor i; T_j is the vegetation factor j; and ε_{ij} is the error. The third 2-way ANOVA was performed to discern the effects of two levels of tree plantation (T: tree and no tree plantation) with two levels of fertilization (F: sludge and mineral) and their interactions on agronomic soil (treatments Agronomic + PMT, Agronomic + PST, Agronomic + PM and Agronomic + PS) where Y_{ij} is the studied variable; μ is the variable mean; F_i is the fertilizer factor i; T_j is the vegetation factor j; and ε_{ij} is the error. Finally A 1-way ANOVA of one factor with three levels of fertilization F (NF, mineral and sludge) was to discern the effects of fertilization on agronomic soil with herbaceous vegetation (treatments NF, Agronomic + PM and Agronomic + PS). The Tukey´s HSD test was used for subsequent pair wise comparisons (*P < 0.05*; a = 0.05). The statistical software package SAS (2001) was used for all analyses.

4.4. RESULTS

4.4.1. Soil

4.4.1.1. *KCl soil pH, CEC, SOM percentage, total N and total and Mehlich P*

The KCl soil pH, the CEC and the total soil levels of N and P and P extracted by Mehlich 3 in 2008 and 2009 are shown in Figure 4.2. The soil pH was significantly affected by the type of soil ($p<0.01$) in *Pasture + Tree* treatments in 2008 (agronomic: 4.91 *vs* forest: 4.44) and 2009 (agronomic: 4.77 *vs* forest: 4.33) as happened with CEC in 2008 ($p <0.001$; agronomic: 8.35 cmol (+) kg $soil^{-1}$ *vs* forest: 3.65 cmol (+) kg $soil^{-1}$)

and 2009 ($p<0.001$; agronomic: 9.44 cmol (+) kg soil^{-1} *vs* forest: 4.01 cmol (+) kg soil^{-1}). Soil total N and P were also significantly affected by the type of soil. Soil total N ($p<0.01$ and $p<0.001$) and P ($p<0.001$ and $p<0.01$) were significantly higher in agronomic soils than in forest soils in 2008 (agronomic: 1.7 g total N kg^{-1} and 0.5 g total P kg^{-1}*vs* forest: 1.1 g total N kg^{-1}and 0.2 g total P kg^{-1}) and in 2009 (agronomic: 2 g total N kg^{-1} and 0.6 g total P kg^{-1} *vs* forest: 1.6 g total N kg^{-1} and 0.3 g total P kg^{-1}). For the SOM percentage, no significant differences between the treatments were found ($p>0.05$ (data not shown)). No differences in soil variables appeared between treatments when only *Forest soils* were taken into account, with the exception of KCl pH, which was significantly reduced when mineral fertilization was applied and tree and pasture was established compared with those treatments planted with trees and receiving sludge fertilization, but without pasture sowing. On the other hand, in the *Agronomic soils*, the soil CEC was significantly higher when the pasture was fertilized with sludge (9.69 cmol(+) kg^{-1} soil) than with mineral (8.65 cmol(+) kg^{-1} soil) in 2009 ($p<0.01$), being P extracted by Mehlich 3 significantly ($p<0.001$) improved when mineral or sludge fertilized was applied in 2008 and compared with no fertilization treatment in agronomic soils.

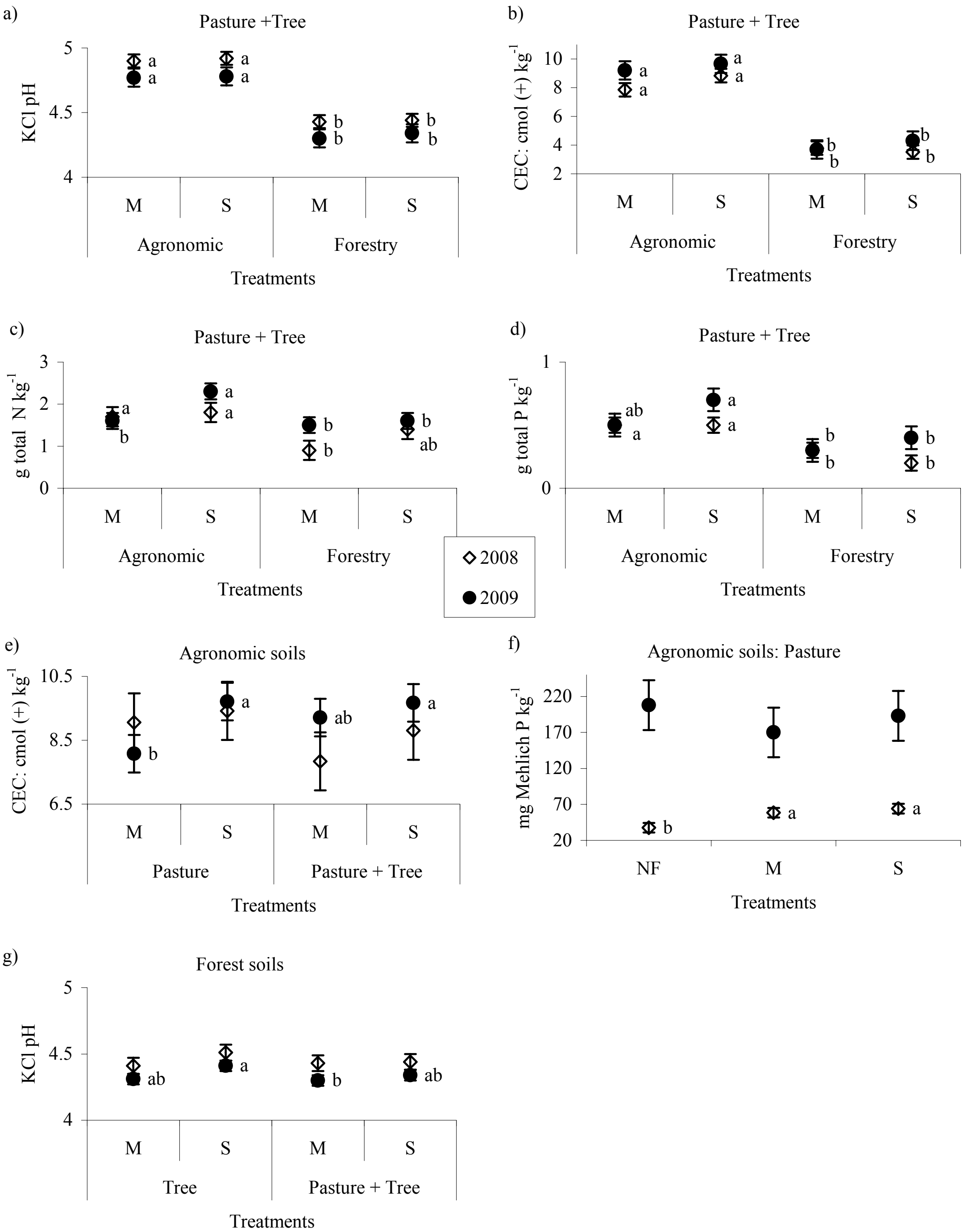

Figure 4.2: Soil pH in KCl (a), CEC (cmol(+) kg^{-1}) (b), and total N (c) and P (d) concentrations in soil (g kg^{-1}) in Pasture + Tree treatments, CEC (cmol(+) kg^{-1}) in agronomic soils (e), P Mehlich 3 (mg kg^{-1}) in no forested agronomic soils (f) and soil pH in KCl in forest soils (g) in 2008 and 2009. M: Mineral fertilization, S: sewage sludge fertilization. Different letters indicate significant differences between treatments within the same year. Vertical lines indicate mean standard error.

4.4.1.2. *Nitrate concentration in leaching water*

The significant effects of the treatments on nitrate concentrations in the leaching water are shown in Figure 4.3. The nitrate concentration in the leaching water of agronomic soils was usually above the maximum set by the EU directive for drinking water in all treatments (<11.3 mg NO^{-}_3 –N l^{-1}) (UE, 1980) in December 2006 and in the first months of 2007. The nitrate concentrations of all treatments measured from May 2007 to 2009 were below the maximum allowed by the EU directive for drinking water (always below 5 mg NO^{-}_3 –N l^{-1}) and without differences between treatments (data not shown). With respect to the effects of the treatments in the *Pasture + Tree* pots at the beginning of the experiment, the results show that by 12 March 2007 the nitrate concentrations in the leaching water were significantly higher in Agronomic treatments sown with pasture and planted with trees than in the same Forest treatments. After this sampling date, no differences were found between treatments. However, the total nitrate leached was significantly higher ($p<0.05$) when sewage sludge (3.05 grams of nitrate until 24 May 2007) was applied in *Forest soils* compared with mineral fertilizers (1.89 grams of nitrate until 24 May 2007). There were not differences between treatments in the subsequent years ($p>0.05$).

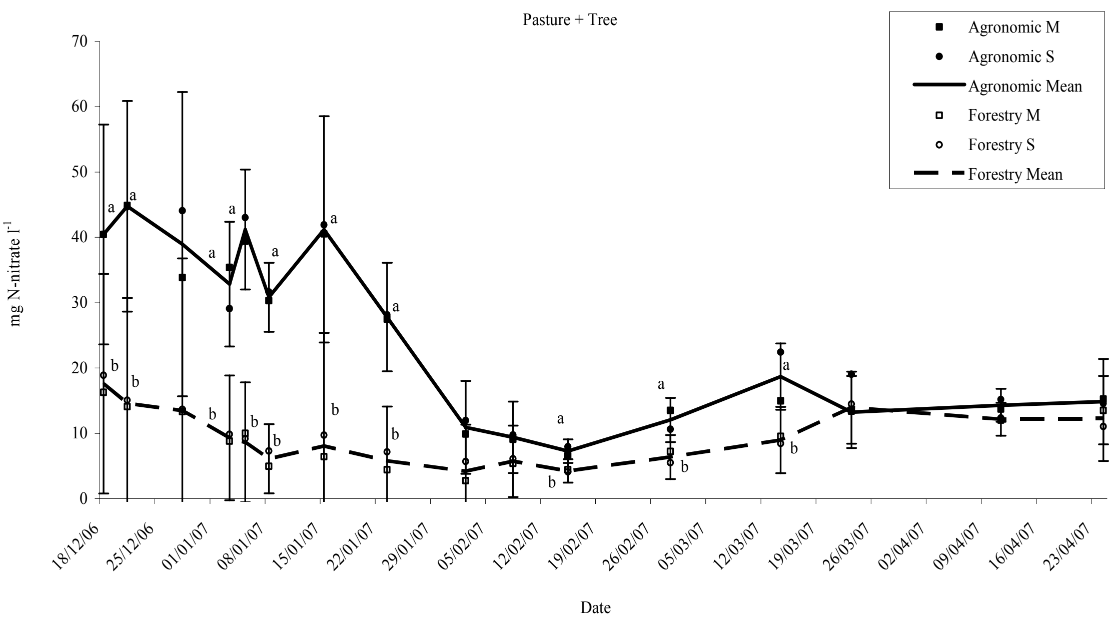

Figure 4.3: Nitrate (mg N-nitrate l^{-1}) concentration in leached water in Pasture+Tree treatments. M: mineral; S: sludge. Different letters indicate significant differences between soil mean treatments. Vertical lines indicate mean standard error.

4.4.2. Tree height and diameter

Tree heights (179 cm) and basal diameters (41.25 mm) for each treatment were not significantly affected by any of the treatments in 2009.

4.4.3. Pasture

4.4.3.1. *Production*

Pasture production for the different fertilization treatments in the spring 2007, 2008, and 2009 can be observed in Figure 4.4. Significant differences were detected between the treatments in all years (p<0.001 in the spring of 2007, 2008, and 2009) with the exception of the spring 2007 and 2008 when the planting of trees were evaluated in agronomic soils and the spring 2008 when the three types of fertilization were compared in treeless pastures established in agronomic soils. The highest levels of pasture production were generally found in the spring of 2008 (2.2-13.4 Mg pasture ha^{-1}), while the lowest values were detected in the spring of 2009 (1.2-10.3 Mg pasture ha^{-1}). When the Agronomic and Forest treatments are compared (*Pasture + Tree* treatments), it is apparent that pasture production was lower in the Forest treatments (Forest + PMT and Forest + PST) fertilized with mineral or sludge than in Agronomic treatments (Agronomic + PMT and Agronomic +PST), with the exception of the first year of the study, when pasture production was similar in those pots fertilized with sewage sludge in forest soils (PST) to both Agronomic treatments (Agronomic + PMT and Agronomic + PST). Within the Agronomic treatments the fertilization with sewage sludge in no planted pots (Agronomic + PS) had higher pasture production than pots fertilized with mineral and planted with trees (Agronomic + PMT) in the last year of the study. On the other hand, in 2007, within the Agronomic treatments, pasture production was lower in no fertilized pots than in those fertilized with sewage sludge (Agronomic + PS). However, mineral fertilization reduced pasture production in the last year of the study compared with no fertilization treatment. Moreover, in the *Forest soils*, the mineral fertilization decreased pasture production (Forest + MT and Forest + PMT), while the fertilization with sewage sludge increased this variable (Forest + ST and Forest + PST) in 2007 and 2009. However, pasture production was reduced in forest soils when pasture was sown, tree was planted and sewage sludge was applied compared with the rest of the treatments in 2008.

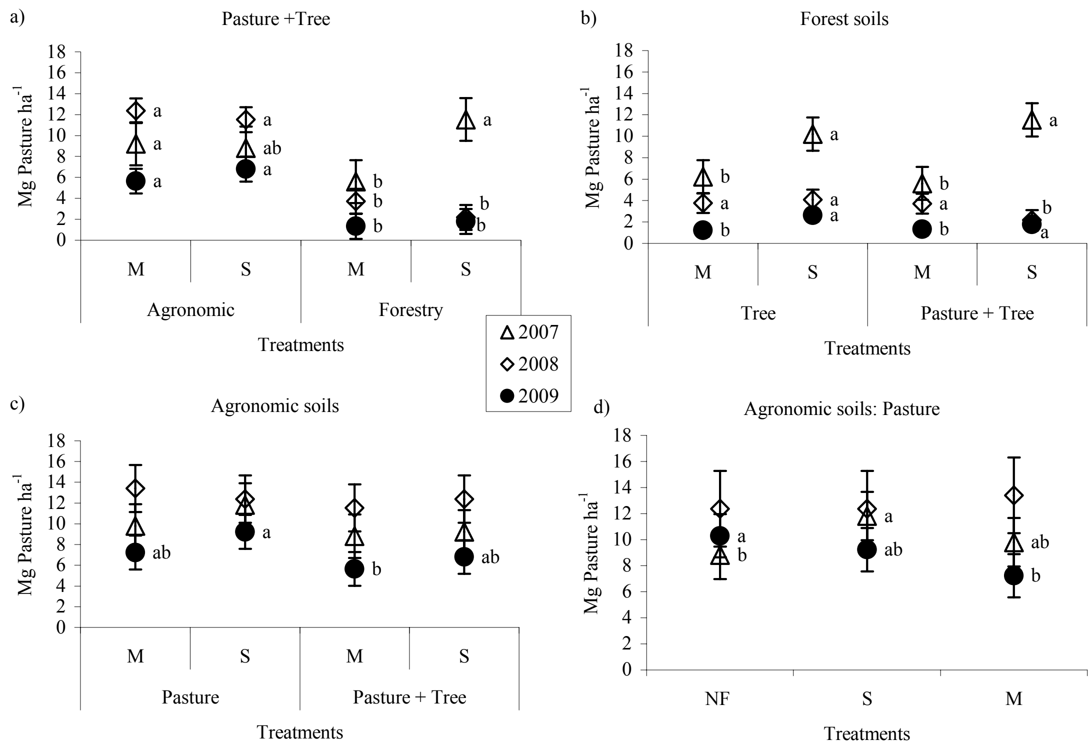

Figure 4.4: Spring pasture production (Mg ha^{-1}) under Pasture + Tree (a), Forest (b), Agronomic (c) and Agronomic not afforested soils (d) in 2007, 2008, and 2009. Different letters indicate significant differences between treatments within the same year. Vertical lines indicate mean standard error.

4.4.3.2. *Botanic composition*

The significant ANOVA results of the weight proportion (% dry matter) of the sown species (cocksfoot, ryegrass, and white clover) and the most representative spontaneous species (chamomile (Cha) [*Chamomilla recutita* (L.)], creeping bentgrass (Crb) [*Agrostis stolonifera* (L.)], narrowleaf plantain (Np) [*Plantago lanceolata* (L.)], and yarrow (Yar) [*Achillea millefolium* (L.)]) in the pasture over the entire study period are shown in Table 4.3. There were other spontaneous species, though their contributions were minimal (data not shown). In general, the percentage of cocksfoot was significantly affected by treatments (Table 43) in all harvests of the spring-summer of 2007 and 2008, with the exception of the first harvest of 2007 and harvests of 2008 when the effect of planting trees within agronomic soils was evaluated. Moreover, all ANOVAs of cocksfoot were also significant in May 2009, but cocksfoot was not affected by treatments when the comparison of two types of soil (agronomic and forest) in *Pasture + Tree* pots was carried out in the first harvest of 2009. Regarding ryegrass, it was only affected by treatments in August 2007 (when 1) agronomic and forest soils were compared in *Pasture + Tree* pots and 2) treatments within forest soils were evaluated), in May 2008 in all treatments (with the exception of the evaluation of tree plantation in agronomic soils) and in May 2009 (when agronomic and forest soils in *Pasture + Tree* pots were compared and the effect of tree planting within agronomic soils was evaluated). The fertilization with mineral or sludge initially increased the proportion of cocksfoot in forest soils compared with agronomic soils in *Pasture + Tree* treatments (Figure 4.5). Moreover, the positive effect of the mineral fertilization compared with the sludge fertilization on cocksfoot percentage in forest soils was more evident as the study advanced. On the contrary, agronomic soils had a significantly higher proportion of clover than forest soils mostly when sludge instead of mineral was applied in August 2007, May 2008 and July 2008.

Within the *Forest soils*, there was a positive effect of pasture sowing on the proportion of sown species (cocksfoot and ryegrass) in all harvests until May 2009 (Figure 4.5). The establishment of cocksfoot in forest soils was better with mineral than with sludge fertilization. On the contrary, creeping bentgrass was the main pasture species found in forest unsown pots in 2007 and 2008 (Figure 4.5). Creeping bentgrass percentage was significantly higher in unsown than sown forest treatments from the start of the experiment to July 2008, when the increment of the percentage of creeping bentgrass in unsown pots compared with sown pots was only significant if mineral fertilization was used. In May 2009, mineral fertilization improved the percentage of creeping bentgrass in unsown pots compared with those pots previously sown and fertilized with sludge.

Within *Agronomic soils*, the percentage of cocksfoot was significantly (Table 3) improved when sewage sludge (12.6 % and 22.3 %) instead of mineral (4.19 and 7.5 %) was used in May and August 2007, respectively. On the contrary, the percentage of cocksfoot was significantly higher in mineral (44.17^{a} %) than sludge (12.5^{b} %) in no afforested pots in May 2009, but no differences were found in agronomic afforested pots regarding the percentage of cocksfoot (26.68^{ab} % and 22.32^{ab} % in mineral and sludge, respectively) in the same harvest (different superscript letters indicate significant differences between treatments). However, in May 2009, the use of sludge (11.42^{a} %) increased the proportion of ryegrass compared with mineral (1.15^{b} %) in no afforested pots, and sludge (1.80^{b} %) in afforested pots, but the percentage of ryegrass was similar in both afforested pots (4.51^{ab} % in mineral afforested pots). Mineral fertilization also increased the percentage of cocksfoot compared with no fertilization in agronomic soils (Figure 4.5).

Regarding the spontaneous species, it was found that chamomile was initially better established in agronomic (21.87 %) than forest soils (0.02 %) in *Pasture + Tree* treatments in May 2007. On the other hand, the percentage of narrowleaf plantain [*Plantago lanceolata* (L.)] was significantly higher when sewage sludge was applied in agronomic soils (5.91^{a} %) compared mineral fertilization (0.05^{b} %) in forest soils in July 2008, but the percentage of this species was similar in mineral fertilized agronomic soils (2.47^{ab} %) and in sludge fertilized forest soils (1.63^{ab} %) in the same harvest. Finally, yarrow was improved within *Forest soils* when no sown was carried out but mineral fertilization was applied (18.81^{a} %) compared with sludge fertilization with sowing (0.55^{b}) but similar to the percentage found in those pots unsown and fertilized with sludge (12.48^{ab} %) or sown and fertilized with mineral (2.88^{ab} %) in July 2008.

Forest soil

Sown species	For	Fert	For*Fert	SEM
Tr May-08	*	*	*	5.2
Tr July-08	ns	*	ns	8.8
Dg May-07	***	ns	*	11.5
Dg June-07	***	ns	ns	6.7
Dg August-07	***	ns	*	12.5
Dg May-08	***	ns	ns	17.3
Dg July-08	***	ns	ns	15.2
Dg May-09	**	ns	ns	16.5
Lp May-07	***	*	*	8
Lp June-07	***	ns	ns	12.8
Lp August-07	***	*	ns	12.8
Lp May-08	***	ns	ns	7.8
Unsown species	**For**	**Fert**	**For*Fert**	**SEM**
Crb May-07	***	ns	ns	24.6
Crb June-07	***	ns	ns	24.8
Crb July-08	**	ns	ns	19.1
Crb May-09	**	ns	ns	21.8
Yar July-08	**	ns	ns	10.2

Pasture + Tree

Sown species	Soil	Fert	Soil*Fert	SEM
Tr August-07	**	ns	ns	5.1
Tr May-08	**	ns	ns	8.5
Tr July-08	***	ns	ns	7.5
Dg May-07	***	**	*	8.6
Dg June-07	**	ns	ns	9.0
Dg August-07	***	***	ns	9.4
Dg May-08	***	ns	ns	17.83
Dg July-08	*	*	ns	17.3
Lp August-07	ns	*	ns	16.3
Lp May-08	**	ns	ns	18.8
Lp May-09	ns	ns	*	7.3
Unsown species	**Soil**	**Fert**	**Soil*Fert**	**SEM**
Crb July-08	*	ns	ns	11.1
Np July-08	*	*	ns	2.9

Agronomic soil

Sown species	Silvo	Fert	Silvo*Fert	SEM
Dg June-07	ns	**	ns	6.7
Dg August-07	ns	*	ns	8.07
Dg May-09	ns	**	**	13
Lp May-09	ns	ns	**	5.5
Unsown species	**Silvo**	**Fert**	**Silvo*Fert**	**SEM**
Crb May-07	**	*	ns	7.5
Crb May-08	ns	**	**	7.8

Agronomic soil

Sown species	Fert	SEM
Dg May-07	**	2.7
Dg June-07	***	2.7
Dg August-07	**	8.4
Dg May-08	*	9.2
Dg July-08	*	6.8
Dg May-09	*	17.6
Lp May-08	*	17.1
Unsown species	**Fert**	**SEM**
Crb May-08	***	7.04

Table 4.3: ANOVA results for treatments with significant effects for sown grasses (cocksfoot (Dg), ryegrass (Lp), and white clover (Tr)), spontaneous species (chamomile (Cha), creeping bentgrass (Crb), narrowleaf plantain (Np) and yarrow (Yar)) in May, June, and August 2007; May and July 2008; and May 2009. SEM: mean standard error, ns: not-significant, *: $p<0.05$, **: $p<0.01$, ***: $p<0.001$.

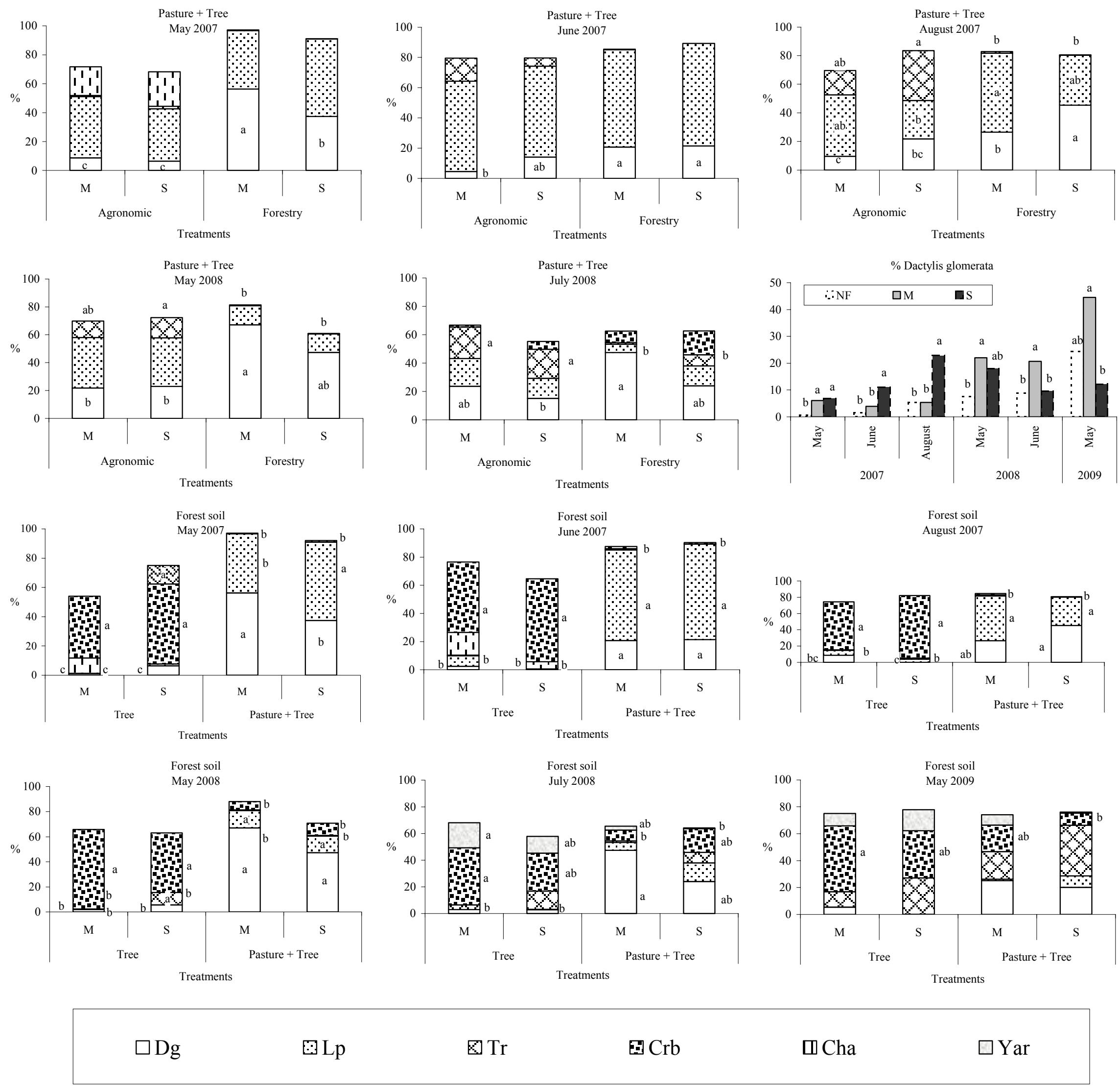

Figure 4.5: Proportion in weight (% dry matter) of sown (Dg: cocksfoot, Tr: white clover, and Lp: ryegrass) and spontaneous species (Cha: chamomile, Crb: creeping bentgrass and Yar: yarrow) in pasture under the different treatments in May, June, and August 2007; May and July 2008; and May and June 2009 of Pasture + Tree and Forest and Agronomic soils treatments. M: mineral, S: sludge, NF: No fertilization: Different letters indicate significant differences between treatments within the same harvest.

4.4.3.3. *Pasture concentrations of CP and P*

The concentrations of CP and P in the pasture in the spring 2007, 2008, and 2009 are presented in Figure 4.6. The concentration of CP was significantly affected by the treatments in August 2007 (soil effect: $p < 0.001$) and in December 2008 (soil effect: $p<0.01$) in *Pasture + Tree* treatments and revealed that CP was higher in agronomic than forest soils, with the exception of pasture grown in agronomic pots fertilized with sludge in December 2008. On the other hand, the concentration of P in pasture was significantly affected by treatments in June (soil effect: $p<0.001$) and August (soil effect: $p<0.001$) 2007 and in December 2008 (soil x fertilization interaction effect: $p<0.05$; soil effect: $p<0.001$) when the effect of the type of soil was evaluated. As happened with CP, the concentration of P was significantly higher in agronomic than forest soils, with the exception of pasture developed in agronomic soils fertilized with sludge which did not differ from the same forest treatment.

Within *Forest soils*, the lack of sowing increased the CP concentration of pasture. Mineral fertilization in unsown treatments significantly increased the concentration of CP compared with sludge and mineral fertilization in sown pots in August 2007 (sown effect $p<0.001$) and only with sludge fertilization in June 2007 (sown effect: $p<0.001$).

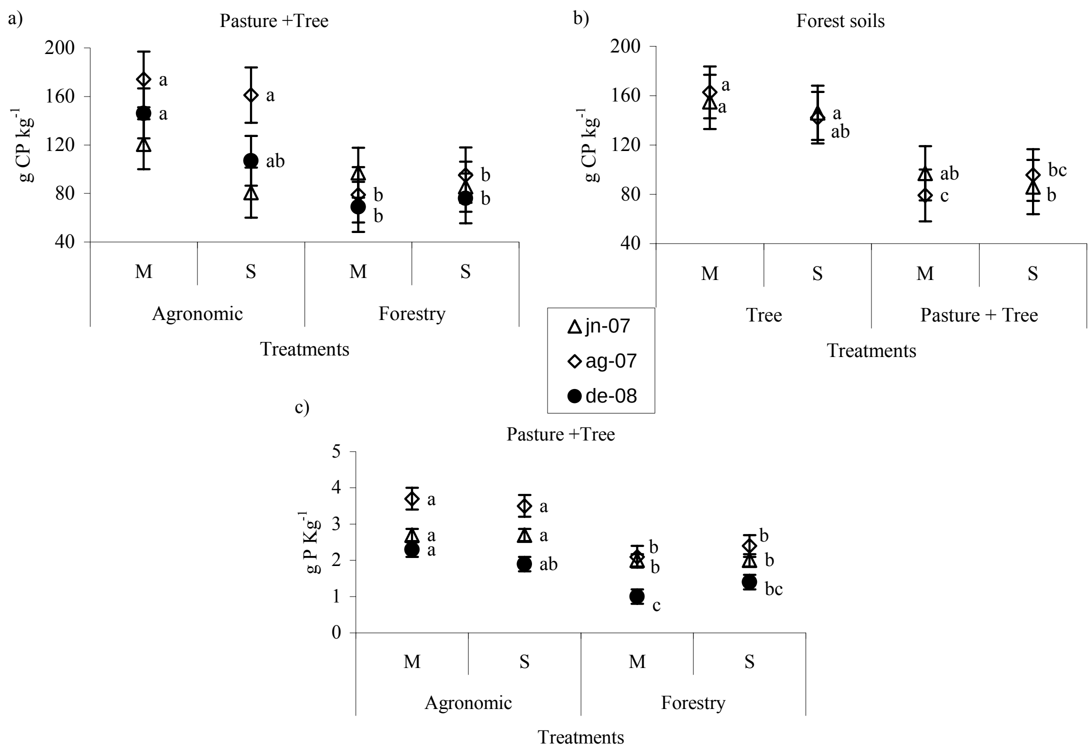

Figure 4.6: Concentrations of crude protein (CP) in pastures (g kg^{-1}) under Pasture + Tree (a) and forest soil (b) treatments and phosphorous in pastures (g kg^{-1}) under Pasture + Tree (c) in the significant harvests of 2007, 2008, and 2009. M: Mineral, S: sludge Different letters indicate significant differences between treatments within each harvest. Vertical lines indicate mean standard error.

4.5. DISCUSSION

All soil studied variables (KCl pH, CEC, total N and P), excluding organic matter, were significantly higher in agronomic compared with forest soils and explains the better initial fertility stage and pasture production found in the Agronomic than in the Forest treatments. Moreover, the highest presence of white clover in the Agronomic treatments probably increased pasture production because *Rhizobium* N fixation is performed by this species (González, 1992; Whitehead, 1995; Green et al., 1999; López-Díaz et al., 2009), which increases the input of N into the soil (Mosquera-Losada et al., 1999), and the subsequent consumption of this nutrient by grasses. Authors such as González (1992) indicate that 30% of white clover in pastures developed in North Western Spain can incorporate up to 250 kg ha^{-1} $year^{-1}$ of N. In any case, pasture production ranges found in the present experiment were similar to those usually described in North Western Spain forest soils (0.5-6 Mg pasture ha^{-1}) (Mosquera-Losada et al., 2001) and in North Western Spain agronomic soils (6-12 Mg ha^{-1}) (Mosquera-Losada and González-Rodríguez, 1999).

Although soil pH was reduced from the first to the last year of the experiment, N and P levels in soil were usually higher if sludge instead or mineral were used in Pasture + Tree treatments developed in agronomic soils. Soil pH in 2009 was reduced in comparison to 2008 probably due to the N mineralization (the step from NH_4^+ to NO_3^- occurs, and H^+ is released into the soil solution media after leaching of NO_3^- by rainfall (Whitehead, 1995)) and tree and pasture cation extractions from the soil (Mosquera-Losada et al., 2006). In the last year of the experiment, soil total N was higher if sludge instead of mineral fertilizer had been previously applied in pasture and tree agronomic soils. The sludge nutrient release rate is slower than those from mineral fertilizers (EPA, 1994; Smith, 1996) and this would explain the extended effect of the sludge in time on soil total N variable. The same tendency was found with total soil P.

Water pollution from nitrate leaching, was only relevant at the start of the experiment when both tree and pasture were at the establishment phase, and therefore they were less efficient taking up soil nutrients like nitrate. Afterwards, the adequate establishment of pasture and tree limited nitrate leaching, even though mineral fertilizer inputs were annually performed. At the beginning of the present experiment (2006), the nitrate-N content of the soil solution exceed the 11.3 mg L^{-1} limit for drinking water set by the EU (UE, 1980). In the following samplings, the concentrations of nitrate in the leached water from the agronomic soils were also above the value of 11.3 mg NO^-_3 –N L^{-1}, and it was higher than that found in forest soils (Knight et al., 1989). High initial soil pH of agronomic soil at the beginning of the experiment would explain a higher mineralization activity, and therefore nitrate leaching, in agronomic than in forest soils, which caused an increase of pasture production and the levels of CP. Moreover, the initial higher percentage of grasses instead of annual species in the Forest treatments than in the Agronomic treatments could also have reduced the leaching of nitrate in the forest soils, while taking into account that perennial grasses have a greater capacity for taking up soil nitrate than weeds (Humphreys et al., 2006; Abberton et al., 2008). Moreover, the initial establishment of dicotyledonous annual species, such as chamomile, could also have increased the nitrate leaching in the agronomic treatments when the annual species died, since dicotyledonous species are richer in N than monocotyledonous species (Hanley et al., 1992; Paré et al., 2006). The absence of annual species in the seed bank of the forest soils was probably the cause of the best initial establishment of the sown species in the forest soils that were simultaneously fertilized and sown than in the agronomic soils, where the annual species showed rapid-growth characteristics (abundant seed – rapid germination) (Grime et al., 2007; Mosquera-Losada et al., 2009a).

There were no appreciably significant differences in the forest soils as a result of the different treatments of fertilization or sowing, mainly because most of the soil's biological activity was greatly restricted as pH was very low in the forest soils (Omil et al., 2007; Djukic et al., 2009). However, a positive effect of sewage sludge inputs on soil pH was detected when the less intensive system (no sowing) was compared with more intensive systems implying mineral fertilization and sown of pasture. The positive effect o sludge applications on soil pH in very acid soils were also described in soils receiving a higher total quantity of sludge than in the present experiment (López-Díaz et al., 2007). The improvement of soil pH caused by sludge applications may explain the increase of the total nitrate leaching and the pasture production in forest soils compared with mineral fertilizations. This result could be firstly explained by the residual effect of organic fertilizers compared with mineral fertilizers described by the EPA (1994) and secondly because the sludge added more Ca, Mg and micronutrients than the mineral fertilizer (Smith, 1996; López-Díaz et al., 2007; Mosquera-Losada et al., 2010b).

Even though the forest soils were only significantly affected by the treatments in terms of KCl pH, they modified the botanical composition and CP in the initial samplings. However, these differences disappeared at the end of the study. The improvement in the percentage of ryegrass and cocksfoot species as a result of sowing caused a lower crude protein percentage than in the pastures of unsown forest soils. Sown species like ryegrass and cocksfoot are not usually adapted to the low fertility of forest soils, being less extractive than weeds in forest soils (Whitehead, 1995), thus reducing the proportion of N in the pasture. This reduction ultimately caused the reduction of the percentage of sown species at the end of the experiment in sown forest soils.

The response of the soil and pasture production variables to treatments within the agronomic soils did appeared two years after the experiment had begun, probably due to the better initial soil fertility. CEC was significantly increased in sludge fertilized treatments with or without tree planting than in mineral when trees were not planted, and it could be explained by the physical soil improvement caused by the sludge (Smith, 1996). Moreover, in 2009, those treatments with sludge fertilization in the agronomic soils without tree planting had higher pasture production than mineral fertilization treatment that had been previously planted. Mineral fertilization caused a direct increase of N, P, and K concentrations in soil, which can reduce other cation levels in soil, thus limiting pasture production (Whitehead et al., 1995). This effect can be seen in the lower CEC of the PM treatment compared with the rest of the agronomic treatments fertilized with the sludge in the last year of the study. Additionally, the presence of trees modified the soil conditions in the agronomic soil with the mineral fertilizer (PMT), which may also have reduced pasture production in this treatment due to the high extractions performed by both crops (tree and pasture), compared with agronomic treeless systems fertilized with mineral.

The initial improvement of soil P availability as a result of fertilization (mineral or sludge) was also previously described (Allen et al., 2006; Nair et al., 2007) and may help to explain the high production of pasture in sewage sludge fertilized pots compared with no fertilization in agronomic soils. Moreover, cation extraction may have reduced pasture production in mineral treatment compared with no fertilization at the end of the study.

In 2009, the Monterey pine heights and diameters varied from 164-193 cm to 49.5-58 mm, respectively. The tree diameters and heights were greater than those described by López-Díaz et al. (2009) in a study carried out in agrarian soils in North Western Spain, with a pH close to neutral (pH 6.3), and by Sánchez-Rodríguez (2000) in the Northwest of Spain. The higher growth of trees in our study could be explained by the high precipitation rate found in the summer during planting, which usually increases initial tree growth (Rigueiro-Rodríguez et al., 2000; Mosquera-Losada et al., 2006) and may have reduced the differences between treatments. A similar result was found by Rigueiro-Rodríguez et al. (2010a), in which fertilization treatments initially modified the Monterey pine response, but differences between the treatments disappeared when a humid summer for tree growth occurred. Tree plantation did not affect nitrate leaching probably because trees were too young to uptake N from soil when the nitrate concentrations in soil was high. However, tree plantation reduced pasture production in agronomic soils fertilized with mineral and KCl pH, and CP concentration in forest soils compared with the less intensive treeless pastures (López-Díaz et al., 2007).

The concentrations of CP (58-183 g kg^{-1}) and P (1.6-4.8 g kg^{-1}) were similar to the concentrations described by Grime et al. (2007) (1.5-4.5 g kg^{-1}) and Whitehead et al. (1995) (80-250 g kg^{-1}), respectively, with the exception of the CP concentrations in those harvests performed in the summer, which were usually lower due to the usual seasonal evolution of CP caused by the different pasture species' phenological growth states (Whitehead et al.,1995). The concentration of CP in the pasture did not reach the minimum requirements for the maintenance of live weight in sheep (94 g kg^{-1}) (NRC, 1985), horses (85 g kg^{-1}) (NRC, 1989), and goats (60 g kg^{-1}) (Lamand, 1981) in the summer harvests of 2007 and 2008. The concentration of P in all other harvests did meet the requirements for maintenance of live weight in sheep (1.6-3.7 g kg^{-1}) (NRC, 1985), horses (2 g kg^{-1}) (NRC, 1989), and goats (2.5 g kg^{-1}) (Lamand, 1981).

4.6. CONCLUSION

Fertilizer management, sowing and plantation practices caused different effects on the agrarian and forestry soils in our study. Agronomic soils were more fertile than the forestry soils due to the high pH and microbial activity of the former, which increases nitrate and P availability, and therefore the risk of leaching. Better nutrient availability in the agronomic soils increased pasture production as initially did sludge instead of mineral fertilization in forest soils due to the inputs of other nutrients done by sewage sludge compared with mineral, which probably increased mineralization in forest soils. Nitrate leaching was only relevant at the start of the experiment when trees and pasture were not enough developed to uptake nitrate. On the other hand, the soil variables evaluated in this study, with the exception of KCl, were not modified by fertilization or sowing treatments in the forest soils. However, the production, botanical composition and quality of the pasture developed in the forest soils were positively by sludge inputs instead of mineral due to the greater amount of nutrients applied with the former and the pH increase that sludge caused. The presence of grasses like cocksfoot or ryegrass was enhanced by sowing in forest soils. However, due to the low soil fertility, the quality of the pasture at the time these species were sown was low, and the sown grass species disappeared shortly after the establishment of the experiment. Finally, soil fertility was better preserved with the sludge than mineral fertilizer within the agronomic soils due to the broad range of nutrient applied with the former. Therefore, the use of the sludge as fertilizer allows nutrient recycling of this residue in poor soils and increases productivity and preserves fertility compared with mineral fertilizer at short (forest soils) and medium (agronomic soils) term.

4.7. ACKNOWLEDGMENTS

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PARTE V

Response to sewage sludge fertilisation in a *Quercus rubra* L. silvopastoral system: soil, plant biodiversity and tree and pasture production

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5.1. ABSTRACT

Silvopastoral systems are sustainable form of land management promoted by European Union. The productivity of the herbaceous and tree components in a silvopastoral system could be limited by soil fertility. The use of adequate doses of organic fertilisers like sewage sludge could enhance the productivity of both pasture and trees and promote biodiversity. The quantification of the best dose of sewage sludge to be applied in a silvopastoral system is important in order to enhance production and biodiversity in a silvopastoral system, while avoiding nitrate contamination of the ground water. This study aims to evaluate the effects of different doses of sewage sludge (100 kg total N ha^{-1}, 200 kg total N ha^{-1} and 400 kg total N ha^{-1}) on different variables of soil (soil pH, effective exchange capacity and the saturation percentage of Al, K, Ca, Mg and Na), tree growth and pasture (production, species richness and botanical composition) as compared to the treatment of no fertilisation in a silvopastoral system under *Quercus rubra* L. Sewage sludge applications initially improved soil nutrient levels (effective exchange capacity and Ca saturation percentage) and subsequently pasture production and tree growth when 200 and 400 kg of total N ha^{-1} were applied. On the other hand, the establishment of pasture and trees improved soil conditions at a medium term due to the organic matter input into the sandy soil, which increased species diversity specially at a 100 kg of total N ha^{-1} dose.

Keywords: agroforestry, afforestation, sowing, waste, dose

5.2. INTRODUCTION

Silvopastoral systems are agroforestry systems in which tree and pasture production is combined. These systems are currently being promoted by the EU (Council Regulation 1698/2005 (UE 2005)). Silvopastoral systems can produce social benefits, as the economic return is obtained earlier than when exclusive forest systems are used. As a result, silvopastoral systems tend to enhance the stabilisation of the rural population. Silvopastoral systems also provide environmental benefits such as the improvement of nutrient recycling, the control of soil erosion, a reduction in fire risk and an increase in carbon sequestration (Rigueiro-Rodríguez et al., 2008a; Howlett et al., 2010). Furthermore, silvopastoral systems can promote biodiversity through the creation of micro-sites within the plantation, such as shaded and unshaded areas. These micro-sites occur as a result of the presence of trees not found in purely agronomic land.

Biodiversity can be further promoted through the reduction of habitat fragmentation (Rois et al., 2006).

*Quercus rub*ra L. is a native species from the Atlantic coast of North America that is widely used for reforestation in Galicia and in other regions of Northern Spain (Álvarez-Álvarez et al., 2000). The exotic *Quercus rubra* L. is preferred over the native *Quercus robur* L. by European foresters because of its faster growth, which yields earlier profits for forest farmers (Renou-Wilson et al., 2008). *Quercus rubra* L. is frequently used in the establishment of silvopastoral systems (Balandier and Dupraz, 1999; Lehmkuhler et al., 2003) because this species possesses an open crown that allows light to reach the pasture surface, making it compatible with pasture production. Furthermore, because it is a deciduous species, it allows better light penetration than perennial conifers during the autumn and early spring, and it provides shading during the summer. Thus, evapotranspiration is reduced, leading to enhanced pasture production as compared to pasture under conifers or on open pasture sites (Álvarez-Álvarez et al., 2000).

In Galician silvopastoral systems, the productivity (of both understory and trees) can be limited by low soil fertility as a result of increased acidity (Zas and Alonso, 2002). The use of fertilisers can modify the productivity of various components of the system (including the pasture and trees) as well as its botanical composition (Mosquera-Losada et al., 2009a). One alternative that has been adopted in various countries around the world is the application of sewage sludge to soil as fertiliser (EPA, 1994). In Europe, this is regulated by the directive 86/278/EEC (UE, 1986). The use of sewage sludge as fertiliser is being promoted by the EU due to its specific organic matter and macronutrients content, particularly nitrogen (N) (MMA, 2006).

The main problem with the agricultural use of sewage sludge is the higher heavy metal concentration of the sludge than in the soil (Smith, 1996). This problem is one of the set of indications regulated in Spain by the R.D. 1310/1990 (BOE, 1990) and by the European Directive 86/278/EEC (UE, 1986). Thus, the agronomic rate that can be safely applied depends both on the heavy metal concentrations in the sludge and on the nitrogen concentration and the proportion of the nitrogen that can readily be mineralised within the first year after application to the soil (Barry et al., 1986; EPA, 1994; Smith, 1996). A sewage sludge application rate exceeding the crop needs could result in nitrate contamination of the ground water by leaching. The sewage sludge should therefore be applied as close to the time of maximum nutrient uptake by crops as feasible (EPA, 1994).

The effects of the application of sewage sludge in silvopastoral systems in northwest Spain established under *Pinus radiata* D. Don (López-Díaz et al., 2009; Rigueiro-Rodríguez et al., 2010a; Mosquera-Losada et al., 2010a), *Populus* x *canadensis* Moench (Mosquera-Losada et al., 2010c) and *Fraxinus excelsior* L. (Rigueiro-Rodríguez et al., 2010b) have been already studied. The objective of the present study is to evaluate the effects of different doses of sewage sludge on changes in soil chemical properties, tree growth, understory production and biodiversity as compared to the treatment of no fertilisation in a silvopastoral system under *Quercus rubra* L.

5.3. MATERIALS AND METHODS

5.3.1. Characteristics of the study site

The experiment was established in autumn 2001 in A Pastoriza (Lugo, Galicia, NW Spain, European Atlantic Biogeographic Region) at an altitude of 480 m above sea level. Figure 5.1 shows the mean monthly precipitation and temperatures for 2002 - 2005 as well as the previous 30 years. The total annual rainfall was 1296 mm, 1111 mm 822.9 mm and 824.3 mm in 2002, 2003, 2004 and 2005, respectively. In general, very low precipitation was observed in these years as compared with the mean for the last 30 years, and this limited pasture production and tree growth. However, in 2002 and 2003, there was a period of high precipitation from October to December. October 2004 and 2005 were also very rainy months (212.9 mm and 146.8 mm, respectively). The annual mean temperature was mild (12 ºC) with low temperatures at the beginning and at the end of the years under study.

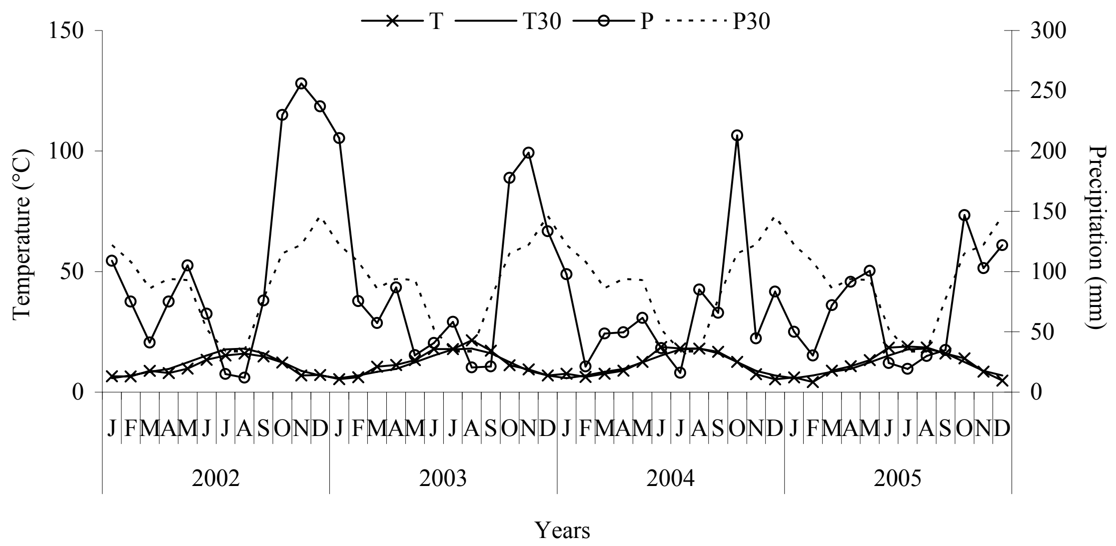

Figure 5.1: Monthly precipitation and mean temperatures for the study area in 2002, 2003, 2004 and 2005 and mean data for the last 30 years. T: mean monthly temperature (ºC), T30: mean temperature over the last 30 years (ºC), P: mean monthly precipitation (mm) and P30: mean precipitation over the last 30 years (mm).

The experiment was carried out on abandoned agricultural land. The soil texture at the beginning of the experiment was sandy (91.81% sand, 4.92% silt and 3.27% clay) with a moderately acidic pH of 5.2 as well as high levels of soil organic matter content, total N and total P (10.34%, 0.25% and 0.09%, respectively). All heavy metal concentrations in the soil (Table 5.1) were below the maximum threshold for using sewage sludge fertiliser as specified by the European Union Directive 86/278/CEE (UE, 1986) and Spanish legislation under R.D. 1310/1990 (BOE, 1990).

	Heavy metal concentrations (mg kg^{-1})					
Soil	Cd	Cu	Cr	Ni	Pb	Zn
Initial soil concentration	-	11.6	-	2.1	37.6	11.9
Spanish law limits	1-3	50-210	100-150	30-112	50-300	150-450

Table 5.1: Heavy metal concentrations in the soil at the beginning of the experiment and the legal limits established by European Directive 86/278 and Spain R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH<7; maximum: soil pH>7). A dash (-) signifies an element concentration below the detection limit of the technique used for its determination.

5.3.2. Experimental design

At the beginning of the experiment in autumn 2001, the soil was ploughed, and the pasture was sown with a mixture of *Dactylis glomerata* L. var. Artabro (12.5 kg ha^{-1}), *Lolium perenne* L. var. Brigantia (12.5 kg ha^{-1}) and *Trifolium repens* L. var. Huia (4 kg ha^{-1}). Naked-root *Quercus rubra* L. plants were planted at a density of 1112 trees ha^{-1}, with 3 m x 3 m between rows. The experimental design was a randomised block with three replicas and four treatments distributed in experimental units of 144 m^2 with 25 trees arranged in a frame of 5 x 5 trees. The treatments consisted of (a) no fertilisation (0N), (b) fertilisation with anaerobically digested sludge with an input of 100 kg total N ha^{-1} in March 2002 and 2003 (100N). (c) fertilisation with anaerobically digested sludge with an input of 200 kg total N ha^{-1} in March 2002 and 2003 (200N) and (d) fertilisation with anaerobically digested sludge with an input of 400 kg total N ha^{-1} in March 2002 and 2003 (400N).

5.3.3. Sewage sludge

Anaerobically digested sludge from a municipal waste treatment plant of Lugo was used. The required amount of sludge was calculated based on the percentage of total N and dry matter content (EPA, 1994), keeping in mind that approximately 25% of the total N from anaerobically digested sewage sludge is available in the first year after application. EU Directive 86/278/CEE (UE, 1986) and Spanish regulation R.D. 1310/1990 (BOE, 1990) regarding heavy metal concentrations in the application of sewage sludge to soil were also considered. The composition of the sewage sludge applied in 2002 and 2003 is summarised in Table 5.2.

Parameters	Values		
	Anaerobic sludge (2002)	Anaerobic sludge (2003)	Spanish law limits
Dry matter, %	22.27	23.82	
pH	7.08	7.49	
N, g kg^{-1}	29.7	25.4	
P, g kg^{-1}	19.9	19.6	
K, g kg^{-1}	3	3.9	
Ca, g kg^{-1}	3	3.2	
Mg, g kg^{-1}	6.2	6.8	
Na, g kg^{-1}	0.7	1.2	
Fe, g kg^{-1}	26	13.9	
Cr, mg kg^{-1}	87.9	81.4	1000-1500
Cu, mg kg^{-1}	242.8	195.8	1000-1750
Ni, mg kg^{-1}	98.7	142.8	300-400
Zn, mg kg^{-1}	364.7	748	2500-4000
Cd, mg kg^{-1}	4.4	10.9	20-40
Pb, mg kg^{-1}	130.6	78.7	750-1200
Mn, mg kg^{-1}	311.7	358.8	

Table 5.2: Chemical properties of the sewage sludge and legal limits established by European Directive 86/278 and Spain R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH<7; maximum: soil pH>7)

5.3.4. Field samplings and laboratory determinations

Soil samples were collected at a depth of 25 cm in March 2003 and in January 2004, 2005 and 2006, as described in R.D. 1310/1990 (BOE, 1990). In the laboratory, soil pH was determined in water (1:2.5) (Faithfull, 2002). Al concentrations in the exchange complex and the exchangeable cations (K, Ca, Mg and Na) were determined by extraction with 0.6 N $BaCl_2$ (Mosquera and Mombiela, 1986). The K, Ca, Mg and Na exchangeable concentrations were measured with a VARIAN 220FS Spectrophotometer using the atomic emissions for K and Na and the absorptions for Ca and Mg. Al concentrations were analysed after valoration with 0.01 N NaOH using phenolphthaleine (1%) in an alcohol-based solution as an indicator (Mosquera and Mombiela, 1986). The effective exchange capacity (EEC) was determined by taking the sum of K + Ca + Mg + Na + Al and the saturation percentage of Al, K, Ca, Mg and Na using the quotients Al/EEC, K/EEC, Ca/EEC Mg/EEC and Na/EEC, respectively (Mosquera and Mombiela, 1986).

Tree height and base diameter were measured with a graduated ruler and a calliper, respectively, in September 2006.

Estimated pasture production and botanical composition was determined by randomly collecting four samples of pasture. The four samples were cut with an electric hand clipper at a height of 2.5 cm per plot (0.3 x 0.3 m^2) in June 2002, July and December 2003, June, July and December 2004 and May, July and December 2005. One week after sampling, all plots were grazed by mature sheep (Galician breed) at a stocking rate of 50 sheep over the whole experimental area (1728 m^2). At the laboratory, two pasture samples were dried for 72 hours at 60ºC and weighed to estimate pasture production. The other two samples were separated by hand to determine the proportions of the different plant species and the senescent material. These two samples were then dried for 72 hours at 60 °C to determine the botanical composition on a dry weight basis. The species richness was determined yearly. Annual abundance diagrams omitting the percentage of senescent material (Magurran, 1988) were completed.

5.3.5. Statistical analysis

The data on tree and pasture variables were analysed using ANOVA (proc glm procedure) following the model $Y_{ij} = \mu + T_i + B_j + \varepsilon_{ij}$, where Y_{ij} is the studied variable, μ is the variable mean, T_i indicates treatment i, B_j is the block j, and ε_{ij} is the error. Soil variables and species richness were analysed with repeated ANOVA (proc glm procedure) following the model $Y_{ijk} = \mu + A_i + T_j + B_k + \varepsilon_{ijk}$, where Y_{ijk} is the studied variable, μ is the variable mean, A_i is the year i, T_j indicates treatment j, B_k is the block k, and ε_{ijk} is the error. The Tukey's HSD test was used for subsequent pairwise comparisons ($P < 0.05$; a = 0.05). The statistical software package SAS (2001) was used for all analyses.

5.4. RESULTS

5.4.1. Soil characteristics

Table 5.3 shows that the soil pH, EEC and Ca saturation percentage in the soil exchange complex were significantly higher in 2006 than at the beginning of the study (2003) ($p<0.001$). However, the saturation percentage of Al and K in the soil exchange complex decreased significantly in 2006 as compared with 2003 ($p<0.001$).

Furthermore, the Na saturation percentage in the soil exchange complex was higher in 2004 and 2006 than in 2003 and 2005 (p<0.001).

Parameter	Year				Year effect	SEM
	2003	2004	2005	2006		
pH	5.4 bc	5.27 c	5.51 b	5.72 a	***	0.04
EEC (cmol(+) kg^{-1})	4.94 c	5.16 c	6.30 b	7.84 a	***	0.21
Al (%)	39.4 a	37.1 a	41.11 a	28.65 b	***	0.01
K (%)	1.36 a	1.36 a	1.15 b	0.75 c	***	0.0004
Ca (%)	41.37 b	41.25 b	39.61 b	52.14 a	***	0.01
Mg (%)	13.00	13.37	12.95	11.28	ns	0.002
Na (%)	4.87 b	6.92 a	5.18 b	7.18 a	***	0.002

Table 5.3: Soil pH, effective exchange capacity (EEC) (cmol(+) kg^{-1}) and saturation percentage of Al, K, Ca, Mg and Na in the soil exchange complex (%) in the years 2003, 2004, 2005 and 2006. Different letters indicate significant differences between years. ns: not-significant, ***: $p<0.001$, SEM: mean standard error.

In 2003, soil EEC and Ca saturation percentage in the soil exchange complex were affected significantly by the applied fertilisation treatments ($p<0.05$) (Figure 5.2). However, in 2004, 2005 and 2006 the effect of the treatments on EEC, Ca saturation percentage, soil pH and saturation percentages of Al, K, Mg and Na was not statistically significant. The application of sewage sludge at a rate of 200 kg total N ha^{-1} (200N) increased soil EEC more than when no fertilisation (0N) was applied. Furthermore, the Ca saturation percentage in the soil exchange complex also increased when sewage sludge inputs had been previously done. The increase in the Ca saturation percentage was larger when low (100N) and medium dose (200N) of sewage sludge were applied than when no fertiliser treatments (0N) were used.

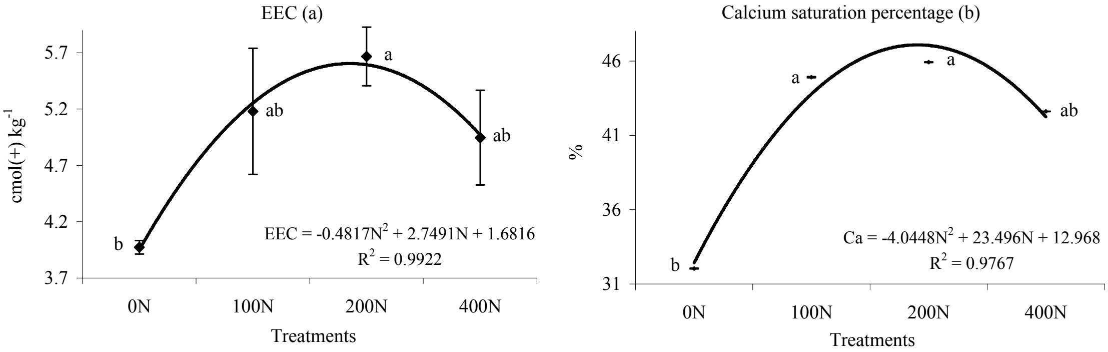

Figure 5.2: Effective exchange capacity (EEC) (cmol (+) kg^{-1}) (a) and calcium saturation percentage in soil exchange complex (%) (b) under each treatment in 2003. 0N: 0 kg total N ha^{-1}; 100N: 100 kg total N ha^{-1}; 200N: 200 kg total N ha^{-1} and 400N: 400 kg total N ha^{-1}. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

5.4.2. Tree heights and diameters

Figure 5.3 shows the tree heights and diameters for each treatment in 2006. During this year, tree heights and diameters were significantly affected by the fertilisation treatment applied (p<0.001). The application of a medium (200N) and a high dose (400N) of sewage sludge increased more the tree heights than the no fertilisation (0N). The tree diameters were larger under the medium (200N) doses of sewage sludge as compared with no fertilisation (0N).

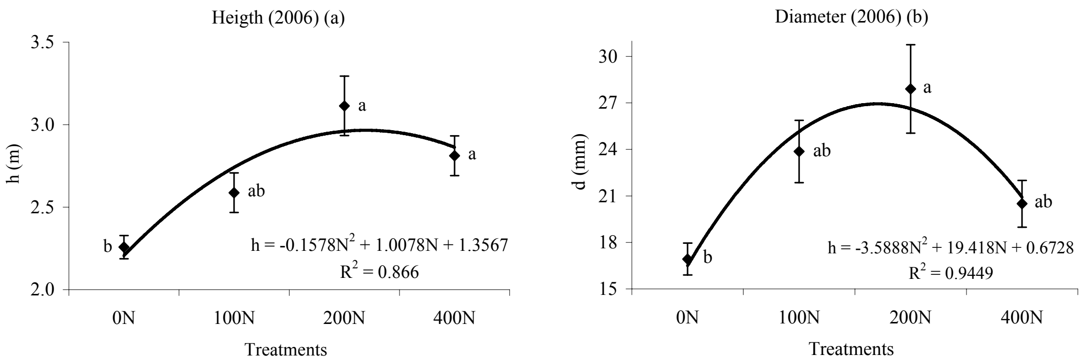

Figure 5.3: Tree heights (m) (a) and tree diameters (mm) (b) under each treatment in 2006. 0N: 0 kg total N ha^{-1}; 100N: 100 kg total N ha^{-1}; 200N: 200 kg total N ha^{-1} and 400N: 400 kg total N ha^{-1}. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

5.4.3. Pasture

5.4.3.1. *Production*

Annual pasture production for the different fertilisation treatments in 2002, 2003, 2004 and 2005 can be observed in Figure 5.4. Pasture production was significantly affected by the dose of sewage sludge in 2003 (p<0.05) and 2004

(p<0.001). In 2002 and 2005, no significant differences were detected between the treatments. The highest levels of pasture production were found in 2004 (9.3-17.5 Mg ha^{-1}), while the lowest values were reported in 2002 (3.9-4.3 Mg ha^{-1}). In 2003 and 2004, the application of medium (200N) and high (400N) sewage sludge treatments increased pasture production more than the no fertilisation (0N) and low fertiliser (100N).

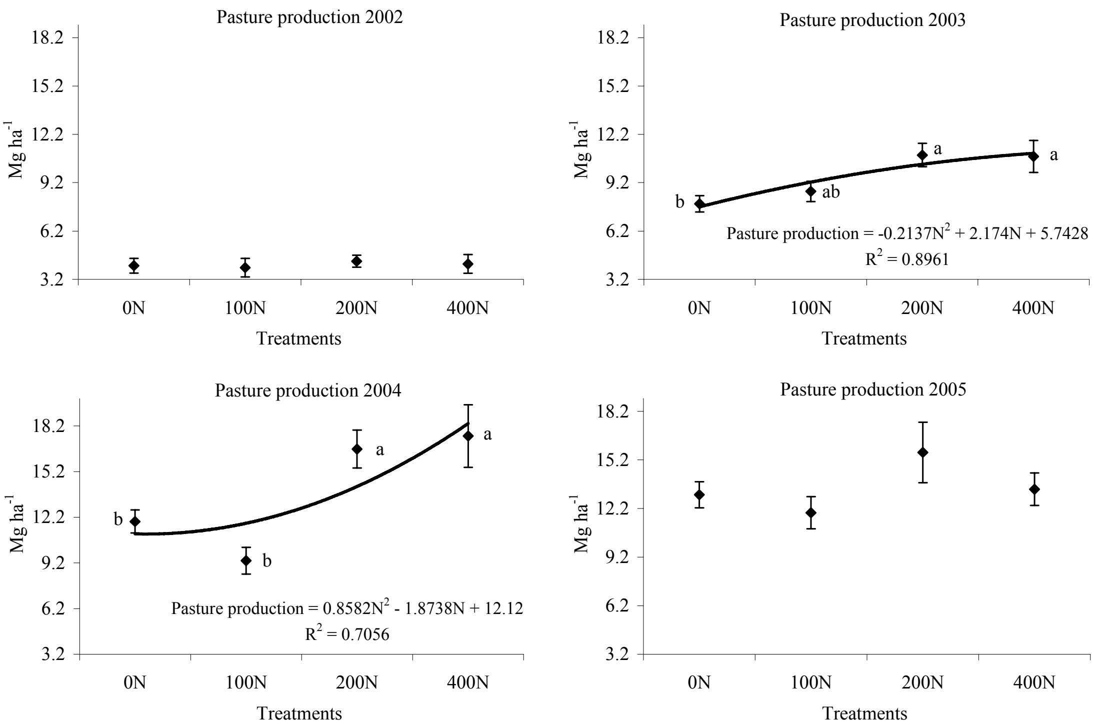

Figure 5.4: Pasture production (Mg ha^{-1}) under each treatment in 2002, 2003, 2004 and 2005. 0N: 0 kg total N ha^{-1}; 100N: 100 kg total N ha^{-1}; 200N: 200 kg total N ha^{-1} and 400N: 400 kg total N ha^{-1}. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error. Only significant regressions were shown.

Cumulative pasture production during the experimental period confirmed an increased pasture production at the medium (200N) and high (400N) sewage sludge treatments in 2004 and 2005 (p<0.001) (Figure 5.5).

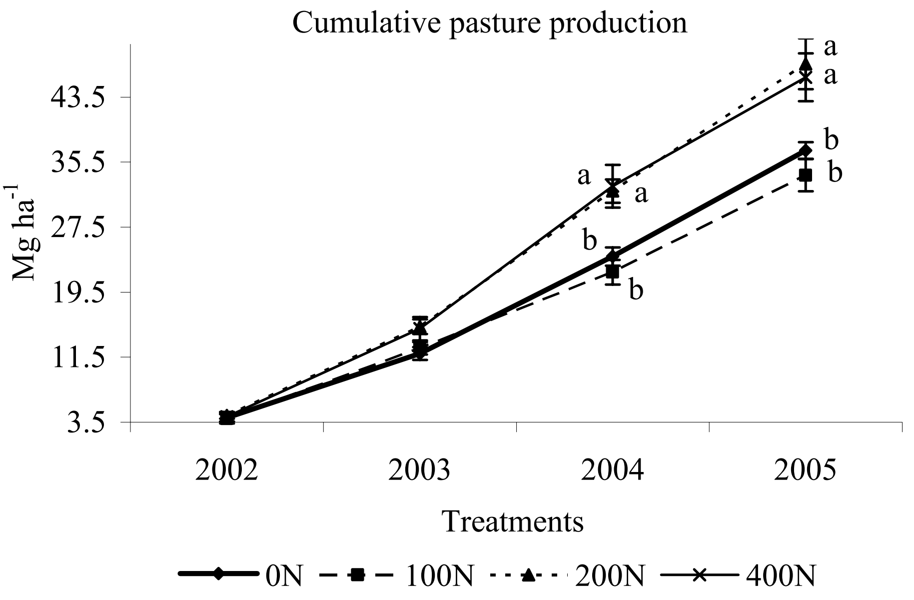

Figure 5.5: Cumulative pasture production under each treatment for the period 2002- 2005 (Mg ha^{-1}). 0N, 0 kg total N ha^{-1}; 100N: 100 kg total N ha^{-1}; 200N: 200 kg total N ha^{-1} and 400N: 400 kg total N ha^{-1}. Vertical lines indicate mean standard error.

5.4.3.2. Species richness

Figure 5.6 shows that species richness was significantly higher at the end of the experiment in 2004 and 2005 than in 2002 and 2003 (p<0.001). In 2002, species richness was significantly affected by the fertilisation treatments (p<0.05). In this year, the low (100N) sewage sludge treatment increased species richness more than the medium fertiliser (200N) treatment. The fertilisation treatments applied also significantly effected (p<0.05) the average species richness in the years 2002, 2003, 2004 and 2005 (0N: 6^{b}; 100N: 8^{a}; 200N: 6^{b}; 400N: 7^{b}) (different superscript letters indicate significant differences between treatments). The average species richness was higher under the low fertiliser (100N) sewage sludge treatment compared with the other treatments (0N, 200N and 400N).

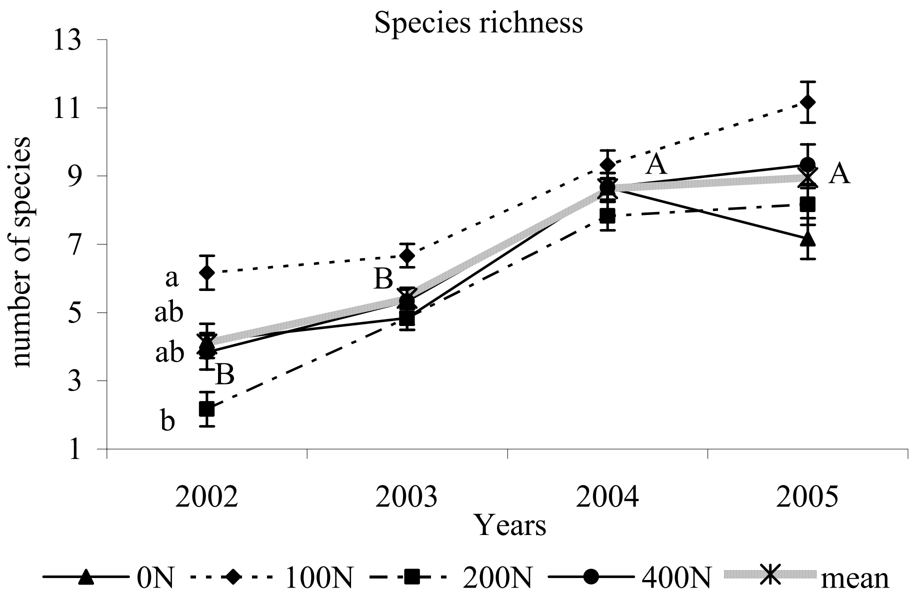

Figure 5.6: Annual evolution of species richness under each treatment in 2002, 2003 2004 and 2005. 0N: 0 kg total N ha^{-1}; 100N: 100 kg total N ha^{-1}; 200N: 200 kg total N ha^{-1} and 400N: 400 kg total N ha^{-1}. Different capital letters indicate significant differences between years, and different lowercase letters indicate significant differences between treatments in each year. Vertical lines indicate mean standard error.

5.4.3.3. Pasture abundance diagrams

Pasture abundance diagrams for the different fertilisation treatments in 2002, 2003, 2004 and 2005 are shown in Figure 5.7. From Figure 5.6, the abundance diagrams confirm that the number of species was higher at the end of the study (2005) than at the beginning (2002). In 2002, the number of dicotyledonous species was higher than the number of monocotyledonous species in all treatments. However, in subsequent years (i.e., 2003, 2004 and 2005), the proportion of dicotyledonous species was higher than the proportion of monocotyledonous species in the no fertiliser (0N) sewage sludge and the low fertiliser (100N) sewage sludge treatments. In the other treatments (200N and 400N), there were more monocotyledonous than dicotyledonous species. *Agrostis capillaris* L. and *Holcus lanatus* L. were present in all treatments and all years. They were also the most dominant species throughout the study. *Agrostis capillaris* L. was the dominant species in the no fertilisation treatment (0N), and in the low (100N) and medium (200N) sewage sludge treatments. However, the proportion of *Holcus lanatus* L. was higher when the high (400N) sewage sludge treatment was applied as compared to the other treatments (0N, 100N and 200N). The presence of sown species (*Dactylis glomerata* L., *Lolium perenne* L. and *Trifolium repens* L.) was low.

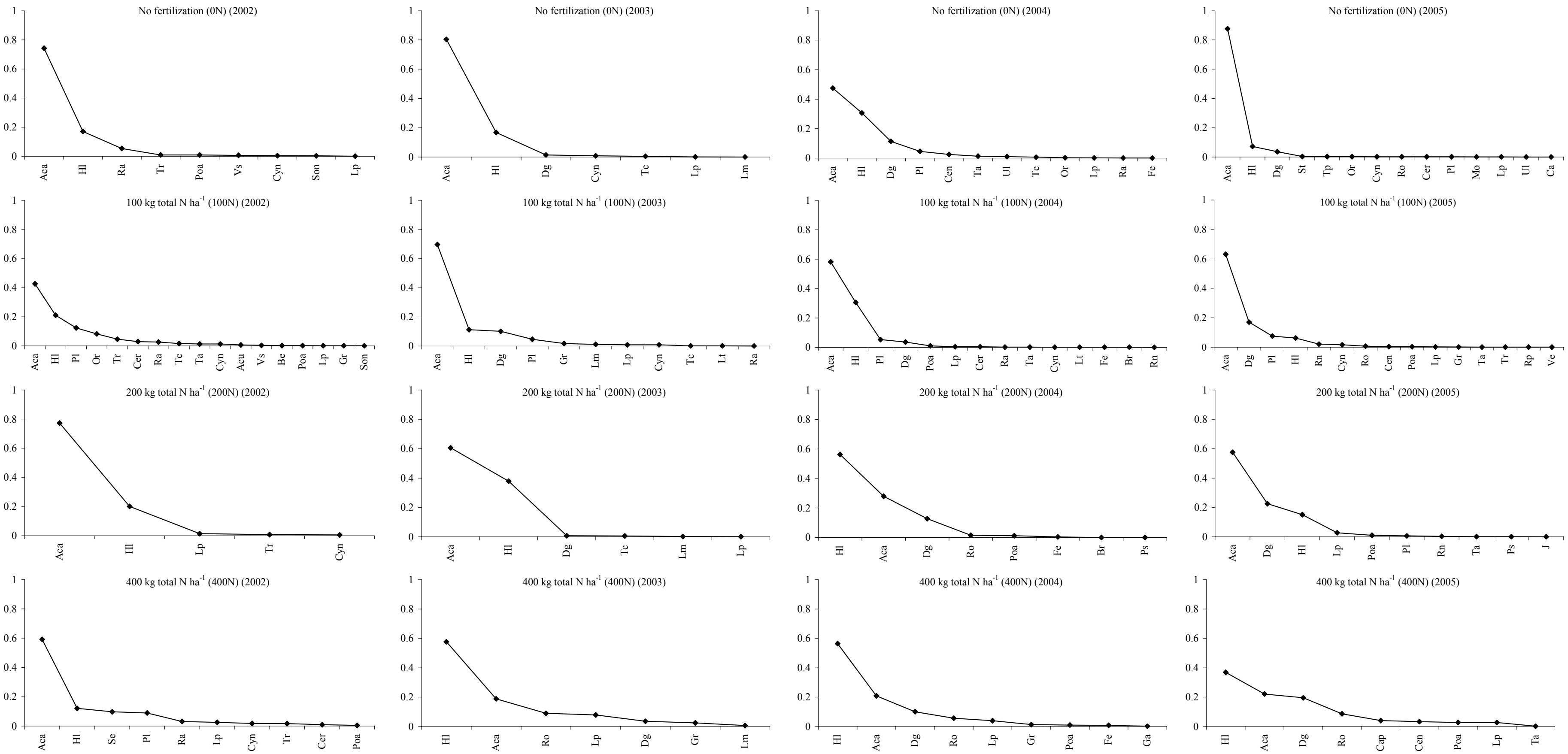

Figure 5.7: Species abundance diagrams for the treatments applied in 2002, 2003, 2004 and 2005. Aca: *Agrostis capillaris* L., Acu: *Agrostis curtisii* Kerguélen, Be: *Bellis perennis* L., Br: *Bromus mollis* L., Ca: *Carduus* sp, Cap: *Capsella bursa-pastoris* L., Cen: *Centaurea limbata* Hoffmanns. / Link, Cer: *Cerastium glomeratum* Thuill, Cyn: *Cynosurus cristatus* L., Dg: *Dactylis glomerata* L., Fe: *Festuca rubra* L., Ga: *Galium* sp, Gr: *Geranium rotundifolium* L., Hl: *Holcus lanatus* L., J: *Juncus effusus* L., Lm: *Lolium multiflorum* Lam., Lp: *Lolium perenne* L., Lt: *Lotus corniculatus* L., Mo: *Molinia caerulea* (L.) Moench, Or: *Ornithopus compressus* L., Pl: *Plantago lanceolata* L., Poa: *Poa pratensis* L., Ps: *Pseudarrhenatherum longifolium* (Thore) Rouy, Ra: *Rumex acetosa* L., Rn: *Ranunculus repens* L., Ro: *Rumex obtusifolius* L., Rp: *Raphanus raphanistrum* L., Se: *Senecio jacobaea* L., Son: *Sonchus oleraceus* L., St: *Stellaria media* L. (Vill), Ta: *Taraxacum officinale* Weber, Tc: *Trifolium campestre* Schreber, Tp: *Trifolium pratense* L., Tr: *Trifolium repens* L., Ul: *Ulex europaeus* L., Ve: *Veronica agrestis* L. and Vs: *Vicia sativa* L.

5.5. DISCUSSION

The use of sewage sludge as a fertiliser is usually described as beneficial for plant growth due its contribution to the improvement of soil properties (Smith, 1996; Egiarte et al., 2009; Mosquera-Losada 2006, 2010c; Rigueiro-Rodríguez et al, 2010a, 2010b). In this study, we found an initial amelioration of different soil properties such as EEC and Ca saturation percentage. The changes in Ca saturation percentage can be explained by the Ca inputs from the sewage sludge application (100N applied 57.5 kg CO_3Ca ha^{-1}, 200N added 115 kg CO_3Ca ha^{-1} and 400N applied 229.95 kg CO_3Ca ha^{-1}). However, these Ca contributions to the soil were not enough to modify other soil properties such as soil pH. López-Díaz et al. (2007) used a silvopastoral system established with *Pinus radiata* D. Don to show that soil fertility increased as the pH and the Ca saturation percentage increased due to the input of Ca from sewage sludge. The change in soil pH observed by López-Díaz et al. (2007) may be due to the high dose of sewage sludge used in comparison to our study. Furthermore, the sewage sludge was applied for three consecutive years, which implies a higher total Ca input into the soil as compared to our study (230 kg CO_3Ca ha^{-1}, 461.4 kg CO_3Ca ha^{-1} and 692.4 kg CO_3Ca ha^{-1}) in the López-Díaz et al. (2007) experiment. The effects of additional Ca on soil improvement under favourable weather conditions are evident in the tree and pasture production results.

In 2006, the ranges of *Quercus rubra* L. heights and diameters were 2.25-3.11 m and 16.92-27.90 mm, respectively. These values were greater than the tree heights (1.38-1.99 m) and diameters (4-10 mm) reported by Kormanik et al. (2005) in a study carried out in North Carolina after four years of experimentation on the same species. The largest tree heights were recorded at the medium (200N) and high (400N) sewage sludge treatments rather than the no fertilisation (0N) treatment. These results can be explained by the fact that these treatments resulted in high-nutrient inputs to the soil. Furthermore, the amount of N applied by these treatments (200N and 400N) may be too high for the pasture to take up but are nevertheless useful for the trees because the nitrate is leached to deeper soil horizons where tree roots are developed. Similar results were also reported by López-Díaz et al. (2009) and Rigueiro-Rodríguez et al. (2010a) regarding silvopastoral systems established under *Pinus radiata* D. Don in agrarian soils in Galicia. A reduction in soil nutrient leaching by root uptake in silvopastoral systems was also reported by Nair et al. (2007), Rigueiro-Rodríguez et al. (2008a) and Mosquera-Losada et al. (2010a) as one of the main advantages of silvopastoral as compared with treeless pasture. Although the high doses (400N) of sewage sludge improved tree heights, it had no evident effect on tree diameters. This is likely due to the higher proportion of *Holcus lanatus* L. in this treatment as compared to the other treatments (0N, 100N and 200N) in which *Agrostis capillaris* L. was the dominant species in the pasture. *Holcus lanatus* L. is a species associated with more fertile soils than *Agrostis capillaris* L. (Grime et al., 2007) and can capture more soil N, which typically limits the diameter growth of trees. In 2008, Laliberté et al. (2008) performed a study by directly seeding *Quercus rubra* L. on recently abandoned pastureland in Canada to find that herbaceous vegetation had no early effect on tree height. However, it slightly decreased diameter, which could have long-term biological significance. In the present experiment it was shown that diameter development depends on the type of dominant herbaceous species growing in the silvopastoral system.

Similar studies carried out in abandoned agricultural soils in Galicia reported that the average production of the pasture was between 6 and 12 Mg ha^{-1} (Mosquera-Losada et al., 1999). In this study, this production range was surpassed in the last two years of the experiment in 2004 and 2005. However, annual pasture production was less than this reported range in 2002 and was similar to this reported range in 2003. The highest levels of annual pasture production in 2004 can be explained by the lack of drought during the summer months of this year, which increased the length of the growing season. This result was also observed by Rigueiro-Rodríguez et al. (2010a) and Mosquera-Losada et al. (2010c) in silvopastoral systems established in the same area under *Pinus radiata* D. Don and *Populus* x *canadensis* Moench, respectively. Moreover, in 2002, the mean monthly temperature was lower than the mean temperature over the previous 30 years, which could have limited pasture production and prevented a response to treatments in terms of pasture production. In 2003 and 2004, the medium (200N) and high (400N) sewage sludge treatments showed higher pasture production than the other treatments. This could be due to a residual effect (EPA, 1994; Smith, 1996) from N mineralisation after sewage sludge application for two consecutive years. Mosquera-Losada et al. (2010c) and Rigueiro-Rodríguez et al. (2010a) also found high pasture production when the dose of the sewage sludge was increased. In 2005, significant differences were not detected between treatments because there was no further residual effect of sewage sludge application.

Even though a clear initial effect on soil fertility was found, pasture and tree development also affected soil characteristics and subsequently impacted species richness over time. In this study, a change in soil pH was observed in 2006 with respect to the beginning of the experiment (2003), but this was not a result of sewage sludge inputs. The change in soil pH can be attributed to the incorporation of pasture litter, including leaves and roots from *Quercus rubra* L., into the sandy soil. The trees were able to take up cations (Ca, K, Mg) that had been previously moved downward in the soil with percolating water and incorporate these cations again into the silvopastoral system soil after leaves fell. Soil improvement following the establishment of a tree plantation as compared to leaving open swards has been previously described by Mosquera-Losada et al. (2010c). Soil EEC in our study was low and below 10 cmol (+) kg^{-1} soil, which can be explained by the high sand fraction in the soil at the experimental site (Brady and Weil, 2008). At the beginning of the study (2003), a change in soil EEC was observed when sewage sludge was applied (Smith, 1996). However, when sewage sludge application ceased, the effect of the treatments on EEC disappeared. This is likely due to the sandy soil in the experimental area, which would allow the cations to be easily leached through the soil profile. Moreover, it was also observed that EEC and thus most available cations were higher at the end of the experiment (2006) than in previous years; however, there were no differences between fertilization dose treatments. The change in EEC can be explained by the establishment of the pasture and trees, which increased the organic matter input into the soil (Nieder et al., 2003); a similar phenomenon was described for *Pinus radiata* D. Don and *Betula alba* L. by Fernández-Núñez et al. (2010a). In other silvopastoral systems established in the same area by Mosquera-Losada et al. (2010c) and Rigueiro-Rodríguez et al. (2010b) using *Populus* x *canadensis* Moench and *Fraxinus excelsior* L., respectively, similar results for pH and EEC were observed. It is important to highlight the increased Ca saturation percentage in the exchange complex and the reduction in the Al saturation percentage in 2006 as compared to 2003, 2004 and 2005 (Smith, 1996). However, the K saturation percentage in the exchange complex was lower at the end of the experiment in 2006 than in the other years of the study, and the Mg saturation percentage followed the same trend. A strong antagonism between Mg and Ca (O'Riordan et al., 1987; Vivekanandan et al., 1991; Rigueiro-Rodríguez et al., 2006) and between K and Ca in the soil has been described, with both Mg and K being preferentially leached (Barber, 1995; Rigueiro-Rodríguez et al., 2006), which could explain these results. Moreover, reduction in K at the end of the experiment in 2006 and the lack of differences between treatments may also be attributed to low K in the sewage sludge (Mosquera-Losada et al., 2010b).

Regarding species richness, it was observed that the number of species increased in 2004 and 2005 as compared with 2002 and 2003. These results can be explained by the change in soil pH because the less restrictive edaphic conditions lead to an increase in the number of species (Mosquera-Losada et al., 2009a). Increasing soil pH and low fertilizer inputs increased species richness mainly due to the increase of dicotyledonous (better developed in cation rich environments) and the reduction of monocotyledonous species, which increases their competitive advantage under more nitrogen rich environment (Grime et al., 2007). Moreover, the silvopastoral system was established under *Quercus rubra* L., which is a deciduous species and allows better light penetration than conifers. Species richness should be higher in this type of silvopastoral system as compared to those established under exotic conifer, such as *Pinus radiata* D. Don, in Galicia once tree canopy closure happens (Mosquera-Losada et al., 2006; Fernández-Núñez et al., 2010b). The low (100N) sewage sludge treatment resulted in higher species richness than the medium (200N) and high (400N) sewage sludge treatments. As reported by Thompson et al. (2001), the increase in fertilisation reduces the invasion of weeds and the number of species. In contrast, *Agrostis capillaris* L. was the most abundant species in the study, probably due to the grazing sheep but mostly when low doses of sewage sludge were applied. *Agrostis capillaris* L. grows vegetatively from stolons (Grime et al., 2007), which spread due to damage during sheep grazing. This result was also found by Krahulec et al. (2001) in a study that took place in the Krkonose Mountains (Czech Republic). However, *Holcus lanatus* L. appeared under the high (400N) sewage sludge treatment because this species is associated with more fertile soils than *Agrostis capillaris* L. (Grime et al., 2007). In this study, a reduction in tree diameter due to the higher uptake of N of *Holcus lanatus* L. as compared to *Agrostis capillaris* L was observed. In the present study, the presence of sown species (*Dactylis glomerata* L., *Lolium perenne* L. and *Trifolium repens* L.) was low because *Dactylis glomerata* L. was not well established due to the drought and grazing sheep. Furthermore, *Lolium perenne* L. and *Trifolium repens* L. require more fertile soils than that found in this study (Rigueiro-Rodríguez et al., 2000; Grime et al., 2007).

5.6. CONCLUSION

While the effect of sewage sludge on pasture production depended on the weather conditions of a specific year, tree growth and pasture production increased when 200 and 400 kg of total N ha^{-1} were applied. Thus, it was advised application of approximately 200 kg of total N ha^{-1} to enhance both components in a silvopastoral system. Sewage sludge applications initially improved soil nutrient levels (EEC and Ca saturation percentage); however, the establishment of pasture and trees improved soil conditions and increased species diversity, showing the most diversity under 100 kg of total N ha^{-1} treatments.

5.7. ACKNOWLEDGMENTS

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PARTE VI

DISCUSIÓN GENERAL

6. DISCUSIÓN GENERAL

6.1. SUELO

En los sistemas silvopastorales establecidos en Galicia, la producción de pasto y el crecimiento del arbolado pueden verse limitados por la baja fertilidad del suelo (Barrecheguren et al., 1992; Mosquera-Losada et al., 1999). El uso del lodo de depuradora como fertilizante se considera normalmente beneficioso para el crecimiento de las plantas debido a la mejora de la fertilidad edáfica que causa su aplicación por lo que su uso se potencia y promueve por diferentes instituciones y autores (UE, 1986; EPA, 1994; Smith, 1996; Hillman et al., 2003). Los tres ensayos que forman parte de esta tesis son representativos de los suelos gallegos, en los que el pH es ácido (pH<6), lo que normalmente conduce a una reducida disponibilidad de cationes y nutrientes y por lo tanto limita la producción del pasto y/o arbolado que se desarrolla sobre ellos (Mosquera, 1992; Whitehead, 2000; Sánchez-Rodríguez et al., 2002; Nilsson, 2003). El pH es uno de los principales indicadores de la fertilidad de un suelo y puede modificarse por los aportes de calcio, bien sea a través de enmiendas calizas (Bailey, 1995; López-Mosquera et al., 1995; López-Díaz et al., 2007), de la quema (Rois et al., 2006; Rigueiro et al., 2010c), por la incorporación de material orgánico rico en este elemento, como son los lodos de depuradora urbana (Smith, 1996; Egiarte et al., 2005; López-Díaz et al., 2007) o la incorporación de hojas y otros residuos vegetales en el suelo (Porta et al, 2003; Moreno-Marcos y Obrador-Olán, 2007; Mosquera-Losada, 2010c). En nuestros estudios hemos encontrado que el pH se ha visto modificado por el aporte de lodo y por la incorporación de materia orgánica procedente del pasto y del arbolado en suelos de carácter arenoso, aunque la magnitud de la respuesta depende del tipo de suelo en el que se aplica el fango y del tipo de estabilización empleada. Así, en el estudio establecido con *Quercus rubra* L, se observó un mayor pH del suelo al final del estudio que al principio, probablemente debido a la incorporación al suelo de los restos del pasto y de las hojas y raíces de *Quercus rubra* L. y de la capacidad del arbolado de tipo caducifolio de consumir los cationes de capas más profundas del suelo en comparación con las superficiales exploradas por el pasto, tal y como ha sido descrito con *Quercus ilex* L. (Moreno-Marcos y Pulido, 2008). Este resultado también fue encontrado por Mosquera-Losada et al. (2010c) en sistemas silvopastorales establecidos en suelos arenosos con la caducifolia *Populus* x *canadensis* Moench. Sin embargo, en los ensayos establecidos con *Pinus radiata* D. Don y *Fraxinus excelsior* L. el establecimiento del pasto y el arbolado no supuso un incremento del pH del suelo con el paso del tiempo, ya que el pH del suelo disminuyó en el último año de estudio con respeto al primero. En estas condiciones, la disminución de pH ocasionada por el incremento en la proporción de H^+ en el sistema edáfico la atribuimos a que las extracciones por parte del pasto y el arbolado son mayores que los insumos realizados con el aporte del lodo y al propio proceso de mineralización del lodo y restos vegetales (NH_4^+ es transformado en NO_3^- y se libera H^+ en el proceso de nitrificación) (Whitehead, 1995). El efecto del lodo sobre la acidez edáfica depende del tipo de suelo en el que se aplica, lo que provoca diferencias en la respuesta en suelos agrícolas (con mayor pH) y forestales (con menor pH). Así, y tal y como sucede en nuestro estudio, la respuesta del pH al aporte de lodos en los suelos forestales con *Pinus radiata* D. Don resultó ser mayor cuando se fertilizó con lodo anaeróbico que cuando se aplicó mineral y se sembró pasto (López-Díaz et al., 2007), debido a la mayor aplicación al suelo de Ca y Mg y micronutrientes con el lodo en comparación con el abonado mineral (Smith, 1996; López-Díaz et al., 2007; Mosquera-Losada et al., 2010b). Este efecto positivo del lodo en comparación con el abonado mineral sobre el pH del suelo también se aprecia en suelos agrícolas de pH próximo a la neutralidad (Mosquera-Losada et al., 2006), en los que la disminución del pH causada por las extracciones del cultivo y las abundantes precipitaciones de Galicia fue menor cuando se fertilizó con lodo que cuando se aplicó el abono mineral. En suelos con pH intermedios no suele modificarse el pH como consecuencia de dosis crecientes de lodo; así, en el sistema silvopastoral con *Quercus rubra* L., no se observó que las diferentes dosis de lodo aplicadas (100 kg de N total ha^{-1}, 200 kg de N total ha^{-1} y 400 kg de N total ha^{-1}) modificaran significativamente el pH del suelo, debido probablemente a que la cantidad de Ca aplicada durante dos años consecutivos (100N aplicó 57,5 kg CO_3Ca ha^{-1}, 200N añadió 115 kg CO_3Ca ha^{-1} y 400N aplicó 229,95 kg CO_3Ca ha^{-1}) con este tipo de fertilización orgánica no fue suficiente como para cambiar el pH del suelo. La ausencia de respuesta del pH del suelo a la fertilización con lodos de lechería también fue observada por Rigueiro-Rodríguez et al. (2000) en sistemas silvopastorales establecidos en suelos agrícolas con *Pinus radiata* D. Don y con pH inicial de 6,8, debido a que al tratarse de suelos agrícolas ya existía una historia de mejora de encalado reciente y a que el lodo utilizado tenía un pH próximo al del suelo. Finalmente, también hemos encontrado que el tipo de lodo también afecta de forma diferenciada al pH para una misma dosis de nitrógeno. Así, en el estudio desarrollado con *Fraxinus excelsior* L., en el que se comparaba el aporte de 320 kg de N total ha^{-1} en forma de lodo compostado, anaeróbico y peletizado, se encontró que el lodo compostado supuso un incremento del pH del suelo en comparación con el lodo anaeróbico, el lodo peletizado y la ausencia de fertilización, debido a los aportes de Ca que se hacen al fertilizar con aquel tipo de lodo, más rico en este elemento que el anaeróbico o el peletizado (Mosquera-Losada et al., 2010b). El mayor efecto del lodo compostado se justifica porque éste tipo de residuo presenta una menor proporción de N en su composición que el lodo anaeróbico o el lodo peletizado y por ello es necesario aplicar dosis más altas de lodo compostado que de los otros dos tipos de lodo para poder alcanzar las mismas dosis de N total. Las mayores dosis de lodo empleadas con el lodo compostado, en comparación con el anaeróbico o peletizado, y la mayor concentración de Ca, K y Mg en el lodo compostado implicó que en el experimento desarrollado con *Fraxinus excelsior*, en suelos de pH intermedio, el lodo compostado aportara al suelo alrededor de 1835,8 kg Ca ha^{-1}, 99,53 kg K ha^{-1} y 541,9 kg Mg ha^{-1}, mientras que sólo 90,51 kg Ca ha^{-1}, 28,66 kg K ha^{-1} y 67,88 kg Mg ha^{-1} fueron añadidos con el lodo anaeróbico y 776.85 kg Ca ha^{-1}, 23.07 kg K ha^{-1} y 158,96 kg Mg ha^{-1} con el lodo peletizado, lo que justifica el mayor incremento del pH edáfico obtenido con el lodo compostado.

La capacidad de intercambio catiónico efectiva (CIC) del suelo es otra característica importante al abordar su fertilidad. Los suelos gallegos se caracterizan por poseer unas CIC reducidas debido a su elevada acidez natural y al importante lavado de los cationes. En los tres ensayos de esta tesis, que incluyen zonas de monte y agrícolas, encontramos que la CIC fue baja, con valores casi siempre inferiores a 10 cmol (+) kg^{-1} de suelo, lo que puede explicarse por la elevada proporción de arena de los suelos (Brady y Weil, 2008). Por lo tanto, la fertilidad edáfica en estos ensayos es reducida al tratarse de suelos ácidos con baja CIC. Por otro lado, la mayor fertilidad de los suelos agrícolas con *Pinus radiata* D. Don en comparación con los suelos forestales con esta especie arbórea, vuelve a ponerse de manifiesto en la mayor CIC que presentaron los suelos agrícolas en comparación con los suelos forestales. En los sistemas silvopastorales establecidos con frondosas, los resultados obtenidos muestran un incremento de la CIC con el paso del tiempo, no siendo tan evidente este efecto en el caso de la conífera establecida tanto en suelos agrícolas como en los forestales. Este aumento de la CIC del suelo, desde el principio al final de los dos ensayos con frondosas, puede explicarse por la baja CIC inicial derivada del laboreo que se realizó en las fincas experimentales al comienzo de los estudios que rompió la estructura de los agregados del suelo (Dexter, 1988). Con el paso del tiempo, la incorporación de restos vegetales y la exploración del suelo por parte de raíces y animales invertebrados del suelo provocaron una mejora de su estructura. Además, la adición de fertilizantes, el establecimiento del pasto y el arbolado y por tanto el consiguiente aporte de materia orgánica al suelo favorecieron el incremento de la CIC bajo frondosas (Nieder et al., 2003), tal y como encontraron Fernández-Núñez et al. (2010a) en estudios con *Betula alba* L. y Mosquera-Losada et al. (2010c) en sistemas silvopastorales establecidos con *Populus* x *canadensis* Moench. El incremento de la CIC del suelo con el paso del tiempo también supuso un aumento de la disponibilidad de la mayoría de los cationes edáficos, siendo importante la mejora del porcentaje de saturación del Ca en el complejo de cambio en detrimento del porcentaje de saturación de Al (Smith, 1996). En concreto la aplicación de lodo compostado supuso un mayor aporte de Ca y Mg al suelo que la de lodo anaeróbico y lodo peletizado, cationes que reemplazan al Al en el complejo de cambio (Smith, 1996, Prasad and Power, 1997; Speir et al., 2004) y producen un incremento del contenido total y disponible del Ca en el mismo (Mosquera-Losada et al., 2001, 2009b; Rigueiro-Rodríguez et al., 2008b) desde el inicio del ensayo hasta el final del estudio. Por otra parte, en el sistema silvopastoral realizado con *Quercus rubra* L. se encontró que al mismo tiempo que se incrementaba el porcentaje de saturación del Ca en el complejo de cambio con el paso del tiempo se reducía el porcentaje de saturación del K y tendía a disminuir el de Mg debido al antagonismo existente entre el Mg y el Ca (O'Riordan et al., 1987; Vivekanandan et al., 1991; Rigueiro-Rodríguez et al., 2006) y entre el K y el Ca, que provocan una pérdida de esos cationes (K y Mg) por lavado (Barber, 1995; Rigueiro-Rodríguez et al., 2006).

El efecto de la dosis de lodo sobre los niveles de CIC fue en general positivo, tal y como obtuvieron López-Díaz (2004) y Rodríguez-Barreira (2007) en sistemas silvopastorales establecidos en suelo forestal con *Pinus radiata* D. Don y Mosquera-Losada et al. (2010c) en sistemas silvopastorales con *Populus* x *canadensis* Moench en suelo agrícola. Así, el incremento de la dosis de lodo anaeróbico aplicando 200 kg de N total ha^{-1} en el sistema silvopastoral establecido con *Quercus rubra* L. ocasionó un incremento de la CIC en comparación con la ausencia fertilización, probablemente debido a los aportes de Ca al suelo que supone la fertilización con lodo de depuradora (Smith, 1996). Sin embargo, no se encontró un efecto residual al año siguiente del cese del aporte de lodo, probablemente debido a que el experimento se estableció en un suelo arenoso en el que los cationes pueden ser lavados fácilmente. La evaluación del efecto de la dosis de lodo anaeróbico sobre los niveles de CIC en el caso de los suelos agrícolas desarbolados reveló una mayor CIC del suelo en comparación con la fertilización mineral, debido a la mejora de las propiedades físicas del suelo que supone la aplicación de lodos de depuradora (Smith, 1996) y a que la fertilización mineral sólo añade N, P y K al suelo, reduciéndose los niveles de otros cationes debido a las extracciones por parte del pasto y el arbolado (Whitehead et al., 1995). En los suelos forestales repoblados con *Pinus radiata* D. Don el reducido pH del suelo limitó la actividad biológica del mismo (Omil et al., 2007; Djukic et al., 2009) y por lo tanto no hubo diferencias significativas apreciables del efecto de los diferentes tratamientos (siembra de pasto y fertilización con mineral o con lodo) sobre la CIC del suelo. Finalmente, las dosis de lodo anaeróbico, compostado y peletizado aplicadas en el sistema silvopastoral establecido con *Fraxinus excelsior* L. fueron bajas, lo que impidió que se observara un efecto claro del aporte de los diferentes tipos de lodo sobre la CIC del suelo. Sin embargo, el efecto positivo sobre la CIC de la fertilización con grandes aportes de residuos orgánicos en suelos con menores contenidos en materia orgánica que los nuestros ha sido recogido por autores como Vivekanandan et al. (1991) y Smith (1996).

Cuando se aplican fertilizantes al suelo es importante hacer un seguimiento de la cantidad de N y P aportada, ya que sólo menos de la mitad del N o del P aplicado con los fertilizantes es usado por los cultivos, siendo el exceso de fertilizante, sobre todo nitrogenado, lavado a través del perfil del suelo, provocando la contaminación de las aguas y la eutrofización de las mismas (Cassman, 1999). La mayor fertilidad de los suelos agrícolas en comparación con los forestales, se detectó en el ensayo con *Pinus radiata* D. Don establecido en ambos tipos de suelos, ya que la concentración de N y P total fue mayor en los suelos agrícolas con *Pinus radiata* D. Don que en los suelos forestales. Además, los suelos agrícolas presentaron una mayor proporción de *Trifolium repens* L. en la composición del pasto, lo que pudo incrementar las entradas de N al suelo, ya que esta especie pratense, asociada a suelos fértiles y con bajos niveles en nitrógeno disponible, es capaz de fijar N atmosférico mediante el establecimiento de relaciones de simbiosis con bacterias del género *Rhizobium* (González, 1992; Whitehead, 1995; Green et al., 1999; López-Díaz et al., 2009). Autores como González (1992) han señalado que en pastos establecidos en el noroeste de Galicia un 30% de *Trifolium repens* L. en el pasto puede incorporar al suelo hasta 250 kg ha^{-1} $año^{-1}$ de N. Si nos centramos sólo en los suelos agrícolas con *Pinus radiata* D. Don se vio como en el último año de estudio la siembra de pasto, la plantación del arbolado y la fertilización con lodo de depuradora anaeróbico incrementó más el N y el P total del suelo que las mismas condiciones pero con fertilización mineral, debido a que los lodos de depuradora se caracterizan por una liberación más lenta de nutrientes que el abonado mineral (EPA, 1994; Smith, 1996) y por lo tanto su efecto se prolonga en el tiempo. En los suelos agrícolas desarbolados también se detectó una mayor cantidad de P disponible cuando se fertilizó con lodo de depuradora aneróbico y abono mineral, en comparación con la ausencia fertilización, tal y como ya habían obtenido previamente en sus estudios Allen et al. (2006) y Nair et al. (2007), lo que permite explicar la mayor producción de pasto en las parcelas fertilizadas con lodo que en las no fertilizadas. A pesar de estos resultados obtenidos en los suelos agrícolas, en los suelos forestales con *Pinus radiata* D. Don no hubo un efecto claro de los tratamientos aplicados (fertilización y siembra de pasto) sobre las siguientes variables edáficas: N y P total y P disponible por el método Mehlich, probablemente debido al bajo pH del suelo, que dificultó la actividad biológica y la incorporación de estos nutrientes al suelo (Omil et al., 2007; Djukic et al., 2009).

En relación al efecto de la dosis de lodo sobre la posible contaminación de las aguas ocasionada por el lavado de nitratos hay que señalar que encontramos que este lavado fue mayor al inicio del estudio, tanto en los suelos agrícolas con *Pinus radiata* D. Don como en los suelos forestales con esta especie arbórea, probablemente debido a la ausencia de vegetación al inicio del ensayo, lo que favoreció el lavado, ya que no había una cubierta vegetal que absorbiera este nutriente. La mayor tasa de mineralización encontrada en los terrenos agrícolas provocó que en ellos se superasen los niveles máximos permitidos por la Unión Europea para el consumo de agua (11, 3 mg NO^-_3 –N L^{-1}) (UE, 1980). Sin embargo, con el transcurso del tiempo el adecuado establecimiento del pasto y del arbolado disminuyó el lavado de nitratos a pesar de los insumos de fertilizante mineral realizados anualmente. En general, tras los tres primeros meses de establecimiento, la concentración de nitratos en el agua lixiviada fue mayor en los suelos agrícolas con *Pinus radiata* D. Don que en los suelos forestales con esta conífera (Knight et al., 1989), no encontrándose diferencias significativas entre tratamientos a partir de esta fecha. Este resultado pudo ser debido al mayor pH inicial de los suelos agrícolas, lo cual implicó una mayor tasa de mineralización y por lo tanto un mayor lavado de nitratos. Además, inicialmente los suelos forestales con *Pinus radiata* D. Don presentaron en el pasto una mayor proporción de especies gramíneas perennes que los suelos agrícolas en los que inicialmente predominaron especies anuales lo cual redujo el lavado de nitratos en los suelos forestales en comparación con los suelos agrícolas, ya que las gramíneas se caracterizan por su mayor capacidad de absorción de nitrato del suelo (Humphreys et al., 2006; Abberton et al., 2008). La presencia inicial de especies anuales dicotiledóneas, como *Chamomilla recutita* L., en el pasto de los suelos agrícolas con *Pinus radiata* D. Don también pudo incrementar el lavado de nitratos en este tipo de suelos, ya que las especies dicotiledóneas son más ricas en N que las especies monocotiledóneas y al morir implican un mayor aporte de N al suelo (Hanley et al., 1992; Paré et al., 2006).

Es importante tener en cuenta el efecto del lodo de depuradora sobre la concentración de Cu y Zn en el suelo y en las plantas, debido que éstos son los metales con mayores insumos cuando se fertiliza con lodos, ya que son los que están presentes en mayor concentración (Smith, 1996; Mosquera-Losada et al., 2010b). En el sistema silvopastoral establecido con *Fraxinus excelsior* L. todos los valores de Cu y Zn total en suelo resultaron ser inferiores a los valores máximos permitidos por la legislación española para el uso de lodo de depuradora como fertilizante en suelos ácidos (Cu: 50 mg kg^{-1} y Zn: 150 mg kg^{-1}) (R.D. 1310/1990) (BOE, 1990). En el ensayo con *Fraxinus excelsior* L. se observó una menor cantidad de Cu y Zn total en el suelo al final del estudio que al principio, debido al lavado de Cu y Zn a través del perfil edáfico y a las extracciones por parte del pasto y del arbolado. Tal y como se vio en el caso del Ca, la aplicación de lodo compostado también supuso una mayor concentración de Cu y Zn en el suelo, probablemente debido a las mayores aplicaciones de Cu y Zn al suelo (4,46 kg Cu ha^{-1} y 27,76 kg Zn ha^{-1}) en comparación con el lodo anaeróbico (3,59 kg Cu ha^{-1} y 26,43 kg Zn ha^{-1}) o el lodo peletizado (1,74 kg Cu ha^{-1} y 14,47 kg Zn ha^{-1}). Además, el lodo compostado incrementó más el pH del suelo que los otros tipos de lodo, lo cual normalmente implica una reducción de la solubilidad del Cu y Zn, debido al incremento de la capacidad de intercambio catiónico (Prasad and Power, 1997). En China, Miao-Miao et al. (2007) también encontraron que el uso del lodo compostado como fertilizante incrementaba la concentración de Cu y Zn en el suelo, pero en ese estudio la concentración de Cu y Zn en la composición del lodo empleado (Cu: 2316 mg kg^{-1} y Zn: 2971 mg kg^{-1}) era muy superior a la de nuestro caso, debido a que procedía de una zona más industrial. Un incremento de la concentración de Cu y Zn en el suelo como resultado del uso de lodo de depuradora como fertilizante también fue encontrado por Yuan (2009) y por Mosquera-Losada et al. (2010b).

6.2. ALTURA Y DIÁMETRO DEL ARBOLADO

En este estudio se ha observado que la fertilización con lodos de depuradora mejora el crecimiento del arbolado, ya que las tres especies forestales utilizadas para el establecimiento de los sistemas silvopastorales (*Pinus radiata* D. Don, *Fraxinus excelsior* L. y *Quercus rubra* L.) presentaron un crecimiento mayor, tanto en altura como en diámetro, cuando fueron fertilizadas con lodo en comparación a cuando no se fertilizó. Este resultado nos muestra, en primer lugar, los beneficios del abonado con lodos de depuradora en el crecimiento de las distintas especies forestales y, en segundo lugar, que sin fertilización el suelo no es capaz de poner a disposición del árbol los nutrientes que éste necesita para su desarrollo óptimo. Además, si se comparan los valores obtenidos de altura y diámetro de *Pinus radiata* D. Don, *Fraxinus excelsior* L. y *Quercus rubra* L. con los obtenidos por otros autores se vuelve a poner de manifiesto que la fertilización, tanto con lodo de depuradora como con mineral, favorece el crecimiento del arbolado, ya que en nuestro estudio los valores de altura y diámetro son superiores a los obtenidos en otros estudios para las mismas especies forestales a la misma edad. En el caso del *Pinus radiata* D. Don, se obtuvo una altura (164-193 cm) y un diámetro en la base (4,95-5,8 cm) mayores que los obtenidos por López-Díaz et al. (2009) (altura: 73,2-87.1 cm y diámetro: 1,69-2,03 cm) en un estudio llevado a cado en suelo agrícola con pH cercano al neutro (pH 6,3) en el que se fertilizó con lodos de depuradora urbana y por Sánchez-Rodríguez (2000) en un estudio también establecido en el noroeste de España. En cuanto al crecimiento del *Fraxinus excelsior* L., la altura (104-135 cm) y el diámetro (10,81-14,04 mm) se encontraron dentro del rango establecido por Mwase et al. (2008) (altura comprendida entre 23–275 cm y diámetro de 15,3 mm) en el Reino Unido a pesar de que en el caso de Mwase et al. (2008) las condiciones climáticas favorecían más el crecimiento del arbolado debido a que normalmente el clima en el Reino Unido se caracteriza por la ausencia de sequía en verano, lo que permite al arbolado crecer en esta estación, a diferencia de lo que ocurre en Galicia. *Quercus rubra* L. también presentó un crecimiento en altura y en diámetro de un 37,12% y 68,76%, respectivamente, mayor al que citan Kormanik et al. (2005) en un estudio establecido en el Norte de Carolina para la misma especie.

En general, la fertilización con lodos de depuradora mejoró el crecimiento del arbolado, pero este efecto fue diferente en función de la dosis empleada y del tipo de lodo utilizado (anaeróbico, compostado o peletizado). Las dosis medias (200 kg de N total ha^{-1}) y altas (400 kg de N total ha^{-1}) de lodo de depuradora implicaron un mayor crecimiento en altura del *Quercus rubra* L. que la ausencia de fertilización, debido a que a estas dosis de lodo se suministran grandes cantidades de N al suelo, que no es utilizado en su totalidad por el pasto y queda a disposición de los árboles, debido a que el nitrato se lava hacia las capas más profundas del suelo en las que se desarrollan las raíces del arbolado (López-Díaz et al., 2009; Rigueiro-Rodríguez et al., 2010a). En relación al tipo de lodo, inicialmente se observó un mayor crecimiento en diámetro de *Fraxinus excelsior* L. cuando se fertilizó con lodo anaeróbico que cuando no se fertilizó, este efecto positivo puede ser atribuido a la mejora de la fertilidad del suelo y al aumento de la capacidad de retención de agua del mismo como consecuencia de la fertilización con este residuo cuando se establece la plantación (Wolstenholme et al., 1992). Sin embargo, al final del estudio, sólo el lodo peletizado mostró un incremento significativo de la altura y el diámetro de los árboles, y el lodo compostado del diámetro, en comparación con la fertilización mineral, lo cual puede ser atribuido a la mayor producción de pasto en el tratamiento mineral, que hace que aumente la competencia entre el pasto y el arbolado, a la aplicación anual del lodo peletizado, que suministra nutrientes de forma continuada, y al incremento de la fertilidad edáfica que provocó el lodo compostado.

Por otro lado, hay que señalar que el crecimiento del arbolado, además de verse influenciado por la fertilización, también va a depender de las condiciones climáticas, tal y como le sucede a la producción de pasto, como se verá más adelante, y por el establecimiento de ciertas especies herbáceas adventicias en el pasto que ejercen competencia por los nutrientes limitando así el crecimiento del arbolado. El efecto del clima sobre el crecimiento de la especie arbórea se pone de manifiesto en el caso del *Pinus radiata* D. Don, que presentó un rápido crecimiento inicial debido a las elevadas precipitaciones registradas durante el verano posterior a la plantación, lo cual redujo las diferencias entre los tratamientos establecidos (Rigueiro-Rodríguez et al., 2000; Mosquera-Losada et al., 2006; Rigueiro-Rodríguez et al., 2010a). En relación a la influencia de la vegetación herbácea sobre el crecimiento del arbolado, se observó que el crecimiento en diámetro de *Quercus rubra* L. era menor en las parcelas que tenían mayor proporción de *Holcus lanatus* L., en comparación con las parcelas en las que predominaba *Agrostis capillaris* L., debido probablemente a que *Holcus lanatus* L. se asocia a suelos más fértiles que *Agrostis capillaris* L. (Grime et al., 2007) y por lo tanto captura más N del suelo, limitando así el crecimiento en diámetro de *Quercus rubra* L. En un estudio llevado a cabo en Canadá en el que se hizo una siembra directa de

Quercus rubra L. también se observó que la presencia de vegetación herbácea disminuía el crecimiento en diámetro del arbolado, no siendo este efecto evidente en el caso del crecimiento en altura (Laliberté et al., 2008).

6.3. PASTO

6.3.1. Producción de pasto

La producción de pasto en los ensayos establecidos en este estudio fue diferente en función de las condiciones climáticas, del tipo de suelo (agrícola o forestal), del tipo de fertilización (mineral o lodo de depuradora), de las dosis de lodo aplicadas (0, 100, 200 y 400 kg de N total ha^{-1}) y del tipo de lodo con el que se fertilizó (anaeróbico, compostado y peletizado).

En los sistemas silvopastorales establecidos en suelo agrícola abandonado y con especies frondosas se observó un efecto muy marcado del clima sobre la producción de pasto. En el ensayo con *Fraxinus excelsior* L., la producción de pasto anual fue menor que la que normalmente se obtiene en la región, ésto pudo ser debido a la sequía registrada en los años 2005, 2006, 2007 y 2008 (precipitación media mensual menor a la precipitación registrada en los 30 años anteriores) y a las bajas temperaturas de principios de año que también limitaron, en el año 2002, la producción de pasto del sistema silvopastoral establecido con *Quercus rubra* L. (Mosquera-Losada et al., 1999). En el estudio con roble americano también se observó que la ausencia de sequía durante el verano del año 2004 le permitió al pasto prolongar su periodo de crecimiento, incrementándose así la producción estacional y anual de pasto. Resultado similar obtuvieron Rigueiro-Rodríguez et al. (2010a) y Mosquera-Losada et al. (2010c) en sistemas silvopastorales establecidos en la misma zona con *Pinus radiata* D. Don y *Populus* x *canadensis* Moench, respectivamente.

En relación al tipo de suelo, la producción de pasto fue mayor en los suelos agrícolas bajo *Pinus radiata* D. Don que en los forestales con esta especie arbórea debido a la mayor fertilidad que en general presentaron los suelos agrícolas, lo cual se pone de manifiesto por el mayor pH del suelo, la mayor CIC y la mayor cantidad de N total y P total. Además, la mayor presencia de *Trifolium repens* L. en el pasto de los suelos agrícolas con *Pinus radiata* D. Don, en comparación con el pasto de los suelos forestales, también pudo incrementar la producción de pasto, ya que esa especie leguminosa fija N atmosférico mediante simbiosis con bacterias del género *Rhizobium* (González, 1992; Whitehead, 1995; Green et al., 1999; López-Díaz et al., 2009), lo cual aumenta las entradas de N al suelo y favorece la producción de pasto. Por otro lado, tanto en los suelos agrícolas desarbolados o con *Pinus radiata* D. Don como en los suelos forestales plantados con esta especie arbórea, la fertilización con lodo incrementó más la producción de pasto que la fertilización con mineral, debido al mayor efecto residual de los fertilizantes orgánicos en comparación con los inorgánicos (EPA, 1994) y a que el lodo añade al suelo mayor cantidad de Ca, Mg y micronutrientes que el fertilizante mineral (Smith, 1996; López-Díaz et al., 2007; Mosquera-Losada et al., 2010b). Por lo tanto, el uso de los lodos de depuradora como fertilizantes permite reciclar gran cantidad de nutrientes a la vez que se incrementa la producción de pasto. Además, si nos centramos sólo en los suelos agrícolas, la plantación de *Pinus radiata* D. Don y la fertilización mineral también disminuyeron la producción de pasto debido a la mayor extracción de cationes del suelo por parte del pasto y del arbolado en comparación con los suelos agrícolas sin arbolado.

Tal y como se observó en el crecimiento de *Quercus rubra* L., las dosis de 200 y 400 kg de N total ha^{-1} incrementaron más la producción de pasto que la no fertilización o la dosis de 100 kg de N total ha^{-1}, este resultado podría deberse al mayor efecto residual de la mineralización del N que presentan las dosis más altas de lodo de depuradora en comparación con las dosis más bajas (EPA, 1994; Smith, 1996). Otros autores como Mosquera-Losada et al. (2010c) y Rigueiro-Rodríguez et al. (2010a) también encontraron en sus estudios establecidos en la zona una mayor producción de pasto al aumentar las dosis de lodo de depuradora. Hay que señalar que el efecto de la dosis de 200 kg de N total ha^{-1} sobre la producción de pasto y el crecimiento del *Quercus rubra* L. fue el mismo que el de la dosis de 400 kg de N total ha^{-1}, por lo que desde un punto de vista económico y medioambiental sería suficiente con aplicar dosis de lodo que implicaran insumos de 200 kg de N total ha^{-1}, ya que las dosis superiores supondrían importantes cantidades de N no utilizadas por el pasto y el arbolado que podrían ser lavadas a través del perfil del suelo provocando la contaminación de las aguas (Nair et al., 2007).

Si nos centramos en el efecto del tipo de lodo empleado para fertilizar en el sistema silvopastoral con *Fraxinus excelsior* L. sobre la producción de pasto, se observó que a pesar de la mejora edáfica inicial que supuso el lodo anaeróbico y el lodo compostado, fue el lodo peletizado el que incrementó más la producción de pasto al final del estudio, debido a la aplicación anual repetida de N que se hizo al fertilizar con este tipo de residuo en comparación con los otros (compostado y anaeróbico). La ausencia de repuesta de la producción de pasto a la fertilización con lodo compostado pudo ser debida a la menor tasa de mineralización y disponibilidad de N en el tratamiento con lodo compostado comparativamente con los tratamientos con lodo anaeróbico o lodo peletizado, tal y como indica la EPA (1994), que cifra en alrededor del 20% la tasa de mineralización del lodo anaeróbico y del 10% la del lodo compostado, en ambos casos en el primer año de aplicación. El efecto del lodo compostado sobre la producción de pasto y el crecimiento en altura del arbolado podría llegar a manifestarse con el paso del tiempo, tal y como sucede en nuestro caso en el diámetro del arbolado. Otros autores, como Warman y Termeer (2004), también encontraron una mejor respuesta inicial de los cultivos a la aplicación de lodo anaeróbico que a la de lodo compostado, para una dosis similar de N total.

6.3.2. Composición botánica del pasto

En general, en los estudios con frondosas se observó que la baja fertilidad del suelo limitó el establecimiento de las especies sembradas (*Dactilys glomerata* L., *Lolium perenne* L. y *Trifolium repens* L.) (Rigueiro-Rodríguez et al., 2000; Grime et al., 2007) y favoreció la presencia de especies espontáneas como *Agrostis capillaris* L. Además, en el estudio establecido en suelos forestales y agrícolas con *Pinus radiata* D. Don, a pesar de la mayor fertilidad de los suelos agrícolas que la de los suelos forestales, hubo un mejor establecimiento inicial de las especies sembradas (*Dactilys glomerata* L. y *Lolium perenne* L.) en los suelos forestales que en los agrícolas, debido a la presencia de especies anuales en el banco de semillas de los suelos agrícolas que desplazan a las especies sembradas, ya que los terófitos se caracterizan por su rápido crecimiento y capacidad de colonización (abundantes semillas y germinación rápida) (Grime et al., 2007; Mosquera-Losada et al., 2009a). Sin embargo, las especies sembradas, como *Dactilys glomerata* L. y *Lolium perenne* L., normalmente no se adaptan bien a la baja fertilidad de los suelos forestales, siendo en este tipo de suelos menos extractivas que las especies no sembradas (Whitehead, 1995) las cuales reducen la proporción de N en el suelo, lo que provoca una disminución de la presencia y abundancia de esas dos especies de siembra al final del estudio en los suelos forestales con *Pinus radiata* D. Don en los que se sembró pasto. La proporción de *Trifolium repens* L. fue mayor en los suelos agrícolas con *Pinus radiata* D. Don que en los suelos forestales con este árbol, debido a que ese trébol se desarrolla mejor en los suelos más fértiles (Grime et al., 2007) y con mayores niveles de potasio en el suelo (Mosquera-Losada y González-Rodríguez, 1997).

En los suelos agrícolas con *Fraxinus excelsior* L. y *Quercus rubra* L., en los que se simuló pastoreo rotacional con ovejas, la especie que apareció en mayor proporción fue *Agrostis capillaris* L., que se adapta bien al pastoreo con estos animales debido a su crecimiento vegetativo mediante estolones (Grime et al., 2007). Este resultado también se observó en otros estudios desarrollados en la República Checa por Krahulec et al. (2001). Otras especies que aparecieron en el pasto pastoreado por ovejas en elevada proporción fueron *Dactylis glomerata* L. y *Holcus lanatus* L., siendo su presencia mayor en aquellas parcelas con mayor fertilidad del suelo, ya que estas especies se asocian a suelos más fértiles que *Agrostis capillaris* L. (Grime et al., 2007), tal y como se ha demostrado en suelos con pH por debajo de 4,97 en el caso de *Dactylis glomerata* L. (Mosquera-Losada et al., 2001).

Si nos centramos en los suelos forestales con *Pinus radiata* D. Don, la proporción de especies sembradas (*Dactilys glomerata* L. y *Lolium perenne* L.) fue mayor cuando se sembró pasto que cuando no se realizó la siembra, siendo *Agrostis stolonifera* L. la especie espontánea presente en mayor proporción en los suelos forestales no sembrados, lo que indica que el banco de semillas de las especies de siembra era reducido en estos suelos. La mejora edáfica del suelo forestal con *Pinus radiata* D. Don que supuso la fertilización con lodo de depuradora o con fertilizante mineral se observa en la primavera del último año de estudio, cuando empieza a ser importante la presencia de *Trifolium repens* L. en el pasto, ya que el trébol necesita suelos fértiles para establecerse (Grime et al., 2007) y una vez que se establece es una especie muy adecuada en el pasto de los sistemas silvopastorales, puesto que compite menos por los nutrientes con el arbolado que las gramíneas, e incluso lo favorece a través del aporte de nitrógeno, favoreciendo además la persistencia de un estrato herbáceo que evita el riesgo de incendios (López-Díaz et al., 2009).

En relación a la riqueza específica del pasto de los suelos agrícolas con *Fraxinus excelsior* L. y *Quercus rubra* L. hay que señalar que el efecto de los tratamientos de fertilización sobre la fertilidad del suelo explica en buena parte el crecimiento del arbolado, la producción de pasto y la biodiversidad. Es importante destacar que en el sistema silvopastoral establecido con *Quercus rubra* L. se encontró una mejora en la riqueza específica o número de especies del pasto al final del estudio debido al efecto positivo del desarrollo del pasto y el arbolado en el suelo arenoso sobre las propiedades físico-químicas del suelo (Mosquera-Losada et al., 2009a). Sin embargo, la riqueza específica se vio reducida en aquellas praderas en las que se aplicaron dosis medias (200 kg de N tota ha^{-1}) y altas (400 kg de N tota ha^{-1}) de lodo de depuradora en comparación con aquellas que recibieron dosis bajas (100 kg de N tota ha^{-1}) de lodo, probablemente debido a que las dosis altas incrementaron más la fertilidad química del suelo, tal y como se pone de manifiesto por el mayor crecimiento del arbolado y la mayor producción de pasto, y en general, un aumento de la fertilidad del suelo suele acompañarse de un menor número de especies invasoras y por lo tanto una disminución del número de especies en el pasto (Thompson et al., 2001). Este mismo resultado fue observado en el sistema silvopastoral establecido con *Fraxinus excelsior* L. inicialmente al aplicar lodo anaeróbico, ya que incrementó en mayor medida la producción de pasto y la fertilidad del suelo, y al final del estudio al aplicar lodo peletizado, debido a que los aportes anuales de N que se hicieron al fertilizar con el lodo peletizado incrementaron más la fertilidad del suelo en los últimos años que los otros tratamientos.

6.3.3. Calidad del pasto

Las concentraciones de proteína bruta (58-183 g kg^{-1}) y P (1,6-4,8 g kg^{-1}) del pasto establecido en suelo agrícola y forestal con *Pinus radiata* D. Don fueron similares a las concentraciones descritas por Whitehead (1995) (80-250 g kg^{-1}) y Grime et al. (2007) (1,5-4,5 g kg^{-1}), respectivamente, con la excepción del pasto correspondiente a las cosechas de verano que presentó menor cantidad de proteína bruta debido a que es la época de floración que se caracteriza por la disminución de la concentración de nitrógeno en planta y un aumento del contenido en fibra (Whitehead, 1995).

Si se comparan los suelos agrícolas con *Pinus radiata* D. Don con los suelos forestales repoblados con esta especie forestal se observa que los niveles de proteína bruta y la concentración de P del pasto fueron mayores en los pastos desarrollados en suelos agrícolas que en los suelos forestales. Este resultado podría ser explicado por la mayor fertilidad (mayor pH, mayor CIC, mayor nivel de cationes) de los suelos agrícolas, lo que implica un aumento de los niveles de P y proteína bruta del pasto en estos suelos.

Si nos centramos en los suelos forestales con *Pinus radiata* D. Don encontramos que la siembra de pasto en este tipo de suelos disminuyó aún más los niveles de proteína bruta en el pasto que la ausencia de siembra, debido a que con la siembra se mejora el establecimiento de *Dactylis glomerata* L. y *Lolium perenne* L., poco adaptadas a suelos poco fértiles como son los suelos forestales, lo que las hace menos extractivas que las especies espontáneas (Whitehead, 1995) y reduce la cantidad de N del pasto.

Por otro lado, el pasto del sistema silvopastoral establecido con *Fraxinus excelsior* L. presentó unas concentraciones de Zn (18,63–49,31 mg kg^{-1}) y Cu (1,4-7,2 mg kg^{-1}) menores que las que normalmente presenta el pasto (Zn: 27–150 mg kg^{-1}) (Cu:10-80 mg kg^{-1}) (Smith, 1996; Loué, 1988) e inferiores a los niveles de Zn (100-400 mg kg^{-1}) y Cu (20-100 mg kg^{-1}) que son considerados excesivos o tóxicos para las plantas (Kabata-Pendías y Pendías, 2001). El tipo de lodo (anaeróbico, compostado y peletizado) influyó en la concentración de Zn (elemento aportado en mayor cantidad con el lodo de entre los metales regulados por el RD 1310/90) en las plantas, mientras que no se observó un efecto claro sobre la cantidad de Cu en el pasto. El lodo anaeróbico tendió a incrementar la concentración de Zn en el suelo y en el pasto, tal y como previamente habían observado Mosquera-Losada et al. (2001) en estudios establecidos en suelos ácidos de Galicia. A pesar de los mayores aportes de Zn al suelo con el lodo compostado, que implicó un incrementó del Zn total y disponible del suelo, no se encontró un efecto claro de este tratamiento sobre la concentración de Zn en el pasto, tal vez debido al aumento del pH del suelo asociado a este tratamiento, que pudo reducir la disponibilidad del Zn para las plantas (Prasad and Power, 1997).

En relación a las necesidades nutricionales de los animales, en el estudio con *Pinus radiata* D. Don, la cantidad de proteína bruta del pasto de las cosechas de verano de los años 2007 y 2008 no fue suficiente para cubrir las necesidades de mantenimiento del ganado ovino (94 g kg^{-1}) (NRC, 1985), equino (85 g kg^{-1}) (NRC, 1989) y caprino (60 g kg^{-1}) (Lamand, 1981). Sin embargo, la concentración de P en el pasto de este estudio fue suficiente para alimentar al ganado ovino (1,6-3,7 g kg^{-1}) (NRC, 1985), equino (2 g kg^{-1}) (NRC, 1989) y caprino (2,5 g kg^{-1}) (Lamand, 1981).

Una concentración de Zn en planta de 500 mg kg^{-1} puede ser considerada tóxica para el ganado vacuno, ovino y equino (Smith, 1996). El pasto del estudio con *Fraxinus excelsior* L. no ha alcanzado este valor, por lo que no se puede considerar que haya riesgo de toxicidad para los animales que lo consuman.

PARTE VII

CONCLUSIONES

7. CONCLUSIONES

✓ En nuestros estudios hemos observado que el **pH** del suelo y la **capacidad de intercambio catiónico** se han visto modificados positivamente por el aporte de lodo y por la incorporación de materia orgánica al suelo procedente del establecimiento del pasto y del arbolado en suelos arenosos, aunque la magnitud de la respuesta va a depender del tipo de suelo (agrícola o forestal), de la dosis de lodo y del tipo de estabilización empleada en el lodo aportado (anaeróbica, compostaje y peletización). El pH del suelo y la capacidad de intercambio catiónico fueron mayores en los suelos agrícolas con *Pinus radiata* D. Don que en los suelos forestales repoblados con esta especie arbórea. El pH del suelo aumentó al fertilizar con lodo compostado que aporta unas mayores cantidades de calcio, mientras que la capacidad de intercambio catiónico fue mayor al aumentar las dosis de lodo de depuradora tal y como se observó en los experimentos desarrollados bajo *Fraxinus excelsior* L y *Quercus rubra* L.

✓ Los suelos agrícolas sin arbolado o con *Pinus radiata* D. Don presentaron una mayor incorporación del lodo que los suelos forestales plantados con esta especie arbórea, debido al mayor pH y actividad microbiana, lo que incrementó la disponibilidad de **nitratos** y **P** en los suelos agrícolas y por lo tanto el riesgo de lavado de estos elementos, siendo más relevante el lavado de nitratos al inicio del experimento, cuando los árboles y el pasto no se habían desarrollado lo suficiente como para absorber los nitratos liberados tras el aporte de lodo en el suelo agrícola.

✓ En el estudio con *Fraxinus excelsior* L., la aplicación de lodo compostado supuso una mayor concentración total de **Cu** y **Zn** en el suelo que el lodo anaeróbico y peletizado debido a los mayores aportes de Cu y Zn al suelo con este tipo de lodo, en todo caso, los valores de Cu y Zn registrados fueron inferiores a los permitidos en suelo por la legislación vigente (Cu: 50 mg kg^{-1} y Zn: 150 mg kg^{-1}) (R.D. 1310/1990) que regula el empleo de lodo de depuradora urbana con fines agrícolas.

✓ La respuesta del crecimiento del arbolado a la fertilización con lodos de depuradora va a depender de las condiciones climáticas, no encontrándose una respuesta positiva cuando el clima limita el crecimiento del arbolado, tal y como sucedió en el estudio con ***Pinus radiata* D. Don**. Cuando las condiciones climáticas lo permitieron, la fertilización con lodos de depuradora mejoró el crecimiento del arbolado pero la magnitud de la respuesta fue diferente en función de la dosis empleada y del tipo de lodo utilizado (anaeróbico, compostado o peletizado). Las dosis medias (200 kg de N total ha^{-1}) y altas (400 kg de N total ha^{-1}) de lodo de depuradora incrementaron el crecimiento del ***Quercus rubra* L.** mientras que el crecimiento del ***Fraxinus excelsior* L.** fue mayor cuando se fertilizó con lodo peletizado que cuando se aplicó lodo anaeróbico o compostado (320 kg de N total ha^{-1}).

✓ La **producción de pasto** se vio afectada por los tratamientos cuando las precipitaciones y temperaturas fueron adecuadas para el desarrollo del pasto. La producción de pasto fue mayor en los suelos agrícolas que en los suelos forestales con *Pinus radiata* D. Don debido a la mayor disponibilidad de nutrientes de los suelos agrícolas y cuando se aplicó lodo de depuradora en comparación con la fertilización mineral. Al igual que en el arbolado, la producción de pasto también aumentó al aplicar dosis medias (200 kg de N total ha^{-1}) y altas (400 kg de N total ha^{-1}) de lodo de depuradora y cuando se fertilizó con lodo peletizado.

✓ En los sistemas silvopastorales establecidos bajo *Fraxinus excelsior* L. y *Quercus rubra* L., la ausencia de fertilización y el pastoreo con ovejas limitó el establecimiento de las **especies sembradas** (*Dactilys glomerata* L., *Lolium perenne* L. y *Trifolium repens* L.) y favoreció la presencia de **especies espontáneas** como *Agrostis capillaris* L. En los suelos forestales con *Pinus radita* D. Don el establecimiento de las especies sembradas (*Dactilys glomerata* L. y *Lolium perenne* L.) fue mejor que en los suelos agrícolas repoblados con esta especie de árbol, en los que se establecieron inicialmente mejor especies anuales dicotiledóneas. Sin embargo, debido a la baja fertilidad de los suelos forestales, el porcentaje de estas dos especies de siembra disminuyó después del establecimiento del experimento.

✓ En los estudios con *Fraxinus excelsior* L. y *Quercus rubra* L. se observó un aumento de la **riqueza específica** del pasto con el paso de los años debido al efecto positivo del desarrollo del pasto y el arbolado en el suelo arenoso sobre la fertilidad física del suelo, pero a su vez también se encontró que la riqueza específica disminuyó al aplicar dosis medias (200 kg de N tota ha^{-1}) y altas (400 kg de N tota ha^{-1}) de lodo de depuradora y al inicio y al final del estudio al fertilizar con lodo anaeróbico y lodo peletizado, respectivamente, ya que estos tratamientos incrementaron la fertilidad química del suelo lo que favoreció el desarrollo de especies de siembra en detrimento de las especies invasoras provocando una disminución del número total de especies.

✓ La menor fertilidad de los suelos forestales plantados con *Pinus radiata* D. Don que de los suelos agrícolas desarbolados o repoblados con esta especie arbórea, así como, el mejor establecimiento de *Dactylis glomerata* L. y *Lolium perenne* L. en los suelos forestales disminuyó los niveles de **proteína bruta** y la concentración de **P** del pasto debido a la menor disponibilidad de nutrientes en los suelos forestales en comparación con los suelos agrícolas y a la mayor capacidad de extracción de nutrientes de las especies sembradas que de las especies espontáneas.

✓ Las concentraciones de **Zn** y **Cu** en el pasto desarrollado bajo *Fraxinus excelsior* L. fueron inferiores a los niveles considerados como excesivos o tóxicos para las plantas. El lodo anaeróbico incrementó más la concentración de Zn en el pasto que los otros tipos de lodo de depuradora.

✓ En los sistemas silvopastorales, la fertilización con lodos de depuradora permite el reciclaje de este residuo a la vez que incrementa la producción de pasto y el crecimiento del arbolado. Las dosis de 200 y 400 kg de N total ha^{-1} incrementaron de forma similar la producción de pasto y el crecimiento del arbolado por lo que desde un punto de vista económico y medioambiental sería suficiente con aplicar dosis de lodo que implicaran insumos de 200 kg de N total ha^{-1}. Debería promoverse el uso del lodo peletizado ya que este tratamiento mejora la producción de pasto y arbolado y, al tener un menor contenido en agua que los lodos compostados y digeridos anaeróbicamente, su aplicación al suelo es más sencilla y su transporte y almacenamiento son más baratos.

PARTE VIII

CONCLUSIONS

8. CONCLUSIONS

- ✓ Soil **pH** and **cation exchange capacity** was positively modified by the inputs of sewage sludge as well as by the establishment of pasture and trees in sandy soils, which increased the organic matter inputs into the soil. The magnitude of the response depended on the type of soil (agronomic or forest), sewage sludge dose and the stabilization type of sewage sludge (anaerobic, composted and pelletized). Soil pH and the cation exchange capacity were higher in the agronomic soils with *Pinus radiata* D. Don than in the forest soils with this tree species. Soil pH was increased when composted sludge was applied (which has a higher level of Ca), being the cation exchange capacity improved as the dose of sewage sludge was increased as observed under *Fraxinus excelsior* L. and *Quercus rubra* L.
- ✓ Sludge incorporation into the agronomic soils planted with *Pinus radiata* D. Don or treeless pastures was better than the forest soils with this tree species due to the higher pH and microbial activity of the former, which increases **nitrate** and **P** availability, and therefore the risk of leaching. Nitrate leaching from soils was only relevant if trees and pasture were not enough developed to uptake nitrate.
- ✓ In the study with *Fraxinus excelsior*, the use of composted sludge as fertilizer increased the total concentration of **Cu** and **Zn** in the soil compared with the anaerobic or pelletized sludge due to the higher input rates of Cu and Zn into the soil made by the composted sludge. In any case, the total soil concentrations of Cu and Zn found in the experiment were always below the maximums set by Spanish regulations for the use of sewage sludge in agriculture for acid soils (Cu: 50 mg kg^{-1} and Zn: 150 mg kg^{-1}) (R.D. 1310/1990)
- ✓ Tree growth response to sludge inputs depended on climatic conditions. No answer to sewage sludge was found when specific weather conditions limited tree growth, as happened in the experiment of ***Pinus radiata* D. Don**. When tree growth was allowed, the fertilization with sewage sludge improved tree growth, but the magnitude of the response depended on the sludge dose and the type of sludge used (anaerobic, composted and pelletized). Medium (200 kg total N ha^{-1}) and high (400 kg total N ha^{-1}) doses of sewage sludge increased the growth of ***Quercus rubra* L.,** while ***Fraxinus excelsior* L.** growth was better with pelletized than anaerobic or composted sludge (320 kg total N ha^{-1}).
- ✓ **Pasture production** was increased by the treatments when the precipitation and the temperatures were adequate to allow pasture development. Pasture production was

also higher in the agronomic than in the forest soils planted with *Pinus radiata* D. Don, due to the better nutrient availability in the agronomic soils, and also pasture production was positively affected by the sewage sludge addition compared with mineral. As in the trees, pasture production was increased by the application of medium (200 kg total N ha^{-1}) and high (400 kg total N ha^{-1}) doses of sewage sludge and when pelletized sludge was applied.

✓ In the silvopastoral systems developed under *Fraxinus excelsior* L. and *Quercus rubra* L., the lack of fertilization and sheep grazing limited the establishment of the **sown species** (*Dactilys glomerata* L., *Lolium perenne* L. and *Trifolium repens* L.) and favoured the presence of spontaneous species such as *Agrostis capillaris* L. The establishment of the sown species (*Dactilys glomerata* L. and *Lolium perenne* L.) was better in forest soils with *Pinus radiata* D. Don than in agronomic soils, where dicot annual species were initially established. However, due to the low soil fertility of the forest soils, the percentage of these grass sown species decreased after the establishment of the experiment.

✓ In the studies under *Fraxinus excelsior* L. and *Quercus rubra* L. it was observed that the pasture **species richness** increased over time due to the positive effect of the establishment of the pasture and trees on the soil physical fertility in sandy soils. However, it was also found that the improvement of the soil chemical fertility reduced the invasion of weeds and therefore the number of species when medium (200 kg total N ha^{-1}) and high (400 kg total N ha^{-1}) doses of sewage sludge were applied and anaerobic and pelletized sludge were used at the beginning and at the end of the experiment, respectively.

✓ The lower fertility of the forest soils with *Pinus radiata* D. Don than the treeless or *Pinus radiata* D. Don planted agronomic soils as well as the best establishment of *Dactylis glomerata* L. and *Lolium perenne* L. in forest soils decreased the levels of **crude protein** and **P** in the pasture due to the lower of nutrient availability in forest soils compared with agronomic soils and the higher extraction capacity of these nutrients by sown than unsown species.

✓ The concentrations of **Zn** and **Cu** in pasture under *Fraxinus excelsior* L. were below the levels that are considered excessive or toxic for plants. Anaerobic sludge increased more the concentration of Zn in pasture than the other types of sewage sludge.

✓ In silvopastoral systems, the fertilization with sewage sludge allows nutrient recycling of this residue while increases pasture production and tree growth. The doses of 200 and 400 kg total N ha^{-1} increased pasture production similarly and the tree growth, therefore, from an economic and environmental point of view would be sufficient to apply 200 kg total N ha^{-1}. The pelletized sludge should be promoted because this type of sludge enhances tree and pasture production, allows for better nutrient recovery due to the split applications and is less costly to apply than the other two treatments.

BIBLIOGRAFÍA

ABBERTON M.T., MARSHALL A.H., HUMPHREYS M.W., MACDUFF J.H., COLLINS R.P. Y MARLEY C.L. (2008) Genetic improvement of forage species to reduce the environmental impact of temperate livestock grazing systems. *Advances in Agronomy*, **98**, 311–355.

AFTA (Association for Temperate Agroforestry) (1999) *Agroforestry practices*. http://www.missouri.edu/afta/Agfo_Practices.html (consultado el 18 de octubre del 2010).

AGROFORESTRY FORUM (2007) *Definitions.* http://www.agroforestry.ac.uk/systems/index.html (consultado el 18 de octubre del 2010).

AGROFORESTRY FORUM (2009) *The Farm Woodland Forum. Silvopastoral National Network Experiment*. http://www.agroforestry.ac.uk/systems/pastoral/snne03.html (consultado el 11 de mayo del 2010).

ALAVALAPATI J.R.R., MERCER, D.E. Y MONTAMBAULT J.R. (2004) Agroforestry systems and evaluation methodologies. En: Alavalapati J.R.R. y Mercer D.E (eds.) *Valuing Agroforestry Systems: Methods and Applications*. *Advances in Agroforestry*, **2**, 1-8. Kluwer Academic Publishers. Dordrecht, Holanda.

ALLEN S., NAIR P.K.R, BRECKE B.J., NKEDI-KIZZA P. Y RAMSEY C.L. (2004) Safety net role of tree roots: experimental evidence from an alley cropping system. *Forest Ecology and Management*, **192**, 395-407.

ALLEN S.C., NAIR V.D., GRAETZ D.A., JOSE S. Y NAIR R.P.K. (2006) Phosphorus loss from organic versus inorganic fertilizers used in alleycropping on a Florida Ultisol. *Agriculture, Ecosystems and Environment*, **117**, 290-298.

ALLOWAY B.J. (1995) *Heavy metals in soils*. Blackie academic and professional. Londres, Reino Unido.

ALONSO J. Y BENTO J. (2005) Adaptation of an agrosilvopastoral system to land-use dynamics: local-level analysis of strategies and practice changes in north-eastern Portugal. En: Mosquera-Losada M.R., McAdam J. y Rigueiro-Rodríguez A. (eds.) *Silvopastoralism and sustainable land management,* pp. 343–345. CABI Publishing. Wallingford, Reino Unido.

ÁLVAREZ-ÁLVAREZ P., BARRIO-ANTA M., DÍAZ-VARELA R.A., HIGUERAS-DE MARCO J., RIESCO-MUÑOZ G., RIGUEIRO-RODRÍGUEZ A., RODRÍGUEZ-SOALLEIRO R. Y VILLARINO-URTIAGA J.J. (2000) *Manual de selvicultura de frondosas caducifolias*. Columella. Escuela Politécnica Superior de Lugo. http://www.agrobyte.com/agrobyte/publicaciones/frondosas/indice.html (consultado el 25 de agosto del 2010).

ÁLVAREZ P., ROSA E., VEGA P. Y RODRÍGUEZ-SOALLEIRO R. (2001) Viveros forestales y uso de planta en repoblación en Galicia. *III Congreso Forestal Español*, **3,** 232-238. Granada, España.

ANDERSON L.S. Y SINCLAIR F.L. (1993) Ecological interactions in agroforestry systems. *Forestry Abstracts*, **54**, 489-523.

ANDRADE-COUCE M.L., BAO-IGLESIAS M., FERNÁNDEZ-FERNÁNDEZ C.J. Y GUITIÁN-OJEA F. (1985) Caracterización química y poder fertilizante de los lodos residuales de la planta depuradora de aguas de Santiago de Compostela. *Anales de Edafología y Agrobiología XLIV*, **44**, 143-156.

AUGUSTO L., RANGERA J., BINKLEY D. Y ROTHE A. (2002) Impact of several common tree species of European temperate forests on soil fertility. *Annals of Forest Science,* **59**, 233-253.

BAILEY J.S. (1995) Liming and nitrogen efficiency: some effects of increased calcium supply and increased soil pH on nitrogen recovery by perennial ryegrass. *Communications in Soil Science and Plant Analysis,* **26**, 1233-1246

BALANDIER P. Y DUPRAZ C. (1999) Growth of widely spaced trees. A case study from young agroforestry plantations in France. *Agroforestry Systems*, **43**, 151-167.

BARBER S.A. (1995) *Soil nutrient bioavailability: a mechanistic approach*, 2nd edn. John Wiley & Sons. New York, USA.

BARRECHEGUREN M.A., CABRERO M., COLLAR J., MORENO J., SANTAOLALLA G. Y ZEA J. (1992) *"100 anos de Investigación Agraria"*. Xunta de Galicia. Santiago de Compostela, España.

BARRY E., KEEFE O. Y AXLEY J. (1986) Evaluation of nitrogen availability indexes for a sludge compost amended soil. *Journal of Environmental Quality*, **15**, 121-128.

BEATON A. Y HISLOP M. (2000) Trees in agroforestry systems. En: HISLOP M. y CLARIDGE J. (eds.) *Agroforestry in the UK.* Forestry Commission Bulletin 122, pp. 31-43. HMSO. Londres, Reino Unido.

BENAVIDES R., DOUGLAS E.G. Y OSORO, K. (2009) Silvopastoralism in New Zealand: review of effects of evergreen and deciduous trees on pasture dynamics. *Agroforestry Systems*, **76**, 327–350.

BELTRÁN E.M., MIRALLES DE IMPERIA L.R., PORCEL M.A., DELGADO M.M., BERINGOLA M.L. Y MARTIN J.V. (2002) Effect of sewage sludge compost application on ammonium-nitrogen and nitrate-nitrogen contents of an olive grove soil. *Proceedings 12th International Soil Conservation*, **3**, 395-402. Tsinghua University Press. Pekin, China.

BOE (Boletín Oficial del Estado) (1990) *Real Decreto 1310/1990 29 del de Octubre de 1990, que regula la utilización de los lodos de depuración.* Ministerio Agricultura, Pesca y Alimentación. Madrid, España.

BRADY N.C. Y WEIL R.R. (2008) *The nature and properties of soils*, 14th edn. Prentice Hall. Upper Saddle River, New Jersey, USA.

BRAN+LUEBBE (1979) *Methods for the chemical analisis of water and wastes*. Bran+Luebbe. Norderwstedt, Alemania.

BREMNER J.M. (1965) Nitrogen availability indexes En: Black C.A., Evans D.D., Esminger L.E. y Clark F.E. (eds.) *Methods of soil analysis. Part. 2. Chemical and Microbiological Properties*, pp. 1324–1345. American Society of Agronomy. Madison, USA.

BURGESS P.J. (1999) Effect of agroforestry on a farm biodiversity in UK. *Scottish Forestry,* **53**, 24-27.

CALMANO W., MANGOLD S., STICHNOTHE H. Y THÖMING J (2001) Clean-up and assessment of metal contaminated soils. En: Stegmann R., Brunner G., Calmano W. y Matz G. (eds.) *Treatment of Contaminated Soil - Fundamentals, Analysis, Applications*, pp. 471-490. Springer Verlag. Heidelberg, Alemania.

CAMPBELL C.D., ATKINSON D., JARVIS P.G. Y NEWBOULD P. (1993) Effects of nitrogen fertilizer on tree/pasture competition during the establishment phase of a silvopastoral system. *Annals of Applied Biology*, **124**, 83–96.

CANET R., POMARES F., ESTELA M. Y TARAZONA F. (1996) Efecto de los lodos de depuradora en la producción de hortalizas y las propiedades químicas del suelo. *Investigación agrícola serie Producción y Protección Vegetal*, **2**, 83-99.

CANET R., POMARES F., TARAZONA F. Y ESTELA M. (1998) Sequential fractionation and plant availability of heavy metals as affected by sewage sludge applications to soil. *Communication of Soil Plant*, **29**, 697-716.

CASSMAN K. G. (1999) Ecological intensification of cereal production systems: yield potential, soil quality, and precision agriculture. *Proceedings of National Academy of Sciences of the USA*, **96**, 5952-5959.

CASTRO P., GONZÁLEZ A. Y PRADA D. (1990) Simultaneous determination of nitrogen and phosphorus in pasture samples. *XXX Reunión Científica de la Sociedad Española para el estudio de los pastos*, **30**, 200–207. San Sebastián, España.

CEM (1994) *Microwave digestion applications manual*. CED. San Francisco, USA.

COSTA-YAGÜE F., HERNÁNDEZ-FERNÁNDEZ M.T. Y MORENO-SÁNCHEZ J.I. (1987) *Utilización agrícola de lodos de depuradora*. CSIC. Alicante, España.

CUCCI G., LACOLLA G. Y CARANFA L. (2008) Improvements of soil properties by application of olive oil waste. *Agronomy Sustainable Development*, **28**, 521-526.

CUTHBERTSON A. Y MCADAM J. (1996) The effect of tree density and species of carabid Beetles in a range of pasture-free agroforestry systems on a lowland site. *Agroforestry forum*, **7**, 17-20.

CBD (2010) *Convention on Biological Diversity*. http://www.cbd.int/ (consultado el 17 de octubre del 2010).

CHANG S.X., AMATYA G., BEARE M.H. Y MEAD D.J. (2002) Soil properties under a *Pinus radiata*-ryegrass silvopastoral system in New Zealand. Part I. Soil N and moisture availability, soil C, and tree growth. *Agroforestry Systems*, **54**, 137-147.

DANS F., FERNÁNDEZ DE ANA-MAGÁN F.J., ROMERO A., CASTRO A., MOLINA F., ROJO A. Y SÁNCHEZ F. (1999) *Manual de selvicultura del Pino Radiata en Galicia*. Proyecto Columella. Escuela Politécnica Superior de Lugo. http://www.agrobyte.com/agrobyte/publicaciones/pinoradiata/cap1_1.html (consultado el 2 de septiembre del 2010).

DENNIS P., SHELLARD L.J.F. Y AGNEW R.D.M. (1996) Shifts in arthropod species assemblages in relation to silvopastoral establishment in upland pastures. *Agroforestry Forun*, **7**, 14-17.

DEXTER A.R. (1988) Advances in characterization of soil structure. *Soil and Tillage Research*, **11**, 199–238.

DJUKIC I., ZEHETNER F., MENTLER A. Y GERZABEK M.H. (2009) Microbial community composition and activity in different Alpine vegetation zones. *Soil Biology and Biochemestry*, **42**,155-161.

EEA (European Environmental Agency) (2000) *Working document on sludge 3rd draft. Brussels. 27 April 2000. ENV.E.3/LM.* http://ec.europa.eu/environment/waste/sludge/pdf/sludge_en.pdf (consultado el 27 de enero del 2010).

EFMA (European Fertilizer Manufacturers Association) (2009) *Annual report 2009. Modern agriculture feeds the world and helps protect the environment.* http://www.efma.org/ (consultado el 3 de enero del 2010).

EGIARTE G., ARBESTAIN M.C., ALONSO A., RUÍZ-ROMERA E. Y PINTO M. (2009) Effect of repeated applications of sewage sludge on the fate of N in soils under Monterey pine stands. *Forest Ecology and Management*, **216**, 257–269.

EICHHORN M., PARIS P., HERZOG F., INCOLL L., LIAGRE F., MANTZANAS K., MAYUS M., MORENO G., PAPANASTASIS V., PILBEAM D., PISANELLI A. Y DUPRAZ C. (2006) Silvoarable Systems in Europe– Past, Present and Future Prospects. *Agroforestry Systems*, **67**, 29-50.

EPA (Environmental Protection Agency) (1994) *Land application of sewage sludge. A guide for land appliers on the requirements of the federal standards for the use and disposal of sewage sludge, 40 CFR Part 503*. Environment Protection Agency. Washington DC, USA.

ETIENNE M. (2005) Silvopastoral management in temperate and Mediterranean areas. Stakes, practices and socio-economics constraints. En: Mosquera-Losada M.R., McAdam J. y Rigueiro-Rodríguez A. (eds.) *Silvopastoralism and sustainable land management,* pp. 299–311. CABI Publishing. Wallingford, Reino Unido.

FAITHFULL N.T. (2002) *Methods in agricultural chemical analysis*. CAB International Publishing. Wallinford, Reino Unido.

FERGUSSON J.E. (1990) *The heavy elements: chemistry environmental impact and health effects*. Pergamon Press. Oxford, Reino Unido.

FERNÁNDEZ-NÚÑEZ E., MOSQUERA-LOSADA M.R. Y RIGUEIRO-RODRÍGUEZ A. (2007) Economic evaluation of different land use alternatives: forest, grassland and silvopastoral systems. *European grassland Federation*, **12**, 508-511.

FERNÁNDEZ-NÚÑEZ E., RIGUEIRO-RODRÍGUEZ A. Y MOSQUERA-LOSADA M.R. (2010a) Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW España. *Ecological Engineering*, **36**, 876-890.

FERNÁNDEZ-NÚÑEZ E., RIGUEIRO-RODRÍGUEZ A. Y MOSQUERA-LOSADA M.R. (2010b) Afforestation of agricultural land with *Pinus radiata* D. Don and *Betula alba* L. in NW Spain: Effects on soil pH, understorey production and floristic diversity eleven years after establishment. *Land Degrade*. “in press”.

GIDDENS K.M., PARFITT R.L. Y PERCIVAL H.J. (1997) Comparison of some soil properties under *Pinus radiata* and improved pasture. *Journal Agricultural Research*, **40**, 409–416.

GIGLIOTTI G., GIUSQUIANI L.D. Y BUSINELLI D. (2001) A long-term chemical and infrared spectroscopy study on a soil amendment with municipal waste sewage sludge. *Agronomie*, **21**, 169-178.

GONZÁLEZ A. (1992) Las praderas de gramínea y trébol y la fertilización nitrogenada en Galicia. En: Barrecheguren M.A., Cabrero M., Collar J., Moreno J., Santaolalla G. y Zea J. (eds.) *“100 anos de Investigación Agraria”*, pp. 383-393. Xunta de Galicia. Santiago de Compostela, España.

GRAVES A.R., BURGUESS P.J., LIAGRE F., PISANELLI A., PARIS P., MORENO G., BELLIDO M., MAYUS M., POSTMA M., SCHINDLER B., MANTZANAS K., PAPANASTIS V.P. Y DUPRAZ C. (2008) Farmer perceptions of silvoarable systems in seven European countries. En: Rigueiro-Rodríguez A., McAdam J.H. y Mosquera-Losada M.R. (eds.) *Agroforestry in Europe, Advances in Agroforestry*, **6**, 155–199. Springer. Los Países Bajos.

GREEN J.J., ALIFRAGIS D., BADDELEY J.A., BAGNARESI U., BOHNE H. Y BULFIN M. (1999) Impact of understorey vegetation on tree production in European agrofororestry systems. *Grassland Science in Europe*, **4**, 171–176.

GRIME J.P., HODGSON J.G. Y HUNT R. (2007) *Comparative plant ecology. A functional approach to common British species, 2nd ed.* Castlepoint press. Dalbeattie, Escocia.

GUITIÁN F. Y CARBALLÁS T. (1976) *Técnicas de análisis de suelos (Soil Analysis Techniques)*. Ed. Pico Sacro. Santiago de Compostela, España.

HAMON R.E., HOLM P.E., LORENZ S.E., MCGRATH S.P. Y CHRISTENSEN T.H. (1999) Metal uptake by plants from sludge-amended soils: caution is required in the plateau interpretation. *Plant and Soil*, **216**, 53-64.

HANLEY T.A., ROBBINS C.T., HAGERMAN A.E. Y MCARTHUR C. (1992) Predicting digestive protein and digestible dry matter in tannin-containing forages consumed by ruminants. *Ecology*, **73**, 537-541.

HAWKE M.F. (1991) Pasture production and animal performance under pine agroforestry in New Zealand. *Forest Ecology and Management*, **45**, 109-118.

HILLMAN J.P., HILL J., MORGAN J.E. Y WILKINSON J.M. (2003) Recycling of sewage sludge to grassland: a review of the legislation to control of the localization and accumulation of potential toxic metals in grazing systems. *Grass and Forage Science,* **58**, 101–111.

HISLOP M. Y SINCLAIR F. (2000) Introduction. In: HISLOP M. Y CLARIDGE J. (eds.) *Agroforestry in the UK.* Forestry Commission Bulletin 122, pp. 1-6. HMSO. Londres, Reino Unido.

HOWLETT D.S., MOSQUERA-LOSADA M.R., RAMACHANDRAN P.K.R., NAIR V.D. Y RIGUEIRO-RODRÍGUEZ A. (2010) Soil carbon storage in silvopastoral systems and a treeless pasture in northwestern Spain. *Journal Environmental quality*. “in press”

HUMPHREYS M.W., YADAV R.S., CAIRNS A.J., TURNER L.B., HUMPHREYS J. Y SKOT L. (2006) A changing climate for grassland research. *New Phytologist*, **169**, 9–26.

IBARRA A., ALBIZU I. Y BESGA G. (2000) La opción del silvopastorismo en el País Vasco. *Sustrati*, **56,** 40-43.

II IFN (1987) *Segundo Inventario Forestal Nacional 1986-1995*. Ministerio de Medio Ambiente. Madrid, España.

III IFN (1998) *Tercer Inventario Forestal Nacional 1997-2006. La transformación histórica del paisaje forestal en Galicia*. Ministerio de Medio Ambiente. Madrid, España.

KABATA-PENDÍAS A. Y PENDÍAS H. (2001) *Trace elements in soils and plants*. CRC Lewis publishers. Boca-Ratón. Florida, USA.

KELLAS J.D., BIRD P.R., CUMMING K.N., KEARNEY G.A. Y ASHTON A.K. (1995) Pasture production under a series of *Pinus radiata*-pasture agroforestry systems in South-west Victoria, Australia. *Australian Journal of Agricultural Research*, **46**, 1285–1297.

KNIGHT D., ELLIOT P.W. Y ANDERSON J.M. (1989) Effects of earthworms upon transformations and movement of nitrogen from organic matter applied to agricultural soils. En: Hansen J.A. y Henriksen K. (eds.) *Nitrogen in Organic Wastes Applied to Soils*, pp. 59-80. Academic Press. London, Reino Unido.

KNOWLES R.L. (1991) New Zealand experience with silvopastoral systems: a review. *Forest Ecology and Management*, **45**, 251-267.

KORMANIK PAUL P., SUNG SHI-JEAN S. Y ZARNOCH STANLEY J. (2005) Is seedling grading beneficial to artificial regeneration of northern red oaks?. En: Weigel D.R., Van Sambeek J.W. y Michler C.H. (eds.) *Ninth workshops on seedling physiology and growth problems in oak plantings*, pp. 18-20. Department of Agriculture, Forest Service, North Central Research Station. St. Paul.

KOWALENKO C.G. (2001) Assessment of Leco CNS-2000 analyzer for simultaneously measuring total carbon, nitrogen and sulphur in soil. *Communications in Soil Science and Plant Analysis*, **32**, 2065-2078.

KRAHULEC F., SKALOVA H., HERBEN T., HADINCOVA V., WILDOVA R. Y PECHACKOVA S. (2001) Vegetation changes following sheep grazing in abandoned mountain Meadows. *Applied Vegetation Science*, **4**, 97-102.

KREBS R., GUPTA S.K., FURRER G. Y SCHULIN R. (1998) Solubility and plant uptake of metals with and without liming of sludge-amended soils. *Journal of Environmental. Quality*, **27**, 18–23.

LALIBERTÉ E., COGLIASTRO A. Y BOUCHARD A. (2008) Spatiotemporal patterns in seedling emergente and early growth of two oak species direct-seeded on abandoned pastureland. *Annals of Forest Science*, **65**, 407-417.

LAMAND M. (1981) *Metabolis et bésoins en oligoelements des chèvres*. Proceedings of International Conference of Nutrition and Feeding Systems. Tours, Francia.

LEHMKUHLER J.W., FELTON E.E.D., SCHMIDT D.A., BADER K.J., GARRETT H.E. Y KERLEY M.S. (2003) Tree protection methods during the silvopastoral-system establishment in midwestern USA: Cattle performance and tree damage. *Agroforestry Systems*, **59**, 35-42.

LOEHR R.C., JEWELL W.J., NOVAK J.D., CLARKSON W.W. Y FRIEDMAN G.S. (1979) *Land application of wastes*. Van Nostrand Reinhold. New York, USA.

LÓPEZ-DÍAZ M.L., MOSQUERA-LOSADA M.R. Y RIGUEIRO-RODRÍGUEZ, A. (1999) Nitrogen and mineralization sewage sludge doses in grassland. *10th Nitrogen Workshop*. Vol 2, I.27. Dinamarca.

LÓPEZ-DÍAZ M.L. (2004) *Aplicación de lodos de depuradora urbana como fertilizante en un sistema silvopastoral*. Tesis doctoral. Universidad Santiago de Compostela. Lugo, España.

LÓPEZ-DÍAZ M.L., MOSQUERA-LOSADA M.R. Y RIGUEIRO-RODRÍGUEZ A. (2007) Lime, sewage sludge and mineral fertilization in a silvopastoral system developed in very acid soils. *Agroforestry Systems*, **70**, 91–101.

LÓPEZ-DÍAZ M.L., RIGUEIRO-RODRÍGUEZ A. Y MOSQUERA-LOSADA M.R. (2009) Influence of pasture botanical composition and fertilization treatments on tree growth. *Forest Ecology and Management*, **257**, 1363-1372.

LÓPEZ-MOSQUERA M.E. (1995) *Enmiendas. Corrección de la acidez: encalado. Gestión de la materia orgánica en explotaciones agrícolas. Monografía docente*. Universidad de Santiago de Compostela. Lugo, España.

LÓPEZ-MOSQUERA M.E., MOIRÓN C. Y SEOANE S. (2002) Changes in chemical properties of an acid soil after applications of dairy sludge. *Investigación agrícola serie Producción y Protección Vegetal*, **17**, 77-86.

LOUÉ A. (1988) *Los microelementos en la agricultura*. Editorial Mundi-Prensa. Madrid, España.

MAGURRAN A.E. (1988) *Ecological diversity and its measurement*. Princeton University Press. London, Reino Unido.

MARM (Ministerio de Medio Ambiente y Medio Rural y Marino) (2008) *Anuario de Estadística 2008.* Ministerio de Medio Ambiente y Medio Rural y Marino. Madrid, España.

McAdam J.H. y Hoppé G.M. (1996) Pasture production between trees in a silvopastoral agroforestry system. *Grassland Science in Europe*, **1**, 119-122.

McAdam J.H. (2000) Environmental impacts. In: Hislop M. y Claridge J. (eds.) *Agroforestry in the UK.* Forestry Commission Bulletin 122, pp. 83-90. HMSO. Londres, Reino Unido.

McAdam J.H. y Sibbald A. (2000) Grazing livestock management. En: Hislop M. y Claridge J. (eds.) *Agroforestry in the UK. Forestry Commission* **122**, 44-57. HMSO. Londres, Reino Unido.

McAdam J, Sibbald A., Teklehaimanot Z. y Eason W.R. (2007) Developing silvopastoral systems and their effects on diversity of fauna. *Agroforestry Systems,* **70**, 81–89.

McAdam J. y McEvoy P.M. (2008) The potential for silvopastoralism to enhance biodiversity on grassland farms in Ireland. En: Rigueiro-Rodríguez A., McAdam J.H. y Mosquera-Losada M.R. (eds.) *Agroforestry in Europe, Advances in Agroforestry,* **6**, 343–358. Springer. Los Países Bajos.

McEvoy P.M. (2004) *Livestock management in grazed woodlands*. Ph.D. thesis. Queen´s. University. Belfast, Reino Unido.

McGrath S.P., Chaudri A.M. y Giller K.E. (1995) Long term effects of metals in sewage sludge on soils, microorganism and plants. *Journal of Industrial Microbiology*, **14**: 94-104.

Mehlich A. (1985) Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant. *Communications in Soil Science and Plant Analysis*, **15**, 1409-1416.

MIAO-MIAO H., GUANG-MING T., XING-QING L., YI-TONG Y., JIAN-YANG W. Y GEN-DI Z. (2007) Effects of two sludge application on fractionation and phytotoxicity of Zn and Cu in soil. *Journal of Environmental Sciences*, **19**, 1482-1490.

MMA (Ministerio de Medio Ambiente) (1999) *Estrategia Forestal Española.* http://www.mma.es/secciones/biodiversidad/montes_politica_forestal/estrategia_monte/pdf/efe_1.pdf (consultado el 18 de octubre del 2010).

MMA (Ministerio de Medio Ambiente) (2006) *Plan integrado de residuos 2007_2015.* http://www.mma.es/secciones/calidad_contaminacion/residuos/planificacion_residuos/pdf/borradorpnir_anexo5.pdf (consultado el 16 de marzo del 2009).

MOLINA RODRÍGUEZ F., DANS DEL VALLE F., FERNÁNDEZ DE ANA MAGÁN F.J., MOLINA MARTÍNEZ B. Y VERDE FIGUEIRAS M.C. (2004) *Guía : tratamentos silvícolas para a producción de madeira. Carballo americano.* Asociación Forestal de Galicia. Santiago de Compostela, España.

MOMBIELA F.A. Y MATEO M.E. (1984) Necesidades de cal para praderas en terrenos a monte.1. Su relación con el Al cambiable en suelos sobre granitos y pizarras de Galicia. *Anales del INIA Serie Agrícola,* **25**, 129-143.

MORENO-MARCOS G. Y OBRADOR-OLÁN J.J. (2007) Effects of trees and understorey management on soil fertility and nutritional status of holms oaks in Spanish dehesas. *Nutrient Cycling in Agroecosystems*, **78**, 253-264.

MORENO-MARCOS G. Y PULIDO F. (2008) The function, management and persistence of Dehesas. In: Rigueiro-Rodríguez A., McAdam J.H. y Mosquera-Losada M.R. (eds.) *Agroforestry in Europe, Advances in Agroforestry,* **6**, 127–160. Springer. Los Países Bajos.

MOSQUERA A. Y MOMBIELA F.A. (1986). Comparison of three methods for determination of soil Al in an unbuffered salt-extract. *Communications in Soil Science and Plant Analysis*, **17**, 30–35.

MOSQUERA A. (1992) Transformación de terrenos de monte gallegos en praderas permanentes. En: Barrecheguren M.A., Cabrero M., Collar J., Moreno J., Santaolalla G. y Zea J. (eds.) *“100 anos de Investigación Agraria”*, pp. 375-382. Xunta de Galicia. Santiago de Compostela, España.

MOSQUERA-LOSADA M.R. Y GONZÁLEZ-RODRÍGUEZ A. (1997) Uso de nitrógeno y potasio para incrementar la producción y persistencia de trébol blanco. Revista de la Sociedad Española para el Estudio de los Pastos, **27**, 207-208.

MOSQUERA-LOSADA M.R. Y GONZÁLEZ-RODRÍGUEZ A. (1999) Pasture production in Northern Spain dairy system. *New Zealand Journal of Agriculture Research*, **42**, 125–132.

MOSQUERA-LOSADA M.R., GONZÁLEZ-RODRÍGUEZ A. Y RIGUEIRO-RODRÍGUEZ A. (1999) *Ecología y manejo de praderas*. Ministerio de Agricultura, Pesca y Alimentación. Madrid, España.

MOSQUERA-LOSADA M.R., RIGUEIRO-RODRÍGUEZ A. Y LÓPEZ-DÍAZ M.L. (2001) Sewage sludge fertilization of a silvopastoral system with pines in northwestern Spain. *Agroforestry Systems*, **53**, 1–10.

MOSQUERA-LOSADA M.R. Y RIGUEIRO-RODRÍGUEZ A. (2005) La biodiversità nei pascoli. En: Pardini A. (eds.) *Gestione dei pascoli e dei territory pascolivi*, pp. 179-188. Aracne Editrice, Roma, Italia.

MOSQUERA-LOSADA M.R., FERNÁNDEZ-NÚÑEZ E. Y RIGUEIRO-RODRÍGUEZ A. (2006) Pasture, tree and soil evolution in silvopastoral systems of Atlantic Europe. *Forest Ecology and Management*, **232**, 135-145.

MOSQUERA-LOSADA M.R. Y RIGUEIRO-RODRÍGUEZ A. (2006) Efecto del empleo de diferentes dosis de lodo procedentes de industria procesadora de algas como fertilizante en plantaciones de *Eucalyptus globulus* Labill. y *Pinus pinaster* Aiton. *Boletín del CIDEU*, **3**, 181-187.

MOSQUERA-LOSADA M.R., MCADAM J.H., ROMERO-FRANCO R., SANTIAGO-FREIJANES J.J Y RIGUEIRO-RODRÍGUEZ A. (2008a) Definitions and componenets of agroforestry practices in Europe. En: Rigueiro-Rodríguez A., McAdam J.H. y Mosquera-Losada M.R. (eds.) *Agroforestry in Europe, Advances in Agroforestry,* **6**, 3–20. Springer. Los Países Bajos.

MOSQUERA-LOSADA M.R., NAIR V.N., NAIR P.K.R., MORÁN D., SANTIAGO-FREIJANES J.J. Y RIGUEIRO-RODRÍGUEZ A. (2008b) *Soil Phosphorous storage in a poplar+silvopastoral system fertilized with sewage sludge in Northwestern Spain.* ASA-CSSA-SSSA, 2010 International Annual Meeting. http://acs.confex.com/crops/2008am/webprogram/Paper41839.html (consultado el 27 de enero del 2010).

MOSQUERA-LOSADA M.R., RODRÍGUEZ-BARREIRA S., LÓPEZ-DÍAZ M.L., FERNÁNDEZ-NÚÑEZ E. Y RIGUEIRO-RODRÍGUEZ A. (2009a) Biodiversity and silvopastoral systems use change in very acid soils. *Agriculture, ecosystems and environment*, **131,** 315-324.

MOSQUERA-LOSADA M. R., LÓPEZ-DÍAZ M. L. Y RIGUEIRO-RODRÍGUEZ A. (2009b) Zinc and copper availability in herbage and soil of a *Pinus radiata* silvopastoral system in Northwest España after sewage-sludge and lime application. *Journal of Plant Nutrition and Soil Science*, **172**, 843-850.b

MOSQUERA-LOSADA M.R., FERREIRO-DOMÍNGUEZ N. Y RIGUEIRO-RODRÍGUEZ A. (2010a) Fertilization in pastoral and *Pinus radiata* D. Don silvopastoral systems developed in forest and agronomic soils of Northwest Spain. *Agriculture, ecosystems and environment*, **139**, 618-628.

MOSQUERA-LOSADA M.R., MUÑOZ-FERREIRO N. Y RIGUEIRO-RODRÍGUEZ A. (2010b) Agronomic characterization of different types of sewage sludge: policy implications. *Waste Management*, **30**, 492-503.

MOSQUERA-LOSADA M.R., MORÁN-ZULOAGA D. Y RIGUEIRO-RODRÍGUEZ A., (2010c) Effects of lime and sewage sludge on soil, pasture production and tree growth in a six year old *Populus canadensis* Moench silvopastoral system. *Journal of Plant Nutrition and Soil Science.* “in press”.

MUYS B., NACHTERGALE L., LUST N., MERCKX R. Y GRANVAL P. (2004) Medium term evaluation of a forest soil restoration trial combining tree species change, fertilization and earthworm introduction. *Pedobiologia*, **47**, 772-783.

MWASE W.F., SAVILL P.S. Y HEMERY G. (2008) Genetic parameterestimates for growth and form traits in common ash (*Fraxinus excelsior* L.) in a breeding seedling orchard at Little Wittenham in England. *New Forest,* **36**, 225-238.

NAIR P.K.R. (1985) Classification of agroforestry systems. *Agroforestry Systems*, **3,** 97-128.

NAIR P.K.R. (1990) Classification of Agroforestry Systems. En: MacDicken K.G. y Vergara N.T. (eds.) *Agroforestry: Classification and Management*, pp. 31-57. Jhon wiley and Sons. New York, USA.

NAIR P.K.R. (1991) State of the art of agroforestry systems. *Forest Ecology and Management*, **45**, 5-29.

NAIR P.K.R (1993) *An introduction to agroforestry*. Kluwer Academic Publishers. Dordrecht, Holanda.

NAIR V.D. Y GRAETZ D.A. (2002) Phosphorus Saturation in Spodosols Impacted by Manure. *Journal Environment Quality*, **31**, 1279-1285.

NAIR P.K.R. Y NAIR V. (2002) Carbon sequestration in agroforestry systems. *17th World Congress Soil Science*, **989**, 1-13.

NAIR V.D. Y GRAETZ D.A. (2004) Agroforestry as an approach to minimizing nutrient loss from heavily fertilized soils: the Florida experience. *Agroforestry Systems*, **61**, 269–279.

NAIR V.D., NAIR P.K.R., KALMBACHER R.S. Y EZENWA I.V. (2007) Reducing nutrient loss from farms through silvopastoral practices in coarse-textured soils of Florida, USA. *Ecological Engineering*, **29**, 192-199.

NAIR P.K.R., GORDON A.M. Y MOSQUERA-LOSADA M.R. (2008) Agroforestry. En: Jorgensen S.E. y Fath B.D. (eds.) *Ecological Engineering, Encyclopedia of Ecology*, pp. 101–110. Elsevier. Oxford, Reino Unido.

NAVARRO-PEDREÑO J., MORAL-HERRERO R., GÓMEZ L. Y MATAIX-BENEYTO J. (1995) *Residuos orgánicos y agricultura*. Universidad de Alicante. Alicante, España.

NIEDER R., BENBI D.K. Y ISERMANN K. (2003) Soil organic matter dynamics. En: Benbi D.K. y Nieder R. (eds.) *Handbook of processes and modelling in the soil-plant system*, pp. 346-408. Haworth Press. Londres.

NILSSON S.I. (2003): Soil acidification. En: Benbi D.K. y Nieder R. (eds.) *Handbook of processes and modelling in the soil-plant system,* pp. 177-199. Haworth Press. Londres.

NRC (National Research Council) (1985) *Nutrient requirements of sheep*. National Academic Press. Washington, USA.

NRC (National Research Council) (1989) *Nutrient requirements of horses*. National Academic Press. Washington, USA.

NUZZO V.A. (1986) Extent and status of Midwest oak savanna: presettlement and 1985. *Natural Areas Journal*, **6**, 6-36.

OMIL B., MOSQUERA-LOSADA M.R. Y MERINO A. (2007) Responses of a Non–N-Limited Forest Plantation to the Application of Alkaline-Stabilized Dewatered Dairy Factory Sludge. *Journal of Environmental Quality*, **36**, 1765–1774.

O'RIORDAN E.G., DODD V.A., FLEMING G.A. Y TUNNEY H. (1987) The fertilizer nutrient value of an anaerobically digested sewage sludge under grassland field conditions. *Irish Journal of Agricultural of Food Research*, **26**, 199–211.

PARDINI A. (2008) Agroforestry systems in Italy: Traditions towards modern management. En: Rigueiro-Rodríguez A., McAdam J.H. y Mosquera-Losada M.R. (eds.) *Agroforestry in Europe, Advances in Agroforestry,* **6**, 255–268. Springer. Los Países Bajos.

PARÉ K., CHANTIGNI M.H., CAREY K., JOHNSTON W.J. Y DIONNE J. (2006) Nitrogen uptake and leaching under annual bluegrass ecotypes and bentgrass species: A lysimeter experiment. *Crop Science*, **46**(2), 847-853.

PÉREZ-CRUZADO C., MANSILLA SALINERO P., SOLLA-GULLÓN F., ALVAREZ-ALVAREZ P., RODRÍGUEZ-SOALLEIRO R. Y MERINO A. (2006) Una red de zonas piloto para evaluar y mejorar a nivel regional los indicadores de gestión sostenible de los bosques de la fachada atlántica del sur de Europa. Proyecto FORSEE. Informe final, parte 4 región de Galicia.

PICCOLO A., ZACCHEO P. Y GENEVINI P.G. (1992) Chemical characterization of humic substances extracted from organic-waste-amendment soil. *Bioresearch Technology*, **40**, 275-282.

POMARES F. Y CANET R. (2001) Residuos orgánicos utilizables en agricultura: origen, composición y características. En: Boixadera J. y Rosa-Teira M. (eds.) *Aplicación agrícola de residuos orgánicos,* pp. 195-213. Universidad de Lleida. Lleida, España.

PONTES L.S., CARRÈRE P., ANDUEZA D., LOUAULT F. Y SOUSSANA J.F. (2007) Seasonal productivity and nutritive value of temperate grasses found in semi-natural pasture in Europe: response to cutting frequently and N supply. *Grass and Forage Science*, **62**, 485-496.

PORTA J, LÓPEZ-ACEVEDO M Y ROQUERO C. (2003) *Edafología para Agricultura y Medio Ambiente.* Ediciones Mundi Prensa. Madrid.

PRASAD R. Y POWER J.F. (1997) *Soil fertility management for sustainable agriculture*. CRC Lewis publishers. Boca Ratón, Florida, USA.

RENOU-WILSON F., KEANE M. Y FARRELL E.P. (2008) Establishing oak woodland on cutaway peatlands: Effects of soil preparation and fertilization. *Forest Ecology and Management,* **255**, 728-737.

RIGUEIRO-RODRÍGUEZ A. (1992) Pastoreo controlado en los bosques gallegos. *El Campo*, **124**, 29-33.

RIGUEIRO-RODRÍGUEZ A., SILVA-PANDO F.J., RODRÍGUEZ-SOALLEIRO R., CASTILLÓN-PALOMEQUE P.A., ÁLVAREZ-ÁLVAREZ P., MOSQUERA-LOSADA M.R., ROMERO-FRANCO R. Y GONZÁLEZ-HERNÁNDEZ M.P. (1997) *Manual de sistemas silvopasotrales*. Columella. Escuela Politécnica Superior de Lugo. http://www.agrobyte.com/agrobyte/publicaciones/silvo/indice.html (consultado el 17 de marzo del 2010).

RIGUEIRO-RODRÍGUEZ A. (2000) Sistemas silvopastorales en la Iberia Atlántica. *XL Reunión Científica de la Sociedad Española para el Estudio de los Pastos*, **7-13**, 649-658. Bragança-A Coruña-Lugo, España.

RIGUEIRO-RODRÍGUEZ A., MOSQUERA-LOSADA M.R. Y GATICA-TRABANINI E. (2000) Pasture production and tree growth in a young pine plantation fertilized with inorganic fertilizers and milk sewage in Northwestern España. *Agroforestry Systems*, **48**, 245–256.

RIGUEIRO-RODRÍGUEZ A., MOSQUERA-LOSADA M.R., ROMERO-FRANCO R., GONZÁLEZ-HERNÁNDEZ M.P. Y VILLARINO-URTIAGA J.J. (2005a) Silvopastoral systems as a forest fire prevention technique. En: Mosquera-Losada M.R., McAdam J. y Rigueiro-Rodríguez A. (eds.) *Silvopastoralism and sustainable land management,* pp. 380–387. CABI Publishing. Wallingford, Reino Unido.

RIGUEIRO RODRÍGUEZ A., MOSQUERA LOSADA M. R., ROMERO FRANCO R., GONZÁLEZ HERNÁNDEZ M. P. VILLARINO URTIAGA J. J. Y LÓPEZ L. (2005b) 25 años de investigación en Galicia sobre sistemas silvopastorales en prevención de incendios forestales. II Conferencia Internacional sobre Estrategias de Prevención de Incendios en el Sur de Europa, Barcelona.

RIGUEIRO-RODRÍGUEZ A., FERNÁNDEZ-NÚÑEZ E. Y MOSQUERA-LOSADA M.R. (2005c) Effect of tree density and species on botanical evolution diversity in a five years experiment in a silvopastoral system. *Grassland Science in Europe*, **10**, 384-387.

RIGUEIRO-RODRÍGUEZ A., MOSQUERA-LOSADA M.R. Y LÓPEZ-DÍAZ M.L. (2006) Mineral concentrations in herbage and soil in a *Pinus radiata* silvopastoral system in north-west Spain after sewage sludge and lime application. *Grass and Forage Science*, **62**, 208-224.

RIGUEIRO-RODRÍGUEZ A.; MOSQUERA-LOSADA M.R. Y LÓPEZ-DÍAZ M.L. (2007) Mineral concentrations in herbage and soil in a *Pinus radiata* silvopastoral system in north-west Spain after sewage sludge and lime application. *Grass and Forage Science,* **62**, 208-224.

RIGUEIRO-RODRÍGUEZ A., FERNÁNDEZ-NÚÑEZ E., GONZÁLEZ-HERNÁNDEZ P., MCADAM J.H. Y MOSQUERA-LOSADA M.R. (2008a) Agroforestry systems in Europe: productive, ecological and social perspectives. En: Rigueiro-Rodríguez A., McAdam J.H. and Mosquera-Losada M.R. (eds.) *Agroforestry in Europe, Advances in Agroforestry,* **6**, 43–66. Springer. Los Países Bajos.

RIGUEIRO-RODRÍGUEZ A., MOSQUERA-LOSADA M.R., LÓPEZ-DÍAZ M.L. (2008b) Effect of sewage sludge and liming on productivity during the establishment of a silvopastoral system in north-west Spain. *New Zealand Journal of Agricultural Research*, **51**, 199-207.

RIGUEIRO-RODRÍGUEZ A., SANTIAGO-FREIJANES J.J., FERNÁNDEZ-NÚÑEZ E. Y MOSQUERA-LOSADA M.R. (2009) Sistemas biológicos de prevención de incendios forestales: los sistemas silvopastorales. En: Vélez-Muñoz R. (eds.) *La defensa contra incendios forestales: fundamentos y experiencias*, 2nd edn, pp. 447–451. McGraw Hill. Madrid, España.

RIGUEIRO-RODRÍGUEZ A., CASTRO S. Y MOSQUERA-LOSADA M.R. (2010a) Effects of dose and period of sewage sludge application on soil, tree and pasture components in a *Pinus radiata* D. Don silvopastoral system. *Agroforesry Syst*ems, **79**, 237-247.

RIGUEIRO-RODRÍGUEZ A., FERREIRO-DOMÍNGUEZ N. Y MOSQUERA-LOSADA M.R. (2010b) The effects of fertilization with anaerobic, composted and pelletized sewage sludge on soil, tree growth, pasture production and biodiversity in a silvopastoral system under ash (*Fraxinus excelsior* L.). *Grass and Forage Science*, **65**, 248-259.

RIGUEIRO-RODRÍGUEZ A., ROIS-DÍAZ M. Y MOSQUERA-LOSADA M.R. (2010c) Integrating Silvopastoralism and Biodiversity Conservation. En: Lichtfouse E. (eds.) Biodiversity, Biofuels, Agroforestry and Conservation Agriculture, **5**, 359-373. Springer. Los Países Bajos.

RODRÍGUEZ-BARREIRA S. (2007) *Crecimiento del arbolado, producción de pasto y efectos edáficos en sistemas silvopastorales fertilizados con lodos de depuradora urbana, efecto residual*. Tesis doctoral. Universidad Santiago de Compostela. Lugo, España.

ROIS-DÍAZ M., MOSQUERA-LOSADA M.R. Y RIGUEIRO-RODRÍGUEZ A. (2006) *Biodiversity indicators on silvopastoralism across Europe*. European Forest Institute Technical Report 21 EFI, Joensuu, Finland. http://www.efi.fi/publications/technical-reports/ (consultado el 25 de agosto del 2010).

ROSSWALL T. (1982) Microbiological regulation of the biogeochemical nitrogen cycle. *Plant and Soil*, **67**, 15-34.

ROZADOS-LORENZO M.J., GONZÁLEZ-HERNÁNDEZ M.P. Y SILVA-PANDO F.J. (2007) Pasture production under different tree species and densities in an Atlantic silvopastoral system. *Agroforestry Systems*, **70**, 53-62.

SÁNCHEZ-RODRÍGUEZ F. (2000) *Growth and productivity of Pinus radiata in Galicia.* Ph. D. thesis. Universidad de Santiago de Compostela. Santiago de Compostela, España.

SÁNCHEZ-RODRÍGUEZ F., RODRÍGUEZ-SOALLEIRO R., ESPAÑOL E., LÓPEZ C.A. Y MERINO A. (2002) Influence of edaphic factors and tree nutritive status on the productivity of *Pinus radiata* D. Don plantations in northwestern Spain. *Forest Ecology and Management*, **171**, 181-189.

SANDER I.L. (1990) *Quercus rubra* L. Northern red oak. In: Burns R.M. y Honkala B.H. (eds.) *Silvics of North America. Hardwoods,* **2**, 727-733. U.S. Department of Agriculture Forest Service. Washington, D.C. http://www.na.fs.fed.us/pubs/silvics_manual/volume_2/quercus/rubra.htm (consultado el 7 de octubre del 2010).

SANDERS J.R., MCGRATH S.P. Y ADAMS T.M.M. (1986) Zinc, Copper and Nickel concentrations in ryegrass grown on sewage sludge-contaminated soils of different pH. *Journal of the Science of Food and Agriculture*, **37**, 961-968.

SAS (2001) *User's Guide: Statistics.* SAS Institute Inc. Cary, NC.

SAWYER J.E., MALLARINO A.P., KILLORN R. Y BARNHART S.K. (2008). *A general guide for crop nutrient and limestone recommendations in Iowa. Iowa State University extension service*. http://www.extension.iastate.edu/Publications/PM1688.pdf (consultado el 27 de enero del 2010).

SHARROW S.H. (1999) Silvopastoralism: competition and facilitation between trees, livestock and improved grass-clover pastures on temperate lands. En: Buck E.D.E.L., Lassoie J.P. y Fernández E.C.M. (eds.) *Agroforestry in sustainable agricultural systems,* pp. 43-66. CRC Lewis publishers. Boca Ratón, Florida, USA.

SIBBALD A.R. (1996) Silvopastoral systems on temperate sown pastures: a personal perspectiva. En: Etiene M. (eds.) *Western European Silvopastoral Systems*, pp. 23-26. INRA editions. París, Francia.

SIBBALD A.R., EASON W.R., MCADAM J.H. Y HISLOP A.M. (2001) The establishment phase of a silvopastoral national network experiment in the United Kingdom. *Agroforestry System*, **39**, 39-53.

SILVA-PANDO F.J. (1988) Aprovechamientos Silvopastorales. *Actas Curs d'Estiu 'Estudis Pirinencs*, **4**, 1-21. Seo de Urgel, Barcelona, España.

SILVA-PANDO F.J. Y ROZADOS-LORENZO M.J. (2002) Agroselvicultura, Agroforestería, Prácticas forestales, Uso múltiple: Una definición y un concepto. En: Silva-Pando F.J., Mosquera-Losada M.R. y Puerto-Arribas G. (eds.) *Actas de la I reunión sobre sistemas agroforestales y I reunión sobre gestión de espacios naturales*, **14**, 9-21. Sociedad española de ciencias forestales. Lugo-Pontevedra, España.

SIMON J.C. Y LE CORRE L. (1992) Fertilization des cultures annuelles et lessivage de l´azote nitrique. *Fourrages,* **129**, 3-10.

SIMPSON K. (1986) *Abonos y estiércoles*. Editorial Acribia. Zaragoza, España.

SLATTERY W.J., CONYERS M.K. Y AITKEN R.I. (1999) Soil pH, aluminium, manganese and lime requirement. En: Peverill K.I., Sparrow L.A. and Reuter D.J. (eds.) *Soil analysis: an interpretation manual,* pp. 103-128. CSIRO Publishing. Collingwood, Victoria, Australia.

SMITH S.R. (1996) *Agricultural recycling of sewage sludge and the environment*, pp. 207-236. CAB International. Wallingford, Reino Unido.

SPEIR T.W., HORSWELL J., VAN SCHAIK A.P., MCLAREN R.G. Y FIETJE G. (2004) Composted biosolids enhance fertility of a sandy loam soil under dairy pasture. *Biology and Fertility of Soils*, **40**, 349-358.

THOMPSON K., HODGSON J.G., GRIME J.P. Y BURKE M.J.W. (2001) Plant traits and temporal scale: evidence from a 5-year invasion experiment using native species. *Journal of Ecology*, **89**, 1054-1060.

TIFFANY M.E., MCDOWELL L.R., O`CONNOR G.A., NGUYEN H., MARTIN F.G., WILKINSON N.S. Y CARDOSO E.C. (2000a) Effects of pasture-applied biosolids on forage and soil concentrations over a grazing season in North Florida. I. Macrominerals, crude protein and In Vitro digestibility. *Communications in Soil Science and Plant Analysis*, **31**, 201-213.

TIFFANY M.E., MCDOWELL L.R., O´CONNOR G.A., NGUYEN H., MARTIN F.G., WILKINSON N.S. Y CARDOSO E.C. (2000b) Effects of pasture applied biosolids on forage and soil concentrations over a grazing season in North Florida. II. Microminerals. *Communications in Soil Science and Plant Analysis,* **3**, 215-227.

TSADILAS C.D., MATSI T.H., BARBAYIANNIS N. Y DIMOYIANNIS D. (1995) Influence of sewage sludge application on soil properties and on the distribution an availability of heavy metal fractions. *Communications in Soil Science and Plant Analysis,* **26**, 2603-2619.

UE (Unión Europea) (1980) *Council Directive of 15 July 1980 relating to the quality of water intended for human consumption (80/778/EEC). Official J. of the Europ. Comm. No. L 229/11-29.* European Commission. Brussels, Belgium

UE (Unión Europea) (1986) *DOCE nº L 181 04/07/1986. Council Directive 86/278/CEE of 12 June 1986 on the protection of the environment and, in particular of the soil, when sewage sludge is used in agriculture (86/278/EEC).* EC Official Journal L 181 04/07/1986. European Commission. Brussels, Belgium. http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:31986L0278:ES:HTML (consultado el 15 de diciembre del 2009).

UE (Unión Europea) (1991) *DOCE nº L 135 30/05/1991. Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment.* http://aragon.ugt.org/mambiente/Normativa/pdf/aguas/aguas_eu_91–271.pdf (consultado el 15 de diciembre del 2009).

UE (Unión Europea) (1992) *Council Regulation (CEE) Nº 2080/92 of 30 June 1992 instituting a Community aid scheme for forestry measures in agriculture*. http://www.legaltext.ee/text/en/T30207.htm (consultado el 4 de octubre del 2010).

UE (Unión Europea) (1999) *Council Regulation (CEE) N^o 1257/1999 of 17 May 1999 on support for rural development from the European Agricultural Guidance and Guarantee Fund (EAGGF) and amending and repealing certain Regulations*. http://www.esf.gov.sk/documents/1999-1257.pdf (consultado el 4 de octubre del 2010).

UE (Unión Europea) (2002) *Decission nº 1600/2002/EC of the European Parliament and of the Council of 22 July 2002*. http://biodiversity-chm.eea.europa.eu/convention/F1117799202/F1122843896 (consultado el 30 de octubre del 2009).

UE (Unión Europea) (2005) *Council regulation (EC) nº 1698/2005 of Septembre 2005 on support for rural development by the European Agricultural Fund for Rural Development*.http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2005:277:0001:0040:EN PDF (consultado el 15 de diciembre del 2009).

UE (Unión Europea) (2009) *Report on implementation of forestry measures under the rural development regulation 1698/2005 for the period 2007-2013*. http://ec.europa.eu/agriculture/fore/publi/report_exsum_en.pdf (consultado el 11 de noviembre del 2010).

UN (1992) *Agenda 21*. United Nations Conference on Environment & Development. Rio de Janeiro, Brasil.

VAN NOORDWIJK M., LAWSON G., SOUMARÉ A., GROOT J.J.R. Y HAIRIAH K. (1996) Root distribution of trees and crops: competition and/or complementarity. En: Ong C.K. y Huxley P. (eds.) *Tree-Crop interations: a physiological approach*, pp. 319-364. CAB International, Wallingford, Reino Unido.

VIRGEL-MENTXAKA S. (2002) *Efecto de la aplicación de escoria siderúrgica LD y fertilizante NPK en sistemas agrarios*. Tesis doctoral. Universidad del País Vasco.

VIVEKANANDAN M., BROWN J.R., WILLIAMS J., CLEVENGER T., BELYEA R. Y TUMBLESON M. E. (1991) Tolerance of forage legumes to lime-estabilized sludge. *Communications in Soil Science and Plant Analysis*, **22**, 449-463.

WAGNER R.G., LITTLE K.M., RICHARDSON B. Y MCNABB K. (2006). The role of vegetation management for enhancing productivity of the world´s forests. *Forestry*, **79**, 57-79.

WARMAN P.R. Y TERMEER W.C. (2005) Evaluation of sewage sludge, septic waste and sludge compost applications to corn and forage: yields and N, P and K content of crops and soils. *Bioresource Technology*, **96**, 955-961.

WEBER-BLASCHKE G. Y REHFUESS K.E. (2002) Correction of Al toxicity with European ash (*Fraxinus excelsior* L.) growing on acid soils by fertilization with Ca and Mg carbonate and sulfate in pot experiments. *Forest Ecology and Management*, **167**, 173-183.

WHITEHEAD D.C. (1995) *Grassland nitrogen*. CAB International. Wallingford, Reino Unido.

WHITEHEAD D.C. (2000) *Nutrient elements in grassland. Soil-Plant-Animal Relationships*. CAB International. Wallingford, Reino Unido.

WIERSUM K.F. (1981) Outline of the agroforestry concept. En: Wiersum K.F. (eds.) *Viewpoints on Agroforestry*, pp. 1-23. Hinkeloord, Agricultural University Wagenigen. Wagenigen, Los Países Bajos.

WILLIAMS P.A., GORDON A.M., GARRETT H.E. Y BUCK, L. (1997) Agroforestry in North America and its role in farming systems. En: Gordon A.M. y S.M. Newman (eds.) *Temperate Agroforestry Systems,* pp. 9-84. CABI Publishing. Wallingford, Reino Unido.

WOLSTENHOLME R., DUTCH J., MONAT A., BAYES C.D. Y TAYLOR C.M.A. (1992) *A manual of good practice for the use of sewage sludge in forestry*. Forestry Commission Bulletin 107. HMSO. Londres, Reino Unido.

XUNTA DE GALICIA (2009) *Anuario de Estatística Agraria*. Dirección Xeral de Investigación, Tecnoloxía e Formación Agroforestal da Consellería do Medio Rural. Santiago de Compostela, España.

YUAN G. (2009) Copper, zinc, and nickel in soil solution affected by biosolids amendment and soil management. *Australian Journal of Soil Research*, **47**, 305-310.

ZAS R. Y ALONSO M. (2002) Understorey vegetation as indicators of soil characteristics in northwest España. *Forest Ecology and Management*, **171**, 101-111.

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The effects of fertilization with anaerobic, composted and pelletized sewage sludge on soil, tree growth, pasture production and biodiversity in a silvopastoral system under ash (*Fraxinus excelsior* L.)

A. A. Rigueiro-Rodríguez, N. Ferreiro-Domínguez and M. R. Mosquera-Losada
Crop Production Department, High Polytechnic School, University of Santiago de Compostela, Santiago de Compostela, Spain

Abstract

In silvopastoral systems, tree growth and the composition and productivity of pasture can be modified by management practices such as initial fertilization when tree seedlings are more sensitive to understorey competition. The aim of this study was to compare the effects of fertilization with different types of sewage sludge (anaerobic sludge, composted sludge and pelletized sludge), using different rates of incorporation and mineralization with traditional treatments (with and without mineral fertilizers) on the growth of newly established ash (*Fraxinus excelsior* L.) and on pasture development, to obtain sustainable management practices that enhance the growth of both components. Soil characteristics, tree growth, sward composition and pasture development were modified differently according to the type of sewage sludge used, and for similar total nitrogen inputs. Anaerobic sludge had a higher initial effect on both tree and pasture productivity. Pelletized sludge sustained better tree and pasture production. Composted sludge was found to be the most appropriate treatment for improving soil characteristics over the long term on sandy soils. It was concluded that pelletized sludge should be promoted because it enhances productivity, allows for better nutrient recovery and is less costly to store and apply compared with anaerobic sludge and composted sludge. No toxic concentrations of Zn or Cu were found in plants or in the soil despite higher concentrations being present in the applied sludge than in soil.

Correspondence to: M.R. Mosquera-Losada, Crop Production Department, High Polytechnic School, University of Santiago de Compostela, Lugo, Spain.
E-mail: mrosa.mosquera.losada@usc.es

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Introduction

Silvopastoral systems are a type of agroforestry system in which trees and grazing are combined, resulting in benefits for wood production (with long-term economic returns) and livestock (with short-term economic returns) (Rigueiro-Rodríguez *et al.*, 2005). Agroforestry systems are sustainable land management techniques that are promoted by the EU [Council Regulation 1698/2005 (EU, 2005)]. Agroforestry systems have also received favourable evaluations from farmers in Europe; for example, a sample of 214 farmers interviewed in fourteen areas of Europe, half indicated they would attempt silvo-arable agroforestry on their farm (Graves *et al.*, 2008).

In the early stages of the development of silvopastoral systems established through afforestation, competition between trees and pasture can be high (Nair and Graetz, 2004; Rigueiro-Rodríguez *et al.*, 2008a). In newly established systems, adequate management should aim to optimize silvopastoral outputs through the selection of trees and pasture species as well as through fertilization inputs. Shrub development should be avoided through frequent clearing in order to avoid shrub–tree competition and fire risk (Rigueiro *et al.*, 2009). Moreover, in early stages of establishment pasture–tree competition should be avoided, either through mulching or by providing forage for grazing animals (Wagner *et al.*, 2006).

The European Ash (*Fraxinus excelsior* L.) is a widely distributed tree species that integrates well into silvopastoral systems in the Atlantic biogeographic region of Europe (McAdam and Hoppé, 1996; McAdam and Sibbald, 2000). European ash trees possess apical

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dominance and deep roots that avoid root competition between trees and pasture; these roots enhance nutrient recovery from the deep soil layer up into the system, making them compatible with silvopastoral systems. Moreover, as a deciduous species, it allows better light penetration than conifers during the autumn and early spring, and provides shading during the summer, thereby reducing evapotranspiration and thus enhancing pasture production when compared with pasture under conifers or on open pasture sites. The open crowns of ash trees also allow light to reach the pasture surface, and they do not intercept much rain (McEvoy, 2004). The most appropriate pasture species for silvopastoral system implementation are those that are well adapted to shading, such as *Dactylis glomerata* L. (Mosquera-Losada *et al.*, 2001, 2006). However, legume species also enhance pasture quality and production as well as tree growth (Whitehead, 1995; López-Díaz *et al.*, 2009).

In Galicia, the natural soils have low fertility due to their acidity (Zas and Alonso, 2002). This acidity implies a high concentration of saturated aluminium in the exchange complex and low cation and phosphorus availability (Prasad and Power, 1997; Rigueiro-Rodríguez *et al.*, 2007). The EU promotes the use of sewage sludge as a fertilizer because of its specific organic matter and content of macronutrients, particularly N (MMA, 2006). However, a higher concentration of heavy metals (mainly Zn and Cu) in sewage sludge than is normally found in soil, as well as long-term sludge loadings, limits the use of sewage sludge according to the Spanish (R.D. 1310/1990; BOE, 1990) and European Directives (86/278/CEE; DOCE, 1986) in order to prevent harmful effects on soil and vegetation, and on animal and human health.

Anaerobic digestion and composting are two sewage sludge stabilization processes which are promoted by the EU (EEA, 2000) before the sludge is used as a fertilizer in agriculture. However, sewage sludge stabilized by these processes contains a high proportion of water. Pelletized sewage sludge is derived from the thermic treatment of anaerobic digested sewage sludge in order to reduce water content to 2%, which consequently reduces storage, transport and spreading costs compared with anaerobic or composted sludge (COM) (Mosquera-Losada *et al.*, 2009). Each type of sewage sludge has different characteristics, nutrient contents (Mosquera-Losada *et al.*, 2009) and rates of incorporation into the soil according to the treatment stabilization (EPA, 1994) and the specific local climate.

The aim of this study was to evaluate the effects of municipal sewage sludge that has been stabilized by either anaerobic digestion, composting or pelletization, on changes in soil chemical properties, tree growth, understorey production, biodiversity in terms of sward botanical composition and quality of pasture compared with treatments (receiving either mineral fertilizers or no fertilization) in a silvopastoral system under *F. excelsior* L. during a 4-year period.

Materials and methods

Characteristics of the study site

The experiment was conducted in A Pastoriza (Lugo, Galicia, NW Spain, European Atlantic Biogeographic Region; t 43° 14′ N, 7° 21′ W; 550 m a.s.l.). Figure 1 shows the mean monthly precipitation and temperatures for 2005, 2006, 2007 and 2008 and the previous 30-year mean. Total annual rainfall was 824·3, 1157·5, 734·4 and 1222·3 mm in 2005, 2006, 2007 and 2008 respectively. Very low precipitation was observed in 2005 and 2007 compared with the 30-year mean. There were periods of drought from April to July 2006 and from June to October 2008, which would have been unfavourable for tree growth and pasture production during these periods. The annual mean temperature was mild (12°C).

The experiment was carried out on abandoned agricultural land. The soil texture at the start of the experiment was sandy (91·81 sand, 4·92 silt and 3·27% clay) and pH (water) was moderately acidic at 5·6. Although the initial soil Mehlich 3-P concentration (35·1 mg kg^{-1}) can be considered high (Sawyer *et al.*, 2008), no risk of phosphorous leaching has been found in the area because of the high soil acidity, which leads to a soil P storage capacity (Mosquera-Losada *et al.*, 2008). All heavy metal concentrations in the soil (Table 1) were below the maximum threshold for using sewage sludge as fertilizer as specified by the European Union Directive 86/278/CEE (DOCE, 1986) and Spanish legislation under R.D. 1310/1990 (BOE, 1990).

Experimental design

At the beginning of the experiment, the soil was double ploughed to a depth of 50 cm, which is traditional practice in the area, and the pasture was sown with a mixture of *D. glomerata* L. var. Artabro (12·5 kg ha^{-1}), *Lolium perenne* L. var. Brigantia (12·5 kg ha^{-1}) and *Trifolium repens* L. var. Huia (4 kg ha^{-1}) in autumn 2004. Bare-rooted 1-year old plants of *F. excelsior* L. (typical height of 25 cm) were planted at a density of 952 trees ha^{-1}, with a distance between rows of 3·0 × 3·5 m. The experimental design was a randomized block with three replicates and five treatments distributed in experimental units of 168 m^2 with twenty-five trees arranged in a frame of 5 × 5 trees. The treatments consisted of (i) no fertilization (NF); (ii) mineral fertilization (MIN) with 500 kg ha^{-1} 8:24:16 compound fertilizer ($N:P_2O_5:K_2O$) at the beginning of the growing

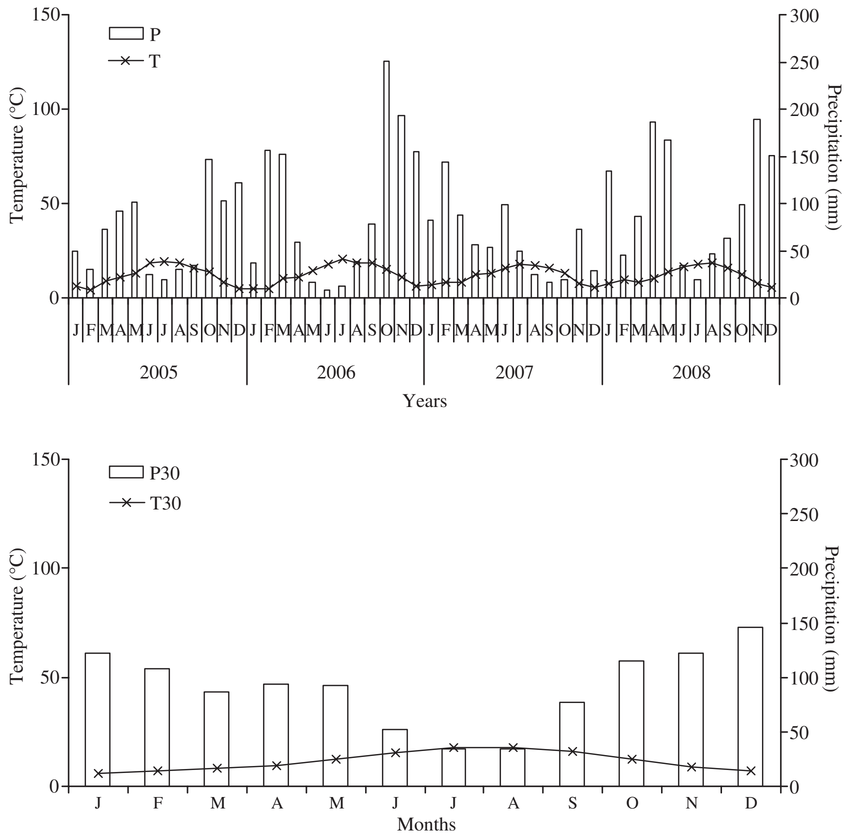

Figure 1 Monthly precipitation and mean temperatures for the study area in 2005, 2006, 2007 and 2008, and mean data for the last 30 years. T: mean monthly temperature (ºC); T30: 30-year mean temperature; P, mean monthly precipitation (mm); and P30: 30-year mean precipitation.

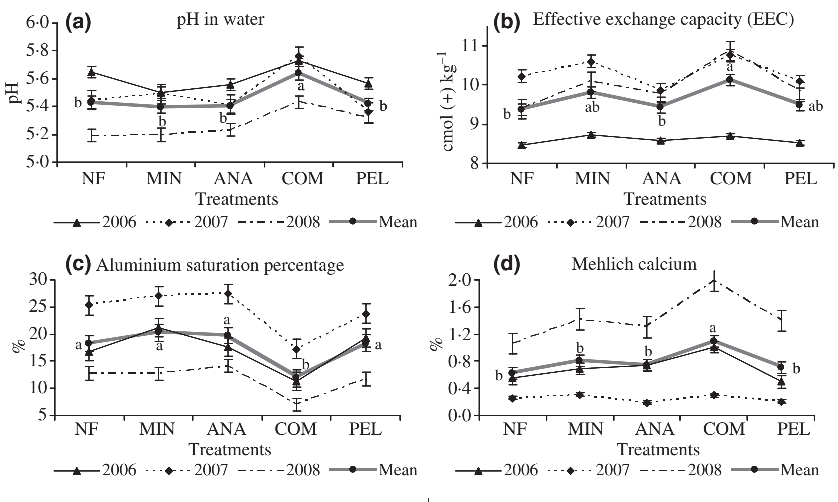

Figure 2 Soil pH in water (a), exchange capacity [cmol (+) kg^{-1}] (b), aluminium saturation percentage in soil exchange complex (%) (c) and amount of Ca extracted by Mehlich (%) (d) in each treatment in the years 2006, 2007 and 2008. NF, no fertilization; MIN, mineral; ANA, anaerobic sludge; COM, composted sludge and PEL, pelletized sludge. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

Table 1 Heavy metal concentrations in the soil at the beginning of the experiment and legal limits established by European Directive 86/278 and Spain R.D. 1310/1990.

	Cd (mg kg^{-1})	Cu (mg kg^{-1})	Cr (mg kg^{-1})	Ni (mg kg^{-1})	Pb (mg kg^{-1})	Zn (mg kg^{-1})
Initial soil concentration	–	5·8	4·1	2·1	–	20·6
Spanish law limits	1–3	50–210	100–150	30–112	50–300	150–450

Limits depend on soil pH (minimum: soil pH < 7, maximum: soil pH > 7).
Concentrations of Cd and Pb were below detection limit of the technique used for determination.

season and 40 kg N ha^{-1} after first harvest (MIN); (iii) fertilization with anaerobically digested sludge with an input of 320 kg total N ha^{-1} before pasture sowing (ANA); (iv) fertilization with composted sewage sludge with an input of 320 kg total N ha^{-1} before pasture sowing (COM) and (e) application of pelletized sewage sludge, which involved a total input of 320 kg total N ha^{-1} split into 134 kg total N ha^{-1} just after pasture sowing in 2004, and 93 kg N ha^{-1} at the end of 2005 and 2006, which correspond to similar total inputs of 320 kg total N ha^{-1} (PEL). Based on previous experiments in the area and EPA (1994) recommendations, it was assumed that approximately 0·25 of the total nitrogen would be mineralized in the first year if compost or anaerobic sludge was added, and therefore approximately 80 kg N ha^{-1} year^{-1} was applied. There was no available information about the pellet fertilizer, but a total similar input of 320 kg total N ha^{-1} during the experiment was used.

Sewage sludge

Anaerobically digested sewage sludge, COM and pelletized sludge (PEL) were taken from municipal waste treatment plants at Lugo, Valladolid and Madrid respectively. The calculation of the required amounts of sludge was conducted according to the percentage of the total nitrogen and dry-matter contents EPA (1994), taking into account the European Union Directive 86/278/CEE (DOCE, 1986) and Spanish regulation R.D.1310/1990 (BOE, 1990) regarding heavy metal concentrations for the application of sewage sludge on to soil. The composition of the sewage sludge is summarized in Table 2. The sludge used in the present experiment had a similar composition to the mean composition of the sludge described for plants all over Spain (Mosquera-Losada *et al.*, 2009). The proportion of N in the sludge was higher than the P and K concentrations in the sludge. As the calculations of the required amounts of sludge were based on N and pasture nitrogen, and as phosphorous needs are similar, contamination by phosphorus is not likely to occur. Furthermore, the acidity of the soil would prevent phosphorous from leaching from the soil (Mosquera-Losada *et al.*, 2008).

Field samplings and laboratory determinations

Soil samples were collected to a depth of 25 cm, as described in the RD 1310/1990 (BOE, 1990) in February 2006, January 2007 and January 2008. In the laboratory, soil pH was determined in water (1:2·5) (Guitián and Carballás, 1976). The aluminium concentration in the exchange complex and the exchangeable cations were determined by extraction with 0·3 M $BaCl_2$. The K, Ca, Mg and Na exchangeable concentrations were measured with a Varian 220FS Spectrophotometer (Varian, Walnut Creek, CA, USA) using the atomic emissions for K and Na and the absorptions for Ca and Mg. Aluminium concentrations were analysed after valoration with 0·01 N NaOH, using phenolphthalein (1%) in an alcohol-based solution as an indicator (Mosquera and Mombiela, 1986). The effective exchange capacity (EEC) was determined by taking the sum of K^+ Ca^+ Mg^+ Na^+ Al and the aluminium percentage saturation using the quotient Al/EEC. The total soil Ca, Cu and Zn concentrations were determined after microwave digestion (CEM, 1994), and the available Ca, Cu and Zn were measured after extraction with Mehlich (1985) with the Varian 220 FS Spectrophotometer using atomic absorption.

Base tree height and diameter were measured with a graduated ruler and a calliper, respectively, at the beginning of 2005, 2006, 2007 and 2008.

Pasture production was determined randomly by taking four samples of pasture at a height of 2·5 cm per plot (0·3 × 0·3 m) using an electric hand clipper in August and December 2005; June and December 2006; April, June and December 2007 and May and December 2008 before all of the plots were grazed by mature sheep (Galician breed Raza ovella galega) at a stocking rate of fifty sheep over the whole experimental area (2520 m^2) 1 week after sampling. Two pasture samples were dried for 48 h at 60°C and weighed to estimate pasture production. The other two samples were separated by hand to determine the proportions of the different plant species and the senescent material, and then dried (60°C for 72 h) to determine the botanical composition on a dry weight basis. Annual abundance diagrams (Magurran, 1988) were made which excluded senescent material. The total Zn in the harvested

Table 2 Chemical properties of the sewage sludge applied, total loadings supplied with the inputs of different types of sludge in this experiment and legal limits established by Spanish directive R.D. 1310/1990.

Parameters	Anaerobic sludge	Composted sludge	Pelletized sludge	Spanish legal limits
Dry matter (%)	29·47	65·19	95·4	
pH	7·25	7·28	7·25	
N (g kg^{-1})	26·2	8·8	35·5	
P (g kg^{-1})	21·4	3·9	10·7	
K (g kg^{-1})	1·9	2·7	1·8	
Ca (g kg^{-1})	6·0	49·8	60·6	
Mg (g kg^{-1})	4·5	14·7	12·4	
Na (g kg^{-1})	1·0	0·4	0·7	
Fe (g kg^{-1})	19·6	12·8	141·5	
Cr (mg kg^{-1})	92·3	3·9	16·6	1000–1500
Cu (mg kg^{-1})	238·5	121·2	136·1	1000–1750
Ni (mg kg^{-1})	69·5	95·3	91·9	300–400
Zn (mg kg^{-1})	1752·3	753·1	1130·4	2500–4000
Cd (mg kg^{-1})	14·4	<0·01	<0·01	20–40
Pb (mg kg^{-1})	281·1	104	58·5	750–1200
Mn (mg kg^{-1})	248·3	90·5	108·8	
	Total heavy metal inputs per treatment			
Parameters	**Anaerobic sludge**	**Composted sludge**	**Pelletized sludge**	**Spanish legal limits†**
Cr (kg ha^{-1})	1·39	0·14	0·21	3
Cu (kg ha^{-1})	3·59	4·47	1·76	12
Ni (kg ha^{-1})	1·04	3·51	1·18	3
Zn (kg ha^{-1})	26·43	27·76	14·63	30
Cd (kg ha^{-1})	0·21	<0·001	<0·0001	0·15
Pb (kg ha^{-1})	3·74	3·33	1·40	15

Limits depend on soil pH (minimum: soil pH < 7, maximum: soil pH > 7). Lower section shows total loadings supplied with the inputs of different types of sludge in this experiment.

†Limit values for amounts of heavy metals which may be added annually to soil, based on a 10-year average (kg ha^{-1} year^{-1}).

pasture herbage was determined by microwave digestion with nitric acid (CEM, 1994).

Statistical analysis

The data were analysed using ANOVA, and the differences between the averages were determined using the LSD test (at the alpha level of 0·05) using the SAS statistical package (SAS, 2001). All soil variables [soil water pH, EEC, Al saturation percentage, total (Cu and Zn) and Mehlich (Ca, Cu and Zn) soil concentrations] were analysed using the ANOVA model ($Y_{ij} = \mu + B_i + T_j + Y_k + BT_{ij} + TY_{jk} + BY_{ik} + \epsilon_{ijk}$), where μ is the mean; B_i is block (two freedom degrees); T_j is treatment (four freedom degrees); Y_k is year (three freedom degrees), BT_{ij} (block treatment interaction (eight freedom degrees), TY_{jk} is treatment year interaction (twelve freedom degrees); BY_{ik} is block year interaction (six freedom degrees) and ϵ_{ijk} is the error term. All tree and pasture variables were analysed using the ANOVA model ($Y_{ij} = \mu + B_i + T_j + \epsilon_{ij}$), where μ is the mean; B_i is block (two freedom degrees); T_j is treatment (four freedom degrees); and ϵ_{ij} is the error term. ANOVA type III errors were taken into account to determine significances.

Results

Soil chemical properties

pH in water, effective exchange capacity (EEC), aluminium saturation percentage and Ca extracted by Mehlich

Composted sludge (COM) increased the mean soil pH ($P < 0.01$) and the amount of mean Ca extracted by

Mehlich ($P < 0{\cdot}05$), but the effect on ECC was not significant ($P > 0{\cdot}05$). Because of these modifications (Fig 2), the mean percentage of aluminium saturation decreased with this type of sludge application ($P < 0{\cdot}01$). There was a significant effect of year ($P < 0{\cdot}0001$) on soil pH (2006: $5{\cdot}6^{a}$; 2007: $5{\cdot}5^{b}$ and 2008: $5{\cdot}27^{c}$); on EEC (2006^{*}: $8{\cdot}6^{b}$; 2007: $10{\cdot}31^{a}$ and 2008: $8{\cdot}6^{b}$ expressed as cmol (+) 100 g^{-1} soil); on aluminium saturation percentage (2006^{*}: $17{\cdot}24^{b}$; 2007: $24{\cdot}15^{a}$ and 2008: $11{\cdot}66^{c}$) and amount of Ca extracted by Mehlich (2006: $0{\cdot}69^{b}$; 2007: $0{\cdot}25^{c}$ and 2008: $1{\cdot}43^{a}$ expressed as mg kg^{-1}) (in all cases different superscript letters indicate significant differences between years). The soil pH and aluminium saturation percentage were lower at the end of the study (2008) than at the beginning of the study (2006), whereas the EEC and the amount of Ca extracted by Mehlich were higher in 2008 than in 2006.

Total and Mehlich extracted Cu and Zn

The total and Mehlich soil levels of Cu and Zn in 2006, 2007 and 2008 are presented in Figure 3. The mean total Cu and Zn were significantly affected by the treatments ($P < 0{\cdot}001$ and $P < 0{\cdot}05$ respectively). There was a significant effect of year on total Cu ($P < 0{\cdot}0001$), total Zn ($P < 0{\cdot}0001$) and on the Zn extracted by Mehlich ($P < 0{\cdot}001$). In the case of the Cu extracted by Mehlich, the interaction of treatment × year was significant ($P < 0{\cdot}01$). All of the variables were increased by COM, and total Zn was increased by anaerobic sludge (ANA) compared with the other treatments. The total soil Cu and Zn found in the experiment were below the maximums set by Spanish regulations for the use of sewage sludge in agriculture for acid soils (Cu: 50 mg kg^{-1} and Zn: 150 mg kg^{-1}) [R.D. 1310/1990, (BOE, 1990)]. The concentrations of total Cu and Zn and Zn extracted by Mehlich were lower in 2008 than in 2006.

Tree height and diameter

The average tree heights for each treatment in 2005, 2006, 2007 and 2008 are shown in Table 3. In all 2 years of the study, tree height was significantly modified by fertilization treatment (Table 4). Initially, the tree height was higher in all treatments that received organic fertilizer (ANA, COM and PEL) than the MIN or NF treatments. Tree height in 2005 and 2006 was increased by anaerobic (ANA) and PEL compared with the mineral fertilization (MIN) and no fertilization (NF) treatments, but this positive effect was only found in those plots receiving PEL in 2007 and 2008. Trees growing in NF and MIN treatments had the lowest height during the 4 years of the experiment.

Tree diameter (Table 3) was significantly modified by fertilization treatment in all 4 years of the study, with the exception of 2007, (Table 4). In 2005, 2006 and 2007, tree diameter was increased by ANA compared with the other treatments (NF, MIN, ANA, COM and PEL). In 2008, the COM and PEL treatments resulted in larger tree diameters than MIN treatment.

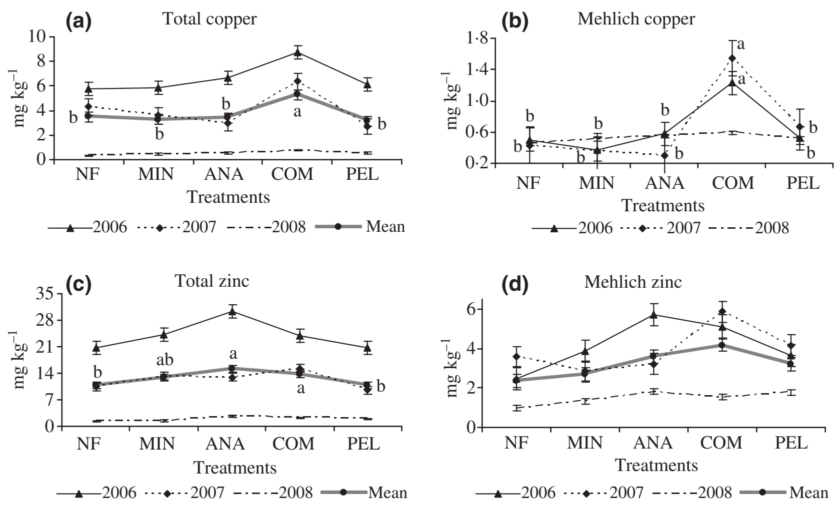

Figure 3 Total Cu (mg kg^{-1}) (a) and total Zn concentration (mg kg^{-1}) (c) in soil and amount of Cu (mg kg^{-1}) (b) and amount of Zn (mg kg^{-1}) (d) extracted by Mehlich in each treatment in the years 2006, 2007 and 2008. NF, no fertilization, MIN, mineral; ANA, anaerobic sludge; COM, composted sludge and PEL, pelletized sludge. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

Table 3 Mean values for tree height (cm), tree diameter (mm) and pasture production (t ha^{-1}) under the different fertilization treatments in the years 2005–2008.

	Year	Treatments NF	MIN	ANA	COM	PEL	s.e.
Tree height (cm)	2005	32·88	32·33	38	34·85	38·05	0·69
	2006	33·65	33·08	39·33	35·8	38·1	0·66
	2007	41·59b	42·58b	48·89ab	47·65ab	53·55a	1·05
	2008	114·36ab	107·56b	103·63b	121·84ab	135a	3·51
Tree diameter (mm)	2005	5b	4·83b	5·89a	4·7b	5b	0·14
	2006	5·12b	5·42b	6·56a	4·8b	5·15b	0·14
	2007	6·29	5·92	7·54	6·45	6·09	0·14
	2008	12·55ab	10·81b	12·29ab	13·68a	14·04a	0·37
Pasture Production t ha^{-1}	2005	1·96b	2·98a	3·17a	2·81ab	3·36a	0·14
	2006	4·91	3·98	5·15	4·82	5·24	0·18
	2007	5·66ab	5·88ab	4·81b	5·53ab	6·55a	0·18
	2008	3·96	4·25	3·37	3·94	4·51	0·14

Different letters indicate differences between treatments within the same year that were significant at $P < 0{\cdot}05$.
s.e., mean standard error; NF, no fertilization; MIN, mineral fertilizer; ANA, anaerobic sludge; COM, composted sludge and PEL, pelletized sludge.

Pasture understorey

Pasture production

The annual pasture production for the different fertilization treatments in 2005, 2006, 2007 and 2008 is summarized in Table 3. Significant differences were detected between the treatments in all years except 2006 and 2008 (Table 4). The highest levels of pasture production were found in 2007 (4·8–6·5 t ha^{-1}), whereas the lowest values were recorded in 2005 (1·9–3·3 t ha^{-1}). In 2005, there was a positive response to organic fertilization (ANA, COM and PEL) and inorganic fertilization (MIN) compared with no fertilization (NF), but this effect was only maintained in the PEL treatment until the end of the experiment. In the final years of the study (2007–2008), the annual pasture production tended to be lower than that of the PEL when the ANA was applied to the soil.

Table 4 ANOVAs for tree height, diameter and pasture production in 2005, 2006, 2007 and 2008.

	Year	Height	Diameter	Pasture production
Treatment effect	2005	ns	*	*
	2006	ns	**	ns
	2007	**	ns	*
	2008	*	*	ns

ns, not significant.
*, $P < 0{\cdot}05$; **, $P < 0{\cdot}01$; ***, $P < 0{\cdot}001$.

Pasture abundance diagrams

Figure 4 shows the abundance diagrams for the different fertilization treatments in 2005, 2006, 2007 and 2008. *Agrostis capillaris* L., *D. glomerata* L., *Holcus lanatus* L. and *L. perenne* L. were present in the sward in all treatments and in all years. The treatments with a high number of species were associated with a higher proportion of dicotyledonous species. *Agrostis capillaris* L. and *D. glomerata* L. were the most dominant species throughout the study. The proportion of *A. capillaris* L. was always over 75% in the NF treatment and approximately 50% in the other treatments and years, with the exception of mineral (MIN) and ANA fertilization in the final year. However, the mineral (MIN) and ANA treatments had higher proportions of *D. glomerata* L. in the first years of the study. In the PEL treatment, codominance between *D. glomerata* L. and *A. capillaris* L. was evident throughout the experiment. In terms of species richness, the ANA treatment had the lowest number of species at the start of the experiment, but the PEL treatment had the lowest number from 2006 onwards. The COM treatment in the second year of the experiment was an exception to this trend, exhibiting the lowest number of species. The PEL treatment was dominated by monocotyledonous species in the final year than the remaining treatments (NF, MIN, ANA and COM).

Zn and Cu concentrations in pasture

The concentration of Zn in the pasture was significantly affected by treatments in April 2007 ($P < 0{\cdot}01$), and

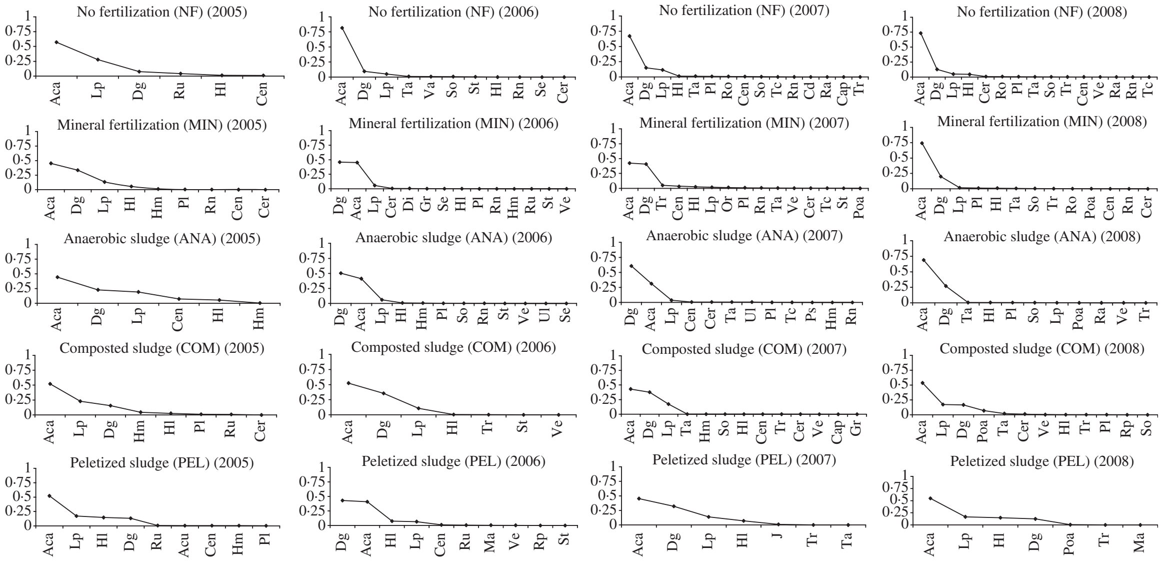

Figure 4 Abundance diagrams for the treatments applied in the years 2005, 2006, 2007 and 2008. Aca: *Agrostis capillaris* L.; Acu: *A. curtisii* Kerguélen; Cd: *Carduus* spp.; Cap: *Capsella bursa pastoris* L.; Cen: *Centaurea limbata* Hoffmanns. /Link; Cer: *Cerastium glomerata* Thuill; Di: *Digitalis purpurea* L.; Dg: *Dactylis glomerata* L.; Gr: *Geranium rotundifolium* L.; Hl: *Holcus lanatus* L.; Hm: *Holus mollis* L.; J: *Juncus effesus* L.; Lp: *Lolium perenne* L.; Ma: *Matricaria* spp.; Mu: Musgo; Or: *Ornithopus compressus* L.; Pl: *Plantago lanceolata* L.; Poa: *Poa pratensis* L.; Ps: *Pseudarrhenatherum longuifolium* (Thore) Rouy; Ra: *Rumex acetosa* L.; Rn: *Ranunculus repens* L.; Ro: *Rumex obtusifolius* L.; Rp: *Raphanus raphanistrum* L.; Ru: *Rubus* spp.; Se: *Senecio jacobea* L.; So: *Sonchus oleraceus* L.; St: *Stellaria media* L. (Vill); Ta: *Taraxacum officinale* Weber; Tc: *Trifolium campestre* Schreber; Tr: *Trifolium repens* L.; Ul: *Ulex europaeus* L.; Ve: *Veronica agrestis* L.

May 2008 ($P < 0{\cdot}05$) (Figure 5). However, Cu levels in the pasture were not affected by any treatments (data not shown). In April 2007, the concentration of Zn in the pasture was higher when the ANA and PEL treatments were applied. In May 2008 the concentration of Zn in the pasture increased in the ANA treatment compared with the COM treatment. There were no responses to the treatments in terms of the pasture concentrations of Zn after harvests made in 2005, 2006, December 2007 and December 2008.

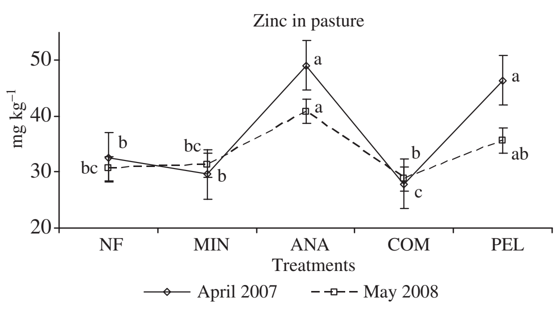

Figure 5 Concentrations of Zn in pasture (mg kg^{-1}) under the different fertilizer treatments in the harvests of April 2007 and May 2008. NF, no fertilization; MIN, mineral; ANA, anaerobic sludge; COM, composted sludge and PEL, pelletized sludge. Different letters indicate significant differences between treatments. Vertical lines indicate mean standard error.

Discussion

This experiment demonstrates that there are very few negative effects of sewage sludge application, either from toxic metal loading or in terms of productivity in the specific edaphoclimatic conditions of this study. The growth of trees and pasture production can be limited by low soil pH, EEC and a high aluminium saturation percentage; however, they can be improved by applications of high pH organic waste and by fertilization (Mosquera-Losada *et al.*, 2009). The soil pH in this experiment ranged from acidic (5·1–5·5) to moderately acidic (5·6–6·0) (Slattery *et al.*, 1999); conditions which usually indicate deficiencies in the availability of cations and, therefore, limit pasture production (Whitehead, 2000). Moreover, the soil EEC was low and usually below 10 cmol (+) kg^{-1} soil, which could be explained by the high sand proportion in the soil of this experimental site (Brady and Weil, 2008). However, the aluminium saturation percentage was below 25%, which indicates the optimum characteristics for pasture and tree growth in Galicia (Mombiela and Mateo, 1984). Aluminium saturation was also found to have a diachronic variation, which usually improved from the start to the end of the experiment. The EEC increase could be explained by the tilling that occurred at the start of the experiment. The soil aggregate structure was probably destroyed by the tilling process (Dexter, 1988), but the structure is likely to have been improved with the addition of fertilizer and with the establishment of pasture and trees, and the potential for increased organic matter input into the soil. The soil pH, however, was lower at the end of the experiment, despite the increase in available Ca and the reduction in available Al caused by sludge incorporation into the soil (Smith, 1996); this could be explained by the increase in the proportion of H^+ related to the higher EEC. The soil acidity increase derived from the H^+ can be explained by the cation extraction of the crops, the mineralization process (NH_4^+ is transformed in $NO3^-$ and H^+ is released in the nitrification process) and the high mean rainfall of the area, which promotes cation leaching (Whitehead, 1995).

Composted sludge (COM) increased soil fertility compared with other treatments. The pH and mean amount of available Ca were improved by the COM treatment, which was expected and is explained by the different composition of the sludge and the rate of mineralization (Mosquera-Losada *et al.*, 2009). Applied doses of sewage sludge with compost were higher in order to meet the nitrogen required by EPA (Environmental Protection Agency) (1994) recommendations because the COM composition revealed a lower concentration of nitrogen than the anaerobic and PEL. The COM also had a low mineralization rate, as indicated by the EPA (Environmental Protection Agency) (1994) (the mineralization rate is approximately 20% for anaerobic compared with 10% for COM in the first year). Moreover, the pH and available Ca were increased in the COM treatment because of the higher concentrations of Ca, K and Mg compared with anaerobic and PEL (excepting Ca). The COM treatment applied approximately 1835·8 kg Ca ha^{-1}, 99·53 kg K ha^{-1} and 541·9 kg Mg ha^{-1}; meanwhile, only 90·51 kg Ca ha^{-1}, 28·66 kg K ha^{-1} and 67·88 kg Mg ha^{-1} were added with the ANA treatment, and 776·85 kg Ca ha^{-1}, 23·07 kg K ha^{-1} and 158·96 kg Mg ha^{-1} were the soil inputs with the PEL treatment. The higher rate of Ca and Mg with the COM also explains the reduction in the aluminium saturation percentage for this treatment (Smith, 1996; Prasad and Power, 1997; Speir *et al.*, 2004). In previous studies, the anaerobic sewage sludge effect on the soil pH and EEC was found to depend on the previous initial soil pH. When the initial soil pH was high, anaerobic sludge inputs increased acidity as extraction was promoted (Mosquera-Losada *et al.*, 2006), but when the soil pH was very low (4·5), as described by López-Díaz *et al.* (2007), a positive effect of sewage sludge application on soil pH was found. In this study, although the soil fertility was enhanced by COM treatment, tree growth and pasture production were not promoted. The lack of enhancement in tree growth and pasture production was probably due to the lower mineralization rate and the N availability of COM treatment compared with that of the anaerobic (ANA) or PEL as described by the EPA (1994). Warman and Termeer (2005) found better crop production in response to ANA than to COM sludge. The mineralization rate depends on the local climate and on the mineralization rates (EPA, 1994), and in our case, the COM treatment seems to have an initial lower mineralization rate compared with the ANA treatment. As a result, the effect of COM on pasture production and tree growth could become apparent over a longer period, as was seen with tree diameter. The soil fertility is differently affected by sewage sludge application, and the effect is dependent on the initial soil pH and on the type of the sewage sludge applied.

At the end of the 4-year study, the heights of ash trees varied from 104 to 135 cm and stem diameters from 10·81 to 14·04 mm. The tree heights were within the range (23–275 cm) reported in a study carried out in UK after 3 years of experimentation (Mwase *et al.*, 2008). The initial tree diameters in our study were also similar to those found by the same authors (15·3 mm). A positive effect was found for the ANA treatment in the two first years of the experiment; this trend has also been described for *Populus* × euroamericana (Rigueiro-Rodríguez *et al.*, 2008b) and *Pinus radiata* D. Don, under soil conditions that were initially either very acid (López-Díaz *et al.*, 2007) or neutral (Mosquera-Losada *et al.*, 2006). In other work, Muys *et al.* (2004) and Weber-Blaschke and Rehfuess (2002) demonstrated that soil fertility improvements led to higher *F. excelsior* growth, relative to the control, on sites with sandy loam and loam soils respectively. In both the present experiment and in these previous studies, the positive effect of the ANA treatment could be attributed to the soil fertility and water retention improvements caused by anaerobic fertilizer applied at the establishment of the plantation (Wolstenholme *et al.*, 1992). However, at the end of the experiment, only the PEL treatment showed a significant increase in tree height diameter and the COM treatment in tree diameter compared with the MIN treatment, which could be explained by the enhancement of pasture growth and increased competition between trees and pasture caused by the MIN treatment as compared with the NF treatment.

Annual pasture production was below the common levels in this region due to the droughts in 2005, 2006, 2007 and 2008 (mean monthly precipitation of these years were lower than mean precipitation over the previous 30 years) and the low temperatures at the beginning of the year. Pasture production was significantly increased by organic or inorganic fertilization as found by Mosquera-Losada *et al.* (2006) and by López-Díaz *et al.* (2009) in agrarian soils in Galicia afforested with *P. radiata* D. Don with a pH close to neutral (pH 6·8) and a slightly acidic pH (pH 6·3) respectively. At the end of this study, only the PEL treatment was found to have positively affected annual pasture production by the PEL treatment because of the annual application; however, no residual effect was found as a result of the COM or ANA treatments (Rigueiro-Rodríguez *et al.*, 2000; López-Díaz *et al.*, 2007).

The lower proportion of *A. capillaris* L. and the higher proportion of *D. glomerata* L. could be associated with low and high soil fertility, as demonstrated in soils with a pH below 4·97 (Mosquera-Losada *et al.*, 2001). Modifications to soil fertility also caused variations in the dominance of these two species, with a high proportion of *D. glomerata* L. initially associated with the MIN and ANA treatments followed by a switch in the botanical composition to over 75% *A. capillaris* L. by the end of the experimental period.

Fertilizer treatment effects on soil fertility explain tree and pasture production, and affect biodiversity. The ANA treatment initially increased soil fertility, as demonstrated by higher tree growth, greater pasture production, a lower number of species and a higher proportion of *D. glomerata* L. (Mosquera-Losada *et al.*, 2001) compared with the other treatments (NF, MIN, COM and PEL). These results are of particular interest in areas in which the initial tree development is important to guarantee tree survival. However, there is a reduction in tree and pasture growth when soil fertility is depleted, probably because the nutrient requirements of trees are greater than in the other treatments and because the capacity for nutrient retention in the ANA treatment is lower than in the other treatments because nutrient leaching occurs. On the contrary, the PEL treatments sustained better tree and pasture production over the long term as the nutrient inputs were supplied as split doses through time. The higher fertility of soils fertilized with PEL probably explains the dominance of monocotyledonous species and the low number of species throughout the experiment.

It is important to be aware of the effects of sewage sludge on the concentrations of Cu and Zn in the soil and in plants, because Cu and Zn are commonly present in higher proportions in the municipal sewage sludge (Smith, 1996; Mosquera-Losada *et al.*, 2009) relative to soil concentrations. As seen with the macronutrients, the COM treatment resulted in higher input rates of Cu and Zn (4·46 kg Cu ha^{-1} and 27·76 kg Zn ha^{-1}) into the soil than the ANA (3·59 kg Cu ha^{-1} and 26·43 kg Zn ha^{-1}) or PEL treatment (1·74 kg Cu ha^{-1} and 14·47 kg Zn ha^{-1}). The COM treatment increased the soil pH compared with the other treatments, which usually implies a reduction of Cu and Zn solubility because of the CEC increment (Prasad and Power, 1997). In China, Miao-Miao *et al.* (2007) also found that the use of COM as fertilizer increased the concentrations of Zn and Cu in the soil, but in that study the COM contained higher concentrations of Cu and Zn due to the effects of local industry (Cu: 2316 mg kg^{-1} and Zn: 2971 mg kg^{-1}) compared with the sludges used in our experiment. ANA also increases the total soil Zn because this type of sludge has a higher concentration of Zn than the COM or the PEL (anaerobic sludge: 1752·3 mg Zn kg^{-1} soil; COM: 478·7 mg Zn kg^{-1} soil and PEL: 753·1 mg Zn kg^{-1}). An increment of Cu and Zn in soil as a result of sewage sludge applications was also found by Yuan (2009). In contrast, in this study the concentrations of Zn and Cu in the soil were lower in 2008 than at the beginning of the experiment. This could be explained by leaching, but is more likely due to pasture and tree extractions.

The range of Zn concentrations in the pasture in this experiment (18·63–49·31 mg kg^{-1}) was at the low end of the concentrations commonly found in pastures (27–150 mg kg^{-1}) and below the levels of 100 and 400 mg kg^{-1} considered excessive or toxic for plants (Kabata-Pendías and Pendías, 1985; Smith, 1996). ANA tended to increase the soil and pasture Zn concentrations in the acidic soils of the Galician region, as described by Mosquera-Losada *et al.* (2001). In spite of the higher Zn inputs with the COM treatment, which increased the total and the available Zn in the soil, no effect was detected in the pastures because of the increased soil pH compared with other treatments, which could in turn reduce the availability of absorbable Zn in the soil (Prasad and Power, 1997). A Zn concentration of 500 mg kg^{-1} is considered toxic for cattle, sheep and horses (Smith, 1996), but this value was not reached or exceeded in the pasture in this experiment.

Conclusion

Soil characteristics, tree growth, pasture species biodiversity and pasture development are modified by the type of sewage sludge used when similar nitrogen inputs are applied. The ANA treatment has a higher initial effect on tree and pasture productivity, but PEL treatment sustains better production as it is applied in several times and the COM treatment improved soil characteristics over the long term in sandy soils. The PEL treatment should be promoted because this treatment enhances productivity, allows for better nutrient recovery and is less costly to apply than the other two treatments. No toxic Zn or Cu concentrations were found in the plants or in the soil in spite of the higher concentrations in sewage sludge than in the soil.

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References

BOE (Spanish Official Bulletin) (1990) *Real Decreto 1310/1990 29 de Octubre de 1990, que regula la utilización de los lodos de depuración (Royal Decree 1310/1990 29th October 1990, that regulates the use of sewage sludge)*. Madrid, Spain: Ministerio Agricultura, Pesca y Alimentación.

Brady N.C. and Weil R.R. (2008) *The nature and properties of soils*, 14th edn. Upper Saddle River, New Jersey: Prentice Hall.

CEM (1994) *Microwave digestion applications manual*. San Francisco: CED.

Dexter A.R. (1988) Advances in characterization of soil structure. *Soil and Tillage Research*, **11**, 199–238.

DOCE (1986) *Council Directive 86/278/CEE of 12 June 1986 on the protection of the environment and, in particular of the soil, when sewage sludge is used in agriculture (86/278/EEC). EC Official Journal L 181 04/07/1986*. Brussels, Belgium: European Commission.

EEA (2000) Working document on sludge 3rd draft. Brussels. 27 April 2000. ENV.E.3/LM. Available at: http://ec.europa.eu/environment/waste/sludge/pdf/sludge_en.pdf (accessed 27 January 2010).

EPA (Environmental Protection Agency) (1994) *Land application of sewage sludge. A guide for land appliers on the requirements of the federal standards for the use and disposal of sewage sludge, 40 CFR Part 503*. Washington, DC: Environment Protection Agency.

EU (European Union) (2005) Council regulation (EC) nº 1698/2005 of September 2005 on support for rural development by the European Agricultural Fund for Rural Development. (EAFRD). Available at: http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2005:277:0001: 0040:en:pdf

Graves A.R., Burguess P.J., Liagre F., Pisanelli A., Paris P., Moreno G., Bellido M., Mayus M., Postma M., Schindler B., Mantzanas K., Papanastis V.P. and Dupraz C. (2008) Farmer perceptions of silvoarable systems in seven European countries. In: Rigueiro-Rodríguez A., McAdam J.H. and Mosquera-Losada M.R. (eds) *Agroforestry in Europe, advances in agroforestry*, 6, pp. 155–199. Netherlands: Springer.

Guitián F. and Carballás T. (1976) *Técnicas de análisis de suelos (soil analysis techniques). Santiago de Compostela*. Spain: Ed. Pico Sacro.(in Spanish).

Kabata-Pendías A. and Pendías H. (1985) *Trace elements in soils and plants*. Boca-Raton, Florida: CRC Lewis publishers.

López-Díaz M.L., Mosquera-Losada M.R. and Rigueiro-Rodríguez A. (2007) Lime, sewage sludge and mineral fertilization in a silvopastoral system developed in very acid soils. *Agroforestry Systems*, **70**, 91–101.

López-Díaz M.L., Rigueiro-Rodríguez A. and Mosquera-Losada M.R. (2009) Influence of pasture botanical composition and fertilization treatments on tree growth. *Forest Ecology and Management*, **257**, 1363–1372.

Magurran A.E. (1988) *Ecological diversity and its measurement*. London, UK: Princeton University Press.

McAdam J.H. and Hoppé G.M. (1996) Pasture production between trees in a silvopastoral agroforestry system. *Grassland Science in Europe*, **1**, 119–122.

McAdam J.H. and Sibbald A. (2000) Grazing livestock management. In: Hislop M. and Claridge J. (eds) *Agroforestry in the UK*, Commission Bulletin 122, pp. 44–57. London, UK: HMSO.

McEvoy P.M. (2004) *Livestock management in grazed woodlands*. Ph.D. thesis, Belfast, UK: Queen's University.

Mehlich A. (1985) Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant. *Communications in Soil Science and Plant Analysis*, **15**, 1409–1416.

Miao-Miao H., Guang-Ming T., Xing-Qing L., Yi-Tong Y., Jian-Yang W. and Gen-Di Z. (2007) Effects of two sludge application on fractionation and phytotoxicity of Zn and Cu in soil. *Journal of Environmental Sciences*, **19**, 1482–1490.

MMA (Spanish Environment Ministry) (2006) Plan integrado de residuos 2007_2015. (Integrated waste plan 2007-2015). Available at: http://www.mma.es/secciones/calidad_contaminacion/residuos/planificacion_residuos/pdf/borradorpnir_anexo5.pdf (accessed 16 March 2009).

Mombiela F.A. and Mateo M.E. (1984) Necesidades de cal para praderas en terrenos a monte.1. Su relación con el Al cambiable en suelos sobre granitos y pizarras de Galicia. (Lime requirements for pasture establishment in forest soils. 1. Al exchangeable relationships in granite and schist soils). *Anales del INIA Serie Agrícola*, **25**, 129–143.

Mosquera A. and Mombiela F.A. (1986) Comparison of three methods for determination of soil Al in an unbuffered salt-extract. *Communications in Soil Science and Plant Analysis*, **17**, 30–35.

Mosquera-Losada M.R., Rigueiro-Rodríguez A. and López-Díaz M.L. (2001) Sewage sludge fertilization of a silvopastoral system with pines in northwestern Spain. *Agroforestry Systems*, **53**, 1–10.

Mosquera-Losada M.R., Fernández-Núñez E. and Rigueiro-Rodríguez A. (2006) Pasture, tree and soil evolution in silvopastoral systems of Atlantic Europe. *Forest Ecology and Management*, **232**, 135–145.

Mosquera-Losada M.R., Nair V.N., Nair P.K.R., Morán D., Santiago-Freijanes J.J. and Rigueiro-Rodríguez A. (2008) Soil Phosphorous storage in a poplar + silvopastoral system fertilized with sewage sludge in Northwestern Spain. ASA-CSSA-SSSA, 2010 International Annual Meeting. Available at: http://acs.confex.com/crops/2008am/webprogram/Paper41839.html (accessed 27 January 2010).

Mosquera-Losada M.R., Muñoz-Ferreiro N. and Rigueiro-Rodríguez A. (2009) Agronomic characterization of different types of sewage sludge: policy implications. *Waste Management*, **30**, 492–503.

Muys B., Nachtergale L., Lust N., Merckx R. and Granval P. (2004) Medium term evaluation of a forest soil restoration trial combining tree species change, fertilization and earthworm introduction. *Pedobiologia*, **47**, 772–783.

Mwase W.F., Savill P.S. and Hemery G. (2008) Genetic parameterestimates for growth and form traits in common ash (*Fraxinus excelsior* L.) in a breeding seedling orchard at Little Wittenham in England. *New Forests*, **36**, 225–238.

Nair V.D. and Graetz D.A. (2004) Agroforestry as an approach to minimizing nutrient loss from heavily fertilized soils: the Florida experience. *Agroforestry Systems*, **61**, 269–279.

Prasad R. and Power J.F. (1997) *Soil fertility management for sustainable agriculture*. Boca Ratón, Florida: CRC Lewis publishers.

Rigueiro A., Santiago J.J., Fernández E. and Mosquera M.R. (2009) Sistemas biológicos de prevención de incendios forestales: los sistemas silvopastorales (Biological systems to prevent forest fires: silvopastoral systems). In: Vélez-Muñoz R. (ed) *La defensa contra incendios forestales: fundamentos y experiencias*, 2nd edn, pp. 447–451. Madrid, Spain: McGraw Hill.

Rigueiro-Rodríguez A., Mosquera-Losada M.R. and Gatica-Trabanini E. (2000) Pasture production and tree growth in a young pine plantation fertilized with inorganic fertilizers and milk sewage in Northwestern Spain. *Agroforestry Systems*, **48**, 245–256.

Rigueiro-Rodríguez A., Mosquera-Losada M.R., Romero-Franco R., González-Hernández M.P. and Villarino-Urtiaga J.J. (2005) Silvopastoral systems as a forest fire prevention technique. In: Mosquera-Losada M.R., McAdam J. and Rigueiro-Rodríguez A. (eds) *Silvopastoralism and sustainable land management, pp*, pp. 380–387. Wallingford, UK: CABI Publishing.

Rigueiro-Rodríguez A., Mosquera-Losada M.R. and López-Díaz M.L. (2007) Mineral concentrations in herbage and soil in a *Pinus radiata* silvopastoral system in north-west Spain after sewage sludge and lime application. *Grass and Forage Science*, **62**, 208–224.

Rigueiro-Rodríguez A., Fernández-Núñez E., González-Hernández P., McAdam J.H. and Mosquera-Losada M.R. (2008a) Agroforestry systems in Europe: productive, ecological and social perspectives. In: Rigueiro-Rodríguez A., McAdam J.H. and Mosquera-Losada M.R. (eds) *Agroforestry in Europe, Advances in Agroforestry, 6, pp*, pp. 43–66. The Netherlands: Springer.

Rigueiro-Rodríguez A., Mosquera-Losada M.R. and López-Díaz M.L. (2008b) Effect of sewage sludge and liming on productivity during the establishment of a silvopastoral system in north-west Spain. *New Zealand Journal of Agricultural Research*, **51**, 199–207.

SAS (2001) *User's guide: Statistics*. Cary, NC: SAS Institute Inc.

Sawyer J.E., Mallarino A.P., Killorn R. and Barnhart S.K. (2008). A general guide for crop nutrient and limestone recommendations in Iowa. Iowa State University extension service. Available at: http://www.extension.iastate.edu/Publications/PM1688.pdf (accessed 27 January 2010).

Slattery W.J., Conyers M.K. and Aitken R.I. (1999) Soil pH, aluminium, manganese and lime requirement. In: Peverill K.I., Sparrow L.A. and Reuter D.J. (eds) *Soil analysis: an interpretation manual, pp*, pp. 103–128. Collingwood, Victoria, Australia: CSIRO Publishing.

Smith S.R. (1996) *Agricultural recycling of sewage sludge and the environment*, pp. 207–236. Wallingford, UK: CAB International.

Speir T.W., Horswell J., Van Schaik A.P., McLaren R.G. and Fietje G. (2004) Composted biosolids enhance fertility of a sandy loam soil under dairy pasture. *Biology and Fertility of Soils*, **40**, 349–358.

Wagner R.G., Little K.M., Richardson B. and McNabb K. (2006) The role of vegetation management for enhancing productivity of the world's forests. *Forestry*, **79**, 57–79.

Warman P.R. and Termeer W.C. (2005) Evaluation of sewage sludge, septic waste and sludge compost applications to corn and forage: yields and N, P and K content of crops and soils. *Bioresource Technology*, **96**, 955–961.

Weber-Blaschke G. and Rehfuess K.E. (2002) Correction of Al toxicity with European ash (*Fraxinus excelsior* L.) growing on acid soils by fertilization with Ca and Mg carbonate and sulfate in pot experiments. *Forest Ecology and Management*, **167**, 173–183.

Whitehead D. (1995) *Grassland nitrogen*. Wallingford, UK: CAB International.

Whitehead D.C. (2000) *Nutrient elements in grassland. Soil-plant-animal relationships*. Wallingford, UK: CAB International.

Wolstenholme R., Dutch J., Monat A., Bayes C.D. and Taylor C.M.A. (1992) *A manual of good practice for the use of sewage sludge in forestry*. Forestry Commission Bulletin, 107. London, UK: HMSO.

Yuan G. (2009) Copper, zinc, and nickel in soil solution affected by biosolids amendment and soil management. *Australian Journal of Soil Research*, **47**, 305–310.

Zas R. and Alonso M. (2002) Understorey vegetation as indicators of soil characteristics in northwest Spain. *Forest Ecology and Management*, **171**, 101–111.

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Fertilization in pastoral and *Pinus radiata* D. Don silvopastoral systems developed in forest and agronomic soils of Northwest Spain

M. Rosa Mosquera-Losada*, Nuria Ferreiro-Domínguez, Antonio Rigueiro-Rodríguez

Departamento de Producción Vegetal, Escuela Politécnica Superior de Lugo, Universidad de Santiago de Compostela, Campus Universitario s/n, 27002 Lugo, Spain

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ABSTRACT

The effects of fertilization, pasture sowing and tree plantation on soil fertility and tree and pasture production can vary depending on the soil type. Tree plantation is recognized as a way to reduce nutrient leaching and increase land profitability in agronomic and forest soils, meanwhile pasture fertilization and sowing is usually associated to better pasture productivity and quality. Fertilization can be performed with mineral fertilizers, which have become expensive in recent times, or with organic fertilizers like sludge, which is being promoted worldwide. This study aims to evaluate the effects of sludge fertilization, tree planting and pasture sowing on different variables of soil (KCl-pH, cation exchange capacity, total N, total and Mehlich P, nitrate and soil organic matter) and pasture (production, botanical composition, crude protein and P concentration) in treeless and agroforestry systems established in forest and agronomic soils. The experimental design was a randomized block following an incomplete factorial design with three replicates and nine treatments including two types of soils (forestry and agronomic), two types of vegetation (natural and sown), two types of fertilization (sludge fertilization and mineral fertilization, with a no fertilizer control) in afforested and treeless pastures. Pasture production and quality was better under agronomic soils, which also had higher levels of KCl-pH, cation exchange capacity, nitrate, total N and P than forest soils. Tree establishment did not modify nitrate or P leaching, probably due to the youth of the trees when most of nitrate was leached at the beginning of the experiment, but reduction of soil KCl pH and pasture crude protein was found in forest soils, when trees and pasture were together established, probably due to the high extractions of these systems compared with unsown forests. Moreover, the sludge inputs increased pasture production better than the mineral fertilizer in the forest soils, probably due to the greater amount of nutrients applied by the former. Sowing enhanced the presence of sown grasses in the forest understory, but their presence reduced pasture quality, and they disappeared within a short period of time. Therefore, the use of the sludge as fertilizer allows nutrient recycling of this residue in soils of low fertility and increases productivity and preserves fertility compared with mineral fertilizer at short (forest soils) and medium (agronomic soils) term.

1. Introduction

Agroforestry systems are sustainable land management techniques that are promoted by the EU (European Union, Council Regulation 1698/2005 [EU, 2005]) and are considered a good management tool that can be implemented by farmers in the different countries of Europe (Graves et al., 2008).

Monterey pine [*Pinus radiata* (D. Don)] is a tree species that is currently used in silvopastoral systems in temperate areas like Australia, New Zealand, and Chile (Hawke, 1991; Knowles, 1991; Benavides et al., 2009) due to its fast growth. The species is widely used in the Atlantic biogeographic region of Europe (mostly in the North of Spain and West of France) in both forestry and farm grassland soils. Adequate fertilization practices in Monterey pine silvopastoral systems should be implemented to increase tree and pasture growth simultaneously at the same time that nutrient leaching risk is reduced. Recent increases in inorganic fertilizer prices along with environmental concerns have reduced the use of inorganic nitrogen fertilizers in the EU (EFMA, 2009), which are currently being replaced by organic fertilizers like sewage sludge as a cheaper nitrogen resource.

In EU countries, sewage sludge production has increased since the early nineties due to the implementation of European Directive 91/271/EEC (EU, 1991), which was enacted to enhance continental water quality. Therefore, it is necessary to find an adequate means of disposal for these residues in compliance with the environmental policies of the EU. One alternative that has been adopted in various countries around the world is the application of sewage sludge to soils as fertilizer (EPA, 1994), which is regulated in Europe by the directive 86/278/EEC (EU, 1986). The use of sewage sludge

* Corresponding author. Tel.: +34 600942437; fax: +34 982285926.
E-mail address: mrosa.mosquera.losada@usc.es (M.R. Mosquera-Losada).

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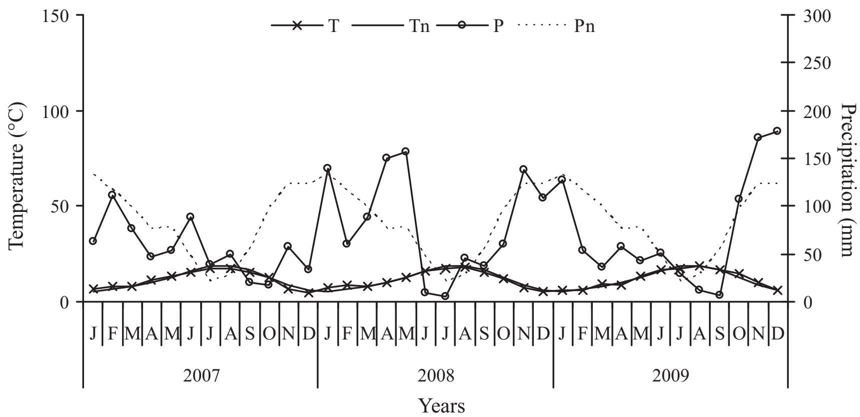

Fig. 1. Mean monthly precipitation and mean monthly temperature in 2007, 2008, and 2009 and mean normal data for the study area. *T*, mean monthly temperature (°C); T_n, mean normal temperature; *P*, mean monthly precipitation (mm); and P_n, mean normal precipitation.

as fertilizer is being promoted because it eliminates waste and reduces environmental pollution while imparting organic matter and macronutrients, particularly N and P, to the soil (Loehr et al., 1979; Rosswall, 1982; Beltrán et al., 2002; Mosquera-Losada et al., 2010). The study of the proportion of nitrogen that is readily mineralizable is important in determining the dose of sewage sludge that should be applied to the soil (Barry et al., 1986) in order to enhance both understory and overstory production in silvopastoral systems and to evaluate nitrate leaching risks (Simon and Le Corre, 1992; EPA, 1994). Sludge mineralization depends on soil types, pH and microbial soil activity (Smith, 1996) and this process is usually faster in agronomic than in more acid forest soils in North Western Spain. Moreover, the impact of sludge mineralization also depends on land use, being nitrate leaching risk usually higher in exclusive agronomic use (grasslands) than in silvopastoral systems due to the presence of the tree that may use the nitrate not employed by the pasture (Nair et al., 2008; Rigueiro et al., 2008).

Monterey pine and pasture growth response to sewage sludge inputs can be modified by the type of soil in which it is applied due to the different N and P availability in forestry and agronomic soils, which also affects nitrate leaching. Northwestern Spain forest soils usually have high organic matter content, which can act as a source of nutrients for crops. However, P availability is usually lower in forest soils due to high Al and Fe levels (Nair and Graetz, 2002). P and N availability can modify tree and pasture development as well as nitrate and P losses and pasture botanical composition (Campbell et al., 1993; Kellas et al., 1995; Nair et al., 2008). The impact of sewage sludge inputs in different soil types on nitrate and P cycling in agroforestry systems compared with treeless systems has not been evaluated in Western Europe.

The aim of this study is to evaluate the soil, productivity (tree and pasture) and environmental (nitrate and, P leaching) response to mineral or municipal sewage sludge inputs in grasslands and silvopastoral systems developed under Monterey pine established in agronomic and forest soils.

2. Materials and methods

2.1. Characteristics of the study site

The experiment was initiated in December 2006 through the use of 27 cilindric pots of about 2 m^3 (144 cm height × 134.5 cm width) that were installed in the town of Piugos (Lugo, Galicia, NW Spain, European Atlantic Biogeographic Region) at an altitude of 470 m above sea level and filled with soils. Soils are gleyic umbrisols (FAO classification) and Umbrept Inceptisols (USDA system). Fig. 1 shows the monthly mean precipitation and temperature values for 2007, 2008, and 2009 and the normal mean precipitation and temperature values of the study area. The total annual rainfall was 658.1, 1000, and 872.7 mm in 2007, 2008, and 2009, respectively. Meanwhile, the rainfall registered in the spring of 2007, 2008, and 2009 was 439.8, 604.5, and 368.4 mm, respectively. In general, these years were drier than the mean year (998.3 mm) for the study area. However, the mean monthly precipitation in 2007 was higher than the mean normal precipitation from June to August, which reduced the drought period found in 2008 and 2009 that limited pasture growth. The annual mean temperature was mild (12 °C).

Fifteen pots were filled with agronomic soil from Sarria (Lugo, Galicia, NW Spain) and the other with forest soil (12 pots) from Bascuas (Condesmo, Lugo, Galicia, NW Spain). In each pot, a lysimeter was installed at a depth of 135 cm to study the leaching of nutrients. The lysimiter was a PVC pipe of 2 cm of diameter introduced after making a hole in the pots. The tube is completely adjusted to the pot and no water is leached outside with the exception of the hole of the PVC pipe.

Initial agronomic soil analyses showed a highly acid KCl pH (4.46) (Faithfull, 2002), low soil organic matter (SOM: 36.3 g kg^{-1}) (Kowalenko, 2001), and a total N (1.9 g kg^{-1}) and P (0.3 g kg^{-1}) (Castro et al., 1990). Meanwhile, the forest soil analyses also had a highly acid KCl pH (4.27) (Faithfull, 2002), higher SOM (72 g kg^{-1}) (Kowalenko, 2001) and total P (0.8 g kg^{-1}) contents and a lower concentration of total soil N (1.8 g kg^{-1}) (Castro et al., 1990) than the agronomic soils. All heavy metal concentrations in both the agronomic and forest soils (Table 1) were below the maximum thresholds for the use of sewage sludge as fertilizer, as specified by the EU Directive 86/278/CEE (EU, 1986) and Spanish legislation under R.D. 1310/1990 (BOE, 1990).

2.2. Experimental design

The experimental design was a randomized block with three replicates and nine treatments. Treatments followed a design that consisted of a fractional factorial design of a 2*p* fully factorial, with "*p*" = 4 factors (2 levels per factor). The treatments established were chosen because they are the most traditional practices in the area of study (agronomic soil without tree, forest soil without pasture, and silvopastoral systems) in forest and agronomic lands. The treatments consisted of the following: (1) Agronomic soil + pasture sowing (Agronomic + P); (2) Agronomic soil + pasture sowing + sewage sludge (Agronomic + PS); (3) Agronomic soil + pasture sowing + mineral (Agronomic + PM); (4) Agronomic soil + pasture sowing + sewage sludge + tree (Agronomic + PST); (5) Agronomic soil + pasture sowing + mineral + tree (Agronomic + PMT); (6) Forest soil + sewage sludge + tree (For-

Table 1
Heavy metal concentrations in the agronomic soil and in the forest soil at the beginning of the experiment and legal limits established by European Directive 86/278 and Spain R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH < 7, maximum: soil pH > 7). –, element concentration below detection limit of the technique used in its determination.

Soil	Heavy metal concentrations (mg kg^{-1})					
	Cd	Cu	Cr	Ni	Pb	Zn
Initial agronomic soil	0.1	1	0.9	–	17.7	28.8
Initial forest soil	0.9	7.8	2	–	–	32.5
Spanish law limits	1–3	50–210	100–150	30–112	50–300	150–450

est + ST); (7) Forest soil + mineral + tree (Forest + MT); (8) Forest soil + pasture sowing + sewage sludge + tree (Forest + PST); and (9) Forest soil + pasture sowing + mineral + tree (Forest + PMT). The following physical parameters were used:

- Pasture sowing (P): the pasture was sown with a mixture of cocksfoot [*Dactylis glomerata* (L.)] var. Artabro (12.5 kg ha^{-1}) (Dg), ryegrass [*Lolium perenne* (L.)] var. Brigantia (12.5 kg ha^{-1}) (Lp), and white clover [*Trifolium repens* (L.)] var. Huia (4 kg ha^{-1}) (Tr) in December 2006.
- Tree (T): a one-year-old Monterey pine tree was planted in January 2007.
- Sewage sludge (S): an anaerobically digested sludge with an input of 320 kg total N ha^{-1} applied in December 2006.
- Mineral (M): in the Agronomic + PM, Agronomic + PMT, Forest + MT, and Forest + PMT treatments, 500 kg ha^{-1} of 8 (% N):24 (% P_2O_5):16 (% K_2O) were applied at the beginning of the years 2007, 2008, and 2009, and 40 kg of N ha^{-1} as calcium ammonium nitrate (26% of N) was applied after each harvest.

2.3. Sewage sludge

Anaerobically digested sludge was taken from the municipal waste treatment plant of Lugo. A calculation of the required amount of sludge was conducted according to its percentage of total N and dry matter content (EPA, 1994) and taking into account that around 25% of the total N from anaerobically digested sewage sludge is available in the first year after application. EU Directive 86/278/CEE (1986) and Spanish regulation R.D. 1310/1990 (BOE, 1990) regarding heavy metal concentrations in the application of sewage sludge to soil were also considered. The composition of the sewage sludge applied is summarized in Table 2.

Table 2
Chemical properties of the sewage sludge applied and legal limits established by European Directive 86/278 and Spain R.D. 1310/1990. Limits depend on soil pH (minimum: soil pH < 7, maximum: soil pH > 7).

Parameters	Values	
	Anaerobic sludge	Spanish law limits
Dry matter (%)	20.47	
pH	7.47	
N (g kg^{-1})	35	
P (g kg^{-1})	17.8	
K (g kg^{-1})	3.5	
Ca (g kg^{-1})	27.1	
Mg (g kg^{-1})	8.4	
Na (g kg^{-1})	1.5	
Fe (g kg^{-1})	17.9	
Cr (mg kg^{-1})	39.4	1000–1500
Cu (mg kg^{-1})	142.7	1000–1750
Ni (mg kg^{-1})	29.4	300–400
Zn (mg kg^{-1})	1248.56	2500–4000
Cd (mg kg^{-1})	0.7	20–40
Pb (mg kg^{-1})	84.4	750–1200
Mn (mg kg^{-1})	6.1	

2.4. Field samplings and laboratory determinations

Soil samples were collected at a depth of 25 cm, as described in R.D. 1310/1990 (BOE, 1990) in March 2008 and in January 2009. In the laboratory, the pH of the KCl was determined as 1:2.5 soil:0.1 M KCl (Faithfull, 2002). The total C content in the soil was determined by oxidation of the total organic matter with potassium dicromate and sulphuric acid. The excess of dicromate was valorated with Mohr salt (Kowalenko, 2001). The percentage of organic matter was calculated by multiplying the total C content of the soil by the de Van Bemmelen factor (1.724). Cation exchange capacity (CEC) was calculated as the sum of the concentrations of Ca, K, Mg, Na and Al expressed as cmol(+) kg^{-1} of soil, after extraction with 0.6 N $BaCl_2$ (Mosquera and Mombiela, 1986). The total soil N and total soil P concentrations were determined after micro-kjeldahl digestion with a TRAACS-800+ autoanalyzer, as described by Castro et al. (1990) (US-786-86 A method, for N and US-787-86 A method, for P (Bran+Luebbe, 1979)). The available P was measured after extraction with Mehlich 3 (Mehlich, 1985) with the TRAACS-800+ autoanalyzer (US-787-86 A method (Bran+Luebbe, 1979)) and water volume measured with a volumetric ware. Total nitrate leached was estimated by multiplying nitrate concentration and water leached in each sampling. Nitrate leached for a period in each sampling was summed to obtain the global period nitrate leached.

Water was extracted each week from the lysimeters unless drought caused a lack of water. Nitrate was determined according to Bremner (1965) using a continuous-flow analytical system (TRAACS-800+).

Tree height and diameter were measured with graduated ruler and caliper, respectively, in September 2009.

To estimate pasture production, botanical composition, and crude protein (CP) and P content of the pasture, two samples of pasture were randomly taken with an electric hand clipper at a height of 2.5 cm per pot (0.3 m^2 × 0.3 m^2) in May, June, and August 2007; in May and July 2008; and in May and June 2009 (autumn data were not used in this study). Later, the samples were labeled and transported to the laboratory, where the samples were weighed and separated by hand according to the different plant species and the senescent material. They were then dried at 60 °C for 72 h to determine the harvest pasture production and the botanical composition weight. The CP and P content of the pasture were determined after micro-kjeldahl digestion with a TRAACS-800+ autoanalyzer, as described by Castro et al. (1990).

2.5. Statistical analysis

The data obtained from soil, tree and pasture variables were analyzed with three 2-way ANOVAs (proc glm procedure) following the model $Y_{ij} = \mu + F_i + T_j + \varepsilon_{ij}$. The first ANOVA was performed to discern the effects of soil type (agronomic vs. forest) with mineral and sludge fertilization in Pasture + Tree (silvopastoral systems) with two levels of fertilization (F: sludge and mineral) × two types of soil (T: Forest and agronomic) and their interactions (treatments Agronomic PMT, Agronomic PST, Forest PMT and Forest PST); where Y_{ij}

is the studied variable; μ is the variable mean; F_i is the fertilizer factor i; T_j is the soil type factor j; and ε_{ij} is the error. The second 2-way ANOVA was performed to discern the effects of two levels of pasture vegetation (S: sown and unsown pasture) with two levels of fertilization (F: sludge and mineral) and their interactions on forest soil (Treatments Forest + PMT, Forest + PST, Forest + ST and Forest + MT) where Y_{ij} is the studied variable; μ is the variable mean; F_i is the fertilizer factor i; T_j is the vegetation factor j; and ε_{ij} is the error. The third 2-way ANOVA was performed to discern the effects of two levels of tree plantation (T: tree and no tree plantation) with two levels of fertilization (F: sludge and mineral) and their interactions on agronomic soil (Treatments agronomic + PMT, agronomic + PST, agronomic + PM and agronomic PS) where Y_{ij} is the studied variable; μ is the variable mean; F_i is the fertilizer factor i; T_j is the vegetation factor j; and ε_{ij} is the error. Finally a 1-way ANOVA of one factor with three levels of fertilization F (NF, mineral and sludge) was to discern the effects of fertilization on agronomic soil with herbaceous vegetation (treatments NF, agronomic PM and agronomic PS). The Tukey's HSD test was used for subsequent pair wise comparisons ($p < 0.05$; $a = 0.05$). The statistical software package SAS (2001) was used for all analyses.

3. Results

3.1. Soil

3.1.1. KCl soil pH, CEC, SOM percentage, total N and total and Mehlich P

The KCl soil pH, the CEC and the total soil levels of N and P and P extracted by Mehlich 3 in 2008 and 2009 are shown in Fig. 2. The soil pH was significantly affected by the type of soil ($p < 0.01$) in *Pasture + Tree* treatments in 2008 (agronomic: 4.91 vs. forest:

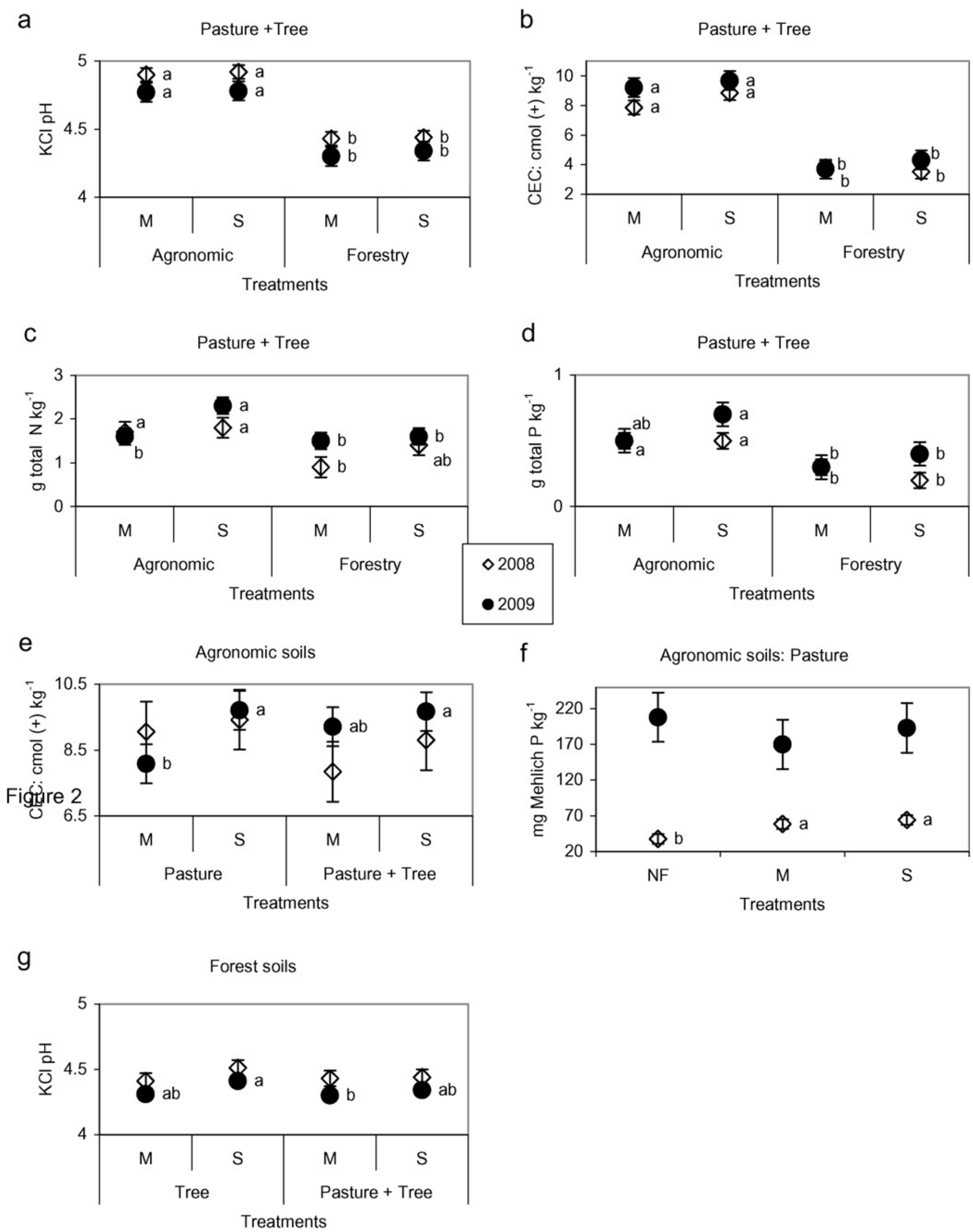

Fig. 2. Soil pH in KCl (a), CEC (cmol(+) kg^{-1}) (b), and total N (c) and P (d) concentrations in soil (g kg^{-1}) in Pasture + Tree treatments, CEC (cmol(+) kg^{-1}) in agronomic soils (e), P Mehlich 3 (mg kg^{-1}) in no forested agronomic soils (f) and soil pH in KCl in forest soils (g) in 2008 and 2009. M, mineral fertilization; S, sewage sludge fertilization. Different letters indicate significant differences between treatments within the same year. Vertical lines indicate mean standard error.

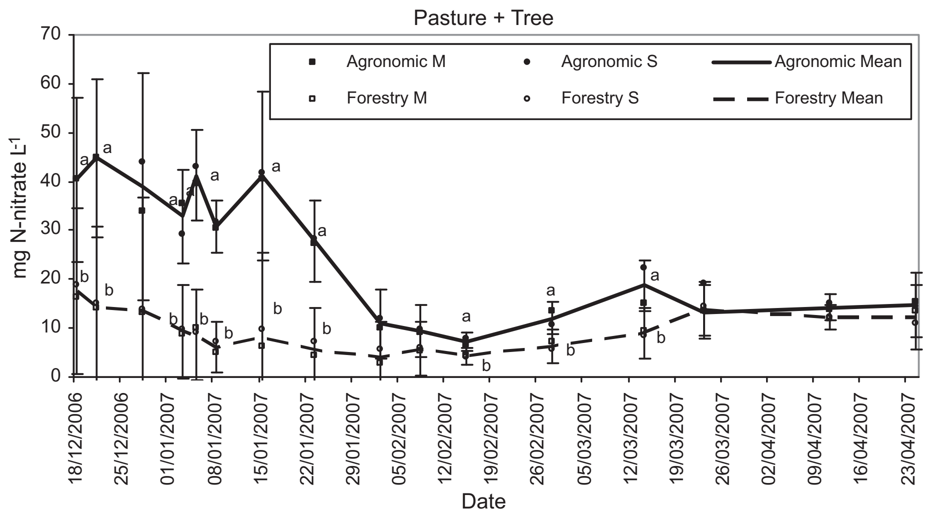

Fig. 3. Nitrate (mg N-nitrate L^{-1}) concentration in leached water in Pasture + Tree treatments. M, mineral; S, sludge. Different letters indicate significant differences between soil mean treatments. Vertical lines indicate mean standard error.

4.44) and 2009 (agronomic: 4.77 vs. forest: 4.33) as happened with CEC in 2008 ($p < 0.001$; agronomic: 8.35 cmol(+) kg $soil^{-1}$ vs. forest: 3.65 cmol(+) kg $soil^{-1}$) and 2009 ($p < 0.001$; agronomic: 9.44 cmol(+) kg $soil^{-1}$ vs. forest: 4.01 cmol(+) kg $soil^{-1}$). Soil total N and P were also significantly affected by the type of soil. Soil total N ($p < 0.01$ and $p < 0.001$) and P ($p < 0.001$ and $p < 0.01$) were significantly higher in agronomic soils than in forest soils in 2008 (agronomic: 1.7 g total N kg^{-1} and 0.5 g total P kg^{-1} vs. forest: 1.1 g total N kg^{-1} and 0.2 g total P kg^{-1}) and in 2009 (agronomic: 2 g total N kg^{-1} and 0.6 g total P kg^{-1} vs. forest: 1.6 g total N kg^{-1} and 0.3 g total P kg^{-1}). For the SOM percentage, no significant differences between the treatments were found ($p > 0.05$ (data not shown)). No differences in soil variables appeared between treatments when only *Forest soils* were taken into account, with the exception of KCl pH, which was significantly reduced when mineral fertilization was applied and tree and pasture was established compared with those treatments planted with trees and receiving sludge fertilization, but without pasture sowing. On the other hand, in the *Agronomic soils*, the soil CEC was significantly higher when the pasture was fertilized with sludge (9.69 cmol(+) kg^{-1} soil) than with mineral (8.65 cmol(+) kg^{-1} soil) in 2009 ($p < 0.01$), being P extracted by Mehlich 3 significantly ($p < 0.001$) improved when mineral or sludge fertilized was applied in 2008 and compared with no fertilization treatment in agronomic soils.

3.1.2. *Nitrate concentration in leaching water*

The significant effects of the treatments on nitrate concentrations in the leaching water are shown in Fig. 3. The nitrate concentration in the leaching water of agronomic soils was usually above the maximum set by the EU directive for drinking water in all treatments (<11.3 mg NO_3^--N L^{-1}) (EU, 1980) in December 2006 and in the first months of 2007. The nitrate concentrations of all treatments measured from May 2007 to 2009 were below the maximum allowed by the EU directive for drinking water (always below 5 mg NO_3^--N L^{-1}) and without differences between treatments (data not shown). With respect to the effects of the treatments in the *Pasture + Tree* pots at the beginning of the experiment, the results show that by 12 March 2007 the nitrate concentrations in the leaching water were significantly higher in Agronomic treatments sown with pasture and planted with trees than in the same Forest treatments. After this sampling date, no differences were found between treatments. However, the total nitrate leached was significantly higher ($p < 0.05$) when sewage sludge (3.05 g of nitrate until 24 May 2007) was applied in *Forest soils* compared with mineral fertilizers (1.89 g of nitrate until 24 May 2007). There were not differences between treatments in the subsequent years ($p > 0.05$).

3.2. *Trees*

3.2.1. *Tree heights and diameters*

Tree heights (179 cm) and basal diameters (41.25 mm) for each treatment were not significantly affected by any of the treatments in 2009.

3.3. *Pasture*

3.3.1. *Production*

Pasture production for the different fertilization treatments in the spring 2007, 2008, and 2009 can be observed in Fig. 4. Significant differences were detected between the treatments in all years ($p < 0.001$ in the spring of 2007, 2008, and 2009) with the exception of the spring 2007 and 2008 when the planting of trees were evaluated in agronomic soils and the spring 2008 when the three types of fertilization were compared in treeless pastures established in agronomic soils. The highest levels of pasture production were generally found in the spring of 2008 (2.2–13.4 Mg pasture ha^{-1}), while the lowest values were detected in the spring of 2009 (1.2–10.3 Mg pasture ha^{-1}). When the Agronomic and Forest treatments are compared (*Pasture + Tree* treatments), it is apparent that pasture production was lower in the Forest treatments (Forest + PMT and Forest + PST) fertilized with mineral or sludge than in Agronomic treatments (Agronomic + PMT and Agronomic + PST), with the exception of the first year of the study, when pasture production was similar in those pots fertilized with sewage sludge in forest soils (PST) to both Agronomic treatments (Agronomic + PMT and Agronomic + PST). Within the Agronomic treatments the fertilization with sewage sludge in no planted pots (Agronomic + PS) had higher pasture production than pots fertilized with mineral and planted with trees (Agronomic + PMT) in the last year of the study. On the other hand, in 2007, within the Agronomic treatments, pasture production was lower in no fertilized pots than in those fertilized with sewage sludge (Agronomic + PS). However, mineral fertilization reduced pasture production in the last year of the study compared with no fertilization treatment. Moreover, in the *Forest soils*, the mineral fertilization decreased pasture production (Forest + MT and Forest + PMT), while the fertilization with sewage sludge increased this variable (Forest + ST and Forest + PST) in 2007 and 2009. However, pasture production was reduced in

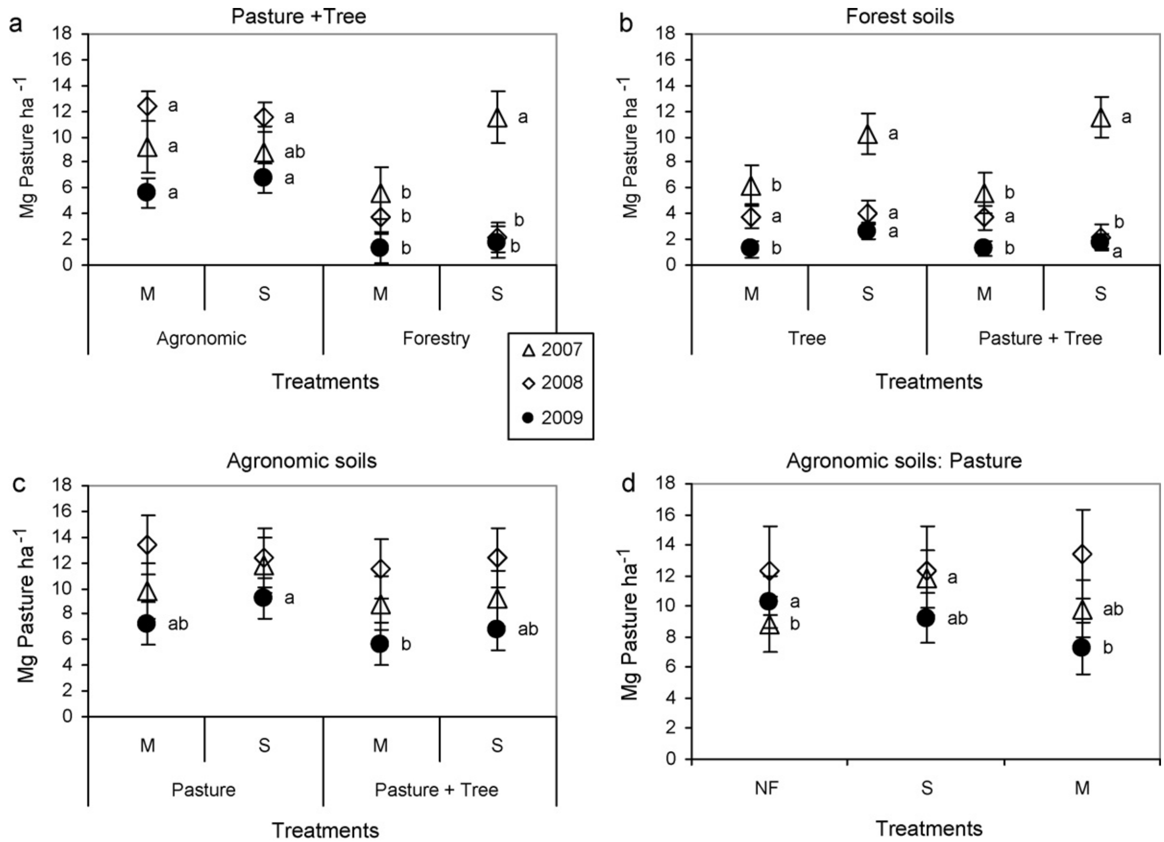

Fig. 4. Spring pasture production (Mg ha^{-1}) under Pasture + Tree (a), Forest (b), Agronomic (c) and Agronomic not afforested soils (d) in 2007, 2008, and 2009. Different letters indicate significant differences between treatments within the same year. Vertical lines indicate mean standard error.

forest soils when pasture was sown, tree was planted and sewage sludge was applied compared with the rest of the treatments in 2008.

3.3.2. Botanic composition

The significant ANOVA results of the weight proportion (% dry matter) of the sown species (cocksfoot, ryegrass, and white clover) and the most representative spontaneous species (chamomile (Cha) [*Chamomilla recutita* (L.)], creeping bentgrass (Crb) [*Agrostis stolonifera* (L.)], narrowleaf plantain (Np) [*Plantago lanceolata* (L.)], and yarrow (Yar) [*Achillea millefolium* (L.)]) in the pasture over the entire study period are shown in Table 3. There were other spontaneous species, though their contributions were minimal (data not shown). In general, the percentage of cocksfoot was significantly affected by treatments (Table 3) in all harvests of the spring–summer of 2007 and 2008, with the exception of the first harvest of 2007 and harvests of 2008 when the effect of planting trees within agronomic soils was evaluated. Moreover, all ANOVAs of cocksfoot were also significant in May 2009, but cocksfoot was not affected by treatments when the comparison of two types of soil (agronomic and forest) in *Pasture + Tree* pots was carried out in the first harvest of 2009. Regarding ryegrass, it was only affected by treatments in August 2007 (when (1) agronomic and forest soils were compared in *Pasture + Tree* pots and (2) treatments within forest soils were evaluated), in May 2008 in all treatments (with the exception of the evaluation of tree plantation in agronomic soils) and in May 2009 (when agronomic and forest soils in *Pasture + Tree* pots were compared and the effect of tree planting within agronomic soils was evaluated). The fertilization with mineral or sludge initially increased the proportion of cocksfoot in forest soils compared with agronomic soils in *Pasture + Tree* treatments (Fig. 5). Moreover, the positive effect of the mineral fertilization compared with the sludge fertilization on cocksfoot percentage in forest soils was more evident as the study advanced. On the contrary, agronomic soils had a significantly higher proportion of clover than forest soils mostly when sludge instead of mineral was applied in August 2007, May 2008 and July 2008.

Within the *Forest soils*, there was a positive effect of pasture sowing on the proportion of sown species (cocksfoot and ryegrass) in all harvests until May 2009 (Fig. 5). The establishment of cocksfoot in forest soils was better with mineral than with sludge fertilization. On the contrary, creeping bentgrass was the main pasture species found in forest unsown pots in 2007 and 2008 (Fig. 5). Creeping bentgrass percentage was significantly higher in unsown than sown forest treatments from the start of the experiment to July 2008, when the increment of the percentage of creeping bentgrass in unsown pots compared with sown pots was only significant if mineral fertilization was used. In May 2009, mineral fertilization improved the percentage of creeping bentgrass in unsown pots compared with those pots previously sown and fertilized with sludge.

Within *Agronomic soils*, the percentage of cocksfoot was significantly (Table 3) improved when sewage sludge (12.6% and 22.3%) instead of mineral (4.19% and 7.5%) was used in May and August 2007, respectively. On the contrary, the percentage of cocksfoot was significantly higher in mineral (44.17^{a}%) than sludge (12.5^{b}%) in no afforested pots in May 2009, but no differences were found in agronomic afforested pots regarding the percentage of cocksfoot (26.68^{ab}% and 22.32^{ab}% in mineral and sludge, respectively) in the same harvest (different superscript letters indicate significant differences between treatments). However, in May 2009, the use of sludge (11.42^{a}%) increased the proportion of ryegrass compared with mineral (1.15^{b}%) in no afforested pots, and sludge (1.80^{b}%) in afforested pots, but the percentage of ryegrass was similar in both

Table 3
ANOVA results for treatments with significant effects for sown grasses (cocksfoot (Dg), ryegrass (Lp), and white clover (Tr)), spontaneous species (chamomile (Cha), creeping bentgrass (Crb), narrowleaf plantain (Np) and yarrow (Yar)) in May, June, and August 2007; May and July 2008; and May 2009.

	Forest soil					Pasture + Tree					Agronomic soil			
	For	Fert	For × Fert	SEM		Soil	Fert	Soil × Fert	SEM		Silvo	Fert	Silvo × Fert	SEM
Sown species														
Tr May-08	*	*	*	5.2	Tr August-07	**	ns	ns	5.1	Dg June-07	ns	**	ns	6.7
Tr July-08	ns	*	ns	8.8	Tr May-08	**	ns	ns	8.5	Dg August-07	ns	*	ns	8.07
Dg May-07	***	ns	*	11.5	Tr July-08	***	ns	ns	7.5	Dg May-09	ns	**	**	13
Dg June-07	***	ns	ns	6.7	Dg May-07	***	**	*	8.6	Lp May-09	ns	ns	**	5.5
Dg August-07	***	ns	*	12.5	Dg June-07	**	ns	ns	9.0					
Dg May-08	***	ns	ns	17.3	Dg August-07	***	***	ns	9.4					
Dg July-08	***	ns	ns	15.2	Dg May-08	***	ns	ns	17.83					
Dg May-09	**	ns	ns	16.5	Dg July-08	*	*	ns	17.3					
Lp May-07	***	*	*	8	Lp August-07	ns	*	ns	16.3					
Lp June-07	***	ns	ns	12.8	Lp May-08	**	ns	ns	18.8					
Lp August-07	***	*	ns	12.8	Lp May-09	ns	ns	*	7.3					
Lp May-08	***	ns	ns	7.8										
Unsown species														
Crb May-07	***	ns	ns	24.6	Crb July-08	*	ns	ns	11.1	Crb May-07	**	*	ns	7.5
Crb June-07	***	ns	ns	24.8	Np July-08	*	*	ns	2.9	Crb May-08	ns	**	**	7.8
Crb July-08	**	ns	ns	19.1										
Crb May-09	**	ns	ns	21.8										
Yar July-08	**	ns	ns	10.2										

Agronomic soil		
	Fert	SEM
Sown species		
Dg May-07	**	2.7
Dg June-07	***	2.7
Dg August-07	**	8.4
Dg May-08	*	9.2
Dg July-08	*	6.8
Dg May-09	*	17.6
Lp May-08	*	17.1
Unsown species		
Crb May-08	***	7.04

SEM, mean standard error; ns, not significant.
* $p < 0.05$.
** $p < 0.01$.
*** $p < 0.001$.

afforested pots (4.51[ab]% in mineral afforested pots). Mineral fertilization also increased the percentage of cocksfoot compared with no fertilization in agronomic soils (Fig. 5).

Regarding the spontaneous species, it was found that chamomile was initially better established in agronomic (21.87%) than forest soils (0.02%) in *Pasture + Tree* treatments in May 2007. On the other hand, the percentage of narrowleaf plantain [*P. lanceolata* (L.)] was significantly higher when sewage sludge was applied in agronomic soils (5.91[a]%) compared mineral fertilization (0.05[b]%) in forest soils in July 2008, but the percentage of this species was similar in mineral fertilized agronomic soils (2.47[ab]%) and in sludge fertilized forest soils (1.63[ab]%) in the same harvest. Finally, yarrow was improved within *Forest soils* when no sown was carried out but mineral fertilization was applied (18.81[a]%) compared with sludge fertilization with sowing (0.55[b]) but similar to the percentage found in those pots unsown and fertilized with sludge (12.48[ab]%) or sown and fertilized with mineral (2.88[ab]%) in July 2008.

3.3.3. *Pasture concentrations of CP and P*

The concentrations of CP and P in the pasture in the spring 2007 and 2008 are presented in Fig. 6. The concentration of CP was significantly affected by the treatments in August 2007 (soil effect: $p < 0.001$) and in December 2008 (soil effect: $p < 0.01$) in *Pasture + Tree* treatments and revealed that CP was higher in agronomic than forest soils, with the exception of pasture grown in agronomic pots fertilized with sludge in December 2008. On the other hand, the concentration of P in pasture was significantly affected by treatments in June (soil effect: $p < 0.001$) and August (soil effect: $p < 0.001$) 2007 and in December 2008 (soil × fertilization interaction effect: $p < 0.05$; soil effect: $p < 0.001$) when the effect of the type of soil was evaluated. As happened with CP, the concentration of P was significantly higher in agronomic than forest soils, with the exception of pasture developed in agronomic soils fertilized with sludge which did not differ from the same forest treatment.

Within *Forest soils*, the lack of sowing increased the CP concentration of pasture. Mineral fertilization in unsown treatments significantly increased the concentration of CP compared with sludge and mineral fertilization in sown pots in August 2007 (tree effect $p < 0.001$) and only with sludge fertilization in June 2007 (tree effect: $p < 0.001$).

4. Discussion

All soil studied variables (KCl pH, CEC, total N and P), excluding organic matter, were significantly higher in agronomic compared with forest soils and explains the better initial fertility stage and pasture production found in the Agronomic than in the Forest treatments. Moreover, the highest presence of white clover in the Agronomic treatments probably increased pasture production because *Rhizobium* N fixation is performed by this species (González, 1992; Whitehead, 1995; Green et al., 1999; López-Díaz et al., 2009), which increases the input of N into the soil (Mosquera-Losada et al., 1999), and the subsequent consumption of this nutrient by grasses. Authors such as González (1992) indicate that 30% of white clover in pastures developed in North Western

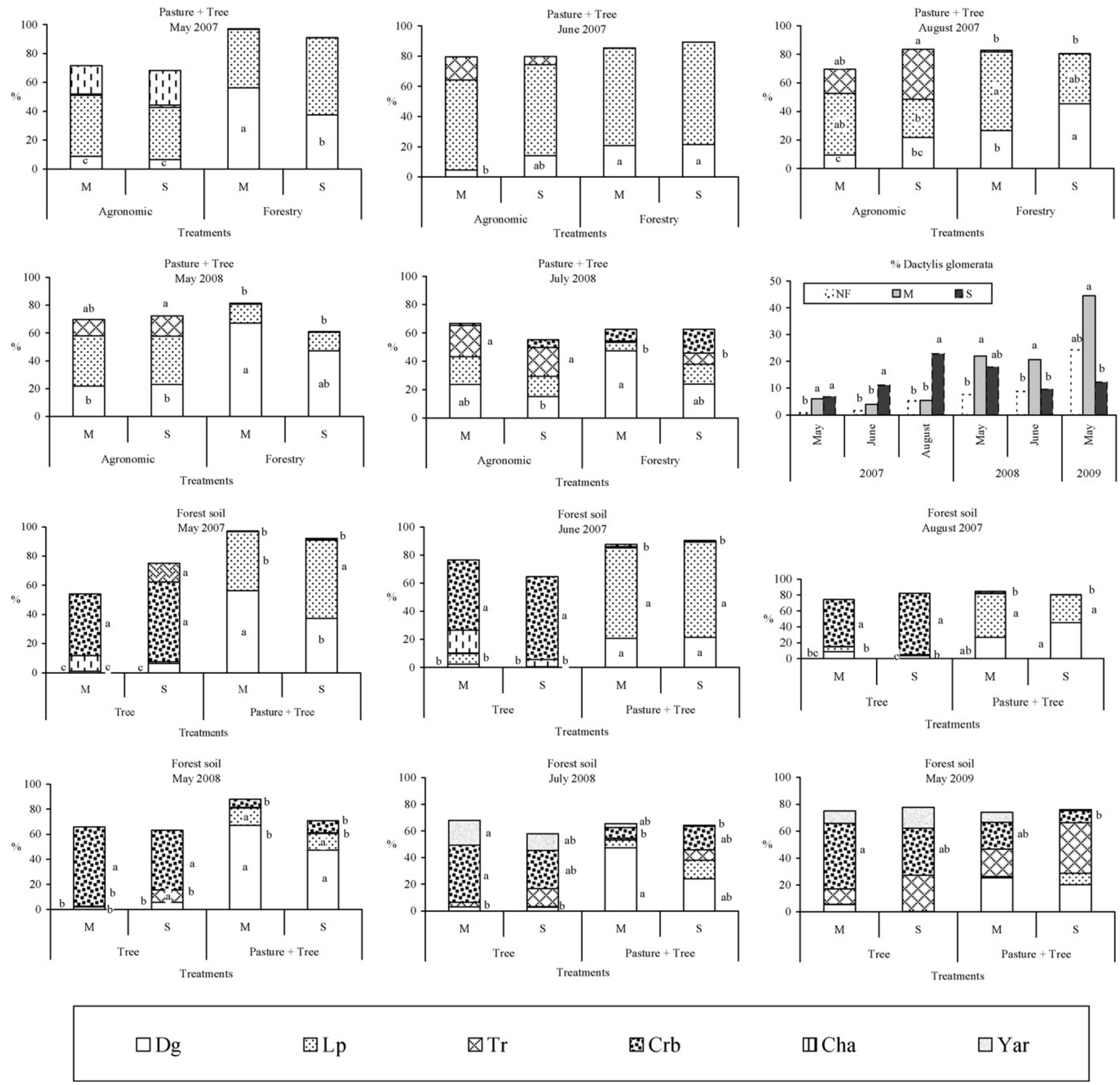

Fig. 5. Proportion in weight (% dry matter) of sown (Dg, cocksfoot; Tr, white clover; and Lp, ryegrass) and spontaneous species (Cha, chamomile; Crb, creeping bentgrass; and Yar, yarrow) in pasture under the different treatments in May, June, and August 2007; May and July 2008; and May and June 2009 of Pasture + Tree and Forest and Agronomic soils treatments. M, mineral; S, sludge; NF, no fertilization. Different letters indicate significant differences between treatments within the same harvest.

Spain can incorporate up to 250 kg ha^{-1} year^{-1} of N. In any case, pasture production ranges found in the present experiment were similar to those usually described in North Western Spain forest soils (0.5–6 Mg pasture ha^{-1}) (Mosquera-Losada et al., 2001) and in North Western Spain agronomic soils (6–12 Mg ha^{-1}) (Mosquera-Losada and González-Rodríguez, 1999).

Although soil pH was reduced from the first to the last year of the experiment, N and P levels in soil were usually higher if sludge instead or mineral were used in Pasture + Tree treatments developed in agronomic soils. Soil pH in 2009 was reduced in comparison to 2008 probably due to the N mineralization (the step from NH_4^+ to NO_3^- occurs, and H^+ is released into the soil solution media after leaching of NO_3^- by rainfall (Whitehead, 1995)) and tree and pasture cation extractions from the soil (Mosquera-Losada et al., 2006). In the last year of the experiment, soil total N was higher if sludge instead of mineral fertilizer had been previously applied in pasture and tree agronomic soils. The sludge nutrient release rate is slower than those from mineral fertilizers (EPA, 1994; Smith, 1996) and this would explain the extended effect of the sludge in time on soil total N variable. The same tendency was found with total soil P.

Water pollution from nitrate leaching, was only relevant at the start of the experiment when both tree and pasture were at the establishment phase, and therefore they were less efficient taking up soil nutrients like nitrate. Afterwards, the adequate establishment of pasture and tree limited nitrate leaching, even though mineral fertilizer inputs were annually performed. At the beginning of the present experiment (2006), the nitrate-N content of the soil solution exceed the 11.3 mg L^{-1} limit for drinking water set by the EU (1980). In the following samplings, the concentrations of nitrate in the leached water from the agronomic soils were also above the value of 11.3 mg NO_3^--N L^{-1}, and it was higher than that found in forest soils (Knight et al., 1989). High initial soil pH of agronomic

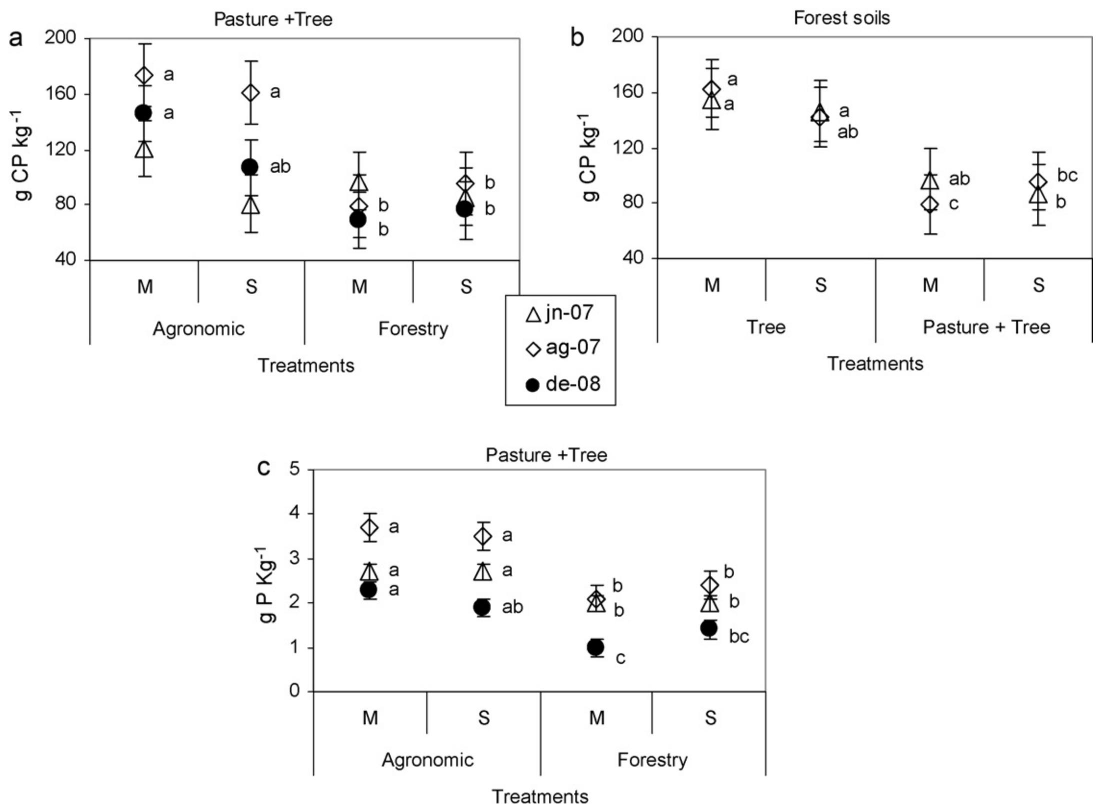

Fig. 6. Concentrations of crude protein (CP) in pastures (g kg^{-1}) under Pasture + Tree (a) and forest soil (b) treatments and P in pastures (g kg^{-1}) under Pasture + Tree (c) in the significant harvests of 2007, 2008, and 2009. M, mineral; S, sludge. Different letters indicate significant differences between treatments within each harvest. Vertical lines indicate mean standard error.

soil at the beginning of the experiment would explain a higher mineralization activity, and therefore nitrate leaching, in agronomic than in forest soils, which caused an increase of pasture production and the levels of CP. Moreover, the initial higher percentage of grasses instead of annual species in the Forest treatments than in the Agronomic treatments could also have reduced the leaching of nitrate in the forest soils, while taking into account that perennial grasses have a greater capacity for taking up soil nitrate than weeds (Humphreys et al., 2006; Abberton et al., 2008). Moreover, the initial establishment of dicotyledonous annual species, such as chamomile, could also have increased the nitrate leaching in the agronomic treatments when the annual species died, since dicotyledonous species are richer in N than monocotyledonous species (Hanley et al., 1992; Paré et al., 2006). The absence of annual species in the seed bank of the forest soils was probably the cause of the best initial establishment of the sown species in the forest soils that were simultaneously fertilized and sown than in the agronomic soils, where the annual species showed rapid-growth characteristics (abundant seed – rapid germination) (Grime et al., 2007; Mosquera-Losada et al., 2009).

There were no appreciably significant differences in the forest soils as a result of the different treatments of fertilization or sowing, mainly because most of the soil's biological activity was greatly restricted as pH was very low in the forest soils (Omil et al., 2007; Djukic et al., 2009). However, a positive effect of sewage sludge inputs on soil pH was detected when the less intensive system (no sowing) was compared with more intensive systems implying mineral fertilization and sown of pasture. The positive effect o sludge applications on soil pH in very acid soils were also described in soils receiving a higher total quantity of sludge than in the present experiment (López-Díaz et al., 2007). The improvement of soil pH caused by sludge applications may explain the increase of the total nitrate leaching and the pasture production in forest soils compared with mineral fertilizations. This result could be firstly explained by the residual effect of organic fertilizers compared with mineral fertilizers described by the EPA (1994) and secondly because the sludge added more Ca, Mg and micronutrients than the mineral fertilizer (Smith, 1996; López-Díaz et al., 2007; Mosquera-Losada et al., 2010).

Even though the forest soils were only significantly affected by the treatments in terms of KCl pH, they modified the botanical composition and CP in the initial samplings. However, these differences disappeared at the end of the study. The improvement in the percentage of ryegrass and cocksfoot species as a result of sowing caused a lower crude protein percentage than in the pastures of unsown forest soils. Sown species like ryegrass and cocksfoot are not usually adapted to the low fertility of forest soils, being less extractive than weeds in forest soils (Whitehead, 1995), thus reducing the proportion of N in the pasture. This reduction ultimately caused the reduction of the percentage of sown species at the end of the experiment in sown forest soils.

The response of the soil and pasture production variables to treatments within the agronomic soils did appeared two years after the experiment had begun, probably due to the better initial soil fertility. CEC was significantly increased in sludge fertilized treatments with or without tree planting than in mineral when trees were not planted, and it could be explained by the physical soil improvement caused by the sludge (Smith, 1996). Moreover, in 2009, those treatments with sludge fertilization in the agronomic soils without tree planting had higher pasture production than mineral fertilization treatment that had been previously planted. Mineral fertilization caused a direct increase of N, P, and K concentrations in soil, which can reduce other cation levels in soil, thus limiting pasture production (Whitehead, 1995). This effect can be seen in the lower CEC of the PM treatment compared with the rest of the agronomic treatments fertilized with the sludge in the last year of the study. Additionally, the presence of trees modified the soil conditions in the agronomic soil with the mineral fertilizer (PMT), which may also have reduced pasture production in this treatment due to the high extractions performed by both crops (tree and pasture), compared with agronomic treeless systems fertilized with mineral.

The initial improvement of soil P availability as a result of fertilization (mineral or sludge) was also previously described (Allen et al., 2006; Nair et al., 2007) and may help to explain the high production of pasture in sewage sludge fertilized pots compared with no fertilization in agronomic soils. Moreover, cation extraction may have reduced pasture production in mineral treatment compared with no fertilization at the end of the study.

In 2009, the Monterey pine heights and diameters varied from 164–193 cm to 49.5–58 mm, respectively. The tree diameters and heights were greater than those described by López-Díaz et al. (2009) in a study carried out in agrarian soils in North Western Spain, with a pH close to neutral (pH 6.3), and by Sánchez-Rodríguez (2000) in the Northwest of Spain. The higher growth of trees in our study could be explained by the high precipitation rate found in the summer during planting, which usually increases initial tree growth (Rigueiro-Rodríguez et al., 2000; Mosquera-Losada et al., 2006) and may have reduced the differences between treatments. A similar result was found by Rigueiro-Rodríguez et al. (2010), in which fertilization treatments initially modified the Monterey pine response, but differences between the treatments disappeared when a humid summer for tree growth occurred. Tree plantation did not affect nitrate leaching probably because trees were too young to uptake nitrogen from soil when the nitrate concentrations in soil were high. However, tree plantation reduced pasture production in agronomic soils fertilized with mineral and KCl pH, and CP concentration in forest soils compared with the less intensive treeless pastures (López-Díaz et al., 2007).

The concentrations of CP (58–183 g kg^{-1}) and P (1.6–4.8 g kg^{-1}) were similar to the concentrations described by Grime et al. (2007) (1.5–4.5 g kg^{-1}) and Whitehead (1995) (80–250 g kg^{-1}), respectively, with the exception of the CP concentrations in those harvests performed in the summer, which were usually lower due to the usual seasonal evolution of CP caused by the different pasture species' phenological growth states (Whitehead, 1995). The concentration of CP in the pasture did not reach the minimum requirements for the maintenance of live weight in sheep (94 g kg^{-1}) (NRC, 1985), horses (85 g kg^{-1}) (NRC, 1989), and goats (60 g kg^{-1}) (Lamand, 1981) in the summer harvests of 2007 and 2008. The concentration of P in all other harvests did meet the requirements for maintenance of live weight in sheep (1.6–3.7 g kg^{-1}) (NRC, 1985), horses (2 g kg^{-1}) (NRC, 1989), and goats (2.5 g kg^{-1}) (Lamand, 1981).

5. Conclusion

Fertilizer management, sowing and plantation practices caused different effects on the agrarian and forestry soils in our study. Agronomic soils were more fertile than the forestry soils due to the high pH and microbial activity of the former, which increases nitrate and P availability, and therefore the risk of leaching. Better nutrient availability in the agronomic soils increased pasture production as initially did sludge instead of mineral fertilization in forest soils due to the inputs of other nutrients done by sewage sludge compared with mineral, which probably increased mineralization in forest soils. Nitrate leaching was only relevant at the start of the experiment when trees and pasture were not enough developed to uptake nitrate. On the other hand, the soil variables evaluated in this study, with the exception of KCl, were not modified by fertilization or sowing treatments in the forest soils. However, the production, botanical composition and quality of the pasture developed in the forest soils were positively by sludge inputs instead of mineral due to the greater amount of nutrients applied with the former and the pH increase that sludge caused. The presence of grasses like cocksfoot or ryegrass was enhanced by sowing in forest soils. However, due to the low soil fertility, the quality of the pasture at the time these species were sown was low, and the sown grass species disappeared shortly after the establishment of the experiment. Finally, soil fertility was better preserved with the sludge than mineral fertilizer within the agronomic soils due to the broad range of nutrient applied with the former. Therefore, the use of the sludge as fertilizer allows nutrient recycling of this residue in poor soils and increases productivity and preserves fertility compared with mineral fertilizer at short (forest soils) and medium (agronomic soils) term.

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References

Abberton, M.T., Marshall, A.H., Humphreys, M.W., Macduff, J.H., Collins, R.P., Marley, C.L., 2008. Genetic improvement of forage species to reduce the environmental impact of temperate livestock grazing systems. Adv. Agron. 98, 311–355.

Allen, S.C., Nair, V.D., Graetz, D.A., Jose, S., Nair, R.P.K., 2006. Phosphorus loss from organic versus inorganic fertilizers used in alleycropping on a Florida Ultisol. Agric. Ecosyst. Environ. 117, 290–298.

Barry, E., Keefe, O., Axley, J., 1986. Evaluation of nitrogen availability indexes for a sludge compost amended soil. J. Environ. Qual. 15, 121–128.

Benavides, R., Douglas, E.G., Osoro, K., 2009. Silvopastoralism in New Zealand: review of effects of evergreen and deciduous trees on pasture dynamics. Agroforest. Syst. 76, 327–350.

Beltrán, E.M., Miralles de Imperia, L.R., Porcel, M.A., Delgado, M.M., Beringola, M.L., Martin, J.V., 2002. Effect of sewage sludge compost application on ammonium-nitrogen and nitrate-nitrogen contents of an olive grove soil. In: Proceedings of the 12th International Soil Conservation. Organization Conference, vol. 3. Tsinghua University Press, Pekin, China, pp. 395–402.

BOE (Spanish Official Bolletin), 1990. Real Decreto 1310/1990 29 de Octubre de 1990, que regula la utilización de los lodos de depuración (Royal Decree 1310/1990 29th October 1990, that regulates the use of sewage sludge). Ministerio Agricultura, Pesca y Alimentación, Madrid, Spain, http://www.boe.es/boe/dias/1990/11/01/pdfs/A32339-32340.pdf (verified 15.12.09).

Bremner, J.M., 1965. Nitrogen availability indexes. In: Black, C.A., Evans, D.D., Esminger, L.E., Clark, F.E. (Eds.), Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties. American Society of Agronomy, Madison, pp. 1324–1345.

Bran+Luebbe, 1979. Methods for the Chemical Analysis of Water and Wastes. Bran+Luebbe, Norderwstedt, Germany.

Campbell, C.D., Atkinson, D., Jarvis, P.G., Newbould, P., 1993. Effects of nitrogen fertilizer on tree/pasture competition during the establishment phase of a silvopastoral system. Ann. Appl. Biol. 124, 83–96.

Castro, P., González, A., Prada, D., (in Spanish) 1990. Simultaneous determination of nitrogen and phosphorus in pasture samples. XXX Reunión Científica de la Sociedad Española para el estudio de los pastos 30, 200–207.

Djukic, I., Zehetner, F., Mentler, A., Gerzabek, M.H., 2009. Microbial community composition and activity in different Alpine vegetation zones. Soil Biol. Biochem. 42, 155–161.

EFMA (European Fertilizer Manufacturers Association), 2009. Annual report 2009. Modern agriculture feeds the world and helps protect environment. http://www.efma.org/ (verified 03.01.09).

EPA (Environmental Protection Agency), 1994. Land application of sewage sludge. A guide for land appliers on the requirements of the federal standards for the use of disposal of sewage sludge, 40 CFR Part 503.

EU (European Union), 1980. Council Directive of 15 July 1980 relating to the quality of water intended for human consumption (80/778/EEC). Official J. Eur. Community L229, 11–29.

EU (European Union), 1986. DOCE n° L 181 04/07/1986. Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment and, in particular of the soil, when sewage sludge is used in agriculture. http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:31986L0278:ES:HTML (verified 15.12.09).

EU (European Union), 1991. DOCE n° L 135 30/05/1991. Council Directive 91/271/EEC of 21 May 1991 concerning urban waste–water treatment. http://aragon.ugt.org/mambiente/Normativa/pdf/aguas/aguas_eu_91–271.pdf (verified 15.12.09).

EU (European Union), 2005. Council regulation (EC) n° 1698/2005 of September 2005 on support for rural development by the European Agricultural Fund for Rural Development. http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2005:277:0001: 0040:EN:PDF (verified 15.12.09).

Faithfull, N.T., 2002. Methods in Agricultural Chemical Analysis. CAB International Publishing, Wallinford, UK.

González, A., 1992. Las praderas de gramínea y trébol y la fertilización nitrogenada en Galicia. In: Barrecheguren, M.A., Cabrero, M., Collar, J., Moreno, J., Santaolalla, G., Zea, J. (Eds.), 100 anos de Investigación Agraria. Xunta de Galicia, Santiago de Compostela, Spain, pp. 383–393.

Graves, A.R., Burgess, P.J., Liagre, F., Pisanelli, A., Paris, P., Moreno, G., Bellido, M., Mayus, M., Postma, M., Schindler, B., Mantzanas, K., Papanastasis, V.P., Dupraz, C., 2008. Farmer perceptions of silvoarable systems in seven European countries. In: Rigueiro-Rodríguez, A., McAdam, J.H., Mosquera-Losada, M.R. (Eds.), Agroforestry in Europe. Advances in Agroforestry, vol. 6. Springer, Netherlands, pp. 67–86.

Green, J.J., Alifragis, D., Baddeley, J.A., Bagnaresi, U., Bohne, H., Bulfin, M., 1999. Impact of understorey vegetation on tree production in European agrofororestry systems. Grassland Sci. Eur. 4, 171–176.

Grime, J.P., Hodgson, J.G., Hunt, R.R., 2007. Comparative Plant Ecology. A Functional Approach to Common British Species, 2nd ed. Castlepoint press, Dalbeattie, Scotland.

Hanley, T.A., Robbins, C.T., Hagerman, A.E., McArthur, C., 1992. Predicting digestive protein and digestible dry matter in tannin-containing forages consumed by ruminants. Ecology 73, 537–541.

Hawke, M.F., 1991. Pasture production and animal performance under pine agroforestry in New Zealand. Forest Ecol. Manage. 45, 109–118.

Humphreys, M.W., Yadav, R.S., Cairns, A.J., Turner, L.B., Humphreys, J., Skot, L., 2006. A changing climate for grassland research. New Phytol. 169, 9–26.

Knowles, R.L., 1991. New Zealand experience with silvopastoral systems: a review. Forest Ecol. Manage. 45, 251–267.

Knight, D., Elliot, P.W., Anderson, J.M., 1989. Effects of earthworms upon transformations and movement of nitrogen from organic matter applied to agricultural soils. In: Hansen, J.A., Henriksen, K. (Eds.), Nitrogen in Organic Wastes Applied to Soils. Academic Press, London, UK, pp. 59–80.

Kellas, J.D., Bird, P.R., Cumming, K.N., Kearney, G.A., Ashton, A.K., 1995. Pasture production under a series of *Pinus radiata*-pasture agroforestry systems in South-west Victoria, Australia. Aust. J. Agric. Res. 46, 1285–1297.

Kowalenko, C.G., 2001. Assessment of Leco CNS-2000 analyzer for simultaneously measuring total carbon, nitrogen and sulphur in soil. Commun. Soil Sci. Plant Nutr. 32 (13–14), 2065–2078.

Lamand, M., 1981. Metabolis et bésoins en oligoelements des chèvres. In: Proceedings of the International Conference of Nutrition and Feeding Systems, Tours, France.

Loehr, R.C., Jewell, W.J., Novak, J.D., Clarkson, W.W., Friedman, G.S., 1979. Land Application of Wastes. Van Nostrand Reinhold, New York, USA.

López-Díaz, M.L., Mosquera-Losada, M.R., Rigueiro-Rodríguez, A., 2007. Lime, sewage sludge and mineral fertilization in a silvopastoral system developed in very acid soils. Agroforest. Syst. 70, 91–101.

López-Díaz, M.L., Rigueiro-Rodríguez, A., Mosquera-Losada, M.R., 2009. Influence of pasture botanical composition and fertilization treatments on tree growth. Forest Ecol. Manage. 257, 1363–1372.

Mehlich, A., 1985. Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant. Commun. Soil Sci. Plan. Anal. 15 (12), 1409–1416.

Mosquera-Losada, M.R., González-Rodríguez, A., Rigueiro-Rodríguez, A., 1999. Ecología y manejo de praderas (Ecology and Management of Grasslands). Ministerio de Agricultura, Pesca y Alimentación, Santiago de Compostela, Spain.

Mosquera-Losada, M.R., González-Rodríguez, A., 1999. Pasture production in Northern Spain dairy system. New Zealand J. Agric. Res. 42, 125–132.

Mosquera, A., Mombiela, F., 1986. Comparison of three methods for the determination of soil Al in an unbuffered salt extract. Commun. Soil Sci. Plant Nutr. 17 (1), 97–113.

Mosquera-Losada, M.R., Rigueiro-Rodríguez, A., López-Díaz, M.L., 2001. Sewage sludge fertilization of a silvopastoral system with pines in northwestern Spain. Agroforest. Syst. 53, 1–10.

Mosquera-Losada, M.R., Fernández-Núñez, E., Rigueiro-Rodríguez, A., 2006. Pasture, tree and soil evolution in silvopastoral systems of Atlantic Europe. Forest Ecol. Manage. 232, 135–145.

Mosquera-Losada, M.R., Muñoz-Ferreiro, N., Rigueiro-Rodríguez, A., 2010. Agronomic characterization of different types of sewage sludge: policy implications. Waste Manage. 30, 492–503.

Mosquera-Losada, M.R., Rodríguez-Barreira, S., López-Diaz, M.L., Fernández-Núñez, E., Rigueiro-Rodríguez, A., 2009. Biodiversity and silvopastoral system use change in very acid soils. Agric. Ecosyst. Environ. 131, 315–324.

Nair, V.D., Graetz, D.A., 2002. Phosphorus saturation in spodosols impacted by manure. J. Environ. Qual. 31, 1279–1285.

Nair, V.D., Nair, P.K.R., Kalmbacher, R.S., Ezenwa, I.V., 2007. Reducing nutrient loss from farms through silvopastoral practices in coarse-textured soils of Florida, USA. Ecol. Eng. 29, 192–199.

Nair, P.K.R., Gordon, A.M., Mosquera-Losada, M.R., 2008. Agroforestry. In: Jorgensen, S.E., Fath, B.D. (Eds.), Ecological Engineering, Encyclopedia of Ecology. Elsevier, Oxford, UK, pp. 101–110.

NRC (National Research Council), 1985. Nutrient Requirements of Sheep. National Academic Press, Washington, USA.

NRC (National Research Council), 1989. Nutrient Requirements of Horses. National Academic Press, Washington, USA.

Omil, B., Mosquera-Losada, M.R., Merino, A., 2007. Responses of a non-N-limited forest plantation to the application of alkaline-stabilized dewatered dairy factory sludge. J. Environ. Qual. 36, 1765–1774.

Paré, K., Chantigni, M.H., Carey, K., Johnston, W.J., Dionne, J., 2006. Nitrogen uptake and leaching under annual bluegrass ecotypes and bentgrass species: a lysimeter experiment. Crop Sci. 46 (2), 847–853.

Rigueiro-Rodríguez, A., Mosquera-Losada, M.R., Gatica-Trabanini, E., 2000. Pasture production and tree growth in a young pine plantation fertilised with inorganic fertilizers and milk sewage in northwestern Spain. Agroforest. Syst. 48, 245–254.

Rigueiro-Rodríguez, McAdam, A., Mosquera-Losada, M.R., 2008. Agroforestry systems in Europe: productive ecological and social perspectives. In: Rigueiro-Rodríguez, A., McAdam, J., Mosquera-Losada, M.R. (Eds.), Advances in Agroforestry Series Agroforestry in Europe, vol. 3. Springer, Dordrecht, The Netherlands, pp. 43–66.

Rigueiro-Rodríguez, A., Castro, S., Mosquera-Losada, M.R., 2010. Effects of dose and period of sewage sludge application on soil, tree and pasture components in a *Pinus radiata* D. Don silvopastoral system. Agroforest. Syst. 79, 237–247.

Rosswall, T., 1982. Microbiological regulation of the biogeochemical nitrogen cycle. Plant Soil 67, 15–34.

Sánchez-Rodríguez, F., 2000. Growth and productivity of *Pinus radiata* in Galicia. Ph.D. Thesis. University of Santiago de Compostela, Spain (in Spanish).

SAS, 2001. SAS/Stat Userĭs Guide: Statistics. SAS Institute Inc., Cary, NC, USA.

Smith, S.R., 1996. Agricultural Recycling of Sewage Sludge and the Environment. CAB International, Wallingford, UK.

Simon, J.C., Le Corre, L., 1992. Fertilization des cultures annuelles et lessivage de lĭazote nitrique, 129. Fourrages, pp. 3–10.

Whitehead, D.C., 1995. Grassland Nitrogen. CAB International, Wallingford, UK.

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5 Crop Production Department. University of Santiago de Compostela. Campus
6 de Lugo. 27002- Lugo

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