Eesti Maaülikool
Estonian University of Life Sciences

THE EFFECT OF PLANT RESIDUES MANAGEMENT AND FERTILIZATION ON HERBAGE GROWTH AND ORGANIC CARBON CONTENT IN SOIL

TAIMEJÄÄTMETE JA VÄETAMISE MÕJU TAIMEDE KASVULE JA ORGAANILISE SÜSINIKU SISALDUSELE MULLAS

KARIN KAUER

A Thesis
submitted in application for the degree of Doctor of Philosophy
in agriculture

Väitekiri
Filosoofiadoktori kraadi taotlemiseks põllumajanduse erialal

Tartu 2012

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LIST OF ORIGINAL PUBLICATIONS

This thesis is based on the following papers, which are refferred to by their Roman numbers.

I Kõlli, R., Köster, T. **Kauer, K**. 2007. Organic matter of Estonian grassland soils. *Agronomy Research*, 5(2), 109-122.

II **Kauer, K**., Raave, H., Viiralt, R., Köster, T., Noormets-Shansky, M., Laidna, T., Keres, I., Parol, A., Selge, A. 2009. Effect of clippings management on turfgrass sward productivity and nitrogen content in the clippings and soil. *Agronomy Research*, 7, 311-316.

III **Kauer, K**., Raave, H., Köster, T., Viiralt, R., Noormets, M., Keres, I., Laidna, T., Parol, A., Selge, A. 2012. The decomposition of turfgrass clippings is fast at high air humidity and moderate temperature. *Acta Agriculturae Scandinavica, Section B – Plant Soil Science*, 62, 224-234.

IV Sammul, M., **Kauer, K**., Köster, T. 2012. Biomass accumulation during reed encroachment reduces efficiency of restoration of Baltic coastal grasslands. *Applied Vegetation Science*, 15, 219-230.

V **Kauer, K**., Kõlli, R., Viiralt, R., Köster, T., Noormets, M., Laidna, T., Keres, I., Parol, A., Varul, T., Selge, A., Raave, H. 2012. The effect of cut plant residues management and fertilization on the dry matter yield of swards and on carbon content in soil. *Communications in Soil Science and Plant Analysis* (accepted for publication).

VI **Kauer, K**., Laidna, T., Keres, I., Köster, T., Noormets, M., Parol, A., Selge, A., Viiralt, R., Raave, H. 2012. Impact of returned clippings on turfgrass growth as affected by rate of N fertilizer, weather and season. Crop Science (submitted).

CONTRIBUTIONS:

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Data collection	RK	**KK**	**KK**	**All**	**KK**	**KK**, HR
Data analysis	**All**	**KK**	**KK**	**All**	**KK**	**KK**, HR
Preparation of manuscript	**All**	**All**	**All**	**All**	**All**	**All**

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ABBREVIATIONS

ANOVA	analysis of variance
BD	bulk density
C	carbon
CI	confidence interval
Cl	clover
CO_2	carbon dioxide
Ctot	total carbon
D	soil sampling depth
DM	dry matter
DMY	dry matter yield
G	grass
GC	grass-clover
K	potassium
LSD	least significant difference
M_d	weight of dry material
M_f	weight of fresh material
M_i	initial plant material dry matter mass (g) in the nylon bag
M_t	plant material dry matter mass (g) in the nylon bag at time t, when bags were removed from the field
N	nitrogen
N_i	initial amount (mg) of N in the bag's decomposing material
N_m	mineralized N
N_t	amount (mg) of N in the bag's decomposing material at time t
Ntot	total nitrogen
NREC	apparent N recovery
$NREC_R$	apparent N recovery from residues
NUE	N-use efficiency
NUE_R	N-use efficiency from residues
NUP	nitrogen uptake by plant
P	phosphorous
RRI	returned plant residues effect
RRM	(plant) residues removed (from plots)
RRT	(plant) residues returned (to plots)
SOC	soil organic carbon
SOM	soil organic matter
TG	turfgrass

1. INTRODUCTION

Agricultural background

Soil organic carbon (SOC) plays a key role in the global carbon (C) cycle. Agricultural soils act as an important C reservoir, containing three times as much C as the atmosphere ($2.25 \cdot 10^{12}$ *versus* $0.75 \cdot 10^{12}$ t C) and five times as much as in forests and other vegetation (Post et al., 1982; Jobbágy & Jackson, 2000). In Estonia, it is estimated that a total of 323 ± 46 million t SOC is retained in mineral soil (Kõlli et al., 2010). Soil organic matter (SOM) has a role in the storage and availability of nutrients, improving tilth, air and water movement, water retention, and decomposition processes in soil (Gregorich et al., 1994). The amount of C stored in agricultural soils depends on soil type, local climate and other site-specific conditions, such as land use and land management policies. To protect or increase the existing SOC pool by sequestration of atmospheric C could prove crucial in mitigating the global greenhouse effect over time (IPCC, 2001; Guo & Gifford, 2002). This positive effect is thought to result from a reduction in the loss of SOC in the absence of soil tillage, while at the same time high C input is maintained through plant fixation. SOC pools are the balance between C input via primary productivity and output via decomposition processes (Amundson, 2001).

Agricultural practices and land use can cause changes in plant cover as well as associated changes in SOC stocks (Post & Kwon, 2000). In general, there is more SOC under grasslands than under cropland (Cole et al., 1993; Jackson et al., 1996) due to several factors, including infrequent soil disturbance, greater plant residue returns, higher root biomass, manure applications, and dung return during grazing.

Roots are the main source of organic matter on production grasslands (e.g., silage, hay, grazing). On set-aside grasslands and turfs where above-ground biomass is not used, plant remains also could be considered a source of organic matter if left on the field to decompose after mowing. It is generally assumed that differences in input amounts (not quality of the input material) are responsible for observed variations in SOC storage (Catovsky et al., 2002; Skinner et al., 2006). On the other hand, it is known that SOC dynamics are affected by the identity and specific structure of chemical substrates entering the soil (Orwin et al., 2006; Meier & Bowman, 2008), which implies that input material quality does have some influence.

Scope of the thesis

Plants return a wide range of C substrates to the soil system. The substrate decomposition rate is determined by their chemical nature, which depends on species composition and growth stages. The overall purpose of this thesis is to investigate the content and stock of SOC in selected species of grassland soils that have been put to various uses. This thesis also studies the effect of various management measures (e.g., fertilization; mowing frequency; removal of plant residues after mowing, or returning them to the surface) on the growth of various grass swards.

2. REVIEW OF THE LITERATURE

2.1. Soil organic matter

Soil organic matter (SOM) is either mixed or associated with the mineral part of soil and thus has a very complex and heterogeneous composition. SOM includes plant, animal, and microbial residues in all stages of decomposition (Post & Kwon, 2000).

Soil organic carbon (SOC) stock plays an important role in the global biochemical C cycle (Schlesinger, 1990; Batjes, 1996). All SOC originally comes from the atmosphere and is captured by plant photosynthesis and converted into plant material. Plant C enters the SOC pool either as above-ground biomass, litter, or root material (Catovsky et al., 2002; Bardgett et al., 2005). Decomposition of these materials leads to the formation of organic material in soil (Swift et al., 1979). The balance between C input and output via decomposition processes controls SOC accumulation within ecosystems (Olson, 1963).

2.2. Agriculture land management

Land management is a key factor controlling biosphere C dynamics (IPCC, 2001; Guo & Gifford, 2002). Changes in land use affect soil properties, including C and nitrogen (N) cycles (Potter et al., 1996; Houghton 1999; Guo & Gifford, 2002). Various land management and tillage methods affect SOC distribution in the soil profile, microbial activity, and the balance between nutrient mineralization and immobilization, thereby changing soil quality.

SOC reduction is known to occur during processes where natural grasslands are converted into arable land (Mann, 1986; Post & Mann, 1990; Davidson & Ackerman, 1993). Decreases in SOC are caused by reduced inputs of organic matter, improved decomposition conditions, and tillage effects that remove the amount of physical protection that otherwise prevents decomposition. Conventional tillage such as ploughing increases SOM reduction by breaking apart soil aggregates and improving soil aeration (Balesdent et al., 1990).

Depending on ecological conditions, every soil type has specific SOC flows throughout the soil cover (Batjes, 1996; Körchens et al., 1998;

Yakimenko et al., 1998; Kõlli et al., 2010). Soil texture plays an important role in stabilizing the SOC effect, as increasing clay and carbonate content decrease C outputs (Paul 1984; Körchens et al., 1998). Low-intensity tillage systems, which leave large amounts of plant residues on soil surfaces, also inhibit SOC reduction. Converting arable land into grasslands is one option for reducing the content of atmospheric carbon dioxide (CO_2) that arises from organic material decomposition (Guo & Gifford, 2002) while increasing SOC accumulation (Lal, 2004). Compared to arable land, higher SOC content in grasslands is explained partly by greater C input to soil (Jackson et al., 1996; Lal & Bruce, 1999) and infrequent soil disturbance (Nyborg et al., 1999).

It is known that converting arable land to grasslands controls both SOC and N distribution in the soil profile (Steinbeiss et al., 2008b). Ploughing arable fields leads to a homogeneous distribution of plant remains, while the input of plant material in grasslands is controlled by above-ground biomass and root distribution, as 70–75% of the root biomass is located in the top 15 cm of soil (Gill et al., 1999). SOC and N concentrations increase in the main rooting zone; reduced root biomass in the deeper horizons cause SOC reduction in deeper soil layers. SOC distribution in soil profiles also varies strongly with vegetation type (Jobbágy & Jackson, 2000).

2.3. Organic matter decomposition

Decomposition of plant residues is not only one of the fundamental processes in agroecosystems, but also is responsible for the recycling of nutrients released from decomposing organic matter, which sustains plant growth and soil quality. Decomposition is a biological process whereby complex organic substances from dead material are physically broken down and transformed into simpler organic and inorganic molecules by saprophytic fungi and bacteria (Juma, 1998). Living organisms use plant C residues as an energy source along with N for building cell structure (Swift et al., 1979; Benbi & Richter, 2002). By breaking down C structures and rebuilding new ones (or storing C into their own biomass), soil biota play the most important role in nutrient cycling processes and provide soil with sufficient nutrients to harvest a healthy crop.

In general, bacteria break down easily-decomposable organic material, which results in nutrients such as N becoming available for uptake by other

organisms; this process is called mineralization. During decomposition, different products besides nutrients also are released: CO_2; energy; water; and resynthesized organic C compounds, which are less decomposable than the original plant material. Fungi break down the less decomposable organic matter and retain those nutrients in the soil as fungal biomass. Just like bacteria, fungal waste products become SOM and are used by other organisms. Subsequent decomposition of dead material and modified organic matter results in the formation of a more complex organic matter called humus (Juma, 1998). Humus affects soil properties because as it slowly decomposes, it darkens soil color, intensifies soil aggregation and aggregate stability, and increases cation exchange capacity (i.e., the ability to attract and retain nutrients).

2.3.1. Factors affecting organic matter decomposition

The decomposition of plant residues is the result of complex processes controlled by the quality of organic matter, environmental factors, contact between plant residues and soil, and decomposing particle size (Swift et al., 1979; Stott et al., 1986; Heal et al., 1997; Martens, 2000).

2.3.1.1. Quality of organic matter

The ability of soil microorganisms to decompose and mineralize organic matter depends on the biochemical composition of the organic material (e.g., N content, C/N ratio, types of C compounds) (Heal et al., 1997; Gunnarsson & Marstrop, 2002; Trinsoutrot et al., 2002). In general, plants contain the same classes of C compounds (e.g., amino acids, organic acids, sugars, fructans, hemicelluloses, cellulose, and lignin), but the proportions of each (which depend on species and maturity) could be different and may influence the degree and rate of decomposition (Martens, 2000; de Neergaard et al., 2002).

The decomposition of organic material begins with simple compounds. At first, non-structural C compounds (e.g., amino acids, organic acids, sugars, fructans) are decomposed followed by compounds with more complex structures (e.g., hemicellulose, cellulose, lignin) (Van Soest, 1982; Gunnarsson & Marstrop, 2002; Gunnarsson et al., 2008). Hemicellulose, cellulose, and lignin are structural polysaccharides, which are the main

components of cell walls. Structural polysaccharides are responsible for conferring rigidity to cell walls, thereby allowing plants to grow as erect structures (Wagner & Wolf, 1999). Hemicelluloses are groups of structural carbohydrates that surround cellulose fibrils and cement them together; they consist of mostly branched chains of cellulose-like sugar units bound together, but with a lower degree of polymerization than cellulose (Wagner & Wolf, 1999; Gunnarsson, 2003). Cellulose is the most abundant carbohydrate in the world, amounting to about 20–40% of all plant dry matter (DM) (Van Soest, 1982). Cellulose has a much higher degree of polymerization compared to hemicellulose (Kögel-Knabner, 2002). Lignin is a large molecule consisting of phenolic groups composed of aromatic rings with three C side chains (Gunnarsson, 2003). Lignin's size and complexity lead to a slower decomposition rate compared to that of cellulose and hemicellulose; thus, its decomposition products have a long residence time in soil, which enhances SOC sequestration through the formation of complexes with other organic molecules (Hättenschwiler & Vitousek, 2000).

Decomposition and mineralization of plant residues also are controlled by N availability and the C/N ratio of the organic matter (Frankenberger & Abdelmagid, 1985; Trinsoutrot et al., 2000). In many studies, the initial N content of residues has been shown to be the main factor predicting decomposition kinetics (Trinsoutrot et al., 2000; de Neergaard et al., 2002). Thus, both initial N content and C/N ratio may be considered as parameters for predicting the decomposability of plant material. Plant materials with a C/N ratio <20 may result in net N mineralization, while those with a C/N ratio >20 tend to cause net immobilization (Schornberg et al., 1994; Wagner & Wolf, 1999).

Incorporation of plant material into soil generally stimulates microbial growth and activity. In early decomposition of plant residues with higher C/N ratios, net immobilization of soil N often occurs (Reinertsen et al., 1984), as more N is needed by developing microorganisms than the substrate provides. If N is the limiting nutrient for microbial growth (and thus for decomposition), then microorganisms compete with plants for N that would be actively taken up from the surrounding environment; thus, the absolute N concentration in plant residues would increase compared to the initial amount.

2.3.1.2. Environmental factors

Temperature and moisture conditions are the main environmental factors influencing the activity of decomposers involved in the decomposition process (Dalias et al., 2001a, b; Pietikäinen et al., 2005). During the decomposition process, an increase in soil temperature and moisture generally results in higher rates of microbial activity along with increased rates of plant residue reduction (Stott et al., 1986). Microbial activity is generally predicted to increase rapidly up to a temperature of about 30 °C. Wang et al. (2000) suggested that higher temperatures accelerate SOC decomposition only when soil moisture is adequate, and inhibits decomposition when soil moisture is limited. An optimal temperature for microbial activity is reached between 35 °C and 45 °C, while the optimal moisture content for organic matter decay is 50–60% (McKinley & Vestal, 1985). At higher temperatures, the temperature influence on decomposition rate is reduced; thus, increasing temperature by the same number of degrees at lower temperatures accelerates decomposition more than the same increase does at higher temperatures (Kirschbaum, 1995; Dalias et al., 2001a).

2.3.1.3. Soil contact and plant residue particle size

Contact between plant residues and soil also affects decomposition (Douglas et al., 1980; Ambus & Jensen, 1997). The decomposition of surface-placed plant residues is generally slower than of buried residues (Seneviratne et al., 1998; Coppens et al., 2006) because moisture is best stored in soil plant residues (Parr & Papendick, 1978). Plant residues spread on the soil surface will normally be exposed to more variable temperature and moisture conditions than will residues buried in the soil (Shomberg et al., 1994). These variable conditions may greatly slow down the decomposition of plant residues on the soil surface.

Decomposition is also influenced by decomposing particle size. There is a positive correlation between decomposition rate and the decreasing particle size of plant material (Bremer et al., 1991; Jensen, 1994). Small particles offer a relatively larger surface area, thereby increasing the possibilities for microbial attack and activity (Angers & Recous, 1997; Gunnarsson, 2003).

2.4. Options for increasing soil organic carbon content

There is a positive correlation between the amount of C input and SOC content (Parton et al., 1995; Karlen & Cambardella, 1996). Thus, increasing grassland productivity could be one important option to improve SOC content (Catovsky et al., 2002; Skinner et al., 2006). However, many long-term field observations show that although plant material is incorporated into soil in large quantities, SOC content does not necessarily increase (Campbell et al., 1991; Körner & Arnone, 1992; Gill et al., 2002). These results suggest a negative relationship between C input and SOC conservation (Gill et al., 2002). The addition of fresh plant residues to soil could cause native SOC decomposition, which is known as the 'priming effect' (Kuzyakov, 2006). The mechanisms involved in the priming effect are unclear (Fontaine et al., 2003); it is commonly accepted that low SOC quality limits the amount of energy available for soil microbes, which, in turn, reduces the SOC mineralization rate (Paul & Clark, 1989). Fresh material is abundant with energy-rich C compounds, so introducing them into soil not only stimulates microbial activity, but also increases microbe biomass (Dalenberg & Jager, 1989); soil microbes typically are limited in C (Smith & Paul, 1990), so the addition of fresh material increases SOM mineralization.

SOM in grasslands originates primarily from root death and decomposition (Gill & Burke, 2002). Root decomposition is a major source of C and nutrient turnover in grasslands (Seastedt, 1988; Dornbush et al., 2002). Through root turnover and rhizodeposition, roots maintain SOC (Anderson & Coleman, 1985) and more complex soil food webs that regulate important nutrient transformations (Neher, 1999). Root material containing lignified tissues and other structural components, which provide a more recalcitrant material than shoots (Rasse et al., 2005). These structures also may physically protect decomposable compounds embedded within them, thereby further decreasing decomposability (Chesson, 1997; Gorissen & Cotrufo, 2000).

Besides roots, SOC input also is mediated through plant residues, especially in set-aside grasslands mowed only 1–2 times per year; reduced mowing prevents brush formation, while plant residues accumulate in large amounts when left to decompose on the sward surface (Harivandi et al., 2001). Mowed biomass contains large amounts of nutrients, so removing plant residues from sward results in major losses of N and other nutrients (Haynes & Goh, 1980). Leaving plant residues onsite not only reduces the

need for additional nutrients (i.e., mineral N) to support sward growth, but also can save significant amount of fossil energy that otherwise would be used for picking up and treating the clippings as waste.

2.4.1. Fertilization of grasslands

Fertilization has been used for centuries to increase forage production in grasslands (Billings et al., 2006). Fertilization results in increased below-ground production as well as above-ground production (Russel & Williams, 1982), which contributes to increased SOC.

Generally, N fertilization may change SOM quality (i.e., C/N ratio) due to biotic and abiotic stabilization of mineral N into SOM (Šimek et al., 1999). The relationship between decomposition and external N availability is not clear (Fog, 1988; Knorr et al. 2005). It is believed that SOC stock increases with the C/N ratio of the material (i.e., roots or plant residues) introduced into the soil (Parton et al., 1995; McGuire et al., 2001); thus, increasing N availability may reduce SOC stock (Hunt et al., 1988; Mack et al., 2004). Some studies, however, have shown how SOC stock increases with higher N availability (Campbell et al., 1991; Magill & Aber, 1998), while other studies have reported that N availability is not influenced by SOC stock (Pastor et al., 1987; Hunt et al., 1988; Prescott, 1995). Across studies, the differential response to N addition partly can be explained by differences in the chemistry of plant residues (Sinsabaugh et al., 2002).

N fertilization is a major component in land management and thus has a considerable impact on soil as well as on soil microbial community. N fertilization may profoundly impact below-ground decomposers by modifying microbial composition, which affects the production of soil enzymes involved in the decomposition of SOM and plant litter (Fog, 1988; Saiya-Cork et al., 2002). Sinsabaugh et al. (2005) found that increasing N availability decreases the quantity of soil enzymes responsible for recalcitrant-C decomposition. Studies conducted in other managed ecosystems showed that N fertilization suppressed soil fungi, which led initially to a bacteria-dominant community (Bardgett et al., 1996), followed by preservation of recalcitrant substances, and then an overall increase in SOC storage. It also has been found that an increase in soil N availability may stimulate the activity of cellulolytic enzymes such as soil cellulose (Fog et al., 1988; Berg & Matzner, 1997) while reducing SOC storage. As a consequence, the N effect on decomposition depends on

the chemical composition of the organic matter (Sinsabaugh et al., 2002). Apparently, the effects of soil N availability on SOM decomposition and SOC storage rely on organic matter chemistry such as the relative abundance of lignin and cellulose (Yao et al., 2009).

2.4.2. Importance of plant species diversity on carbon and nitrogen turnover in grasslands

In grasslands, recent research has demonstrated that the diversity of plant species plays an important role for C transfer into the soil and is able to modify SOC storage under given land-use schemes (Tilman et al., 2006; Steinbeiss et al. 2008b). The interactions of various plant species with soil have been shown to influence primary productivity, SOC, and N sequestration (Nyborg et al., 1999; Catovsky et al., 2002; Tilman et al., 2006; Fornana & Tilman, 2008; Steinbeiss et al., 2008a, b). If plant species diversity influences the size of any one of these pools (i.e., either through a change in the quantity of material entering the pool or in the turnover rate of the pool), then the entire ecosystem productivity could be significantly affected.

Several studies have addressed diversity-function relationships and found positive effects of species richness on productivity and soil processes (Hooper et al., 2005). However, these effects often were driven by plant community composition rather than by plant species or functional group richness (Hooper et al., 2005; De Deyn et al., 2008, 2009). De Deyn et al. (2009) found that changes in SOC and N pools were related more to above-ground biomass or the presence of specific plant species than to plant species richness or total community biomass. For instance, diversity effects on primary productivity in grasslands appeared to depend strongly on the presence of N-fixing legumes (Spehn et al., 2002; Fornana & Tilman, 2008). Legumes increase soil organic N through symbiotic N fixation with rhizobial bacteria (Guretzky et al., 2004). N-fixing legumes (e.g., *Trifolium pratense*, which is typically associated with unfertilized species-rich grasslands) are widely recognized as keystone grassland plant species that influence both soil N availability and overall plant community production (Hopkins & Wilkins, 2006; Van der Heijden et al., 2008). However, their role in grassland SOC storage is less clear; it has been suggested that N-fixing legumes have the potential to promote SOC sequestration (Soussana et al., 2004; Fornana & Tilman, 2008; De

Deyn et al., 2008, 2009). The presence of legumes significantly increased root biomass production through net soil N accumulation, and it is likely that legume-derived N is qualitatively important for building up SOM and storing more C (Drinkwater et al., 1998). Legumes have high litter quality (i.e., low C/N), a high litter decomposition rate, and low nutrient-use efficiency; furthermore, because of symbiotic relationships, legumes have large effects on N availability and N supply rates in many N-limited natural and agricultural systems (Chapin et al., 1986).

The varying effect of different plant species on SOC content is due to differences in chemical composition (especially C compounds). The total concentration of hemicelluloses can vary from 15% to 35% in both legumes and grasses depending on age, species, and the morphology of plant parts (Nelson & Moser, 1994). On the other hand, legumes such as *Trifolium pratense* are known to contain more branched hemicelluloses and large amounts of pectic substances (Aman, 1993). According to Gunnarsson & Marstorp (2002), branched hemicelluloses decompose more rapidly than rigid hemicelluloses. Compared to legumes, grasses generally contain higher concentrations of cellulose (Chesson et al., 1985); however, lignin content has been found to be higher in legumes than in grasses (de Neergaard et al., 2002; Gunnarsson et al., 2008).

3. HYPOTHESIS AND AIMS OF THE STUDY

The main hypothesis of this thesis is that plant residues left to decompose on sward surface not only affect plant growth and SOC content, but also are influenced by various land management measures (e.g., fertilization, mowing frequency) as well as the composition of plant species. The different effect of plant residues is based on variations in their quality, which affects residue decomposition rates and the amount of recycled C and N.

The aims of the study were:

1. To investigate the decomposition of various plant residues left on sward surfaces as well as N mineralization during decomposition.

2. To explain how the decomposition dynamics of plant residues is influenced by their initial chemical composition (i.e., N content and C/N ratio).

3. To investigate how leaving plant residues to decompose on sward surfaces affects herbage growth and SOC content.

4. To explain the integrated effects of plant residues and fertilization on SOC content.

4. MATERIAL AND METHODS

4.1. Background of experimental site

The experiment was carried out from 2003 to 2008 at the Eerika Experimental Station of the Estonian University of Life Sciences (58°23'32" N, 26°41'31" E, elevation 60 m above sea level). According to the World Reference Base for Soil Resources (WRB) classification system (FAO, 1998), the experimental field soil was *Stagnic Luvisol* with a sandy loam soil texture. The humus horizon (A) content was 64.0% sand (2.0–0.02 mm), 7.3% silt (0.02–0.002 mm), and 28.7% clay (<0.002 mm); its depth was 25 cm (Figure 1). Beneath the humus layer were four horizons arranged sequentially as follows: (i) a yellowish-brown Bw ferralic-accumulation horizon (depth: 25–40(46) cm); (ii) a whitish eluvial Ewg horizon (depth: 40(46)–60(78) cm); (iii) an illuvial clay-accumulation Bt horizon (depth: 60(78)–84(90) cm); and (iv) a parent material of horizon C (depth: 84(90) – 100+ cm).

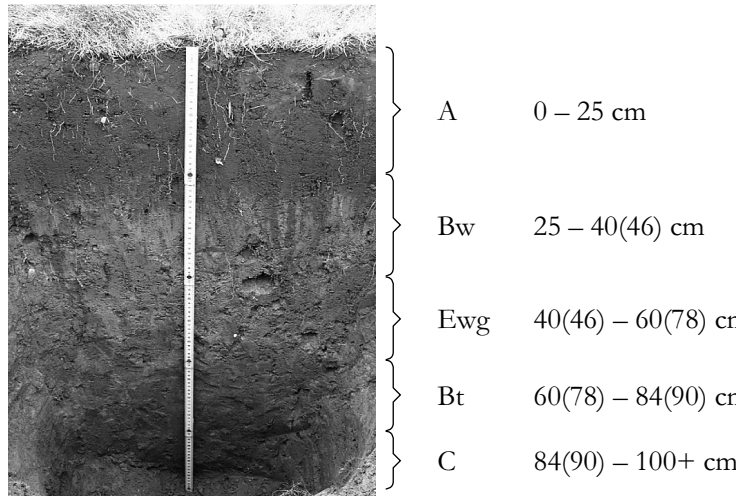

Figure 1. The experimental field soil profile.

The field was previously under barley for three years. Each autumn, straw was harvested and the field was ploughed afterwards. In May 2003, before establishing the swards (i.e., sowing), soil samples from depths up to 20 cm were collected and analyzed. At the beginning of the experiment, the total nitrogen (Ntot) content was 1.49 mg g^{-1}, SOC was 14.7 mg g^{-1}, pH_{KCl} was 5.5, available phosphorous (P) was 39.6 mg kg^{-1}, and potassium (K) was 79.7 mg kg^{-1}. P and K soil contents were determined by the AL-method.

Before seeding the experimental field, the site was cultivated, stones were removed, and the field was smoothed by rolling. Two seed mixtures were sown: (i) a turfgrass (TG) mixture (hereafter 'TG sward') consisting of *Festuca rubra rubra* c 'Kauni' and *Poa pratensis* c'Esto' and (ii) a mixture of grass (G) and white clover (Cl) (hereafter 'GC sward' (*Phleum pratense c*' 'Tika 34%, *Lolium perenne c* 'Raidi' 38% and *Trifolium repens c* 'Tooma' 28%). All cultivars were bred in Estonia. The TG mixture sowing rate was 200 kg ha^{-1} (germinating seed), with each species contributing 50% in terms of germinating seed numbers. The GC mixture sowing rates were 5.5 kg ha^{-1} for *Phleum pratense*, 6.1 kg ha^{-1} for *Lolium perenne*, and 4.6 kg ha^{-1} for *Trifolium repens*. The swards was unfertilized between sward establishments (i.e., the period from May 2003 until May 2004).

4.2. Experimental design

The TG sward experiment was arranged as a 2 x 6 factorial and set out in a randomized complete block design with four replicates. The factors were as follows: (i) two plant residues treatments, with residues removed (RRM) or residues returned (RRT); and (ii) seven applied fertilizer rates (kg ha^{-1}). Both RRM and RRT treatments occurred in two adjacent blocks separated by a 1 m band of *Festuca rubra commutata*. Individual treatment plots were 10 m^2. No watering was applied.

The GC sward experiment was arranged as a 2 x 2 factorial and set out in a randomized complete block design with four replicates. The factors were as follows: (i) two plant residues treatments (RRM or RRT); and (ii) two applied fertilizer rates (kg ha^{-1}). Both RRM and RRT treatments occurred in two adjacent blocks separated by a 1 m band of *Festuca rubra commutata*. As with TG sward, individual treatment plots were 10 m^2 and no watering was applied.

4.3. Management of swards

4.3.1. Fertilization

The TG sward fertilizer treatments were as follows: $N_0P_0K_0$ (hereafter 'TGN0', or unfertilized control), $N_{80}P_{11}K_{48}$ (TGN80), $N_{160}P_{22}K_{96}$ (TGN160), $N_{240}P_{34}K_{144}$ (TGN240), $N_{320}P_{45}K_{192}$ (TGN320) and $N_{400}P_{56}K_{240}$ (TGN400). NH_4NO_3, $Ca(H_2PO_4)_2$, and KCl were used as the sources of N,

P, and K fertilizers, respectively. Depending on the ratio of N, P, and K, fertilizers were applied by hand to plots in two to four splits; N fertilizer was applied in four splits: (i) at the beginning of May (one week after the start of the growing season), and during (ii) the first 10 days of June, (iii) the first 10 days of July, and (iv) the first 10 days of August. P fertilizer was applied in two splits: (i) at the end of April, and (ii) at the end of September. K fertilizer was applied in three splits: (i) at the end of April, (ii) during the first 10 days of June, and (iii) at the end of September.

The GC sward experimental treatments were as follows: $N_0P_0K_0$ (hereafter 'GCN0', or unfertilized control) and $N_{80}P_{26}K_{50}$ (GCN80). N fertilizer rates were applied in July seven days after the second cutting. P and K fertilizers were applied in spring just after beginning of the growing season (i.e., at the end of April).

4.3.2. Mowing

TG sward was cut 15–20 times at a height of 5 cm during the growing season by using traditional Estonian lawn mowing techniques. Cutting took place when the sward height was approximately 7.5 cm, and carried out on average once a week, except during periods of drought and in autumn, when cutting occurred less frequently (i.e., about once every two weeks). A rotary mulching lawn mower (Partner 5553 CMDEW) with a bag attachment (for collection of plant residues) was used to cut the plots. After each cutting, the harvested material was removed from the bag and weighed. Subsamples (100 g) were taken for determination of DM and N content (but only during the fourth year, 2007). For RRT treatment plots, fresh plant residues were returned immediately after weighing, spread evenly by hand over the area where they had been removed, and then mulched with a lawn mower. The end results were intended to resemble how the residues would have looked had they been mulched throughout mowing.

GC sward were cut 3–5 times during the growing season with a frontal bar mower (i.e., MF-70). Sward was cut for the first time when the grasses were at the end of tillering, and leaf tubes were beginning to form. Cutting took place when the sward height was approximately 20 cm. After each cutting, the harvested material was collected and weighed. Subsamples (100 g) were taken for determination of DM and N content (but only during the fourth year, 2007). For RRT treatment plots, fresh plant

residues were returned immediately after weighing and spread evenly by hand over the area where they had been removed.

4.4. Measurements, analyses, and calculations

4.4.1. Dry matter yield

Swards dry matter yield (DMY) was measured between 2004 and 2008. In this thesis, only mowed plant material is considered under DMY (excluding material left as stubble and root biomass). After mowing, cut material from experimental plots (including RRT and RRM treatments) was weighed. Subsamples were collected for DM content measurement, which was performed in four replicates.
The DM content (%) of biomass was determined by drying the sample in a forced-drought oven for six hours at 105 °C and calculated using the following formula:

$$DM = (M_d * 100)/M_f$$

where M_d is the weight of dry material (g), and M_f is the weight of fresh material (g).
The DMY (kg ha^{-1}) was calculated by using the DM content and the mass of fresh biomass.

4.4.2. Decomposition of plant residues

For TG sward, the decomposition of plant residues was studied from 2006 to 2007, during four periods of 8–10 weeks each (referred to as Period I, II, III, and IV; Table 1). Periods were chosen so that residue decomposition could be studied at different times during the growing season. For TG sward, decomposition was investigated under four fertilization variants (i.e., TGN0, TGN80, TGN160, and TGN400) and without separating the grass species. Plant lengths studied during decomposition were approximately 2–3 cm.

Table 1. Swards, experimental duration, and sampling weeks (2006–2007).

Species/ Mixtures	Period	Duration of experiment	Year	Total number of exposed bags	Sampling weeks
TG[1]	I (8)[5]	15 May–10 July	2006	64	2, 4, 6, 8
TG	II (8)	13 Sept–8 Nov	2006	80	1, 2, 4, 6, 8
TG	III (10)	26 Oct–4 Jan	2006	48	2, 5, 10
TG	IV(8)	16 May–10 July	2007	48	2, 4, 8
GC[2]	IA (8)	30 May–25 July	2006	64	2, 3, 5, 8
Cl[3]	IVA (8)	31 May–27 July	2007	64	2, 4, 6, 8
G[4]	IVA (8)	31 May–27 July	2007	64	2, 4, 6, 8

[1]TG = turfgrass; [2]GC = grass-clover; [3]Cl = white clover; [4]G = grass; [5]the number of weeks per decomposition period is given in parentheses.

For GC sward, the decomposition of plant residues also was studied in 2006 and 2007. In both years (referred to as Period IA for 2006 and Period IVA for 2007), plant residues were left on the sward to decompose for eight weeks after the first cut. The decomposition time intervals studied are presented in Table 1. In 2006, G and Cl decomposition was studied in a mixture containing the same ratio of G and Cl residues as when originally grown on the sward. In 2007, G and Cl residues were separated and their decomposition dynamics studied separately. Plant lengths studied during decomposition depended on their length at the first cutting (i.e., ~15 cm).

At the beginning of each period (i.e., 15 May, 13 Sept, and 26 Oct 2006, and 16 May 2007 for TG sward; 30 May 2006 for GC sward; and 31 May 2007 for separated G and Cl residues) and directly after cutting, a 100-g sample of fresh herbage was collected. For GC sward, an average of 10 samples (one handful each) was taken from random cut places.

From each sample, a subsample of 20 g of fresh plant residues was put into 20×20 cm nylon bags with a 1.5 mm mesh size (Figure 2).

Figure 2. Placement of nylon bags used for studying plant residue decomposition.

Each filled nylon bag was fixed with clamps and placed into the same plot thatch layer from where it had been harvested. Per treatment, the number of nylon bags used in different periods was 12–20 and depended on how many times the bags were planned to be removed from plots. In Period I, for example, bags were removed at four different times over eight weeks so that every fertilizing variant received 16 bags (i.e., four bags per replication).

At the beginning of the experiment, the remaining plant sample parts were used to determine the Ntot and total carbon (Ctot) content and the

C/N ratio (TG sward in Table 2; GC sward in Table 3). The cellulose and lignin content in TG sward residues also were measured, but only in Period IV.

Table 2. Ntot, Ctot, and C/N ratio of TG sward residues (Period I – IV); cellulose and lignin content (Period IV)

Parameter	TGN0	TGN80	TGN160	TGN400
		Period I		
Ntot, mg g^{-1}	24.3^{a1}	29.3^{b}	30.5^{b}	39.2^{c}
Ctot, mg g^{-1}	418.6^{a}	418.7^{a}	420.9^{a}	417.6^{a}
C/N	17.3^{c}	14.3^{b}	13.8^{b}	10.6^{a}
		Period II		
Ntot, mg g^{-1}	42.3^{a}	46.6^{a}	53.9^{b}	58.5^{c}
		Period III		
Ntot, mg g^{-1}	38.1^{a}	42.1^{ab}	48.2^{c}	44.4^{bc}
		Period IV		
Ntot, mg g^{-1}	23.3^{a}	38.5^{b}	38.0^{b}	45.2^{c}
Ctot, mg g^{-1}	425.5^{a}	n.d.^{2}	n.d.	433.5^{a}
C/N	18.2^{a}	n.d.	n.d.	9.3^{b}
Cellulose, g kg^{-1}	178.2^{a}	n.d.	185.0^{a}	175.1^{a}
Lignin, mg g^{-1}	12.6^{a}	n.d.	12.2^{a}	12.2^{a}

[1]Within each row, mean values with different letters are significantly different ($p<0.05$); [2]n.d. = not determined.

Table 3. Ntot, Ctot, and C/N ratio of GC mixture residues in spring 2006 (Period IA) and G and Cl residues in 2007 (Period IVA).

	2006		2007			
	GC		G		Cl	
Parameter	GCN0	GCN80	GCN0	GCN80	GCN0	GCN80
Ntot, mg g^{-1}	33.0^{b1}	34.0^{b}	20.3^{a}	18.6^{a}	38.8^{c}	42.4^{d}
Ctot, mg g^{-1}	n.d.^{2}	n.d.	450.8^{a}	444.3^{a}	453.7^{a}	447.6^{a}
C/N	n.d.	n.d.	22^{b}	24^{b}	12^{a}	11^{a}

[1]Within each row, mean values with different letters are significantly different ($p<0.05$); [2]n.d. = not determined.

Nylon bags were removed from the field according to the time intervals specified in Table 1. Weeks were counted from the first day when bags

were placed on an experimental plot; bags always were removed on the seventh day of the week. For each fertilization treatment, four bags were removed simultaneously (i.e., one bag per treatment replicate).
Upon removal, the contents of each nylon bag were carefully examined and visible soil particles removed. The residue was oven-dried for six hours at 105 °C and individually weighed to record the DM. The biomass residue remaining in the bags was expressed as a percentage of the initial dry weight. The weight loss (%) for each period was calculated using the following formula:

$$\text{Weight loss} = 100 * (M_i - M_t)/M_i$$

where M_i is the initial plant material dry matter mass (g) in the nylon bag; and M_t is the plant material dry matter mass (g) in the nylon bag at time t, when bags were removed from field.

4.4.3. Nitrogen mineralization from decomposing plant residues

Based on plant residues left in the nylon bags, Ntot was determined. The amount of mineralized N (N_m) (% of initial amount) at a time interval (t) was calculated using the following formula:

$$N_m = 100 * (N_i - N_t)/N_i$$

where N_i is the initial amount (mg) of N in the decomposing material; and N_t is the amount (mg) of N in the decomposing material at time t.

4.4.4. Weather conditions

The climate of Estonia is almost maritime in the west and slightly continental in the east. The winter period (when average air temperature is permanently below 0 °C) lasts on average 115 days; the average mean temperature of the coldest months is -5.5 °C. The average duration of the vegetation period (air temperature permanently above 5 °C) is 175–190 days. The average period without night frosts is four months, during which time the average midsummer (July) temperature is 16–17 °C. Mean annual precipitation is 550–700 mm; the average precipitation in the wettest months (April to the end of October) is 350–500 mm (Keppart & Loodla, 2006). Throughout the experimental period, micrometeorological

conditions were monitored at the experimental site using Metos Model MCR300 weather stations (Pessl Instruments GmbH, Weiz, Austria). The sensors were positioned 2 m above the ground.
In order to investigate the effect of weather conditions on decomposition, information about average air temperature (°C), relative air humidity (%), and precipitation (mm) was gathered at different times. Weather parameters are presented for TG sward (Periods I–IV) and GC sward (Period IA and IVA) for every week during which decomposition was studied (Table 4).

Table 4. Weekly weather parameters during decomposition periods.

Week	Mean temperature, °C	Mean relative air humidity, %	Precipitation, mm
		Period I	
1	9.7 (1.5)1	70.8 (17.7)	16.0
2	11.1 (1.7)	77.5 (6.9)	14.2
3	11.5 (1.6)	83.8 (10.0)	29.0
4	11.3 (2.1)	82.6 (7.5)	7.8
5	18.2 (2.2)	69.6 (6.9)	0.2
6	20.5 (3.5)	78.3 (11.3)	10.0
7	17.5 (1.5)	77.9 (6.9)	0.0
8	22.1 (2.8)	66.3 (6.8)	10.0
		Period II	
1	12.0 (3.4)	86.7 (2.9)	0.0
2	13.8 (1.0)	93.5 (3.3)	6.0
3	13.2 (0.7)	97.3 (1.4)	6.4
4	10.4 (1.7)	98.6 (0.5)	20.4
5	7.0 (1.2)	97.8 (1.3)	10.4
6	8.8 (3.6)	98.4 (1.1)	43.0
7	3.1 (4.3)	96.0 (2.4)	31.4
8	-2.0 (5.1)	98.3 (1.0)	4.0
		Period III	
1	3.5 (4.1)	96.6 (2.3)	33.4
2	-2.5 (4.2)	98.3 (1.0)	2.0
3	0.8 (2.1)	98.8 (0.4)	1.4

Week	Mean temperature, °C	Mean relative air humidity, %	Precipitation, mm
4	4.6 (1.6)	98.8 (0.5)	9.2
5	7.0 (0.7)	99.0 (0.0)	2.2
6	6.6 (1.6)	98.7 (0.5)	13.6
7	4.9 (2.6)	98.4 (0.5)	22.0
8	0.3 (3.1)	98.4 (0.8)	13.6
9	0.1 (2.8)	96.3 (3.3)	1.0
10	1.8 (1.5)	99.0 (0.0)	16.0
Period IV			
1	14.0 (3.4)	74.9 (8.0)	11.0
2	18.6 (3.9)	85.1 (8.2)	42.4
3	16.3 (2.8)	79.9 (8.2)	9.2
4	18.1 (0.8)	74.7 (1.6)	0.0
5	14.8 (1.4)	84.0 (9.2)	8.0
6	15.8 (1.3)	83.1 (8.3)	17.4
7	16.2 (2.7)	84.9 (7.8)	9.8
8	16.5 (1.6)	93.9 (16.5)	31.8
Period IA			
1	11.8 (1.6)	85.5 (8.7)	7.6
2	12.2 (3.8)	79.1 (11.2)	0.2
3	18.6 (2.6)	70.7 (5.6)	10
4	19.8 (3.5)	80.0 (10.4)	2
5	17.8 (1.8)	76.3 (9.4)	0
6	22.4 (2.9)	65.8 (7.2)	10
7	19.3 (2.5)	70.7 (7.0)	0
8	16.7 (2.1)	73.1 (4.3)	1.6
Period IVA			
1	16.3 (2.8)	79.5 (9.2)	9.2
2	18.1 (0.7)	75.8 (3.3	0.8
3	14.2 (0.3)	83.6 (9.4)	7.2
4	15.7 (1.6)	84.7 (8.6)	18.8
5	17.3 (2.6)	83.5 (7.3)	8.4

Week	Mean temperature, °C	Mean relative air humidity, %	Precipitation, mm
6	16.0 (0.6)	95.6 3.9)	31.8
7	17.6 (1.5)	86.4 (5.6)	15.2
8	16.9 (1.6)	86.1 (7.5)	7.2

[1]Standard deviations are given in parentheses.

4.4.5. Nitrogen uptake by plant and use efficiency

In 2007 (fourth year of the experiment), Ntot was determined from every cut of both swards. For TG sward, Ntot was measured in samples taken from six fertilization variants (TGN0, TGN80, TGN160, TGN240, TGN320, TGN400); for GC sward, samples were taken from two variants (GCN0 and GCN80). For both swards, samples included RRT and RRM treatments. N uptake by plants (NUP), apparent N recovery (NREC), and N-use efficiency (NUE) were calculated from sward DMY under the experimental conditions studied. Additionally, the apparent N recovery from residues ($NREC_R$) and the N-use efficiency from residues (NUE_R) were calculated using the following formulas:

Returned plant residues impact (RRI), kg ha^{-1} y^{-1}, indicates the DMY (kg) difference between the RRT and RRM treatments per ha (ha^{-1}) and per year (y^{-1}):

$$RRI = DMY_{RRT} - DMY_{RRM},$$

where DMY is the dry matter yield of sward (kg ha^{-1}); and the indices RRT and RRM indicate the specific residue treatments.

NUP (kg ha^{-1} y^{-1}) indicates the total N amount (kg) taken up by plants per ha (ha^{-1}) and per year (y^{-1}):

$$NUP_{RRT, RRM} = DMY * Ntot/1000$$

where NUP_{RRT} is the N amount (kg ha^{-1} y^{-1}) returned with plant residues; NUP_{RRM} is the N amount (kg ha^{-1} y^{-1}) removed with plant residues; and Ntot is N content in plant residues (mg g^{-1}).

NREC (%) indicates the percentage of N applied with fertilizer that was recovered in the yield:

$$NREC = \{(NUP \text{ at } N_x - NUP \text{ at } N_0)/ N_x\} * 100$$

where N_x is the applied N rate (kg ha^{-1}), x=0, 80, 160, 240, 320 and 400 for TG sward; 0 and 80 for GC sward; and N_0 is the unfertilized (control) variant.

NUE (kg kg^{-1} y^{-1}) indicates how much DMY (kg) was produced per kg (kg^{-1}) N fertilizer and per year (y^{-1}):

$$NUE = (DMY \text{ at } N_x - DMY \text{ at } N_0)/N_x.$$

On RRM treatment apparent NREC and NUE indicates with inorganic fertilizer applied N recovering and N-use efficiency.

On RRT treatment apparent NREC and NUE indicates with inorganic fertilizer and with residues applied N recovering and N-use efficiency. NUE_R (kg kg^{-1}) indicates the yield (kg) produced per (kg) returned residues N:

$$NUE_R = \{(DMY_{RRT} \text{ at } N_x - DMY_{RRT}Y \text{ at } N_0) - (DMY_{RRM} \text{ at } N_x - DMY_{RRM} \text{ at } N_0)\}/ NUP_{RRT} \text{ at } N_x$$

$NREC_R$ (%) indicates the percentage of N applied with residues that were recovered in the yield:

$$NREC_R = \{(NUP_{RRT} \text{ at } N_x - NUP_{RRT} \text{ at } N_0) - (NUP_{RRM} \text{ at } N_x - NUP_{RRM} \text{ at } N_0)\}/ NUP_{RRT} \text{ at } N_x * 100$$

$$NREC_R \text{ at control treatment} = (NUP_{RRT} - NUP_{RRM})/ NUP \text{ at } N_0 * 100$$

4.4.6. Soil sampling

In June 2006, soil profile pits were dug on the experimental field. Soil samples were collected at depths of 0–5, 5–10, 10–20, 20–30, 30–40, 40–50, 50–60, 60–80, and 80–100 cm. In September 2008, soil samples were taken from various fertilization variants (TGN0, TGN80, TGN160 and TGN400 for TG sward; GCN0 and GCN80 for GC sward), and from RRT and RRM treatments at different depths (0–5 and 5–20 cm).

Soil was air dried and passed through a 2-mm sieve. SOC and Ntot were determined from soil samples.

4.4.7. Soil organic carbon stock

SOC stock (t ha^{-1}) for two depths (0–5 and 5–20 cm) was calculated for TG sward at four fertilization variants (TGN0, TGN80, TGN160 and TGN400) and for GC sward at two fertilization variants (GCN0 and GCN80); RRM and RRT treatments were included as before.
SOC stock (t ha^{-1}) was calculated using the following formula:

$$SOC\ stock = BD * SOC * D / 10$$

where SOC is measured in mg g^{-1}; BD is bulk density (g cm^{-3}); and D is the soil sampling depth (cm), with 5 cm representing a depth of 0–5 cm and 15 cm representing a depth of 5–20 cm.
BD (g cm^{-3}) was estimated using the Adams (1973) equation:

$$BD = 100/\{(SOM/10/0.244) + ((100 - (SOM/10)) / 1.64)\}$$

where SOM is the soil organic matter content (mg g^{-1}), with an assumption that SOM is 58% of SOC (Mann, 1986).

4.4.8. Chemical analyses

All soil and plant analyses were carried out at the laboratory of the Department of Soil Science and Agrochemistry, Estonian University of Life Sciences.

Ntot concentration in soil and plants and Ctot in plants was analyzed by the dry combustion method in a varioMAX CNS elemental analyzer (ELEMENTAR, Germany).

SOC content was determined by wet oxidation with potassium dichromate ($K_2Cr_2O_7$) according to Tjurin (Vorobyova, 1988).

4.4.9. Sward thatch layers

The thickness of TG sward thatch layers was measured at the end of the experiment in 2008. Ten samples from randomly selected places in every plot were taken using a 3.5-cm diameter soil drill. The thatch layer thickness was measured with liner.

4.5. Statistical analysis

The Statistica version 7.0/9.1 (StatSoft Inc.) software package was used for all statistical analyses.

Factorial analysis of variance (ANOVA) and one-way ANOVA were applied to test the effect of various factors (e.g., RRM or RRT; fertilization) on the DMY, Ntot, SOC, and SOC stock. Fisher's least significant difference (LSD) test for homogeneous groups was used for testing significance differences between fertilization and plant residue treatments.
The level of statistical significance was set at $p<0.05$.

5. RESULTS

5.1. Influence of species on plant residue decomposition

When comparing the decomposition of various plant residues (i.e., TG sward, GC sward, G and Cl separately) over an eight-week period, the fastest decomposition occurred with Cl residues (73% of initial mass) and the slowest with G residues (49%) (Figure 3). The addition of Cl residues did not enhance the decomposition of GC residues (48%). For all four periods and fertilization variants, 64% of TG sward residues were decomposed over eight weeks.

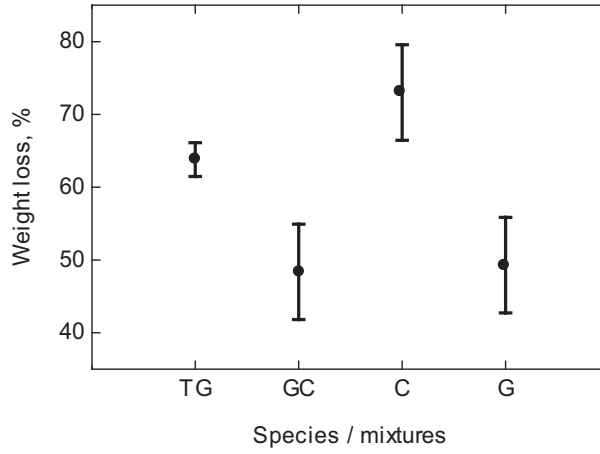

Figure 3. Weight losses of various plant residues as percentage (%) of the initial DM weight depended on species or mixtures; TG = turfgrass sward; GC = grass-clover sward; Cl = white clover; G = grass. Vertical bars denote 0.95 confidence interval (CI).

5.1.1. Turfgrass sward

The decomposition rates of TG residues varied during Periods I–IV (Figure 4). The fastest decomposition (76%) occurred in Period II, when residues were left on growing plots in September. The slowest decomposition (57%) occurred in Period III, when residues were left on plots at the end of October. By comparison, TG residues left in the middle of May were slower (i.e., 62% in Period I and 60% in Period IV).

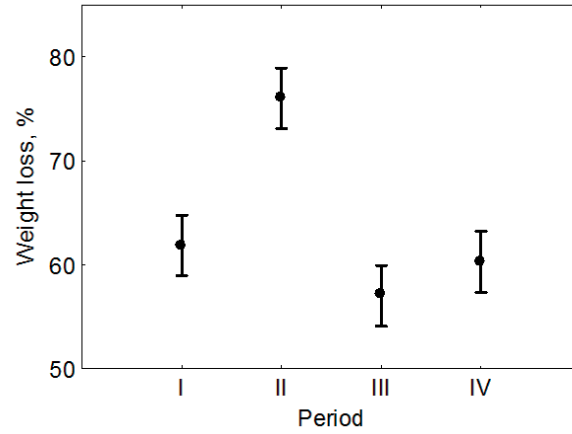

Figure 4. Weight losses of TG sward residues as a percentage (%) of the initial DM weight, during decomposition in spring-summer 2006 (Period I), in autumn (Period II), autumn-winter (Period III) and in spring-summer 2007 (Period IV). Vertical bars denote 0.95 CI.

5.1.2. Influence of weather conditions on plant residue decomposition

Air temperature was the only measured weather parameter that showed a significant influence on the decomposition of TG residues during an eight-week period (R^2=0.97; Figure 3 in **III**). In general, increasing air temperature to 10 °C resulted in an increased decomposition rate, but further increases in air temperature resulted in slower decomposition rates (Figure 3 in **III**). There were no significant relationships between relative air humidity or precipitation and weight loss over an eight-week period (P>0.05; Table 4 in **III**).

In the spring periods of two years (i.e., Period I and Period IV), weather conditions were similar at the start of the decomposition process. Over eight weeks in 2006, the average air temperature was 15.2 °C and relative air humidity was 75.9%; in 2007, those same parameters were 16.3 °C and 82.6%, respectively. Under similar weather conditions, weight losses over eight weeks were also similar, with decomposition measured at 62% and 60%, respectively. In the late summer period (Period II), the average air temperature over eight weeks was lower (8.3 °C) and relative air humidity was higher (95.8%) compared to the spring periods; in these weather conditions, the decomposition rate was the fastest. In Period III,

the average air temperature over ten weeks was 2.7 °C and relative air humidity was the highest (98%) of all periods; under these conditions, 57% of material had decomposed.

The effect of weather conditions on the decomposition of GC sward residues, however, could not be assessed. In 2006, the decomposition of GC sward residues was studied together, but in 2007 G and Cl were studied separately.

At the beginning of the decomposition process for TG sward residues, N content varied significantly within fertilization variants (Table 2), although it did not have any effect on decomposition (Figure 5). Only in Period III did TGN0 variant material decompose slightly faster than fertilized variants over eight weeks.

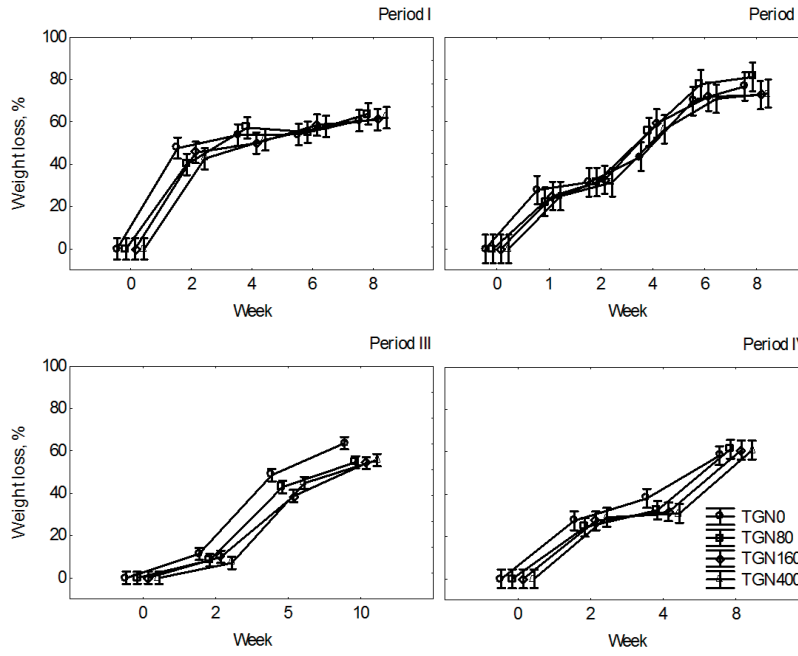

Figure 5. Weight losses of TG sward residues as a percentage (%) of initial DM weight) depended on fertilization and period. Vertical bars denote 0.95 CI.

5.1.3. Grass-clover sward

The N content of GC sward residues from two fertilization variants did not vary significantly (Table 3), although residue decomposition dynamics were different. The GC sward residues from the GCN80 variant decomposed slower compared to residues from the unfertilized control variant (Figure 6). Over eight weeks, 54% of material from unfertilized variants decomposed compared to 43% from fertilized variants.

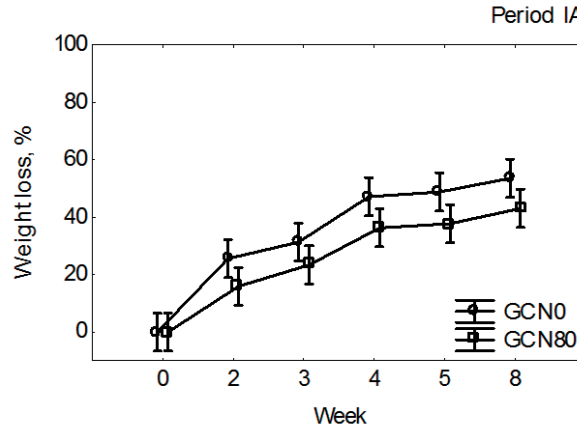

Figure 6. Weight losses of GC sward residues as a percentage (%) of the initial DM weight depended on fertilization (Period IA). Vertical bars denote 0.95 CI.

5.1.4. Decomposition of grasses and white clover separated residues

In 2007, when the decomposition of G and Cl residues were studied separately, Cl residues decomposed faster (Figure 7). For both fertilization variants, 49% of G residues and 73% of Cl residues decomposed over an eight-week period.

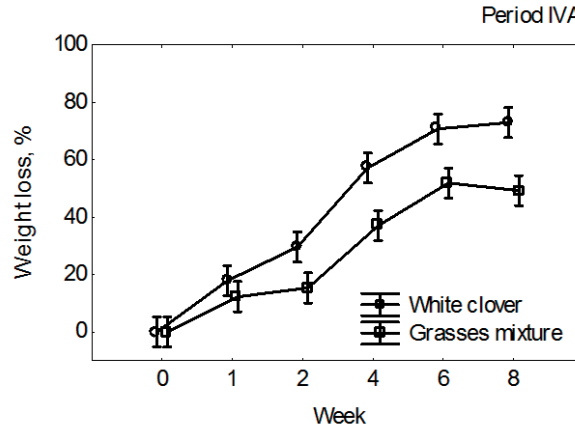

Figure 7. Weight losses of G and Cl residues as a percentage (%) of the initial DM weight; mean of GCN0 and GCN80 (Period IVA). Vertical bars denote 0.95 CI.

For G residues, 55% of the GCN0 variant sample and 44% of the GCN80 variant sample were decomposed (Figure 8). By comparison, the weight loss of Cl residues in both fertilization variants was 78% and 68%, respectively.

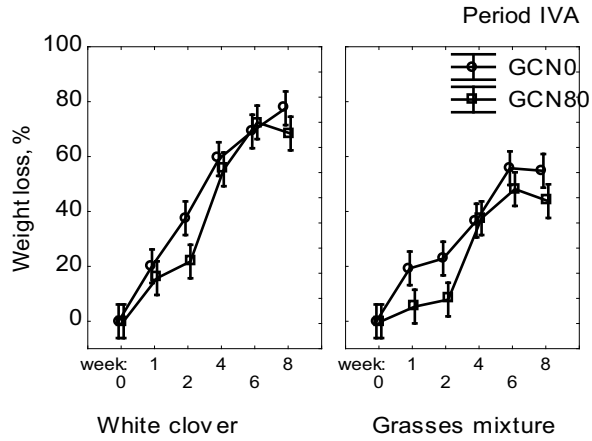

Figure 8. Weight losses of G and Cl residues as a percentage (%) of the initial DM weight depended on fertilization (Period IVA). Vertical bars denote 0.95 CI.

5.2. Nitrogen mineralization

5.2.1. Turfgrass sward

For TG residues, the amount of N_m directly depended on N_i content (Figure 9), as higher N content led to more N mineralized. Depending on the decomposition period, 41–65% (TGN0) and 64–74% (TGN400) of N_i content was mineralized. N_m was largest in autumn (Period II) when its decomposition rate also was the highest.

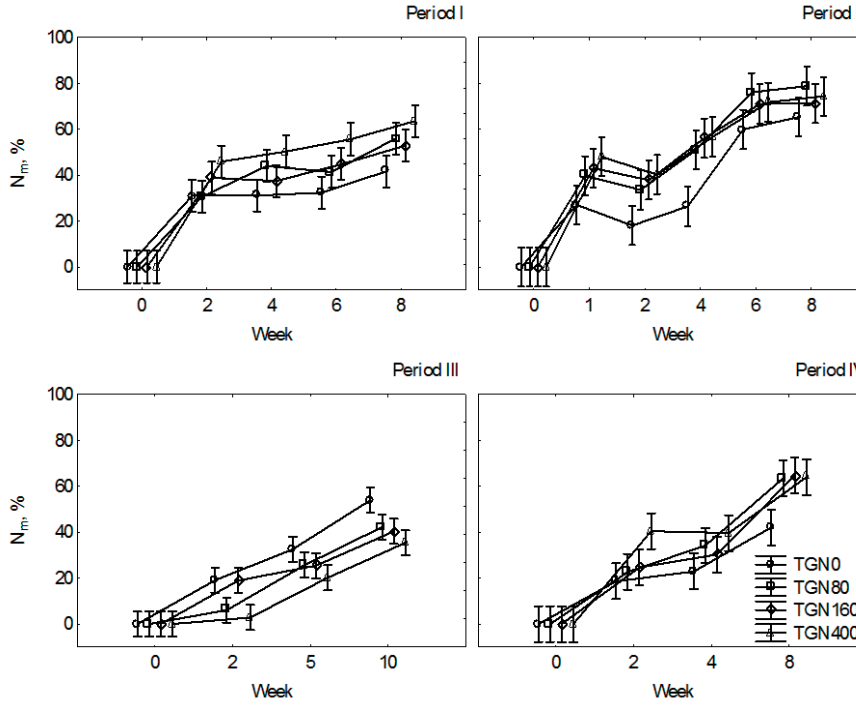

Figure 9. Nm from TG sward residues depended on fertilization and period; Nm = % from Ni. Vertical bars denote 0.95 CI.

5.2.2. Grass-clover sward

For GC swards, N_m was not significantly affected by fertilization, although less N was mineralized in the unfertilized control variant than in the fertilized variant (Figure 10). Over eight weeks, 46% of the GCN0 variant's N_i was mineralized compared to 37% of the GCN80 variant.

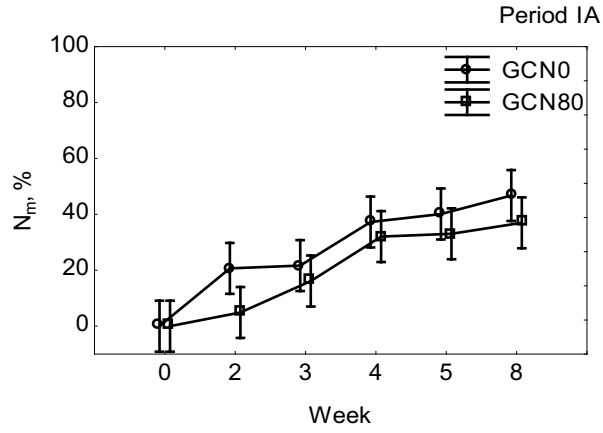

Figure 10. In 2006, Nm for GC sward residues depended on fertilization (Period IA); Nm = % from Ni. Vertical bars denote 0.95 CI.

The N_m rate and amount of N_m were higher in Cl residues than in G residues. Within the first two weeks, 42% of Cl residues in the GCN0 variant was mineralized from N_i compared to 31% in the GCN80 variant (Figure 11; white clover). After eight weeks, 78% of Cl residues from both variants was mineralized from N_i.

On the other hand, 39% of G residues in the GCN0 variant was mineralized from N_i within two weeks compared to the increase of N content by 3.4% in the GCN80 variant (Figure 11; grasses mixture). After eight weeks, 52% of G residues in the GCN0 variant was mineralized from N_i compared to 33% in the GCN80 variant.

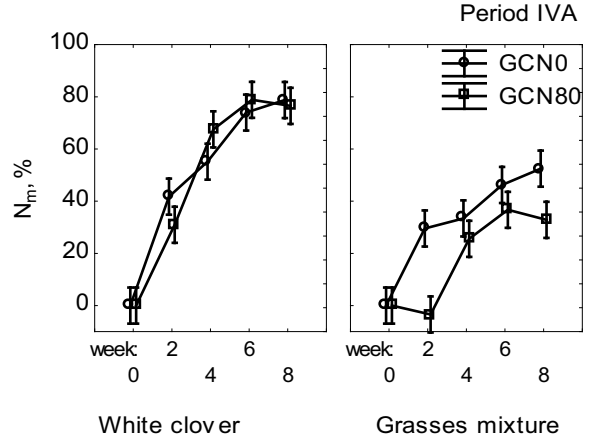

Figure 11. In 2007, Nm from Cl and G residues depended on fertilization (Period IVA); Nm = % of Ni. Vertical bars denote 0.95 CI.

5.3. Effects of plant residues

5.3.1. Above-ground dry matter yield

In 2004–2008, RRT significantly increased GC sward DMY ($P < 0.05$) (Table 5). The effect was highest in the unfertilized variant, where DMY increased by 52%; in the fertilized variant, RRI had a smaller effect on yield, as DMY only increased by 39% with RRT treatment. Due to the integrated effect of plant residues and N fertilization, the DMY increase was 70% for the GCN0 variant with RRM, and 22% for the GCN80 variant with RRM treatment.

During the same experimental period, TG sward had a significantly lower average DMY than that of GC sward. The DMY of TG sward varied between 1524 kg ha^{-1} and 5834 kg ha^{-1} depending on the fertilization variant. The DMY of TG sward was not affected by residues management ($P > 0.05$). The only exceptions were fertilization variants TGN160 and TGN240, which showed significant DMY increases with RRT treatment ($P < 0.05$).

Table 5. Swards DMY, RRI, and RRT:RRM yield ratios (2004–2008).

Fertilization treatment	DMY, kg ha^{-1}			Ratio RRT:RRM
	RRT[1]	RRM[2]	RRI	
TG sward (n=316)				
TGN0	1412^{a3A4}	1524^{aA}	-112^{A}	0.93
TGN80	2840^{aB}	2678^{aB}	162^{A}	1.06
TGN160	4050^{bC}	3436^{aC}	614^{B}	1.18
TGN240	5498^{bD}	4630^{aD}	868^{D}	1.19
TGN320	5784^{aD}	5366^{aE}	418^{C}	1.08
TGN400	5962^{aE}	5834^{aF}	128^{A}	1.02
GC sward (n=84)				
GCN0	5751^{bA}	3778^{aA}	1973^{B}	1.52
GCN80	6417^{bA}	4611^{aB}	1806^{A}	1.39

[1]RRT = plant residues returned to plots; [2]RRM = plant residues removed from plots; [3]Within each row, different small letters indicate a significant influence ($P < 0.05$) of RRT and RRM on sward DMY; [4]Within each column, different capital letters indicate a significant influence ($P < 0.05$) of N fertilization on DMY per sward type.

5.3.2. Nitrogen uptake by plant and use efficiency

For TG sward residues, Ntot depended on the fertilization variant as well as on whether RRM or RRT treatments were followed (Table 6). NUP decreased for TGN0 with RRT, but otherwise was higher for all other fertilization variants. In general, NUP significantly increased with RRT treatment until a fertilization rate of 240 kg N ha^{-1} was applied; beyond that point, increasing the N fertilization rate resulted in only modest increases in NUP. For RRM treatment, however, NUP increased steadily until 400 kg N ha^{-1}.

With RRM treatment, NREC and NUE reached their highest levels at the lowest N application rate applied (i.e., TGN80). When the N rate was increased to 160 kg ha^{-1}, both NREC and NUE decreased and successively higher fertilization rates did not have any significant impact on either parameter. With RRT treatment, NREC and NUE reached optimum values at an N rate of 240 kg ha^{-1}; beyond this point, applying higher N rates resulted in sharp decreases for both parameters. For TG sward, the impact of plant residues on DMY was the highest for the TGN160 and TGN240 variants (Table 5). $NREC_R$ also was the highest with TGN160 and TGN240 variants (Table 7).

Table 6. DMY, Ntot content, NUP, NREC, and NUE of harvested plant residues from TG and GC swards (2007).

Fertilization treatment	DMY, kg ha^{-1}		Ntot, mg g^{-1}		NUP, kg ha^{-1}		NREC, %		NUE, kg kg^{-1}	
	RRM[1]	RRT[2]	RRM	RRT	RRM	RRT	RRM	RRT	RRM	RRT
TG sward										
TGN0	1144	990	30	31	34	31	-	-	-	-
TGN80	2317	2511	34	34	79	85	56.2	67.5	14.8	19.0
TGN160	2903	3541	37	38	107	135	45.6	65.0	11.0	15.9
TGN240	3747	5159	40	43	150	222	48.3	79.6	10.9	17.4
TGN320	4366	5086	43	45	188	229	48.1	61.9	10.1	12.8
TGN400	5155	5481	43	43	221	236	46.7	51.2	10.0	11.2
GC sward										
GCN0	4469	7787	24	24	107	190	-	-	-	
GCN80	5210	7723	24	26	125	204	22.5	17.5	9.3	-0.9

[1]RRT = plant residues returned to plots; [2]RRM = plant residues removed from plots.

For GC sward, the effect of RRT treatment on DMY was statistically significant for both fertilization variants (i.e., GCN0, GCN80), although the impact of plant residues was significantly smaller at GCN80. NUP was higher with RRT treatment, but similar for both variants. Fertilization variants had no effect with RRT treatment, so NUE was negative (i.e., -0.9) at GCN80 (Table 6). $NREC_R$ was 43.7% of Ntot at GCN0 variant, but was reduced by -2.3% at GCN80 (Table 7).

Table 7. $NREC_R$ and NUE_R of harvested plant residues from TG and GC swards (2007).

Fertilization treatment	**$NREC_R$, %**	**NUE_R, kg kg^{-1}**
TG sward		
TGN0	-9.7	-4.9
TGN80	10.5	4.0
TGN160	23.0	5.9
TGN240	33.8	7.0
TGN320	19.2	3.8
TGN400	7.6	2.0
GC sward		
GCN0	43.7	17.4
GCN80	-2.3	-4.0

5.4. Distribution of soil organic carbon and nitrogen content in soil profile

In 2006, SOC and Ntot accumulated in the upper soil layers of the experimental field.

SOC content in the top layer (0–5 cm) was 16.0 mg g^{-1} (Figure 12). At a depth of 5–20 cm, SOC content was 14.7 mg g^{-1}. At lower layers, SOC content decreased significantly from 6.6 mg g^{-1} at 20–30 cm to 1.8 mg g^{-1} at 80–100 cm.

Ntot in the top layer (0–5 cm) was 1.75 mg g^{-1}. At a depth of 5–20 cm, Ntot was 1.60 mg g^{-1} and decreased to 0.40 mg g^{-1} at 50–60 cm.

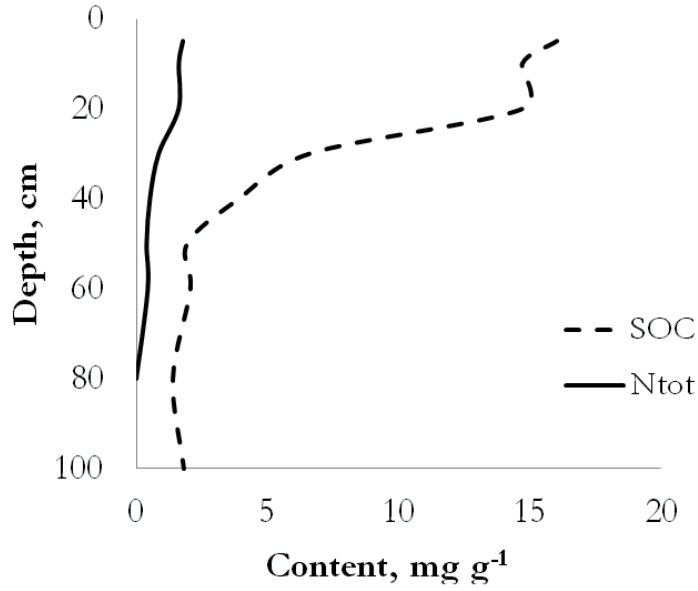

Figure 12. SOC and Ntot (mg g-1) in the experimental field soil profile.

5.5. Soil organic carbon and total nitrogen content in swards soil

For both swards, SOC concentration increased significantly during the five-year experimental period (Table 8). In the upper soil layer (depth: 0–5 cm) of TG sward, SOC concentration increased with RRT treatment (by 21.6%) as well as with RRM treatment (by 7.2%). In the upper soil layer (depth: 0–5 cm) of GC sward, SOC concentration also increased with RRT (by 42.9%) and with RRM (by 32.0%.)

For TG sward, SOC content decreased at lower layers (depth: 5–20 cm), but did not change significantly in GC sward compared to initial concentrations measured in 2003. Fertilization variants did not influence SOC content for TG swards. In GC sward, fertilization variants had an impact at a soil depth of 5–20 cm, where SOC decreased with RRM treatment.

Table 8. Ntot, SOC, and C/N ratio in TG and GC sward soil in 2008[1].

Fertilization treatment	Ntot, mg g^{-1}		SOC, mg g^{-1}		C/N	
	RRT[2]	RRM[3]	RRT	RRM	RRT	RRM
TG sward (0–5 cm)						
TGN0	1.52^{bA}	1.08^{aA}	18.0^{bA}	15.9^{aA}	11.9^{aA}	14.7^{bB}
TGN80	1.54^{bA}	1.27^{aB}	17.6^{bA}	15.7^{aA}	10.8^{aA}	12.0^{bA}
TGN160	1.58^{bA}	1.39^{aB}	17.4^{bA}	15.8^{aA}	11.0^{aA}	11.3^{aA}
TGN400	1.56b^{A}	1.28^{aB}	18.4^{bA}	15.5^{aA}	11.8^{aA}	12.1^{aA}
GC sward (0–5 cm)						
GCN0	2.09^{bA}	1.84^{aA}	20.8^{bA}	19.3^{aA}	10.0^{aB}	10.5^{aB}
GCN80	2.22^{bB}	1.96^{aB}	21.2^{bA}	19.5^{aA}	9.5^{aA}	9.9^{aA}
TG sward (5–20 cm)						
TGN0	1.18^{aA}	1.16^{aA}	13.4^{bA}	12.2^{aA}	11.4^{bB}	10.5^{aB}
TGN80	1.17^{aA}	1.18^{aA}	13.2^{bA}	12.5^{aA}	11.3^{bB}	10.6^{aB}
TGN160	1.22^{aA}	1.10^{aA}	13.6^{bA}	12.3^{aA}	11.2^{aB}	11.2^{aB}
TGN400	1.23^{aA}	1.17^{aA}	13.1^{bA}	12.7^{aA}	10.7^{aB}	10.9^{aB}
GC sward (5–20 cm)						
GCN0	1.49^{aA}	1.48^{aA}	14.5^{aA}	14.1^{aB}	9.7^{bA}	9.5^{aA}
GCN80	1.55^{bA}	1.42^{aA}	14.7^{bA}	13.5^{aA}	9.5^{aA}	9.5^{aA}

[1]In 2003 the initial Ntot and Corg contents were 1.49 and 14.7 mg g^{-1}, respectively, the initial C/N ratio was 9.9; [2]RRT - plant residues were returned to the plots; [3]RRM - plant residues were removed from the plots; [4]Different small letters within each row indicate significant influence ($P < 0.05$) of returning plant residues on soil Ntot and SOC concentrations and C/N ratio; [5]Different capital letters within each column indicate significant influence ($P < 0.05$) of fertilization on soil Ntot and SOC concentrations and C/N ratio within the sward type and soil depth.

For TG sward (depth: 0–5 cm), Ntot did not change significantly with RRT treatment when compared to its initial content, but decreased with RRM treatment (15.7%, $P < 0.05$). At this same depth, Ntot was significantly lower at TGN0 than with other fertilized variants. For GC sward (depth: 0–5 cm), Ntot increased in GCN0 with RRM (by 23.5%) and with RRT (by 40.3%). For GCN80 at the same depth, Ntot increased by 31.5% with RRM treatment and by 49.0% with RRT treatment.

For TG sward at lower layers (depth: 5–20 cm), Ntot decreased in both residue treatments. For GC sward (depth: 5–20 cm), Ntot at GCN0 was not influenced by RRT treatment; at GCN80, however, Ntot increased

with RRT treatment and did not change with RRM treatment compared to its initial content.

The C/N ratio of TG sward soil was higher compared to GC sward soil and did not vary significantly over the experimental period.

From 2003 to 2008, SOC stock increased more in GC swards (depth: 0–5 cm). With RRT treatment, SOC stock increased by 0.73–0.78 t $ha^{-1}y^{-1}$ (Table 9). With RRM treatment, the increase was 0.56–0.59 t ha^{-1} y^{-1}.

Table 9. SOC stock and changes with sward type and fertilization treatments[1].

Fertilization treatment	**SOC stock, t ha^{-1}**		**Change in SOC stock, t ha^{-1} y^{-1}**	
	RRT[2]	**RRM[3]**	**RRT**	**RRM**
TG sward (0–5 cm)				
TGN0	12.6^{cA}	11.3^{bA}	0.41^{bA}	0.15^{aA}
TGN80	12.3^{cA}	11.1^{bA}	0.35^{bA}	0.13^{aA}
TGN160	12.2^{cA}	11.2^{bA}	0.33^{bA}	0.14^{aA}
TGN400	12.8^{cA}	11.0^{bA}	0.45^{bA}	0.10^{aA}
GC sward (0–5 cm)				
GCN0	14.1^{cB}	13.3^{bB}	0.73^{bB}	0.56^{aB}
GCN80	14.4^{cB}	13.5^{bB}	0.78^{bB}	0.59^{aB}
TG sward (5–20 cm)				
TGN0	29.1^{bA}	26.7^{aA}	-0.48^{aA}	-0.96^{bC}
TGN80	28.8^{bA}	27.3^{aA}	-0.54^{aA}	-0.84^{bC}
TGN160	29.5^{bA}	27.0^{aA}	-0.40^{aA}	-0.90^{bC}
TGN400	28.6^{bA}	27.8^{aA}	-0.59^{aA}	-0.74^{bC}
GC sward (5–20 cm)				
GCN0	31.2^{bB}	30.5^{aB}	-0.51^{aB}	-0.20^{bA}
GCN80	31.5^{bB}	29.3^{aB}	0.13^{aB}	-0.44^{bB}

[1]In 2003 the initial SOC stock was 10.5 t ha^{-1} in 0-5 cm and 31.5 t ha^{-1} in 5-20 cm; [2]RRT = plant residues returned to plots; [3]RRM = plant residues removed from plots; [4]Within each row, different small letters indicate a significant influence ($P < 0.05$) of RRT on SOC content and change in SOC stock; [5]Within each column, different capital letters indicate a significant influence ($P < 0.05$) of fertilization on SOC content and change in SOC stock at the same soil depth.

For TG swards (depth: 0–5 cm), SOC stock increased by 0.33–0.45 t $ha^{-1}y^{-1}$ with RRT treatment and by 0.10–0.15 t ha^{-1} y^{-1} with RRM treatment. At lower soil layers (depth: 5–20 cm), SOC stock decreased according to sward type. For GC swards, SOC stock decreased by 0.20–0.44 t $ha^{-1}y^{-1}$ with RRM treatment, while in TG swards the decrease was 0.74–0.96 t ha^{-1}. Overall, SOC reduction was lower with RRT treatment.

5.6. Thickness of thatch layer

At the end of the experiment, the thickness of the TG sward thatch layer was significantly higher with RRT treatment and with fertilization rates of 160–400 kg N ha^{-1} (Figure 13). At lower N application rates, the thickness of the thatch layer was not affected by RRT treatment.

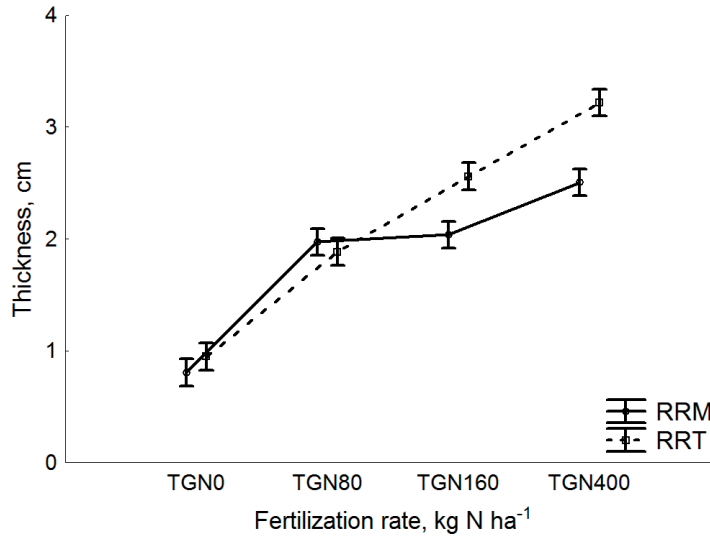

Figure 13. The thickness of the TG sward thatch layer depended on fertilization variants and management of TG residues (2008). Vertical bars denote 0.95 CI.

When the N fertilization rate was increased above 160 kg ha^{-1}, the thickness of the thatch layer also increased and was similar with RRT or RRM treatments. For the TGN160 variant, the difference in thickness was 0.6 cm (i.e., 2.6 cm with RRT treatment and 2.0 cm with RRM treatment). At TGN400, the difference in thickness was 0.7 cm (i.e., 3.2 cm with RRT treatment and 2.5 cm with RRM treatment).
There was no thatch layer formed in GC sward.

6. DISCUSSION

6.1. Decomposition of plant residues

6.1.1. Influence of species and cutting frequency

In our experiment, the decomposition rates of plant residues (TG sward: *Festuca rubra rubra* and *Poa pretense*; GC sward: *Phleum pratense*, *Lolium perenne,* and *Trifolium repens)* varied between 48% and 78% over an eight-week experimental period in unfertilized variant. Cl residues had the fastest decomposition rate, and GC residues the slowest. Variations among plant species likely occurred due to differences in C-compound content. Initially, the decomposition rate is strongly related to the content of easily-decomposable soluble compounds (Gartner & Cardon, 2004). Over time, however, the content of recalcitrant compounds starts having an impact on the decomposition rate (Berg & Staaf, 1980; Berg, 2000). Compared to G, it has been found that Cl contains more easily-decomposable compounds (Martens, 2000; de Neergaard et al., 2002). Cl generally has much less neutral detergent fiber content in the total cell wall than G, but a higher content of lignin (Thomson et al., 1985; Ulyatt et al., 1988; Søegaard et al., 2008). The lowest neutral detergent fiber content was found in Cl leaves. In some cases, cellulose content or acid detergent fiber content can be the same in G and in Cl (Thomson et al., 1985; Ulyatt et al., 1988).

Comparing decomposition rates among plant species is complicated because a proper comparison depends on the decomposition environment (i.e., location, climate) in the experiment as well as the residue chemical composition. A plant's chemical composition depends on species and growth stage (i.e., cutting frequency). By comparing decomposition dynamics after the first cut, our experiment concluded that TG residue weight loss after the first two weeks was 23% (average of 2006 and 2007) and increased to 64% within eight weeks. G weight loss in GC sward was 15% after two weeks and 49% over eight weeks. Thus, the content of easily-decomposable C compounds must have been lower in the G residues of GC sward because they had been cut at a later growth stage than TG sward. During seasonal plant development, the content of easily-decomposable C compounds decreases (Wilman & Wright, 1983) while the content of recalcitrant compounds (e.g., lignin) increases. Higher recalcitrant compound levels inhibit the decomposition rate of plant residues, which results in slower decomposition at later growth stages. Therefore, frequently cut TG plants are in earlier growth stages; furthermore, TG sward residues contain sizeable fractions of

easily-decomposable compounds that decompose at a faster rate (Shi et al., 2006b). Willms & Beauchemin (1990) found that increasing cutting frequency caused lignin content to decrease, which was associated with a decreasing age of plant tissue harvested.

TG sward decomposition dynamics has not been studied thoroughly, and most experiments have been performed in the United States (Kopp & Guillard, 2004; Shi et al., 2006a,b). For example, laboratory soil incubations with bermudagrass (*Cynodon*. *dactylon* X *transvaalensis*) residues showed that 20–30% of C and N clippings were mineralized within seven days (Shi et al., 2006a). Kopp & Guillard (2004) investigated above-ground residues decomposition in Utah; their data indicates that the amount of material decomposed after four weeks was 70% of the initial material, which does not agree with our results. Over our eight-week experimental period, 57–76% of TG sward residues decomposed, and the rate of decomposition depended significantly on weather conditions during decomposition **(III)**.

6.1.2. Effect of air temperature and humidity on decomposition (Paper III)

Plant residues left on sward surfaces decompose at varying rates depending on environmental conditions. For example, our results indicated that the optimal air temperature for fresh material decomposition on ground surfaces was about 10 °C, with a reduction in rates either above or below that temperature. Wang et al. (2000) suggested that higher temperatures accelerate organic matter decomposition only when moisture content is adequate, and inhibits decomposition when air humidity is limited. Flanagan & Veum (1974) showed that organic matter decomposition can be limited by low air temperature as well as by low moisture content, and increasing only one of those factors does not compensate fully for the influence of another limiting factor.

In our experiment, the dynamics of moisture content in decaying material was not determined, but we assumed that increasing air temperature above 10 °C turned low moisture content into the limiting factor for decomposition. At air temperatures below 10 °C, the limiting factor was air temperature, which proved most important at the beginning of the decomposition process (i.e., within the first two weeks) when decomposition rates were the highest. Above 10 °C, weight loss by

decomposing was slowed down, and this effect was most likely caused by the fast drying of plant material. For fast decomposing of TG sward residues, our results suggested that it was important to maintain the initial moisture content because rainwater cannot fully compensate for its loss. In Period IV (i.e., spring-summer 2007), the average air temperature during the first two weeks of the decomposition process was 16 °C, which was relatively higher than in other experimental decomposition periods. Even though the amount of precipitation was also at its highest in Period IV, weight loss stayed at a lower level compared with other decomposition periods.

Henriksen & Breland (1999) found that plant residues decomposed intensively even though the average temperature during the entire investigational period generally stayed below 0 °C and never rose above 2.4 °C. Our research results (**III**) indicated that at an air temperature of about 0 °C (i.e., Period III, autumn-winter 2006), the decomposition rate of plant residues was significantly slower than at higher temperatures. We were unable to determine, however, if the decomposition process continued at 0 °C (or even lower temperatures) because during Period III there were days when the average air temperature exceeded 5 °C. It is possible that most decomposition occurred during those short periods when air temperatures were higher than 0 °C. Due to variability in air temperatures, average temperature cannot be considered a good indicator to evaluate its influence on decomposition during different seasonal periods. This assumption is true for late autumn, winter, and early spring periods when average air temperatures remain about 0 °C, even though there were wide variations between night and day temperatures.

6.1.3. Influence of fertilization on decomposition of plant residues and nitrogen transformations (Paper III)

As a result of different N fertilization rates, the C/N ratio of TG sward residues used in our experiment was <20; despite the two-fold variation, there were no significant differences in decomposition dynamics **(III)**. This is consistent with the results of Kopp & Guillard (2004), who indicated that different N content due to fertilization did not have any impact on the decomposition rate of TG clippings. Our results also confirmed findings by Quemada & Cabrera (1995) and Wagner & Wolf (1999), who found that with a similar C/N ratio <20, further increasing

N content (along with a concomitant decrease in C/N ratio) did not affect the decomposition rate of plant material.

It is well-known in the literature that a decrease in concentration of easily-decomposable plant compounds will occur due to accelerated herbage growth promoted by fertilization (Wilman & Wright, 1983; Jones & Wilson, 1987). In our experiment, fertilization did not have any impact on the content of cellulose and lignin in TG swards (**III**); this may be explained by the high cutting frequency of TG sward, which inhibits the effect of fertilization on the growth and development of plants compared to swards cut less frequently.

In our experiment, as well as the one by Kopp & Guillard (2004), N_m was affected by N_i content in TG sward residues. More N was mineralized where the N_i content was higher. Our results also showed that although TG sward residues are easily decomposable, N immobilization (i.e., increase of N content in decaying material) can occur at the start of the decomposition process. In Period II (i.e., summer-autumn 2006), N content decreased significantly during the first week of decomposing, but all samples showed a tendency to increase during the second week. Microbes can suffer due to the shortage of N when the largest share of N compounds already have decomposed. After the first week of decomposition, the concentration of N in material was on average 38 mg g^{-1}. High N content indicates a low C/N ratio; if the average C content in residues (i.e., 420 mg g^{-1}) is used to calculate the C/N ratio, then the C/N ratio would be 11.

Although such a low ratio ordinarily indicates that there is enough N available for the decomposition process, in reality the N immobilization occurs. The N immobilization within a material of low C/N ratio also has been mentioned in other studies (Jensen, 1994; de Neergaard et al., 2002; Gunnarsson & Mastrop, 2002). According to Swift et al. (1979), soluble substances and labile compounds, which form the biggest proportion in TG residues (Gunnarsson & Marstrop, 2002), are rapidly degraded during early-phase decomposition by fast-growing microorganisms that require a high N concentration (Quemada & Cabrera, 1995; Gunnarsson & Marstrop, 2002). The N immobilization of plant residues was thought to be caused by high C/N ratios of easily-decomposable compounds that constitute the major C sources of early-phase decomposition of plant materials (Jensen, 1997; Andersen & Jensen, 2001; Gunnarsson & Marstrop, 2002; Shi et al., 2006a).

Similar to studies by Chesson et al. (1985), Cadisch et al. (1998), and de Neergaard et al. (2002), our results also indicated that N_m is affected by plant chemical composition. We conclude that N immobilization also happens with high N concentration and a low C/N ratio; thus, preliminary N concentration and C/N ratio do not describe the decomposition processes in detail, or determine if immobilization takes place or not. Those processes could be more accurately predicted if we knew the content of easily-decomposable compounds. It is possible that similar N immobilization happened during other experimental periods (i.e., I, III, IV), but those results were not recorded because analyses were made only from the start of the second week. For TG swards, the total N_m was unaffected by N immobilization occurring during decomposition; therefore, during the eight experimental weeks (and including an average of various periods under different variants), 51–60% of N_i content was mineralized.

Decomposition of GC residues was studied after the first cutting. Before then, only P and K fertilizers (hereafter 'PK') were applied. N fertilizer was applied after the second cutting. GC sward residues with the GCN80 variant decomposed slower than the GCN0 variant, even though the N_i content and C/N ratio in both variants were the same. Thus, by applying PK fertilizers, it is possible that the development and growth rate of grasses and legumes can be affected by reducing the content of easily-decomposable C compounds (Wilman & Wright, 1983). In our experiment, changes in chemical composition may have contributed to the slower decomposition of residues from PK-fertilized variants. Slowdown of the decomposition process also may have occurred because there was not enough N available for decomposers at the start of the decomposition process. Compared to the GCN0 variant, the N amount was probably higher that is related to less decomposable compounds (i.e., translocation of N). In later growth stages, N is bound to decomposable compounds (Berg, 1986; Peyraud & Astigarraga 1998) that need more N to decompose (Aber & Melillo, 1982). This attribute is shown clearly by differences in N content dynamics during residue decomposition. The N content of GC sward residues in variants where PK fertilizers were applied decreased less than in the GCN0 variant despite having identical N_i at the start of the decomposition process.

By separating the G and Cl residues, variations occurred in the decomposition and N_m of GC sward residues that also were observed by de Neergaard et al. (2002) and Rasmussen et al. (2007). By applying PK

fertilizers, there was no effect on decomposition and N_m of Cl residues, but the decomposition of G residues was slower in the fertilized variant (similar to GC sward residues). Compared to G residues, the N content of Cl residues was higher; this probably explains why in the PK-fertilized variant the decomposition process did not slow down as much as with G residues. It is also possible that PK fertilizers act differently on the content of C compounds in legumes and grasses, but this subject was not studied in our experiment.

The addition of Cl residues to G residues had an impact on the decomposition and N_m of G only at the start of the decomposition process (i.e., within the first two weeks). At the start of the process, this shows that the decomposition of G residues may have been limited by low N content. During the first two weeks, the decomposition rate of GC mixture residues was higher than that of G residues, which may be explained by considering how N released from Cl residues likely was used in G decomposition. Similar results also were published by Gunnarsson & Marstorp (2002), who showed that combining plant materials with different carbohydrate and protein compounds affects N_m; furthermore, by varying the quality of C compounds but keeping the same C/N ratio, it is possible to change the course of N release, which either leads to rapid initial immobilization or rapid mineralization. Whether N_m or immobilisation occurs during decomposition depends on when C compounds in plant residues are decomposed (i.e., before or after N_m or immobilization).

6.1.4. Influence of returned plant residues on sward dry matter yield (Papers II, V, VI)

In TG swards consisting mainly of G, RRT treatment did not have any clear impact on herbage growth. In TGN0, however, the effect of RRT treatment was negative because NUP also was reduced. Probably not all N content in residues was available for plant use; instead, a large proportion of N was incorporated into SOM. This phenomenon was confirmed by soil analysis data during the fifth experimental year, when Ntot content was significantly higher when TG residues were returned and not removed (**V**). This effect also was supported by negative NUE_R and $NREC_R$ values.

The finding that RRT treatment in TGN0 does not increase TG sward DMY is not in accordance with results from Kopp & Guillard (2002).

In their experiment, the authors studied the influence of RRT treatment on sward DMY at two experimental fields with different soil types. In one experimental field, the authors found DMY was similar between the nonfertilized variant with RRT treatment and the fertilized variant with RRM (392 kg N ha^{-1}). In the other experimental field, however, the effect of plant residues on DMY also was significant, but smaller than in the first experimental field. Differences may have been due to variations in soil water-holding capacity and SOC content; for example, the content of SOC in both experimental fields was higher than in our study (i.e., 89 and 73 mg g^{-1}, compared to 51 and 42 mg g^{-1}), while the SOC content of our experimental field was 14.7 mg g^{-1}. The effect of returning TG residues was highest in soil with the highest SOC content and lowest in soil with the lowest SOC content. Earlier studies have shown that C and N mineralization rates are lower in soils with a lower SOC content (Shi et al., 2006b), which is probably why the effect of RRT treatment is smaller. It is known that plant and soil C/N ratios are key variables affecting net soil N_m rates (Manzoni et al., 2008; Meier & Bowman, 2009).

RRT treatment in GC swards increased DMY significantly. Significantly more N with residues was returned to the GC sward compared to the TG sward. The amount of released N was sufficient for microbes to use in decomposition as well as for DMY formation.

Thus, the effect of RRT treatment on herbage growth depends on the choice of sward species. The effect of RRT treatment was especially high in GC sward, but not significant in TG sward because the N amount released from residues was used for decomposition and not for DMY production.

6.2. Integrated effects of returned plant residues and fertilization on sward dry matter yield (Paper VI)

6.2.1. Turfgrass sward

In fertilized variants where TG sward residues were returned, NUP, NREC, and NUE all increased and were accompanied by N_m in soil. This result agrees with findings reported by Starr & Deroo (1981) and Kopp & Guillard (2002). The availability of soil N for plant use is determined largely by N_m during the decomposition of organic matter (Swift et al. 1979).

Previously, we concluded that returning TG sward residues had no significant effect on herbage growth. RRI was significant only when fertilization rates of 160 kg N ha^{-1} or 240 kg N ha^{-1} were used. Thus, plants appear only to use excess N not needed by microbes for decomposition (Chapin et al., 2002). TG sward residues with varying N content appeared to decompose at the same rate, although more N was mineralized from residues where N_i content was higher. The N_i content of residues depended on fertilization and increased with higher N rates; thus, the amount of N available for plants also increased. In the TGN160 variant, 135 kg N ha^{-1} was returned to grassland, and the effect of plant residues on herbage growth was significant only when 295 kg N ha^{-1} (with fertilization and RRT treatment) was applied to TG sward.

Kopp & Guillard (2002) showed that the RRT effect increases linearly with higher N fertilization rates (i.e., from 0 to 392 kg N $ha^{-1}y^{-1}$). In our experiment, the RRT effect at first increased with increasing N rates, but then started to decline from 240 kg N ha^{-1} (TGN240) onward Above TGN240, the effect of this decline nevertheless weakened Ntot and RRT even at higher N rates. In our experiment, TG swards were established with seed mixtures of *Festuca rubra rubra* and *Poa pratensis*, which have basic N requirements of 160 kg ha^{-1} and 400 kg ha^{-1}, respectively, under Estonian climate conditions (Raave & Hein 1989). In the TGN320 and TGN400 variants, the N amount probably exceeded plant requirements because DMY increased significantly at both N rates with RRM treatment. These results suggest that RRT efficiency depends on the specific N need of the plant species, as RRT treatment is efficient only until the point when N_m and N released from plant residues does not exceed plant requirements (**VI**).

6.2.1.1. Effect of turfgrass sward characteristics on impact of the returning the residues

RRI on TG sward growth also could have been influenced by the thatch layer on the soil surface. Thatch accumulation occurs when the production of organic matter exceeds the decomposition rate (Beard, 1973). TG sward was cut on average of once per week. Continuously returning fresh material to TG sward prevents microbes from decomposing older material as sufficient N is available from fresh material (Moorhead & Sinsabaugh, 2006). The decomposition rates of TG residues in different fertilization

variants were the same. Thus, the thatch layer was significantly larger with more frequent RRT treatments. Thicker thatch layers decreased the contact between fresh TG residues and soil; thus, decomposition occurred mainly on top of the thatch layer (Figure 14).

Other studies suggest that thatch layers are rich in lignin (Yao et al., 2009). Starr & DeRoo (1981) studied thatch layer formation on TG sward that had been fertilized with 195 kg N ha^{-1} and found that a thatch layer formed not only with RRT treatment, but also with RRM treatment. According to their measurements, the thatch layer contained 280 kg N ha^{-1} with RRM treatment and 510 kg N ha^{-1} with RRT treatment. As residue decomposition takes place on top of the thatch layer, it is reasonable to assume that N released from residues would be immobilized into the thatch layer and unable to reach the plant.

Our results indicate that RRI on TG sward is influenced by sward density and cutting height (**VI**). Returning TG sward residues had an impact only in May, as the sward was still sparse after the winter. During the following months, RRI was not significant because the sward was denser. In previous similar studies (Starr & Deroo, 1981; Heckman et al., 2000; Kopp & Guillard, 2002), the cutting height (3.8–4.4 cm) was lower than in our experiment.

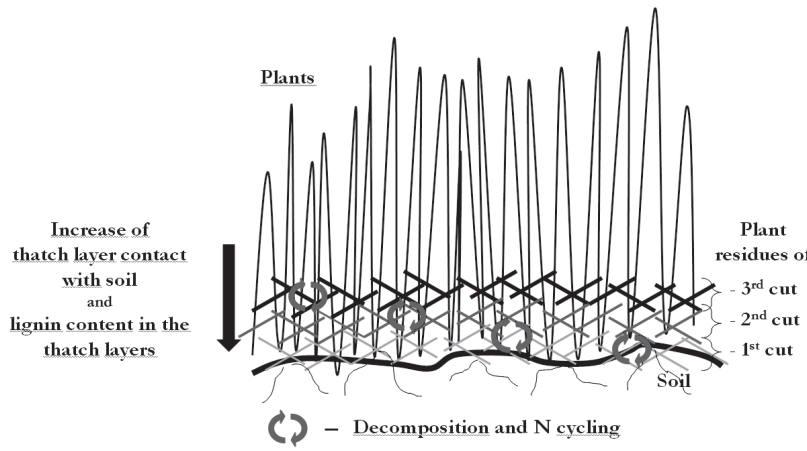

Figure 14. Formation of a thatch layer on TG sward.

The combined effect of taller grass in the sward after cutting, longer grass residues, and denser sward may have hindered residues from reaching the soil surface. Several studies have shown that early-phase decomposition of plant residues is positively influenced by the degree of contact between

plant residues and soil (Jensen, 1994; Sørensen et al., 1996). Close contact with soil usually will increase the microbial decomposition of organic matter (Douglas et al., 1980; Cogle et al., 1989) and this is mainly due to the higher moisture content in residues (Parr & Papendick, 1978). Our study showed that the rate of decay was influenced by the length of the period when grass residues stayed moist after cutting (**III**). Grass residues left on turf dry quickly in the sun and wind, which means their decomposition occurs slowly. Consequently, it can be concluded that RRI may be influenced by the length and density of the TG sward, and that RRI is greater in swards that are shorter and less dense.

6.2.2. Grass-clover sward

The amount of plant residues left on GC sward surfaces was significantly larger than the amount left on TG sward surfaces. For GC sward, RRI affected DMY in both the fertilized (GCN80) and nonfertilized variant (GCN0), as approximately 200 kg N ha^{-1} was returned to the sward by residues in both variants. Thus, the N amount in residues was sufficient for decomposition as well as for DMY formation. In the GCN80 variant, an additional 80 kg N ha^{-1} was applied, but this amount did not have any effect on DMY compared to the GCN0 variant. One possible reason may be that there was more N in the GCN80 variant soil than plants could uptake, as root nodule bacteria also increase the amount of N returned to soil. Another possible reason may be that some N was used to decompose residues.

In the literature, various studies have reported how residues from fertilized variants contain more recalcitrant compounds than residues from nonfertilized variants, and thus require more N for decomposition. Some N also may have been bound in recalcitrant compounds that was unavailable for plant uptake, and thus became part of the SOM. In soil of the fertilized variant, this effect could be seen as a higher Ntot. With RRT treatment, NREC and NUE decreased significantly compared to RRM treatment. Previous research has shown that fertilizer N recovery from perennial grassland soils is often low compared to that of cultivated crops (Power, 1981).

Various reasons for poor fertilizer recovery have been postulated. Perennial grasses have an abundance of carbonaceous root material, which may contribute to N immobilization from fertilizers (Power, 1980).

Similarly, a considerable amount of N may be immobilized in microbial biomass (Amato & Ladd, 1980) or incorporated into cyclic-N compounds, which are relatively resistant to microbial decomposition (Legg et al., 1971; Alien et al., 1973). Several publications suggest that 3–5 years of fertilization is required for the N cycle to establish new steady state levels (Clark, 1977; Power, 1981; Power & Legg, 1984).

In summary, our experiment showed that the herbage growth of both TG and GC swards increased with varying fertilization rates. RRT treatment increased sward herbage growth only when the N amount from fertilization or plant residues was sufficient for decomposition and DMY formation. The RRI on DMY of various swards was seen when the N amount from fertilization or residues was at least 200 kg N ha^{-1} or higher.

6.3. Soil organic carbon (Papers I, IV, V)

Our experiment was conducted on soil classified according to the WRB as *Stagnic Luvisols* (FAO, 2006), which are suitable for a wide range of agricultural uses. *Stagnic Luvisols* are medium fertility soils that are mainly used as arable land (74%), forested (22%), or under grasslands (4%) (Kõlli et al., 2010). The humus layer of *Stagnic Luvisols* is generally of medium or low content (i.e., 19–24 mg g^{-1}), which implies low fertility. In order to improve SOM stock in *Stagnic Luvisols*, short-term grasslands are used in crop rotation.

Information is scarce about SOC grasslands stock established on *Stagnic Luvisols* (**I**; Kõlli et al., 2009). Based on Kõlli et al. (2009), the average SOC of *Luvisols* in epipedon (humus cover) (21.9 cm) is 45 t ha^{-1}. At the start of our experiment, soil at a depth of 0–20 cm contained 42 t SOC ha^{-1}. Our results showed that grasslands with different species influence SOC accumulation in soil differently. Over the five-year experimental period, SOC stock at a depth of 20 cm increased significantly more in GC sward (43 t ha^{-1}) than in TG sward, where SOC stock decreased to 39 t ha^{-1}. With RRT treatment, however, SOC stock also increased in TG sward.

In Estonia, where upland grassland soils have an automorphic moisture regime, SOC pools in epipedon are 40–114 kg ha^{-1} and higher in soils with higher carbonate content (**I**). The average SOC stock of grasslands

epipedon in Estonia is 70 t ha^{-1} (Kõlli et al., 2009). Unfortunately, SOC stock results from other studies are difficult to compare because different soil layer thicknesses were used for SOC measurements. In order to overcome this problem, one option is to calculate SOC stock for separate profiles (i.e., epipedon layer (or topsoil, or humus cover) or soil cover). A second option is to calculate SOC stock at different soil depths; typically, depths of 0–20 cm are widely used, but also data for depths of 0–30 cm may be found. According to Lal et al. (1998) for mean grassland ecosystems, SOC stock is given as 116 t ha^{-1} at a depth of 1 m. Arrouayes et al. (2001) found that land under permanent grassland contained average SOC stocks of nearly 70 t C ha^{-1} at depths of 0–30 cm.

Apart from soil type, water regimens also have a significant impact on SOC accumulation (**I**). SOC stock of epipedon in hydromorphous mineral soils is 117.0 t ha^{-1} in grasslands and 184 t ha^{-1} in wetlands (**I;** Kõlli et al., 2009). In natural coastal grasslands, decomposition and mineralization are reduced due to moist conditions, resulting in SOC accumulation and the formation of thin C-rich humus horizons (**I; IV;** Kõlli et al., 2009).

In our experiment, different swards were established on the same soil type. Although soil type and soil characteristics were similar, our results were affected mostly by plant species and sward management measures. Plant species composition may have an impact on SOC stock because below-ground C input varies significantly in different plant mixtures. Wedin et al. (1995) presented 25-fold differences in below-ground net primary productivity, whereas above-ground productivity only showed two-fold variation. In GC sward, SOC content increased significantly from roots compared to TG sward; the extent of this effect is due to the positive impact of legumes on SOC and Ntot content that were noted in other studies (Spehn et al., 2002; Fornana & Tilman, 2008; Drinkwater et al., 1998). Adding legumes to sward increases N_m in soil (De Deyn et al., 2009), which results in an increase of above-ground grassland DMY as well as below-ground biomass (Fornara & Tilman, 2008).

From a morphological point of view, G and Cl root systems differ quite markedly. G generally profit from a larger root biomass (Hebeisen et al., 1997; Castle et al., 2002). It has been found that Cl root turnover is significantly faster compared to G root decomposition (Rasmussen et al., 2007). In GC sward, SOC originates mainly from G (Wardle et al., 1999; Eriksen et al., 2004; Rasmussen et al., 2007) and N from Cl (Rasmussen et al., 2007). N_m during decomposition of Cl roots is used by microbes

in the decomposition of G roots, where N immobilization probably will occur, while N released from Cl will be bound to SOM and increase Ntot content. Consequently, the Cl root pool is a key component in the build-up of the soil N pool (Drinkwater et al., 1998).

In our experiment, SOC content in various swards increased mainly in the top layer after arable land had been turned into grassland (**V**). At a depth of 5–20 cm, SOC content decreased, but significantly more in TG sward soil, which indicates a small root effect at that depth. In TG sward soil, a dense root system is formed at 0–5 cm (Qian et al., 2010). Although SOC content decreased with both RRT and RRM treatments, the decrease was slower with RRT. Decreases in SOC content at lower soil layers indicate the lack of (or low) C input at that depth, along with SOC mineralization that causes decreases in both SOC and Ntot content. Mineralized C and N probably will be leached to deeper layers of the soil profile. Steinbeiss et al. (2008b) attributed SOC content decreases in deeper horizons to C leaching into deeper soil layers and concluded that plant-derived C is preferentially mineralized and adsorbed to soil particles, while mobilized soil C is transported further down the soil profile. SOC leaching also has been mentioned by Jobbágy & Jackson (2000). In GC sward, however, SOC content at lower soil layers did not change over the five-year experimental period. At depths of 5–20 cm, root turnover probably occurred because G and Cl roots extend deeper than those of TG roots.

Various grassland species have different responses to fertilization as well as to SOC stock. It is generally assumed that a larger amount of SOM accounts for higher SOC accumulation (Nyborg et al., 1999), and to increased G root growth in response to N fertilization (Anderson & Coleman 1985; Malhi & Gill, 2002). In our experiment, the DMY of TG sward varied significantly among fertilization variants (i.e., 1333–5924 kg KA ha^{-1}), but the SOC content at depths of 0–5 cm did not vary.

Returning different amounts of residues to the TG sward surface following fertilization did not have any impact on SOC content because the addition of fresh material may have created a 'priming effect' by activating the decomposition of SOM (Fontaine et al., 2004). The mechanisms involved in the priming effect are not fully understood (Kuzyakov et al., 2000; Fontaine et al., 2003). A high content of easily-decomposable SOC can lead to fast microbial growth, which likely results in higher microbial biomass and activity (Schenk et al., 1995). In

turn, stimulation of soil microbiological activity may increase the SOM decomposition rate (Kuzyakov et al., 2000). Steinbeiss et al. (2008b) observed that RRT treatment not only did not increase SOC content, but also that the amount of C added to soil could be attributed to the amount mineralized in soil. Starr & Deroo (1981) found that the N amount from SOM increased significantly after fertilization possibly due to an enhanced rate of fertilizer mineralization (i.e., the priming effect) (Kuzaykov et al., 2006). Another possible reason why SOC content was unaffected by increased C fertilization input may be that N fertilization (i.e., external N availability) was affected by below-ground decomposers that modified microbial community composition and led to the production of soil enzymes involved in the depolymerization of SOM and plant litter (Fog, 1988; Saiya-Cork et al., 2002; Yao et al., 2009). Increasing N availability may inhibit the activity of oxidative enzymes that decompose recalcitrant compounds and increase SOC storage (Kirk & Farell, 1987), while stimulating the activity of cellulolytic enzymes such as soil cellulase (Fog, 1988; Waldtrop et al., 2004) reduces SOC storage. As a consequence, the N effect on decomposition depends on the chemical composition of SOM (Sinsabaugh et al., 2002). The cellulose content of plants is generally high (Bandaranayake et al., 2003; Qian et al., 2003; Yao et al., 2009), which can lead to an assumption that higher N availability may produce a positive impact on SOC mineralization of TG sward.

In our experiment, the same amount of residues was left on GC sward surfaces for both GCN0 and GCN80 variants. SOC contents with various fertilization rates were similar, indicating that N availability in the soil (i.e., fertilization) in GC sward did not have any effect on SOC content (i.e., SOC mineralization). In our nylon bag decomposition experiment with GC sward, we showed how residues of the GCN80 variant decomposed slower resulting in a larger input of SOM compared to the GCN0 variant. SOC content did not increase in this soil, which shows that higher N availability increased SOC decomposition (mineralization).

The effect of GC sward residues on SOC stock at a depth of 0–5 cm (i.e., top layer) was significantly lower compared to those of TG sward. This variation may be due to differences in the chemical composition of plant residues that became a part of SOM. In TG sward, a thatch layer was formed with lower layers that contained mostly recalcitrant compounds (e.g., lignin) that were in better contact with the soil surface (Yao et al., 2009). Lignin decomposition products favor an increase in SOC content (Takeda, 1998).

GC sward plant residues had better decomposition conditions. Compared to TG sward, GC sward is less dense and creates better contact between plant residues and soil; thus, decomposing material did not dry so easily, organic matter input did not exceed plant residue decomposition, and a thatch layer did not form on the surface. The residues that became part of SOM were of better quality compared to the TG sward thatch layer. High-quality plant residues (i.e., high N, low lignin concentrations) mineralize rapidly, but may not contribute much to the maintenance of SOM (Handayanto et al., 1997). Therefore, larger amounts of lignin-rich material become a part of SOM in TG sward compared to GC sward, and SOC content also increased more in TG sward soil.

CONCLUSIONS

- The decomposition rate of plant residues depended on sward species composition and their developmental stage during cutting. White clover residues decomposed faster than grasses residues. Turfgrass sward residues at earlier developmental stages during cutting also decomposed faster than grass residues in grass-clover sward cut at later stages.
- Differing residue N content due to fertilization did not affect decomposition rate, indicating that N content at the beginning of decomposition did not affect the process. Residues with similar N content may decompose at different rates because the content of C compounds is different. The N content of residues is a suitable indicator for predicting the amount of mineralized N because more N was mineralized from residues with a higher initial N content.
- The effect of residues on sward herbage growth depended on species composition and sward management (i.e., fertilization and cutting frequency). The residues effect was higher in grass-clover sward that was cut less frequently, and dry matter yield was higher compared to turfgrass sward. For turfgrass sward, returning of residues is effective only during the first part of the vegetation period when sward is sparse and shorter, as contact between residues and soil surface is enhanced; this implies that returning of residues may be significantly affected by characteristics such as density and plant height. With dense sward, residues remain on top of herbage, where they dry and nutrients cannot reach plants.
- Sward herbage growth increased due to residues only if the N released by residues was sufficient for decomposers as well as for sward growth. The N amount released by residues depended on fertilization rate, the amount of residues left to decompose, and sward species composition. For grass-clover sward, the largest residues returning effect occurred in the nonfertilized variant. For turfgrass sward, residues returning effect was noticeable only if the N fertilizer rate was 160 kg N ha^{-1} or 240 kg N ha^{-1}.

- Residues returning effect on sward SOC content and stock depended on species composition and sward management (i.e., fertilization and cutting frequency). Turfgrass residues increased Soil organic carbon content in the top layer (i.e., 0–5 cm) significantly more than grass-clover sward residues; residues returning effect was significantly smaller at lower soil layers (i.e., 5–20 cm). Fertilization and cutting frequency had an impact on plant chemical composition that, in turn, affected the chemical composition and decomposition of soil organic matter.
- Frequent cutting of turfgrass sward caused the formation of a thatch layer on top of the soil that prevented contact between easily-decomposable turfgrass residues and soil. As a result, soil N input decreased, which inhibited decomposer activity while increasing the amount of recalcitrant organic material going into soil. No thatch layer formed on the grass-clover sward soil surface. The turfgrass sward thatch layer decomposed less than grass-clover sward residues, which increase soil organic carbon content more than easily-decomposable material.

Application of the study results

Our research results can be used to give advice on optimal sward management (i.e., fertilization, cutting frequency, returning or removing of residues) for turfgrass and grasslands that are out of use. This area of research is important in Estonia where cutting subsidies are available. Previously, it was not known what happens to grassland soil if plant residues are left to decompose on sward surfaces. Our experiment confirmed that residues can be left on turfgrass sward surface only in the first part of summer while the sward is still sparse. In case of dense sward, residues remain on the surface and nutrients cannot reach growing plants; the residue layer eventually damages turfgrass sward by providing spaces and making it less sparse. If residues are left on the turfgrass sward surface, cutting either should be performed before rainfall or the sward should be irrigated after cutting to maintain moisture content for decomposition and to improve the percolation of residues to the soil surface. When residues are returned, the turfgrass sward should be fertilized or the N in residues will be bound to SOM and be unavailable for plant use; this is especially important during the first years after turfgrass is established. As turfgrass

sward ages, organic matter accumulates in the soil, and SOC and Ntot content increase, leading also to increases in the mineralization of SOM nutrients. Older turfgrass swards need less N fertilization, as applying higher fertilization rates has no effect on herbage growth as the amount of residues formed after cutting increases. In such cases, the amount of added residues may exceed their decomposition rate, which results in the formation of a thatch layer on the soil surface.
The addition of legumes to the seed mixture has a positive impact on plant growth and SOM content. It is not necessary to apply mineral fertilizers to grass-clover grassland when cut residues have been left to decompose, as the N amount is sufficient for decomposition of residues as well as for plant growth. By leaving residues to decompose on sward surfaces, dry matter yield and SOC content increases.
The following hypotheses arose during our experiment and need further study:

- To investigate why leaving plant residues to decompose on turfgrass sward surfaces did not have any effect on herbage growth in all fertilization variants (especially in the control variant where the effect of plant residues on dry matter yield was rather negative).
- To investigate why the effect of plant residues left on the sward seen in our experiment was significantly different from results of analogous studies. Is it because of different soil types (i.e., SOC content, texture), or weather conditions?
- To specify N movement released during the decomposition of residues left on sward surfaces in the plant-soil system by using the stable isotopes (15N) method.

SUMMARY IN ESTONIAN

TAIMEJÄÄTMETE JA VÄETAMISE MÕJU TAIMEDE KASVULE JA ORGAANILISE SÜSINIKU SISALDUSELE MULLAS

Sissejuhatus

Mulla orgaaniline süsinik (C_{org}) on olulise tähtsusega globaalses süsiniku ringluses ja muld on oluline süsiniku talletaja. Ta sisaldab kolm korda rohkem süsinikku võrreldes selle sisaldusega atmosfääris ($2.25 \cdot 10^{12}$ versus $0.75 \cdot 10^{12}$ t C) ja viis korda rohkem, kui seda on taimestikus (Post et al., 1982; Jobbágy & Jackson, 2000). Mullas oleva C_{org} sisaldus sõltub mitmetest teguritest: ilmastikust, maakasutusest, majandamisest jne. Võrreldes põllumullaga on sisaldus suurem rohumaa mullas (Cole et al., 1993; Jackson et al., 1996), kuna mulda mineva orgaanilise aine (varis, taimede juured) kogused on seal suuremad ja lagunemine pärsitud. Saagi saamise eesmärkidel kasutatavatel rohumaadel (hein, silo tootmine, loomade karjatamine) on peamiseks orgaanilise aine allikaks taimede juured. Tootmisest väljas olevatel rohumaadel, kuid samuti ka murudel, võivad neile lisanduda ka pärast niitmist taimiku pinnale jäävad taimsed jäätmed.

Mulda mineva orgaanilise aine koguse ja lagunemisprotsessi kiiruse vaheline tasakaal iseloomustab seda, kui suur on mullas orgaanilise aine varu (Amundson et al., 2001). Orgaanilise aine lagunemise kiirust mõjutavad peamiselt kolm faktorit: mullaorganismid (lagundajad), lagunemiskeskkond ja orgaanilise aine keemiline koostis (Swift et al., 1979; Stott et al., 1986; Heal et al., 1997; Martens, 2000). Süsinikühendite sisaldus on oluline orgaanilise aine lagundatavuse indikaator, samuti sõltub lagundajate lagundamisvõime süsinikühendite keemilisest struktuurist (Heal et al., 1997; Gunnarsson & Marstrop, 2002; Trinsoutrot et al., 2002). Orgaanilise aine lagunemine sõltub veel lämmastiku kättesaadavusest ning lagundatava materjali C/N suhtest (Frankenberger and Abdelmagid, 1985; Trinsoutrot et al., 2000).

Käesolev doktoritöö keskendub erinevate taimikutega rohumaade majandamisvõtete mõju uurimisele taimede kasvule ja mulla orgaanilise süsiniku sisaldusele.

Käesolevas doktoritöös uuritakse majandamisvõtete mõju erineva taimikuga rohumaade taimede kasvule ja mulla orgaanilise süsiniku sisaldusele.

Töö peamine hüpotees: niitmisejärgselt lagunema jäetud taimejäätmete mõju taimiku edasisele kasvule ja mulla orgaanilise süsiniku sisaldusele sõltub rohumaa liigilisest koosseisust ja rohumaa majandamisest (väetamisest ja niitesagedusest). Neist tegureist sõltuvad taimejäätmete omadused, nende lagunemiskiirus ja mineraliseeruva lämmastiku kogus. Uurimustöö eesmärgid:

- Uurida taimejäätmete lagunemist taimiku pinnal ja lämmastiku mineraliseerumist.
- Selgitada, kas taimejäätmete keemiline koostis (N sisaldus ja C/N suhe) mõjutab lagunemisdünaamikat ning avaldab toimet taimede kasvule.
- Uurida taimejäätmete mõju, samuti ka jäätmete ja väetamise koosmõju mõju mulla orgaanilise süsiniku sisaldusele.

Metoodika

Uurimaks taimejäätmete mõju taimede kasvule ja mulla C_{org} sisaldusele, rajati 2003. a. Eesti Maaülikooli Eerika katsejaama põllule kahe erineva seemneseguga katsed:

1. Murusegu: punane aruhein (*Festuca rubra rubra*) 50% ja aasnurmikas (*Poa pratensis*) 50%;
2. Valge ristiku ja kõrreliste segu: valge ristik (*Trifolium repens*) 28%, põldtimut (*Phleum pratense*) 34% ja karjamaa raihein (*Lolium perenne*) 38%.

Enne katsete rajamist määrati mullas orgaanilise süsiniku (C_{org}) ja üldlämmastiku ($N_{üld}$) sisaldus. Taimejäätmete mõju taimede kasvule uuriti aastatel 2004 – 2008. Murutaimikut niideti kasvuperioodi jooksul. keskmiselt 15 – 20 korda, valge ristiku ja kõrreliste seguga taimikut 3 – 5 korda. Pärast igat niitmist ja saagi kaalumist tagastati osadele lappidele taimejäätmed kasvukohale (RRT), teistelt need eemaldati (RRM). Mõlemat varianti väetati ühesuguste väetusnormidega:

1. Murutaimik: $N_0P_0K_0$ (TGN0, TG – murutaimik), $N_{80}P_{11}K_{48}$ (TGN80), $N_{160}P_{22}K_{96}$ (TGN160), $N_{240}P_{34}K_{144}$ (TGN240), $N_{320}P_{45}K_{192}$ (TGN320) ja $N_{400}P_{56}K_{240}$ (TGN400);

2. Valge ristiku ja kõrreliste segu taimik: $N_0P_0K_0$ (GCN0, GC – valge ristiku ja kõrreliste seguga taimik) ja $N_{80}P2_6K_{50}$ (GCN80).

Aastatel 2006 – 2007 uuriti 8 nädala jooksul pärast niitmist taimikule tagastatud taimejäätmete lagunemist ja sealt N mineraliseerumist. Murutaimikul uuriti taimejäätmete lagunemist pärast niitmist, mis toimusid 15. mai, 13. septembril ja 26. oktoobril 2006. aastal ja 16. mail 2007. aastal. Murujäätmete lagunemise uurimisel kõrreliste liike ei eraldatud. Valge ristiku ja kõrreliste jäätmete lagunemist uuriti pärast esimest niidet (30. mail 2006. aastal ja 31. mail 2007. aastal). Lagunemise uurimisel 2006. aastal taimi liigiti ei eraldatud ning proov sisaldas valget ristikut ning kõrrelisi samas vahekorras nagu see kasvas taimikus, 2007. aastal määrati lagunemisaste eraldi nii valge ristikul kui kõrrelistel.
Jäätmete lagunemise uurimiseks võeti vahetult pärast niitmist igast variandist 100 g niidetud materjali. Igast proovist kaaluti 20 grammi, mis pandi 20 * 20 cm nailonkotti (koti augu läbimõõt 1.5 mm). Proovi ülejäänud osast määrati kuivaine ja tehti keemilised analüüsid. Nailonkotid taimejäätmetega viidi samale katselapile, kust materjal pärit oli ning kinnitati klambritega taimiku pinnale. Kotid eemaldati lappidelt 2 – 3 nädalase intervalliga 8 nädala jooksul (Table 1). Kotis olev materjal kuivatati 105 °C juures, kaaluti ja arvutati välja kaalukadu võrreldes kotti pandud proovi esialgse massiga. Kaalukao (%) arvutamiseks kasutati valemit:

$$\text{Kaalukadu} = 100 * (M_0 - M_t)/M_0$$

kus M_0 on proovi esialgne kaal (g);
M_t on proovi kaal ajahetkel t, kui nailonkott katselt eemaldati.

Nailonkotti allesjäänud taimejäätmetest määrati $N_{üld}$ sisaldus. Mineraliseerunud N (N_m) kogus (% esialgsest kogusest) ajahetkel t, mil nailonkotid katselt eemaldati, arvutati kasutades valemit:

$$N_m = 100 * (N_i - N_t)/N_i$$

kus N_i on esialgne N kogus (mg) taimejäätmetes;
N_t on N kogus (mg) proovis ajahetkel t.

Kõikidelt katsevariantidelt võeti 2008. aasta septembris 0 – 5 ja 5 – 20 cm sügavuselt mullaproovid. Neis määrati C_{org} ja $N_{üld}$ sisaldus, mille põhjal arvutati C_{org} varu (t ha^{-1}) kahel erineval sügavusel kasutades valemit:

$$C_{org} \text{ varu} = BD * C_{org} * D / 10$$

kus C_{org} on mulla orgaanilise süsiniku sisaldus (mg g^{-1});
BD on mulla lasuvustihedus (g cm^{-3});

D on mullakihi tüsedus (cm), mille kohta C_{org} varu arvutati; 5 cm mullakihi 0 – 5 cm ja 15 cm 5 – 20 cm jaoks.

Lasuvustihedus (g cm^{-3}) arvutati välja kasutades Adams (1973) valemit:

$$BD=100/((OA/10/0.244) + ((100 - OA/10) / 1.64))$$

kus OA on orgaanilise aine sisaldus mullas (mg g^{-1}); me eeldame, et mulla orgaanilise aine sisaldab 58% C_{org} (Mann, 1986).

Tulemused ja arutelu

Niidetud taimejäätmete lagunemisdünaamika taimiku pinnal sõltus taimeliigist ja taimiku niitmiseaegsest arengufaasist. Kõige kiiremini lagunes 8 nädala jooksul valge ristik (73%). Kõige aeglasemalt aga kõrreliste segu (49%). Murujäätmetest lagunes sama aja jooksul 64%. Erinev lagunemiskiirus võrreldes murutaimiku jäätmetega olid tingitud: (i) kõrreliste ja valge ristiku erinevast keemilisest koostisest (erinevate süsinikühendite sisaldus) ja (ii) valge ristiku ja kõrreliste segu taimikult pärinevate kõrreliste jäätmete vanemast arengufaasist. Taimede arenedes suureneb neis raskemini lagunevate süsinikühendite sisaldus ja lagunemine aeglustub.

Erinevatest väetusvariantidest pärit murujäätmed olid küll erineva N sisaldusega, kuid lagunesid 8 nädala jooksul sama kiirusega. Valge ristiku ja kõrreliste segu jäätmete N sisaldus oli lagunemisprotsessi alguses kontrollivariandis ja PK-väetisega väetatud variandis (lämmastikväetist ei antud enne esimest niidet) ühesugune, kuid väetatud variandilt pärit jäätmed lagunesid kontrollvariandiga võrreldes aeglasemalt. Sellest järeldub, et jäätmete lagunemiskiirus ei sõltu N sisaldusest, mistõttu see näitaja ei sobi jäätmete lagundatavuse indikaatoriks. Fosfor- ja kaaliumväetisega (PK) väetamine kiirendas taimede arengut ning selle käigus muutus erinevate süsinikühendite sisaldus. Kontrollvariandiga võrreldes suurenes

PK-väetisega väetatud variandis raskesti lagunevate ühendite sisaldus, mistõttu oli seal ka suurem kogus taimedes sisalduvast lämmastikust seotud raskemini lagunevate ühenditega. N kättesaadavuse vähenemine lagundajatele põhjustas lagunemise aeglustumise.

Taimejäätmete tagastamine suurendas oluliselt taimiku saaki vaid ristiku-kõrrelise katses. Mõju oli suurim väetamata variandis, kus saak suurenes 52%, väetatud variandis aga 39%. Võrreldes väetamata ja taimejäätmeteta variandiga suurenes väetamise ja taimejäätmete mõjul saak 70%. Ainult väetise mõjul kasvas taimiku saak 22%. Murutaimikul avaldus taimejäätmete mõju ainult siis, kui taimikut väetati N normiga 160 ja 240 kg ha^{-1}. Taimejäätmete mõju ilmnes alles nii kõrge N normi korral tõenäoliselt seetõttu, et suur osa jäätmetega tagastatud lämmastikust ei jõua muru puhul taimedeni. Seda näitasid tulemused murutaimiku väetamata variandis ja normiga 80 kg N ha^{-1} väetatud variandis, kus tõenäoliselt kogu taimejäätmetest vabanenud lämmastik kulus taimejäätmete lagundamiseks. Kui aga kasutati suuremaid norme kui 240 kg N ha^{-1}, siis väetisega antud ja taimejäätmetega tagastatud lämmastiku kogus oli suurem kui kõrreliste N omastamise võime. Seda kinnitavad ka tulemused, mis saadi variandis, kus jäätmeid ei tagastatud, sest seal suurenes saak ka 240 kg N ha^{-1} suuremate normide korral. Valge ristiku ja kõrreliste segu katses oli taimejäätmete mõju saagile väetamata variandis suurem, kui väetatud variandis, ehkki neis mõlemas tagastati taimejäätmetega ühesugune kogus lämmastikku (190 – 204 kg N ha^{-1}). Põhjuseks oli nähtavasti see, et väetatud variandist pärit taimejäätmete lagundamisel vabanes vähem lämmastikku, sest need jäätmed sisaldasid rohkem raskemini lagundatavaid ühendeid. Nendega seoti ka osa jäätmetes sisalduvast lämmastikust. Seetõttu tarvitati väetisega antud lämmastikust osa ka taimejäätmete lagundamiseks ning väetisega antud N mõju taimiku saagile jäi väikseks.

Mõlemas katses olnud taimikutüübi mullas suurenes C_{org} sisaldus 5. aastaga oluliselt. Selle kasv oli suurem valge ristiku ja kõrreliste seguga mullas, mis oli tingitud liblikõielise positiivsest mõjust mulla süsiniku ja lämmastiku sisaldusele. Ristik suurendas mulla orgaanilisest ainest lämmastiku mineraliseerumist, mille tulemusena suurenes nii taimiku maapealne kui maa-alune biomass. Kokkuvõttes mõjus see positiivselt mulla orgaanilise aine sisaldusele.

Taimejäätmete jätmine taimiku pinnale, suurendas mulla C_{org} sisaldust mõlemas katsevariandis. Valge ristiku ja kõrreliste segu jäätmete mõju C_{org} varule pindmises kihis (0 – 5 cm) oli oluliselt väiksem võrreldes muru

jäätmete mõjuga. Mõju erinevus võis olla tingitud sellest, et taimejäätmete keemiline koostis, mis sai mulla orgaanilise aine osaks, oli kahel taimiku tüübil erinev. Murutaimikul moodustus mulla pinnale kõdu, mida valge ristiku ja kõrreliste seguga taimikul ei tekkinud. Mulla pinnaga oli kontaktis kõdu alumine kiht, mis tõenäoliselt sisaldas palju ligniini (Yao et al., 2009). Ligniini lagunemisproduktid soodustavad C_{org} sisalduse suurenemist mullas (Takeda, 1998). Valge ristiku ja kõrreliste seguga taimiku taimejäätmetel olid paremad lagunemistingimused. Ristiku-kõrreliste taimik oli murutaimikuga võrreldes hõredam, mistõttu taimejäätmete kontakt oli mullaga parem, lagunev materjal ei kuivanud nii kergesti ning taimikule ei moodustunud kõdukihti. Lagunemissaadused, mis said mulla orgaanilise aine osaks, olid parema kvaliteediga kui murutaimikul tekkinud kõdukihi saagis. Parema kvaliteediga (kõrge N sisaldus, madal ligniini sisaldus) materjal laguneb kiiresti. Lagunemise käigus tekib vähe selliseid laguprodukte, mis suurendavad mulla orgaanilise aine sisaldust (Handayanto et al., 1997). Murutaimikul sai mulla orgaanilise aine osaks suurem kogus ligniinirikkamat materjali võrreldes ristiku-kõrreliste taimikuga ja C_{org} kasv oli seetõttu murutaimiku mullas suurem.

Väetamine ei mõjutanud C_{org} ja $N_{üld}$ murutaimiku mullas ega ka C_{org} sisaldust valge ristiku ja kõrreliste segu taimiku mullas, sest väetisega antud N kiirendas mulla orgaanilise aine lagunemist. Väetamine suurendas ristiku-kõrreliste segu katses mulla pindmises kihis (0 – 5 cm) $N_{üld}$ sisaldust, sest osa taimejäätmetes olevast N-st oli seal seotud süsinikühenditega, mis ei lagunenud ja said mulla orgaanilise aine osaks.

Kokkuvõte

Taimejäätmete lagunemiskiirus sõltus taimiku liigilisest koosseisust ja taimede arengufaasist niitmishetkel. Valge ristik lagunes kõrrelistega võrreldes kiiremini. Niitmishetkel nooremas arengufaasis olevate murutaimiku kõrreliste jäätmed lagunesid kiiremini, kui hilisemas arengufaasis niidetud kõrreliste jäätmed. Väetamisest tingitud suurem lämmastiku sisaldus jäätmetes nende lagunemise kiirust ei mõjutanud. Sellest järeldub, et N sisaldus lagunemisprotsessi alguses ei iseloomusta jäätmete lagunemisprotsessi kulgu. Ühesuguse N sisaldusega jäätmed võivad laguneda aga erineva kiirusega, sest süsinikühendite sisaldus on neis erinev. Kuivõrd lämmastik mineraliseerus rohkem neist jäätmetest, mille N sisaldus oli suurem, siis on see näitaja sobiv indikaator mineraliseeruva lämmastikukoguse ennustamiseks.

Taimejäätmete mõju taimede kasvule sõltus taimiku liigilisest koosseisust ja taimiku majandamisest. Mõju oli suurem ristiku-kõrreliste segu variandis. Seda niideti harvemini ja saagid olid suuremad kui murutaimiku variandis. Murutaimiku puhul oli jäätmete tagastamine efektiivne ainult vegetatsiooniperioodi esimesel poolel kui taimik oli hõredam. See näitab, et jäätmete tagastamisest saadav efekt sõltub taimiku tihedusest ja kõrgusest. Jäätmete tagastamine murule on efektiivne hõredama ja lühema taimiku korral, sest nende imbumine mullapinnale on siis vähem takistatud. Tiheda taimiku korral jäävad jäätmed taimedele kuivama ja neis sisalduvad toitained ei jõua mullapinnale.

Taimejäätmed soodustasid taimede kasvu ainult siis, kui neis sisalduva lämmastiku kogus oli piisav nii lagundajate tegevuseks kui ka taimede kasvuks. Jäätmetega tagastatud lämmastiku kogus sõltus taimiku väetamiseks kasutatud N normist, lagunema jäetud jäätmete kogusest ja taimiku liigilisest koosseisust. Kõige suurema efekti andis jäätmete tagastamine valge ristiku ja kõrreliste segu väetamata variandis. Muru puhul osutus jäätmete tagastamine efektiivseks ainult siis kui taimiku väetamisel kasutati väetisenorme 160 ja 240 kg N ha^{-1}.

Taimiku pinnal lagunevate taimejäätmete mõju mulla C_{org} sisaldusele (ja varule) sõltus taimiku liigilisest koosseisust ja taimiku majandamisest (väetamisest ja niitmissagedusest). Murutaimiku jäätmed suurendasid mulla orgaanilise süsiniku sisaldust pindmises kihis oluliselt rohkem, kui valge ristiku ja kõrreliste segu jäätmed. Alumises kihis oli mõju oluliselt väiksem mõlema taimiku puhul. Väetamine ja niitmissagedus mõjutasid taimede keemilist koostist. See mõjutas mulda mineva orgaanilise aine keemilist koostist ja lagunemist. Murutaimiku sage niitmine tekitas mullapinnale kõdukihi, mis takistas kergestilagunevate murujäätmete kontakti mullaga. Selle tagajärjel vähenes mulda mineva lämmastiku kogus. See pärssis lagundajate tegevust ja suurendas mulda mineva süsinikurikka orgaanilise aine kogust. Ristiku ja kõrreliste segu katses kõdukihti ei tekkinud, jäätmed olid lämmastikurikkamad ning mullapinnal lagunes rohkem orgaanilist ainet.

Uurimustöö tulemuste kasutamine

Uurimistulemuste baasil on võimalik anda teaduslikult põhjendatud soovitusi, kuidas murusid ja tootmisest väljasolevaid rohumaid optimaalselt majandada (väetamine, niitmissagedus, taimejäätmete jätmine taimiku pinnale lagunema või eemaldamine taimikult). Kuivõrd Eestis on võimalik

taotleda rohumaade niitmistoetust, siis on selline teave väga oluline. Enamasti ei teatagi, mis juhtub mullaga siis, kui kulusid kokku hoides jäetakse niitmisjäätmed rohustu pinnale lagunema. Antud tööst järeldub, et taimejäätmed võib jätta murule lagunema ainult suve esimesel poolel kui taimik on veel hõre. Tiheda taimiku korral jäävad jäätmed taimikule ning neis sisalduvad toitained ei jõua taimedeni. Tüse jäätmete kiht taimiku pinnal kahjustab muru, muutes selle hõredaks ja tekitades taimikusse tühikuid. Niita tuleks muru enne vihma või seda pärast niitmist kasta, et säilitada jäätmetes niiskust ja kiirendada jäätmete imbumist mulla pinnale. Jäätmete tagastamisega koos tuleks muru kindlasti väetada, sest vastasel korral seotakse jäätmetes olev lämmastik mulla orgaanilise aine koosseisu ja see ei ole taimedele omastatav. Eriti oluline on see just esimestel aastatel pärast muru rajamist. Muru vananedes toimub orgaanilise aine kuhjumine mulda, C_{org} ja $N_{üld}$ sisaldus suurenevad, selle tulemusena suureneb ka toitainete mineralisatsioon orgaanilisest ainest ning lämmastikväetise vajadus väheneb. Liiga suure N normiga väetamine kiirendab küll taimede kasvu ja suurendab niitmisel tekkivat jäätmete kogust, kuid juurde tulev jäätmete kogus võib ületada lagunemisvõime ning tekitada mullapinnale kõdukihi.

Liblikõielise lisamine seemnesegusse avaldab positiivset mõju taimede kasvule ja mulla C_{org} sisaldusele. Jättes valge ristiku ja kõrreliste seguga taimikul jäätmed taimiku pinnale lagunema, ei ole mineraalväetisega väetamine vajalik, sest taimejäätmetega tagastatavast lämmastikust piisab nii jäätmete lagundamiseks kui ka taimede kasvuks. Lagunevad jäätmed suurendavad saaki ja mulla C_{org} sisaldus.

Edasist uurimist vajavad küsimused:

- Uurida, miks murutaimiku pinnal lagunevad taimejäätmed ei avaldanud enamuses väetusvariantides mõju taimede kasvule? Miks kontrollvariandis oli mõju kuivaine saagile pigem negatiivne?
- Uurida, miks meie uurimustöö tulemused erinesid oluliselt teistes analoogsete uurimustöödede tulemustest? Kas põhjused võivad olla meie muldades (C_{org} sisalduses, lõimis) ja/või kliimas?
- Täpsustada taimikule lagunema jäetud taimejäätmete lagunemise käigus vabanenud lämmastiku liikumist taim-muld süsteemis, kasutades stabiilsete isotoopidega (15N) märgistamismeetodit.

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I can't think about that right now. If I do, I'll go crazy.
I'll think about that tomorrow.

(Scarlett O'Hara)

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I

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Organic matter of Estonian grassland soils

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Abstract. Soil organic carbon (SOC) and soil organic matter (SOM) contents of Estonian grassland soils are analysed in 20 soil groups using data from the database PEDON and CATENA. The SOC and SOM concentrations (g kg^{-1}) and pools (Mg ha^{-1}) for upland mineral soils (*Leptosols, Cambisols, Luvisols, Albeluvisols, Regosols;* total 9 groups), lowland mineral soils (*Gleysols, Fluvisols;* 9 groups) and wetland organic soils (*Histosols;* 2 groups) are given separately for humus cover (HC) and soil cover (SC). The SOC and SOM pools for the entire Estonian grasslands were calculated on the basis of different soil types, morphological characteristics and distribution superficies. It was concluded that in Estonian grasslands SC 39.9±8.0 Tg of organic carbon is sequestered, 76.2% of which is found in HC and 23.8% in subsoils. Grassland SOC is sequestered in 69.1±12.6 Tg of SOM. A quality analysis of humus covers of grassland soils (evaluated from the pedo-ecological perspective) distinguished 5 quality groups and 15 subdivisions.

Key words: grassland soils, humus status of grassland soil, quality of humus cover, carbon sequestration, SOC and SOM concentration and pools

INTRODUCTION

The sequestration of soil organic matter (SOM) and the soil organic carbon (SOC) in soil organic matter are widely recognized as agents of soil formation and functioning (Lal et al., 1998a; Pulleman et al., 2000; Shaffer & Ma, 2001).

Quantification of SOM and SOC flow and sequestration in soil has tremendous importance (Kern et al., 1998; Bernoux et al., 2002; Nemeth et al., 2002; Zhou et al., 2003). SOC may be sequestered in soil horizons in different forms and in variable relations with nitrogen (Batjes, 1996; DeBusk et al., 2001). The SOM (as well as SOC) flow throughout the soil begins with litter falling on the surface or into the soil, continues with its disintegration, transformation into humus and ends with the disappearance from the soil by its consumption, by soil organisms, by complete mineralization or by illuviation into subsoil or eluviation out of the soil profile. Each soil type has specific characteristics (input => acting and sequestration => output) of SOC flow (Körchens et al., 1998; Yakimenko, 1998; Neill et al., 1998; Genxu et al., 2002). Depending on the soil type and land use, the sequestered carbon may have varying fabrics, properties, quality and residence time in soil the complexity of which can be related to types of humus layers (Kõlli, 1992, 1994).

To determine SOC and SOM sequestered into different grassland soils, a macro-morphological quantitative approach based on horizon samples was used in our

research. We have previously determined SOC and SOM pools accumulated into Estonian arable soils (Kõlli & Ellermäe, 2003) and forest soils (Kõlli et al., 2004).

The main tasks of the present work, which fulfils SOC and SOM research gaps in relation to Estonian semi-natural grasslands, were the following: (1) to determine SOC and SOM pools in Estonian grasslands' soil cover (SC); (2) to analyse the humus cover (HC) and subsoil roles in SOC and SOM sequestration into grasslands SC by different soil groups, and (3) to elucidate pedo-ecological regularities of the HC quality of grassland soils.

MATERIALS AND METHODS

The quantitative characteristics of grassland soils originate mainly from the soil profile horizons database PEDON which contains data of 82 grassland experimental areas. PEDON was compiled mainly during 1967–85 and was updated in 1986–95 and in 1999–2002. Data of organic soils were later completed using the humus status research transect data from the database CATENA formed during field studies in 1987–1992.

For the present work, the data on soil morphology (fabric and thickness of soil horizons), bulk density and SOC and SOM concentrations (in the fine earth, ø <1 mm) of humus (A), raw-humuous (AT), histic (T), eluvial (E) and illuvial (B) horizons were used. The carbon concentration for each soil horizon was determined by the Tjurin method (Vorobyova, 1998) based on soil samples taken during field research. For calculation of SOC and SOM pools in the HC and SC of individual profiles (by research areas), the SOC and SOM concentration, soil bulk density and content of coarse fragments in each horizon of the soil profiles were taken into account. The role of rock fragments in soil horizons was determined by their volume in field conditions. The bulk density samples were taken from approximately one third of the profiles. Later the information was generalised and used in the calculation of SOC and SOM pools in different soil horizons and the SC as a whole.

In the present work the pools of SOC and SOM by soils were estimated for two SC layers: (1) HC or epipedon, which consists of humus, raw-humous and/or peat *(histic)* horizons and (2) SC or solum as a whole, the depth of which reaches from the surface to the unchanged parent material or to C horizon. Therefore the SC consists of HC and subsoil including eluvial (E) and illuvial (B) horizons. The thickness of SC was determined by the depth of the boundary between B and C horizons. In the presence of BC horizon, the solum thickness was measured from the surface to the middle of the BC horizon.

The area of Estonian natural grasslands was 299.5 10^3 hectares during the years 2000-2001, forming 6.6% of the total land or 20.9% of agricultural land of Estonia (Statistical…, 2003). For the calculation of means and for the analysis of variance, the PC program MS STATISTICA 7 was used. The soil group names and codes are given in the system of the World Reference Base for Soil Resources (WRB; FAO et al., 1998). The correlation between the Estonian Soil Classification (ESC) and the WRB for Estonian soils is shown in Table 1 by soil codes.

RESULTS AND DISCUSSION

Overall, the thickness of grasslands sola varies between 25 and 77 cm, with standard deviation 5–20 cm (Table 1). Only the average thickness of *Leptosols (skeletic, rendzic)* and *Fluvisols (salic)* formed on coastal areas is smaller. In most cases, HC thickness is between 19 and 29 cm; but the humus horizons of very young coastal and drought-prone skeletal soils are much thinner. It must be mentioned that for *Histosols,* the unique HC (30 cm) and SC depth (50 cm) was taken arbitrarily.

The average SOC and SOM pools in HC and SC by soil groups were calculated on the basis of profile data of different research areas (Table 2).

In upland grassland soils with automorphic moisture regime, SOC pools in HC are between 40–114 Mg ha^{-1}, and are higher in soils with higher carbonate content. SOC pools that are remarkably lower are accumulated into drought-prone skeletal soils (32 Mg ha^{-1}). SOC pools higher than in automorphic soils are characteristic of *Gleysols*. But the highest pools are those in *Sapric* and *Fluvic Histosols,* the HC of which is peat (*hemic, sapric).* Intensely variegated SOC and SOM pools may be sequestered into the HC and SC of *Fluvisols.* The largest quantities are characteristic of the *Histic Fluvisols* which are situated in riverside areas but are remarkably reduced in coastal *Fluvisols.* For *Histosols* the SOC and SOM pools were recalculated to arbitrary HC (30 cm) and SC depths (50 cm).

Unfortunately, up to now, there has been an absence of exact data about soil distribution on Estonian semi-natural grasslands. However, there is data available from the inventory of grasslands by plant associations (Aug & Kokk, 1983), by land cover types (Meiner, 1999), by wet lands (Paal et al., 1999) and others (Arold, 2005) which help to receive approximate superficies and the precise relative importance of different soil groups on grasslands. With data re: soil distribution for the whole mapped area as well as for forested and arable lands, by R. Kokk (1995), it was possible to find superficies of soils which are used mainly as grasslands. Such soils formed 56% of the total grassland area. The superficies of coastal grasslands, alvars and riverside areas matched well. More problematic are areas with *Cambisols, Luvisols, Albeluvisols* as well as areas influenced by erosion, as these soils may be reforested or turned into arable land. Soil distribution data used in calculation (percentage and superficies) by different grassland soil groups are presented in Table 3. The calculations of SOC and SOM pools of 20 soil groups show that the main SOC accumulators are *Sapric Histosols, Histic Gleysols, Cambisols* and *Luvisols* in Estonian grassland SC.

The main quantitative parameters of soil humus status are HC thickness and morphology (fabric), SOC and SOM concentrations and pools by soil horizons, and HC quality (type). In connection with the absence of thickness and of SOC and SOM pools (Mg ha^{-1}) data for five grassland soil groups in our research areas (see Table 2), the gaps in calculation of total SOC and SOM pools (see Table 3) were filled by using the data presented in Table 4. The data of *Luvisols (cutanic, endogleyic)* and *Saprihistic Gleysols* were used as weighted averages (Mg ha^{-1}) of arable and forest soils (Kõlli, & Ellermäe, 2003; Kõlli et al., 2004).

Table 1. Groups of Estonian grassland soils and mean thickness of soil cover layers.

Group No	Soil or soil association by WRB	Soil code		Profiles n	Thickness (M±SD)[2], cm	
		by WRB	by ESC[1]		HC	SC
I	*Rendzic Leptosols*	LP rz	Kh	4	24±2.4	24±2.4
II	*Skeletic Leptosols*	LP sk	Kr	4	16±9.9	16±9.9
III	*Calcaric&Endogleyic Cambisols*	CM ca gln	K Kg	3	22±3.6	28±5.1
IV	*Mollic&Endogleyic Cambisols*	CM mo gln	Ko Kog	7	27±6.5	53±18.1
V	*Haplic&Endogleyic&Glossic Albeluvisols*	AB ha gln gs	Lk Lkg LP	8	21±2.7	77±15.9
VI	*Calcaric Gleysols*	GL ca	Gk	3	25±8.3	25±8.3
VII	*Mollic Gleysols*	GL mo	Go	4	26±5.7	43±10.6
VIII	*Calcic Gleysols*	GL cc	G(o)	10	29±6.6	39±18.8
IX	*Luvic&Spodic Gleysols*	GL lv sd	GI LkG	3	19±9.1	37±6.6
X	*Epigleyic Fluvisols*	FL glp	AG	7	27±7.8	37±5.7
XI	*Histic Fluvisols*	FL hi	AG1	7	25±7.3	37±19.7
XII	*Salic Gleysols*	GL sz	Gr	4	15±2.5	26±4.2
XIII	*Salic Fluvisols*	FL sz	ArG	4	4±1.3	5±0.4
XIV	*Sapric Histosols*	HS sa	M	3	30±0	50±0
XV	*Fluvic Histosols*	HS fv	AM	8	30±0	50±0

1) For correspondence of ESC soil names and soil codes see Kõlli, Ellermäe, 2003; 2) M - mean, SD - standard deviation, HC - humus cover, SC - soil cover.

Table 2. SOC and SOM sequestration capacity (Mg ha^{-1}, M±SE[1]) of different grassland soil groups.

Group No	Soil code by WRB	n	SOC pools Mg ha^{-1}		SOM pools Mg ha^{-1}	
			HC	SC	HC	SC
I	LP rz	4	114±30	114±30	193±52	193±52
II	LP sk	4	32±8	32±8	61±20	61±20
III	CM ca gln	3	90±18	97±16	156±31	168±28
IV	CM mo gln	7	97±24	120±28	168±41	208±49
V	AB ha gln gs	8	40±4	60±6	67±7	104±11
VI	GL ca	3	97±52	97±52	167±89	167±89
VII	GL mo	4	141±48	157±49	243±84	272±84
VIII	GL cc	10	115±18	119±17	197±32	206±30
IX	GL lv sd	3	74±43	92±37	128±73	158±83
X	FL glp	7	100±12	114±11	172±20	197±18
XI	FL hi	7	125±17	131±25	215±28	225±43
XII	GL sz	4	71±6	84±4	122±6	144±7
XIII	FL sz	4	22±8	24±8	42±14	44±12
XIV	HS sa	3	203±43	338±77	357±44	594±81
XV	HS fv	8	126±16	210±27	218±26	363±45

1) M - mean, SE - standard error, HC - humus cover, SC - soil cover.

Table 3. Total SOC and SOM pools (in Gg) by grassland soil groups (I–XX).

Group No	Soil code by WRB	% from grassland area	Superficies in ha	Sum of SOC pools[1] in Gg ha^{-1}		Sum of SOM pools in Gg ha^{-1}	
				HC	SC	HC	SC
I	LP rz	1.9	5690	649±171	649±171	1098±296	1098±296
II	LP sk	0.3	898	29±7	29±7	55±18	55±18
III	CM ca gln	2.5	7488	674±135	726±120	1168±232	1258±210
IV	CM mo gln	11.6	34742	3370±834	4169±973	5837±1424	7226±1702
V	AB ha gln gs	11.2	33544	1342±134	2013±201	2247±235	3489±369
VI	GL ca	4.3	12878	1249±670	1249±670	2151±1146	2151±1146
VII	GL mo	3.3	9884	1394±474	1552±484	2402±830	2688±830
VIII	GL cc	3.2	9584	1102±172	1140±163	1888±307	1974±288
IX	GL lv sd	5.9	17670	1308±760	1626±654	2262±1290	2792±1467
X	FL glp	2.9	8685	868±104	990±96	1494±174	1711±156
XI	FL hi	2.9	8685	1086±148	1138±217	1867±243	1954±373
XII	GL sz	1.7	5092	362±30	428±20	621±30	733±36
XIII	FL sz	1.6	4792	105±38	115±38	201±67	211±58
XIV	HS sa	9.9	29650	6019±1275	10022±2283	10585±1305	17612±2402
XV	HS fv	3.2	9584	1208±153	2013±259	2089±249	3479±431
XVI	LP gln glp	4.9	14676	1101±220	1101±235	1893±382	1893±411
XVII	LV ct gln	11.3	33844	2369±271	3181±305	4061±474	5483±508
XVIII	GL hi	9.6	28752	4859±1208	5923±949	8367±2070	10178±1639
XIX	RG ai am	4.1	12280	356±12	479±37	614±25	823±61
XX	CM&LV&AB erd CM&LV&AB& GL del	3.7	11082	909±44	1352±133	1563±66	2327±233

1) Soil group area x mean pool in one ha; HC - humus cover, SC - soil cover.

Table 4. Grassland soil groups. Humus status characterization was adapted from other sources.

Group No	Soil or soil association By WRB	% from grass-land area	n	Soil code		Thickness (M±SD)[1], cm		SOC		SOM	
								mean (Mg ha^{-1}) ±SE			
				by WRB	by ESC	HC	SC	HC	SC	HC	SC
XVI	*Endogleyic&Epigleyic Leptosols*	4.9	-	LP gln glp	Khg Gh	20±10	25±12	75±15	80±16	129±26	138±28
XVII	*Cutanic&Endogleyic Luvisols*	11.3	20	LV ct gln	KI KIg	25±5	72±19	70±8	94±9	120±14	162±15
XVIII	*Saprihistic Gleysols*	9.6	6	GL his	G1	22±5	48±12	169±42	206±33	291±72	354±57
XIX	*Aric&Anthric Regosols Cambisols& Luvisols&Albeluvisols (eroded)* [1]	4.1	168	RG ai am CM&LV&AB erd	E1 E2 E3	25±6	35±12	29±1	39±3	50±2	67±5
XX	*Cambisols&Luvisols& Albeluvisols&Gleysols (deluvial)*[2]	3.7	154	CM&LV&AB& GL del	D Dg DG	42±8	96±32	82±4	122±12	141±6	210±21

1) XIX soil group includes severely eroded *Regosols (aric, anthric)* and weakly to moderately eroded *Calcaric Cambisols*, *Humic Luvisols* and *Spodic Albeluvisols;* 2) the XX soil group is composed of deluvial (buried or formed by accumulation of eroded sediments) soils or by WRB from *pachic, cumulic, thaptohumic(-histic)* or *endogleyic Cambisols&Luvisols&Albeluvisols&Gleysols.*

Table 5. Generalized data on SOC and SOM pools in Estonian grassland soils.

Characteristic, land use	Unit	Upland mineral soils	Lowland mineral soils	Wetland organic soils	All soils
Grassland superficies	10^3 ha	154.3	106.0	39.2	299.5
	%	51.5	35.4	13.1	100.0
Grassland SOC pools in Tg[1]:	Tg				
- in soil cover		13.7±2.2	14.2±3.3	12.0±2.5	39.9±8.0
- in humus cover		10.8±1.8	12.4±3.6	7.2±1.4	30.4±6.8
- in subsoil		2.9	1.8	4.8	9.5
Grassland SOM pools in Tg[1]:	Tg				
- in soil cover		23.7±3.8	24.3±6.0	21.1±2.8	69.1±12.6
- in humus cover		18.5+3.2	21.2±6.2	12.7±1.5	52.4±10.9
- in subsoil		5.2	3.1	8.4	16.7
Average SOC pool[2]:					
- in soil cover	Mg ha^{-1}	89	134	306	133
- in humus cover		70	117	184	102
- in subsoil		19	17	122	32
Average SOM pool[2]:					
- in soil cover	Mg ha^{-1}	154	229	538	231
- in humus cover		120	200	324	175
- in subsoil		34	29	214	56

1) ± Sum of soil groups SE; 2) Weighed (by area) average.

Table 6. Outlines of grasslands HC quality characteristics and distribution.

HC characterization and subdivisions with mean percentage formula by superficies	% from grasslands superficies	% from grasslands SOC pools in		Dominating soils	Characterization of HC on the group level
		HC	SC		
A. Mild- and calci(pebble)-humous: dry:fresh:moist - 10:47:43	21	18	15	LP rz sk gln CM ca mo gln RG ca ai am	Pebble rich (or episkeletic), calcareous or neutral, rich in humus, on sloping areas may be influenced by weak to sever erosion (decrease in humus content[1] > 15%)
B. Mull-moder-humous, with light eluviation features: dry:fresh:moist - 7:44:49	14	11	11	LV ct gln RG eu ai am LV ph del	Slightly acid from superficial layers, under HC features of light eluviation, transition HC between mild-humous and acid-humous HC, on sloping areas may be influenced by weak to severe erosion (decrease in humus content > 15%) or deluvial (colluvial) sediments
C. Moder-humous, moderately or strongly acid: dry:fresh:moist - 14:43:43	14	5	6	AB ha gs gln RG oh ai am AB ph del	Moderately or strongly acid, clear features of podzolization, have Bhf or Ea horizons, on sloping areas may be influenced by weak to severe erosion (decrease in humus content > 15%) or deluvial (colluvial) sediments
D. Raw-humous or wet HC: calcaric:eutric:moder:entic – 27:23:38:12	26	23	20	GL ca mo cc GL lv sd ph sz FL glp sz LP glp	Superficial horizons are organo-mineral or peaty, mainly eu- or mesothrophic character, some areas pebble rich or calcic, periodically inundated areas may contain alluvial or deluvial (colluvial) sediments, on coastal areas may be very shallow (< 3–5 cm)
E. Peats thin:thick - 48:52	25	43	48	HS sa fv GL his FL hi	Superficial horizons are sapric (eutric) or hemic (mesotrophic) peat, which is moderately or well decomposed, on riverside areas may contain alluvial sediments, the thickness of thin peats is 10–30 cm, thick peats > 30 cm (mostly more than 1m)

1) Estimated by concentration (g kg^{-1}) or by pools (Mg ha^{-1}) .

The data about soils influenced by erosion areas (eroded and deluvial soils) were taken from our unpublished work and data about hydromorphic *Leptosols (endo-* and *epigleyic)* from previously generalized postlithogenic soil matrices (Kõlli et al., 2004).

In total, 39.9±8.0 Tg SOC is sequestered (Table 5) in Estonian grasslands SC. Of that, 76.2% is accumulated into the active layer or into HC (i.e. incorporated into stabilised soil humus, raw-humous material or in peat); 23.8% of SOC is located in the passive layers (in E and B horizons) or in subsoil and is characterized therefore by a long turnover time. The generalised SOC and SOM pools (Table 5) are given separately for three sets of soil groups. The role of these three grassland soil group sets in the sequestration of total SOC pools in grasslands is 34.5, 35.3 and 30.2%, respectively; the role of these sets for the total grassland area, (51.5, 35.4 and 13.1%, respectively).

In Estonian grasslands 69.1±12.6 Tg SOM is accumulated; approximately half (31.3 Tg or 45.3%) is peat. More than half, 54.7%, of grasslands' total SOM may be qualified as humus with different quality and available for soil edaphon. The majority, 78% (29.5 Tg), of total grasslands' humus is situated in active HC and 22% (8.3 Tg) in passive part or in subsoil. The high proportion of peat in the SOM of Estonian grasslands (approximately half) is caused by the high amount of *Histosols* (13.1%) and *Histic Gleysols & Fluvisols* (12.5%).

The generalised (weighted by area) data about SOC and SOM pools (Mg ha^{-1}) in HC and SC are also presented in Table 5. The comparison of three grassland soil sets shows that subsoils of upland and lowland mineral soils have approximately equal SOC and SOM sequestration capacities, but the average sequestration capacity of SOC in lowland mineral soils' HC (in Mg ha^{-1}) is more than 1.6 times higher than in upland soils. Due to the subsoil richness in SOC, the most powerful SOC accumulators are *Histosols* sola, where an average per one hectare's 50 cm layer sequesters 306 Mg SOC.

In the World Soil Resources Report (FAO, 2001) mean SOC amounts of 0.3 m and 1.0 m soil layers in Boreal Agro-Ecological Zones are 98–102 and 231–240 Mg ha^{-1}, respectively; the 0.3 m layer SOC pool matches our grasslands soil HC weighted average (Table 5). The mean grassland ecosystem soil organic pools according to Lal et al. (1998b) is given as 116 Mg ha^{-1} which is similar to our lowland mineral soils HC pools, and is close to the weighted mean of Estonian grassland SOC amounts.

In the Brazilian Amazon Basin the mean SOC amounts to a depth of 1 m (Rosell & Galantini, 1998) are in a similar range with our data, in *Alfisols* 76–120, *Inceptisols* - 68–76 and in *Mollisols* 95–156 Mg ha^{-1}, if we compare them respectively with *Albeluvisols*, *Lepto-&Cambisols* and *Mollic Cambisols* (Table 2). However, the depth of our sola is thinner, as is characteristic of northern areas.

Comparative studies of meadow and forest soils in the forest zone of Russia (Yakimenko, 1998) have demonstrated the ability of grassland ecosystems to accumulate more SOC in a 50 cm soil layer than in forest ecosystems. For example, 67–88 Mg ha^{-1} SOC is accumulated in grassland soil in the Middle Urals, 66-90 in Leningrad province and in Moscow province 52–81 Mg ha^{-1} SOC, which are accordingly 2–21, 12–22 and 8–29 Mg ha^{-1} more than in the same soils under the forests.

The experiments with annual application of nitrogen and sulphur fertilizers on hayed native grasslands in Saskatchewan, on a *Boralfic Boroll* with sandy loam to

sandy clay texture (Nyborg et al., 1998), clearly demonstrated enhancement of SOC storage in grassland soil superficial 37.5 cm layer up to 8 Mg organic carbon per one hectare.

Comparison of Estonian grassland soil SOC pools to a 50 cm layer of soils in the northwestern United States in Mg per ha (Kern et al., 1998) demonstrates the variability of SOC pools with similar limits (CV limits of 20–60%), indicating higher amounts (84–110 Mg ha^{-1}) in *Rendolls, Udolls* and *Borolls* compared with *Udalfs* and *Boralfs* (56–86 Mg ha^{-1}). Soils with *aquic* water conditions in the northwestern United States tend to be similar to pools of Estonian *Gleysols* (varying from 90-200 Mg SOC ha^{-1}).

The humus status of *Histosols* (Tarnocai, 1998) reveals that SOC pools of our *Sapric Histosols* match well with C. Tarnocai's surface (0–30 cm depth) carbon content of *Saprists, Hemists* (*Mesisols* and *Humisols* according to the Canadian Soil Classification) with average SOC amounts of 182 and 217 Mg SOC ha^{-1} respectively. C. Tarnocai (1998) estimated for the Canadian Grassland Ecoclimatic Province a mean SOC content of 122 Mg ha^{-1} which is slightly lower than the value (133 Mg ha^{-1}) found by Post et al. (1982). It is interesting that this is equal to our value for Estonian grassland SOC amounts (133 Mg ha^{-1}; Table 5). But it is clear that the weighted average SOC content of an estimated area depends largely on the presence of *Histosols*.

At present different sources concerning the distribution of SOC and SOM in European soils are available (Rusco et al., 2001; Van-Camp et al., 2004; Zdruli et al., 2004) but in most cases the total SOC and SOM stocks for different countries are computed indirectly and must be updated from time to time. For example, the SOC for Estonian topsoil (0–30 cm) is computed as 1.5 Gt (Van-Camp et al., 2004), however the sources refer to the lack of geo-referenced, measured, harmonised data on SOC available in Europe.

Comparison of SOC and SOM retaining (sequestration) capacity of grassland HC and SC by soils groups and soil sets with those for arable and forest soils (Kõlli & Ellermäe, 2003; Kõlli et al., 2004) enables us to elucidate some pedo-ecological regularities. First of all, arable, forest and grasslands clearly differ by their soil types and texture composition. On arable land, more fertile upland mineral soil types (with loamy texture) are dominant (altogether 72%); on forest lands there is a greater share of organic (37%) and lowland mineral soils (39%); consequently both differ from grassland composition (see Tables 3 and 5). A clear difference is observed in HC thickness, which is highest in arable soil, and in the fabric of HC where a clearly formed forest floor is observed in forest soils. In arable soils the organic superficial layer is absent all together, but may occur on some grasslands that have low biological activity. With regard to grasslands SOC and SOM amounts, their weighted averages are slightly higher on upland mineral and lowland mineral soils when we compare them with arable and forest lands.

Our study reveals that we must not decide carbon sequestration capacity only on SOC and SOM concentration, but first of all on the basis of SOC or SOM. Many researchers have clarified (Kern et al., 1998; Körchens et al., 1998; Percival et al., 2000; FAO, 2001) that SOC- and SOM-retaining capacity depends on the soil moisture regime, physical clay and carbonate content in fine earth, and soil management character. Land use and/or tillage technology have a substantial influence mainly on

the humus status of superficial soil layers. SOC sequestration in subsoils depends greatly on the thickness of the solum. In subsoils of mineral grasslands, an average of 17–19 Mg ha^{-1} SOC or 29–34 Mg ha^{-1} SOM may be found. That may be treated as a buried resource. Thick *Histosols* and various soils with *pachic, cumulic* and *thaptohumic* epipedons formed in mineral soils by accumulation of eroded (deluvial) and alluvial sediments are especially rich in sequestered SOC.

The characteristics of humus quality are presented in Table 6, where a rough estimation of the share of different HC types is shown. The first three divisions (A, B and C) belong to upland mineral soils (see Table 5). Raw-humous HC is developed on lowland mineral soils; the exceptions are *Histic Gleysols* and *Histic Fluvisols,* the HC of which is peaty. By area, the peat type HC (25%) can be divided almost equally between thin peat (peaty soils) and thick peat (real organic (peat) soils). A remarkable share of HC (23% by pools and 26% by area) belongs to raw-humous or wet HC, which is potentially fertile, but suffers from water logging during spring and autumn. These HC are relatively well humified in *Gleysols* with calcareous and neutral reaction, rich in nutrition elements. The portion of acid raw-humous HC with features of podzolization is not high (< 6% by area) but the quality of this kind of SOM is low from the ecological, and especially from the edaphic, viewpoint. Although the soils of upland grasslands form more than half, (51.5%), of the total grassland, their SOM pools account for only about one third, (34.3 %). A comparison of qualities of Estonian forest and grassland HC's show that biologically more active epipedons or HC are characteristic of grassland.

CONCLUSIONS

Sequestration capacities for each soil type of grassland characteristic SOC and SOM have developed. They are determined mainly by soil thickness, moisture regime, as well as by carbonate and clay content. Depending on composition of individual site specific soil properties, the SOC and SOM contents and pools in humus cover and sola may vary greatly.

In Estonian grassland soils 39.9±8.0 Tg SOC is sequestered. The latter is accumulated as 69.1±12.6 Tg of SOM (humus, raw-humous material, peat) in different soil horizons and layers. 76.2% of SOC is located in the biologically active humus cover and 23.8% in less active subsoil. Epipedons formed on grasslands are biologically more active and have better ecological quality than on forest lands.

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Effect of clippings management on turfgrass sward productivity and nitrogen content in the clippings and soil

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Abstract. The maintenance of turfgrass sward includes mowing and fertilization. Every year turfgrass sward produces a sizeable amount of clippings containing large amounts of nutrients which will be available for plants during the decomposition process. The aim of this research was to study clippings decomposition speed, the effect of returned clippings to the turfgrass sward's clippings yield and total nitrogen content in clippings and soil. The study was carried out on turfgrass sward (seed mixture composition *Festuca rubra rubra* 50% and *Poa pratensis* 50%). The turfgrass clippings were either removed after cutting or returned to the plots. The clippings yield and nitrogen content in the clippings were measured after every cutting. The soil samples from different plots were analyzed for total nitrogen at the beginning and the end of the growing season. The decomposition dynamics of clippings was studied using the litterbag technique. Also the nitrogen mineralization from decaying material and the concentration changes of cellulose and lignin were studied during 12 weeks.

The results showed that the turfgrass clippings mass and the content of nitrogen decreased during the decomposition process very quickly. The degradation of cellulose takes place after about 30% of initial weight decomposition. During the 12 week study period we did not fix the beginning of lignin decomposition. Higher productivity was obtained in treatments where clippings were removed. N content did not differ in plant from plots where clippings returned or removed but N content in soil of plots with clippings returned decreased compared to N content in soil of plots where clippings were removed.

Key words: clippings, turfgrass, decomposition, N mineralization, cellulose, lignin

INTRODUCTION

Environmentally friendly agriculture should move towards to a more closed nutrient cycle. This would mean the decreasing use of mineral fertilizers which have been dominant so far. Organic fertilizers as well as nutrients released from plant remains will be as a good replacement for plant nutrition and elevating soil fertility. The decomposition of organic matter has been investigated mainly in soil, and organic matter left above-ground for decomposition has not received so much attention. In amenity grasslands and set-aside fields, the returning of clippings to the site would be economically the cheapest possibility of management of those areas. On one hand the

soil will be enriched with humus substances and on the other, the nutrients released during the mineralization process are ready for usage by plants.

Several investigations have shown that organic matter returned as mulch, will start to accumulate in grasslands (Meinhold et al., 1973; Murry & Juska, 1977). The reason could be the lack of micro-organisms which could decompose the organic matter above ground. At the same time opposite results can be found. The trials of Kopp & Guillard (2002) have shown that from returned clippings a remarkable amount of nitrogen (N) will be released, which would considerably lessen the need for mineral fertilizers. In their trial the returning of clippings did not cause the decomposition of thatch layer which was noticed in aforementioned investigations.

The goal of our work was to explain the decomposition dynamics of clippings left on the sward, the release of N from the clippings and its effect on sward productivity and N content of clippings and soil. Also the changes in concentrations of cellulose and lignin studied during decomposition.

MATERIALS AND METHODS

The experiment was carried out at the Estonian University of Life Sciences in the experimental station Eerika (58°23'32" N latitude, 26°41'31" E longitude) in 2007. The site had been seeded in 2003 with a turfgrass mixture (*Festuca rubra rubra* 50% and *Poa pratensis* 50%).

The soil of the experimental field was *Stagnic Luvisol* according WRB classification (FAO, 1998) and the humus horizon contained 1.6% organic carbon and 1.63 mg N g^{-1}.

The experiment was conducted on unfertilized sward in four replications with plot size 1x7 m. The swards were cut 14 times at a height of 5 cm during the growing season. For the cutting a lawn mower with a bag attachment was used. After every cutting the material was removed from the bag and weighed. After the weighing procedure the clippings of the turfgrass were either returned (hereafter CRT) to the plots or removed (hereafter CRM). The returned and removed clippings were analysed by total N and the amount of N (nitrogen uptake) removed or returned by clippings was calculated using the yields of the plots multiplied by N concentration.

After the first cutting, the decomposition dynamics of clippings was investigated using the litterbag technique. A total of 20 g of fresh biomass equivalent to about 5 g dry biomass was put into 20×20 cm polyester litterbags with a 1.5 mm mesh size. At certain time intervals the bags were collected and the material was removed from bags, dried (105°C, 4 hours), weighed and the weight loss was calculated. The biomass residue remaining in the litterbags was expressed as a percentage of the initial dry weight. The remaining percentage of mass (RPM) for each period was determined using this formula:

$$RPM\ (\%) = (100 \times M_t)/M_0,$$

where M_0 is the initial plant material dry matter mass in the litterbag and M_t is plant material dry matter mass in bag in time t, when litterbags removed from field. The litterbags material was analyzed for N and carbon:nitrogen ratio (C:N).

The remaining percentage of N (RPN) at the time t was calculated:

$$RPN\ (\%) = (100 \times N_t)/N_0,$$

where N_0 is initial N amount in sample and N_t is N amount in sample at time t, when the litterbag was removed from the plot. Van Soest's method used to measure the changes of lignin and cellulose concentration in decomposing clippings (Van Soest, 1963).

At the beginning of the vegetation period (May 2007) and at the end (September 2007) of the vegetation period, the total N content of soil samples (0-5 cm) of both management variants (CRM and CRT) was determined according to Kjeldahl.

The statistical package Statistica version 7.0 (StatSoft.Inc) was used for all the statistical analyses. Factorial ANOVA was applied to test the effect of the treatments on the yield and N content in the clippings and the soil.

RESULTS AND DISCUSSION

The turfgrass clippings decomposition and N mineralization

The decomposition of clippings on the sward was rapid in the first weeks. Already by the first 2 weeks, 27.4% of initial material was decomposed (Fig. 1).

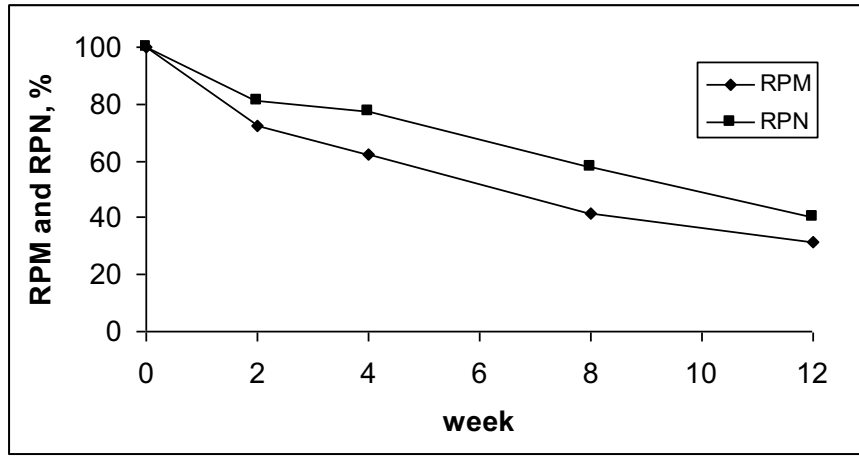

Fig. 1. The remaining percentage of mass (RPM) and nitrogen (RPN) (%) in turfgrass clippings during decomposition process.

After 12 weeks of decomposition, 31.2% of clippings initial mass remained. The initial concentration of cellulose in the clippings was 178.2 mg g^{-1}. During first two weeks the concentration of cellulose increased to 249.0 mg g^{-1} but by the end of week 8 had decreased down to 103.3 mg g^{-1}. The cellulose started to decompose after week 4 and approximately 30% of the initial material had decayed. Analyzing the decomposition process of clippings indicated that approximately 60% of the turfgrass clippings consisted of easily soluble compounds and the remaining 40% was the material needed for that year's decomposition. Such a long-term degradation is caused

by the increased lignin concentration. The clippings initial lignin concentration was 12.6 mg g^{-1}. During the studied decomposition period the lignin decomposition did not occur and the lignin concentration increased after week 8 to 150.0 mg g^{-1}.

Living organisms using the plant residue's carbon as a source of energy and the nitrogen for building cell structure cause the decomposition and mineralization of organic matter. The plant cell is mainly composed of different water soluble carbohydrates, cellulose and lignin. Plant material chemical composition determines the availability of plant carbon to the soil decomposers, and will therefore have a crucial influence on the dynamics of N mineralization during decomposition (Gunnarsson, 2003; Trinsoutrot et al., 2000). The most easily decomposable compounds are water-soluble carbohydrates, and then cellulose and the most difficultly decomposable compounds is lignin.

At the beginning of the decomposition process, the content of N in the clippings was 31.5 mg g^{-1} and after 12 week 59.5% of that was mineralized. Nitrogen released from the clippings did not affect the content of total N in either the soil or the clippings. At the same time the variants where the clippings were removed the soil total N content was increasing during the growing season (Table 1).

Table 1. Total N (mg g^{-1}) content in soil (0-5 cm) from plots with clippings were returned (CRT) or removed (CRM) plots in spring and autumn.

	CRT		CRM	
	spring	autumn	spring	autumn
N, mg g^{-1}	1.37a	1.34a	1.43b	1.61c

Different letters within line indicate significant difference of the mean values at $p<0.05$.

The C:N ratio of decomposing clippings at the beginning was 18 in our study and decreased down to 12 throughout the decomposition period. There is a wide assessment that plant materials with a C:N ratio less than 20 may result in net N mineralization and those with a C:N ratio greater than 20 tend to cause net immobilization (Quemada & Cabrera, 1995). Immobilization (i.e. increasing N amount in decaying sample) did not occur during the decomposition process in our trial (Fig.1). When the relationship of used C:N ratio of different week is calculated, it appears that during first two weeks, at the beginning of decomposition process, the used C and N ration is 27:1. This proves that there was insufficient N for microbial decomposition and the immobilization took place. After week 2 the consumed C:N ratio was 54:1 and after week 4, when the cellulose was degraded, the used C:N was in relation 19:1. The ratio was narrower, because the N which was earlier linked to cellulose was liberated during the mineralization and made available to bacteria. Thus, according also to Andersen & Jensen (2001) the C:N ratio in decomposable material does not influence the decomposing process but the C:N ratio in decomposing compounds is the crucial factor for explaining the decomposition process. The C:N ratio in easily decomposable compounds can be broader than in plant material total and therefore immobilization can occur eventhough the C:N ratio in decomposable organic matter in total is relatively narrow (Andersen & Jensen, 2001).

According to Swift et al. (1979) soluble substances and labile compounds, which form the biggest proportion in turfgrass clippings, are rapidly degraded in the early stages of decomposition by fast growing micro-organisms that require a high

concentration of N which may cause its initial immobilization. To decompose the wide C:N ratio compounds the missing amount of N will be taken from soil. The soil total N did not increased, although during the vegetation period the big amount of N from clippings was returned to the sward after every cutting.

The impact of returned and removed clippings on sward productivity and N uptake

The total N content in the clippings was similar in most cuttings of CRT and CRM variants, a significant difference occurred in first cut where the yield of CRM variant contained much more N compared with CRT variant (Table 2). The average yield of clippings was 13.5% higher in CRM variant compared to the CRT variant but the difference was not statistically significant.

The average yield of the growing period was largely influenced by the yield of the first cutting, which was 42% larger in CRM variant compared to CRT variant. The following cuttings of these two variants were similar. N uptake was greater in CRM variant, referring to the larger amounts of N removed by clippings (34.8 kg N ha^{-1}), compared to the N returned by clippings in CRT variant (30.5 kg N ha^{-1}). If the data of first cutting is omitted from the data analysis, the results will be the opposite, but here the differences between two investigated variants will be statistically minor.

Table 2. The average total N content (mg g^{-1}) in clippings by different cuttings, dry matter yield (kg ha^{-1}) and N uptake (kg ha^{-1}) of turfgrass swards plots with clippings removed (CRM) and with clippings returned (CRT).

	Total N content in clippings, mg g^{-1}		DM yield of sward, kg ha^{-1}		N uptake by clippings, kg N ha^{-1}	
Cut	CRM	CRT	CRM	CRT	CRM	CRT
1st cutting	40.5b	31.5a	462.3a	268.7a	18.7a	8.46a
2-14 cuttings	29.6a	30.7a	681.9b	721.5b	20.2a	22.2b
All cuttings (1-14)	30.4a	30.8a	1144.2c	990.2c	34.8b	30.5c

Different letters within column indicate significant difference of the mean values at $P < 0.05$.

The results of our investigation differed remarkably from the findings of Kopp & Guillard (2002). According to their investigation the productivity of the unfertilized variant was in the context of returned clippings equal to the productivity of the variant receiving 392 kg N ha^{-1} where the clippings were removed. In our trial the amount of N returned by the clippings was 30.5 kg N ha^{-1} but did not have any significant influence on soil or plant N content. Earlier investigations have shown that by the decomposition of clippings a remarkable loss of N occurs by the volatilisation process in NH_3, which can reach up to 10-20% of total mineralizable N (Janzen & McGinn, 1991). Also N immobilization by microbe-decomposers is the reason for N decreasing in soil as we discussed earlier. According to the research of Kuzyakov et al. (2000) the so called *priming effect* will take place in plant-soil systems, which activates an extra decomposition of indigenous soil organic matter, and therefore the soil N content is decreasing. Immobilization took place in our trial because the total N content in CRT variant in upper 0-5 cm soil layer was stable in spring as well as in autumn. At the same time the total N in CRM variant was increasing during the growing season. The

higher content of soil total N was presumably caused by the decomposition of plant roots and released N was combined to the soil organic matter, instead of being used by the microbes in decomposition of clippings as happened in CRT variant.

CONCLUSIONS

We can conclude that over 60% of the turfgrass clippings which stayed on site after mowing consisted of easily decomposable material, which is mostly influenced by the easily soluble compounds in the clippings. A large number of easily decomposable compounds in decaying material caused the nitrogen deficit and missing nitrogen will be taken from soil. The returning of clippings during the growing season did not have any effect on soil and clippings total N content, as the amount of N released from clippings was so small and most was used by the microbe-decomposers. The N content in soil was higher in plots where clippings were removed.

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III

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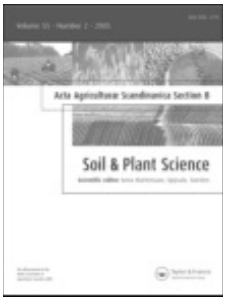

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The decomposition of turfgrass clippings is fast at high air humidity and moderate temperature

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ORIGINAL ARTICLE

The decomposition of turfgrass clippings is fast at high air humidity and moderate temperature

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Abstract
In grassland areas where herbage production has no economic value, the cut grass is often left on the sward surface where its decomposition is influenced by weather conditions. Although the influence of temperature and humidity on decomposition has been investigated under controlled lab conditions, experimentation has generally been under ideal moisture conditions that have not tested the combinations of climatic limitations that might occur in the field. The decomposition of mown turfgrass clippings deposited at different times of vegetation period was studied *in situ* using nylon bags during the first 8 weeks after deposition to investigate the effect of weather conditions (the air temperature, relative humidity, precipitation) on decomposition. Decomposition is the highest in the case of high air humidity and temperature of 10°C. Limiting factors for decomposition at temperatures above 10°C is the air humidity and below 10°C the air temperature. The general tendency was that the rate of decomposition increased with increasing air temperature up to 10°C, but with further increases of air temperature the decomposition rate slowed down. Relative air humidity had a variable impact (at the beginning of the decomposition process (weeks 1–2) the influence was negative, during weeks 3–8 of the decomposition process the effect was positive), and hence had no generalized relationship with decomposition over the studied decomposition period (weeks 1–8). The most significant influence of weather conditions on the decomposition rate was recorded directly after cutting. If the cutting was done during hot weather conditions, the material was drying fast and therefore decomposed slowly. Our results indicate that for fast decomposition of clippings it is important to maintain the freshness of material**.** Lower decomposition rates occurred during conditions of hot and dry weather, and also cooler (temperature near to 0°C) weather, and can be compensated as soon as favourable weather arrives.

Keywords: *Air temperature, nylon bags, precipitation, relative air humidity, sward.*

Introduction

In amenity grassland where there is a requirement to mow grassland for maintenance, the mown grass clippings often have no economic value and are left on the sward surface, where their decomposition is influenced by weather conditions. In temperate zones the weather conditions change during the year and could have a direct influence on the rate of decomposition of organic matter. During periods of unfavourable conditions for decomposition, residues of grass clippings left on the sward surface could tend to accumulate, partly because their deposition exceeds decomposition rate. To avoid these problems, and to ensure good turfgrass management in the absence of agronomic productivity, it is important to know the rates of decomposition of grass residues left on the sward surface during different growing stages.

Temperature and moisture conditions are the main factors influencing the activity of decomposers involved in the decomposition process (Paul and Clark 1996, Dalias et al. 2001a, 2001b, Pietikäinen et al. 2005, Uvarov et al. 2006). During the process of decomposition of organic matter an increase in soil temperature and moisture generally results in greater rates of microbial activity and thus increased rates of reduction of plant residue (Stott et al. 1986,

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Donnelly et al. 1990). Microbial activity is generally predicted to increase rapidly up to a temperature of about 30°C. An optimal temperature for microbial activity is reached between 35 and 45°C, and the optimal moisture content for organic matter decay is 50–60% (McKinley and Vestal 1985, Chen et al. 2000). It is found that at higher temperatures the temperature influence to the decomposition rate is reduced i.e. increasing temperature by the same number of degrees at lower temperatures accelerated decomposition more than the same increase at higher temperatures (Kirschbaum 1995, Dalias et al. 2001a).

The influence of temperature and humidity on decomposition has been investigated mostly under controlled laboratory conditions and in field experiments where plant residues are incorporated into soil (Kirschbaum 1995, Henriksen and Breland 1999, Dalias et al. 2001a). Laboratory experiments that focus on temperature responses have been generally conducted under ideal moisture conditions, so that the various combinations of climatic limitations to decomposition that might occur in the field have not necessarily been tested. Field tests carried out in soil have shown there are strong interactive effects of temperature and moisture on litter respiration during litter decomposition (Flanagan and Veum 1974, Clark and Gilmour 1983, Doel et al. 1990, O'Connell 1990).

Plant residues spread on the soil surface will normally be exposed to more variable temperature and moisture conditions than will residues buried in the soil, and these variable conditions may greatly slow down the decomposition of organic residues on the soil surface. For example, Curtin et al. (1998) found that CO_2 evolution was 36–62% less from soil being exposed to drying/wetting cycles compared with soil having adequate constant moisture content. In natural conditions decomposition of organic matter on the surface of soil is influenced by air temperature, air relative humidity and precipitation, in addition to the influences of soil temperature and moisture content, which as a rule are different from the respective parameters for air (Quemada and Cabrera 1995). Close contact with soil will usually increase the microbial decomposition of organic matter (Douglas et al. 1980, Cogle et al. 1989, Havstad et al. 2010) and this is mainly due to higher moisture content in residues (Parr and Papendick 1978).

Even though the influence of temperature on decomposition has been the focus of many investigations, the decomposition of mown plant residues at different periods in the growing season has received little attention. There is also little information about the effect of weather conditions on decomposition of fresh organic material. The moisture content in fresh grass material is high at the starting point of decomposition and it is probably very favourable material for the microbes so that they can start the decomposition process immediately. Therefore, we can assume that at the beginning of decomposition the rate of decomposition depends only on temperature, because the moisture needed for the decomposition process is adequate. The influence of temperature can occur in two ways: temperature activates the decomposers, and at the same time it also dries the material causing moisture to become a limiting factor as temperatures increase. The objective of the research was to investigate the influence of air temperature, relative air humidity and precipitation on the dynamics of decomposition. We hypothesized that turfgrass clippings mowed at different times during the growing season will decompose at different rates, and that these rates would be mainly influenced by different weather conditions during these periods.

Materials and methods

Background of experimental site

The field experiment was carried out at the Experimental Station Eerika of the Estonian University of Life Sciences (58°23′32″ N latitude, 26°41′31″ E longitude). The soil of the experimental field was a *Stagnic Luvisol* according to WRB classification (FAO, ISSS, ISRIC 1998). Soil analyses by establishing the trial showed that the humus horizon contained 16.0 mg organic carbon g^{-1} and 1.63 mg N g^{-1}. The sward had been established in June 2003 with a turfgrass mixture of *Festuca rubra rubra* (198 kg ha^{-1} germinating seed) and *Poa pratensis* (52 kg ha^{-1}), the mixture providing a 50:50 ratio of germinating seeds by seed number of *Festuca rubra rubra* and *Poa pratensis*. The sward was unfertilized during the period between the sward establishments in 2003 and in May 2004 when in those plots the research was initiated to demonstrate the influence of returned clippings to the yield and growth of sward plants.

The experimental design was a randomized complete block with four replicates of each of four fertilization treatments, with a plot size 1×7 m for each treatment plot. The fertilizer treatments were as follows: $N_0P_0K_0$ (N0) as control, $N_{80}P_{11}K_{48}$ (N80), $N_{160}P_{22}K_{96}$ (N160) and $N_{400}P_{56}K_{240}$ (N400) kg ha^{-1}. The N as $(NH_4)NO_3$ and K as KCl fertilizer was applied by hand to the plots (a plot size 7 m^2) in 2 to 4 splits depending on the ratio during vegetation period. The P fertilizer as $Ca(H_2PO_4)_2$ was applied to the plots in 1 split at the beginning of May. The

turfgrass sward was mown with a rotary lawn mower (Partner 5553 CMDEW) with a bag attachment at 5 cm height when the height of grass was approximately 7–8 cm high. The frequencies of mowing depended on the rates of grass growth. The clippings from each plot were returned to that plot and then mulched with a lawn mower.

The research about decomposition process of clippings was carried out in years 2006 and 2007. We analysed the decomposition process of clippings in four different fertilization treatments (N0, N80, N160, N400) having four replications in each treatment. The research on clippings decomposition was simultaneously implemented together with research on clippings influence on sward and therefore the methodology of the trial was the same in both cases.

Arrangement of experiment

There were four experimental periods, each of 8–10 weeks duration (referred to as Period I, II, III and IV) (Table I).

At the beginning of each period (15 May, 13 Sept, 26 Oct in 2006 and 16 May in 2007), directly after cutting, a sample of 100 g of fresh herbage was collected. From each sample, a subsample of 20 g of fresh herbage (the length of the plant species was 2–3 cm) was put in to 20 × 20 cm nylon bags with a 1.5 mm mesh size. Each of the bags with the fresh clippings were placed and fixed with clamps into the thatch layer of the plot from which it had been harvested. The number of bags used, per treatment, was between 12 and 20 in different periods and it depended on how many times the bags were planned to be removed from the plots. For example in period I the bags were removed four different times during 8 weeks that means every fertilizing variant received altogether 16 bags (4 bags for every replication). Weekly mowing took place even when studying the decomposition of clippings. The nylon bags didn't disturb the mowing because they were fixed tightly to the soil surface. The mowing height was 5 cm and the mower did not touch the nylon bags. Bags were removed according to a certain timetable (Table I). Weeks were counted since the day when the bags were placed on the experimental plot. The removal of bags was taking place on the 7th day of the week. Simultaneously four bags were removed from each fertilization treatment (1 bag per replicate of a treatment).

Measurements analysis

The remainder part of the sample at the beginning of experiment was used to determine (i) dry matter, (ii) total nitrogen, (iii) total carbon, (iv) cellulose and (v) lignin content.

Table I. Time of experiment, the duration of decomposition periods and sampling weeks.

Period	Duration of the experiment	Year	Total number of bags exposed	Sampling weeks
I (8 weeks)	15 May–10 July	2006	64	2, 4, 6, 8
II (8 weeks)	13 Sept–8 Nov	2006	80	1, 2, 4, 6, 8
III (10 weeks)	26 Oct–4 Jan	2006	48	2, 5, 10
IV (8 weeks)	16 May–10 Jul	2007	48	2, 4, 8

At retrieval, the content of each bag was carefully examined and visible soil particles were removed continued by determination of cellulose and lignin content of the remaining plant residue.

The dry matter content was determined by drying the sample in a forced-draught oven for 6 hours at 105°C. Total nitrogen and carbon content were analysed by dry combustion method in a vario MAX CNS elemental analyser (ELEMENTAR, Germany). Van Soest's method was used to measure the cellulose and lignin content on a dry matter basis (Van Soest 1963).

Equations for calculations

Dry matter

$$\text{Dry matter}\ (\%) = (\text{weight of dry material}\ (\text{g}) \times 100(\%))/\text{weight of fresh material}\ (\text{g})$$

The weight loss (decomposition rate) for each period was calculated using the following formula:

$$\text{weight loss}\ (\%) = 100 \times (M_0 - M_t)/M_0$$

where:
M_0 is the initial plant material dry matter mass in the bag;
M_t is plant material dry matter mass in bag in time t, when bags were removed from field.

Weather

The climate of Estonia is almost maritime in the west and slightly continental in the east. The winter period (average air temperature permanently below 0°C) lasts on average 115 days with an average mean temperature of the coldest months of −5.5°C. The average duration of the vegetation period (air temperature permanently above 5°C) is 175–190 days. The average period without night frosts is four months, during which time the average midsummer

(July) temperature is 16–17°C. Mean annual precipitation is 550–700 mm; the average precipitation in the wettest months (April to the end of October) is 350–500 mm (Keppart and Loodla 2006). Throughout the experiment period we monitored the meteorological conditions at the experimental site using Metos Model MCR300 weather stations (Pessl Instruments GmbH, Weiz, Austria). The sensors were positioned two metres above the ground.

In the current research we gathered the information about average day air temperature (°C), relative air humidity (%) and precipitation (mm). By results the weather parameters are presented for periods I–IV and decomposition weeks (weeks 1–2, weeks 3–8, weeks 1–8) (Table III).

Statistical analysis

Statistical analyses were carried out using software Statistica version 7.0 (StatSoft Inc.). Analysis of variance (ANOVA) for a randomized complete design was used to test the influence of N fertilization on decomposition. In the figures presented the value of standard error (SE) was found by using the two-way unweighted means analysis, where the factors were: (i) dates of sampling and (ii) rates of nitrogen fertilization.

The linear and multiple regression analysis with backward stepwise were performed to evaluate the relationships between the weight loss and different weather parameters (average air temperature, air relative humidity and precipitation of each period).

Results

Chemical composition of turfgrass clippings

The DM content and chemical compositions of turfgrass clippings during the different periods are presented in Table II.

Weather

Summarized weather parameters for the whole decomposition process and different periods of the decomposition process are presented in Table III.

Decomposition of turfgrass clippings

The decomposition process of clippings from four different fertilization treatments was similar in all analysed experimental periods. The fertilization rate did not affect the decomposition process significantly ($F_{(3,343)} = 0.024$; $p = 0.99$).

Table II. The dry matter (DM) content, nitrogen (N), carbon (C), cellulose and lignin initial concentration and C:N ratio in clippings at the beginning of Periods I–IV.

	N0[†]	N80	N160	N400
		Period I		
DM,%	28.3a[‡]	26.4a	26.4a	26.3a
N, mg g^{-1}	24.3a	29.3b	30.5b	39.2c
C, mg g^{-1}	418.6a	418.7a	420.9a	417.6a
C:N	17.3c	14.3b	13.8b	10.6a
		Period II		
DM,%	17.7b	16.9b	14.3a	14.4a
N, mg g^{-1}	42.3a	46.6a	53.9b	58.5c
		Period III		
DM,%	28.7c	25.0b	21.8a	21.0a
N, mg g^{-1}	38.1a	42.1ab	48.2c	44.4bc
		Period IV		
DM,%	28.6c	24.2b	23.1a	22.4a
N, mg g^{-1}	23.3a	38.5b	38.0b	45.2c
C, mg g^{-1}	425.5a	n.d.[§]	n.d.	433.5a
C:N	18.2a	n.d.	n.d.	9.3b
Cellulose, mg g^{-1}	178.2a	n.d.	185.0b	175.1a
Lignin, mg g^{-1}	12.6a	n.d.	12.2a	12.2a

[†]N0, N80, N160 and N400 means 0, 80, 160 and 400 kg N ha^{-1}.
[‡]Different letters within each line indicate significant difference of the mean values at $p < 0.05$.
[§]n.d., not determined.

Decomposition of turfgrass clippings during different periods showed the same pattern of decomposition but the decomposition rates were different ($p < 0.05$). The fastest decomposition during the 8-week period was recorded for material cut in late summer on 13 September (Figure 1). The slowest rate was observed during the decomposition which started in October, which is middle of the autumn in Estonia. The difference in decomposition rates was most apparent during the first two weeks of each decomposition period. During the following weeks the variations between decomposition periods were smaller. In all compared periods the decomposition rates were highest directly after deposition of clippings and slowed down during the following weeks. When approximately 40% of the initial material still remained in the bag the decomposition process slowed down considerably. The material from harvested plots of different fertilization treatments decomposed at similar rates (Figure 1).

The cellulose and lignin decomposition was studied for the herbage during Period IV only. The cellulose in the turfgrass clippings started to decompose 4 weeks after cutting, when on average 33% of

Table III. Weather parameters for test periods.

	Mean temperature, °C			Mean relative humidity, %			Precipitation, mm		
Week	Average	Min	Max	Average	Min	Max	Average	Min	Max
				Period I					
1–2	10.4	7.8	13.9	74.1	52.6	94.5	30.2	0.0	13.4
3–8	17.1	8.2	26.6	76.2	58.0	97.0	57.0	0.0	23.8
1–8	15.3	7.8	26.6	75.7	52.6	97.0	87.2	0.0	23.8
				Period II					
1–2	12.9	7.2	15.9	90.3	81.0	98.0	6.0	0.0	6.0
3–8	6.4	−9.7	14.0	97.9	92.0	99.0	115.6	0.0	19.0
1–8	8.1	−9.7	15.9	95.9	81.0	99.0	121.6	0.0	19.0
				Period III					
1–2	0.05	−9.7	10.1	97.4	92.0	99.0	35.4	0.0	14.6
3–10	3.3	−4.8	10.0	98.4	90.0	99.0	79.0	0.0	11.2
1–10	2.6	−9.7	10.1	98.2	90.0	99.0	114.4	0.0	14.6
				Period IV					
1–2	16.6	9.6	24.4	80.1	66.0	98.0	53.6	0.0	23.0
3–8	16.3	12.6	20.3	83.7	70.0	98.0	76.2	0.0	14.0
1–8	16.3	0.0	23.0	82.5	66.0	98.0	129.8	0.0	23.0

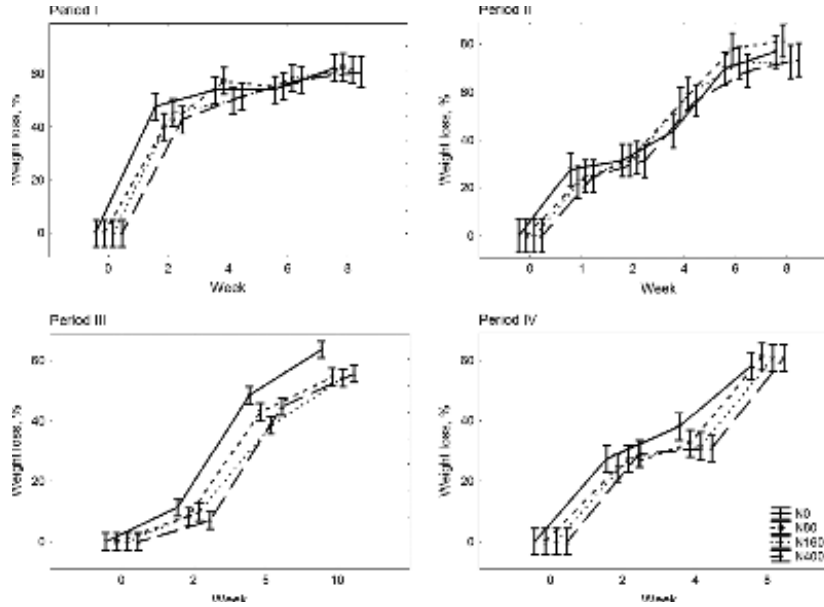

Figure 1. Turfgrass clippings weight loss, as percentage (%) of the initial dry matter (DM) weight, during decomposition in spring–summer 2006 (Period I), in autumn (Period II), autumn–winter (Period III) and in spring–summer 2007 (Period IV). Bars indicate confidence limits at $p < 0.05$.

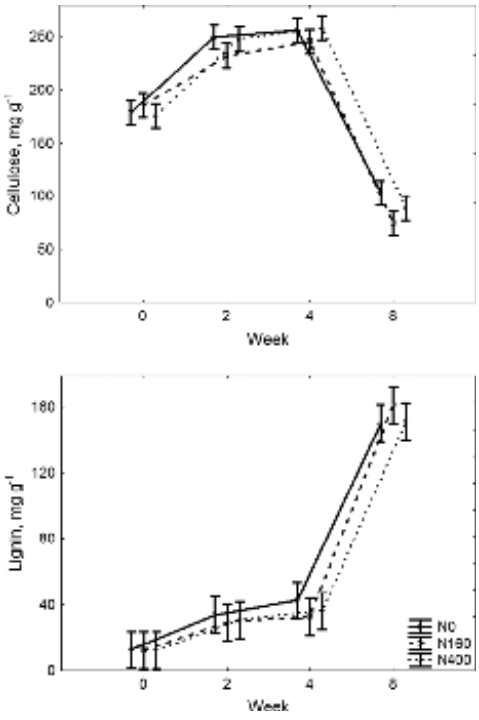

Figure 2. Concentration of the lignin and cellulose in turfgrass clippings during decomposition (Period IV). Bars indicate confidence limits at $p < 0.05$.

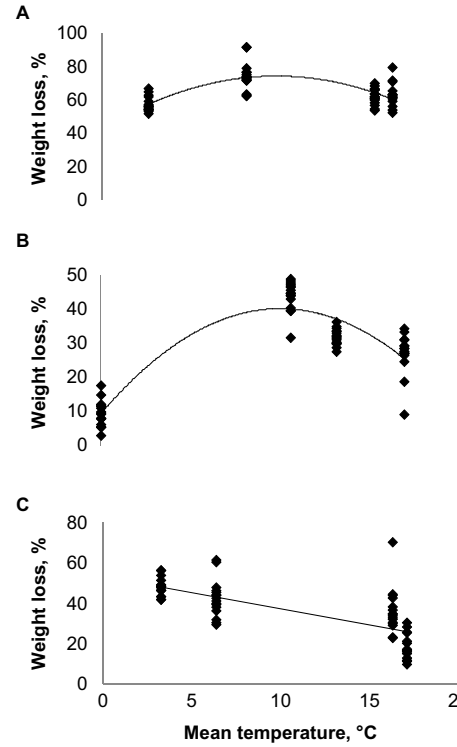

Figure 3. Regressions of mean temperature and weight losses at different times of the decomposition process. The points at different temperatures indicate weight losses of the decaying material from different fertilization treatments. A: Week 1–8 $y = -0.32x^2 + 6.33x + 43.08$ $R^2 = 0.46$*** (polynom). B: Week 1–2 $y = -0.32x^2 + 6.30x + 9.15$ $R^2 = 0.83$* (polynom). C: Week 3–8 $y = -1.58x + 53.09$ $R^2 = 0.48$* (linear). * Significant at $p < 0.05$. *** Significant at $p < 0.001$.

the initial mass had already decomposed (Figure 2). The initial content of cellulose (179.4 mg g^{-1}) had decreased by the end of the investigated period to 88.8 mg g^{-1}. The initial content of lignin in the decomposing material was 12.4 mg g^{-1}. During 8 weeks (the period of decomposition in this trial) the lignin was not decomposing and its concentration was increasing during successive weeks. By week 8 the content of lignin had increased up to 154.0 mg g^{-1}.

Influence of weather conditions on decomposition of clippings

Air temperature was the only measured weather parameter that showed a significant influence on the decomposition of turfgrass clippings during an 8-week period ($R^2 = 0.97$; Figure 3). The general tendency was that increasing the air temperature to 10°C resulted in an increase in the rate of decomposition, but further increases in air temperature resulted in a slowing down of the decomposition rate (Figure 3). There were no significant relationships between relative air humidity or precipitation and weight loss over an 8-week period ($p > 0.05$; Table IV).

In the spring periods of the two years (Period I and Period IV) at the start of the decomposition process the weather conditions were similar for both 8-week periods. In 2006, the average air temperature over 8 weeks was 15.4°C and relative air humidity was 76%. In 2007, those parameters were 16.5°C and 82%, respectively. In similar weather conditions the weight losses over 8 weeks were also similar, and in those years by 8 weeks after the cut the proportions of material that had decomposed were 61.9% and 60.3%, respectively. In the late summer period (Period II) the average air temperature over 8 weeks was lower (8.2°C) and relative air humidity compared with the spring periods was higher (96%), and

Table IV. Regressions of mean relative humidity and precipitation with weight losses at different times of the decomposition process.

Parameter (Climatic variables)	Linear regression	R^2
	1–2 week	
Mean relative humidity, %	Weight loss = − 1.15x + 126.36	0.71*
Precipitation, mm	Weight loss = −0.18x + 33.94	0.063ns†
	3–8 week	
Mean relative humidity, %	Weight loss = 1.16x − 67.82	0.95***
Precipitation, mm	Weight loss = 0.39x + 3.39	0.53***
	1–8 week	
Mean relative humidity, %	Weight loss = 0.21x + 45,63	0.074ns
Precipitation, mm	Weight loss = 0.084x + 54.33	0.034ns

*Significant at $p < 0.05$.
***Significant at $p < 0.001$.
†ns, not significant.

in these weather conditions the rate of decomposition was the fastest. In period III the average air temperature during 8 weeks was 2.2°C, relative air humidity was the highest (98%) and under these conditions 57% of material had decomposed.

By 1–2 weeks of the decomposition process the relationship between air temperature and weight loss was similar to the relationship between air temperature and weight loss of the whole investigated period, and the optimal air temperature for decomposition of clippings was 10°C on average. The mean air temperatures of Period I weeks 1–2 were 10.4°C and of Period IV weeks 1–2 16.3°C. The weight loss during weeks 1–2 in the cooler temperatures of spring–summer 2006 was one-third greater than in the warm temperatures of spring–summer 2007, 43.9% and 27.1% of initial dry matter, respectively.

The rate of decomposition was also decreased under conditions of low temperatures. The slowest was the decomposition during late autumn (Period III) when the average air temperature of weeks 1–2 was 0.1°C. In Period II the average temperature of weeks 1–2 was 8.2°C and weight loss 32%, which was similar to Period IV.

The relationship between weight loss and relative air humidity was negative during weeks 1–2 (Table IV). The highest relative air humidity (91–97%) in this period was characteristic of the late summer and autumn periods (Periods II and III) and the lowest was the air humidity (74–80%) during spring periods. The significant relationship between precipitation and weight loss during weeks 1–2 was not observed.

After the second week of each period the decomposition rate was faster during the autumn periods (Period II and III). The average air temperature of weeks 3–8 of Period III was higher than during first 2 weeks, thereby resulting in a remarkable increase in decomposition rate. The slowest was the decomposition in weeks 3–8 of Period I, when the average air temperature was 17°C. Increasing the relative air humidity and precipitation had a positive influence on weight loss (Table IV).

Decomposition models

Climate parameters (temperature, humidity and precipitation) were tested to predict plant residues decomposition with a multiple regression model (Table V). By weather parameters weight loss models were fitting better if every 8-week period was divided by two different time periods: 1–2 and 3–8 weeks.

Table V. Results of multiple regression analysis with backward stepwise of the test between weight losses of different weeks and mean temperature, mean relative humidity and precipitation.

Week	Intercept	Mean air temperature,°C	Mean air relative humidity,%	Precipitation, mm	Standard error of estimate	R^2	p
1–2	154.770 (6.442)†	0.191 (0.088)	−1.352 (0.064)	−0.392 (0.040)	3.292	0.942	<0.0000
3–8	−41.294 (18.057)	1.225 (0.562)		0.687 (0.129)	7.527	0.703	<0.0000
1–8	−82.776 (15.695)	1.766 (0.234)	1.994 (0.176)	−0.380 (0.038)	4.342	0.736	<0.0000

†Standard deviations are in parentheses.

The multiple regression backward stepwise analysis revealed that air temperature will have a strong positive influence on decomposition process. The model that contained the average air temperature, relative air humidity and precipitation described well the speed of decomposition in weeks 1–2 and 1–8. In weeks 3–8 the weight loss model was the best if the average air temperature and precipitation were included in the model.

Discussion

The influence of air temperature and humidity on decomposition of fresh organic material

The rate of decomposition of turfgrass clippings depends mainly on the weather conditions during the decomposition period. The initial content of nitrogen and different C:N ratio in plant material did not affect the decomposition process through the whole trial period. Quemada and Cabrera (1995) and Wagner and Wolf (1999) have concluded that an increase in nitrogen content does not influence the process of decomposition if the C:N ratio of decomposable material is initially less than 20. In our trial the initial C:N ratio was less than 20 in all fertilization treatments, which was also the presumable cause for a similar decomposition process for the material derived from different fertilization plots. The influence of N fertilization rate on the clippings decomposition has been dealt in our previous paper (Kauer et al. 2007).

Air temperature and relative air humidity were important factors influencing the decomposition of turfgrass clippings on the sward surface. Previous investigations have revealed that by increasing temperatures up to 35°C the rate of organic matter decomposition will increase, and after that it stabilizes and then starts to decrease (McKinley and Vestal, 1985, Chen et al. 2000). Our results indicated that during decomposition of fresh material on the ground surface the optimal air temperature for decomposition was lower than previously reported and did not exceed 10°C. There was a reduction in the rate of decomposition at temperatures above and below 10°C. The research of Flanagan and Veum (1974) showed that decomposition of organic matter can be limited by low air temperature as well as low moisture content, and increasing only one of these factors does not compensate fully for the influence of other limiting factors. Flanagan and Veum (1974) also indicated that, in the case of decomposing material which had low moisture content ($<50\%$ of dry weight), temperature increases had little effect on decomposition, but at higher moisture content, respiration was more responsive to temperature changes. Similarly, they noted that moisture changes had little effect on litter decomposition at lower temperatures (<5°C), while at higher temperatures (10–15°C), decomposition was more responsive to moisture changes. Too little or too much water inhibited, or even stopped, litter decomposition due to matric limitation or oxygen diffusion limitation, respectively (Flanagan and Veum 1974, Clark and Gilmour 1983). Our research gave similar results. In our research the dynamics of moisture content in decaying material was not determined, but still we presume that by increasing the temperature above 10°C the low moisture content in decomposable material was becoming the limiting factor for the decomposition process. The fast drying of clippings is indispensable if they remain on the sward surface. To slow the drying process we must implement activities that promote rapid infiltration of the clippings into the canopy where the moisture content is higher. This is contributed by more frequent mowing. In this case the clippings which sift to the canopy are shorter in length and their reaching the soil surface is less hindered.

At temperatures below 10°C the limiting factor was the air temperature. The influence of air temperature on the decomposition process was the most important at the beginning of decomposition process (during first 1–2 weeks) when the rate of decomposition was highest. By increasing air temperature from the optimal (10°C), the weight loss by decomposing was slowed down and this effect was most likely caused by fast drying of material. Our results indicate that for fast decomposing of turfgrass clippings it is important to maintain the initial moisture content in the material because rainwater can not fully compensate for it. In Period IV the average air temperature during the first 2 weeks of decomposition process was 16°C. This was relatively higher than in other investigated decomposition periods. Even though the amount of the precipitation was also the highest, the weight loss stayed at a lower level compared with the other decomposition periods. The impact of moisture content of returned clippings at the beginning of decomposition is also indicated by the relationship between relative humidity and decomposition. At the beginning of the decomposition period (the first 1–2 weeks) when the material was still fresh, the relationship between air humidity and decomposition rate was negative (Table V). But it was only an apparent effect because the highest rates of air humidity were determined in late autumn when the air temperature was very low and therefore the weight loss by decomposition was very small.

The second reason why the relationship between air humidity and decomposition rate seems negative

is due to differences of the air humidity in spring and autumn. In spring (after winter) the content of the air humidity during the first 2 weeks of studied decomposition process is lower (74.1% Period I and 79.4% Period IV) than in autumn (90.3% for Period II and 97.4% for Period III). In spring the higher air temperature causes the higher air humidity due to the higher rate of evaporation from the soil. In summer and autumn the higher temperature decreases the air humidity. Due to the different temperature effect on the air humidity at the different times of the year the relationship between air humidity and decomposition rate was negative during the first 2 weeks of the decomposition process in our study. In summer and autumn the air humidity is more dependent on the air temperature and precipitation, which is why the decomposition during weeks 3–8 of the decomposition process is described mainly with the air temperature and precipitation. This means that to study the influence of weather conditions on the decomposition we should also take into account seasonal effects at different times of the year because the relationships between temperature and humidity may vary at different time of year.

The negative effect of air humidity on the decomposition process in the first weeks (1–2) and weeks 3–8 was the reason why the influence of air humidity was not significantly important on decomposition process in the whole period. From the beginning of the third week, when decomposable material had already dried, the relationship between weight loss and precipitation became positive, showing that in the case of dry material the moisture content is one of the important factors limiting the decomposition process. Henriksen and Breland (1999) found that plant residues were decomposing intensively even though the average temperature of the whole investigation period stayed under the 0°C and did not rise above 2.4°C. Our research results indicated also that by temperatures around zero degrees (Period III) the decomposition process continued but the rate of decomposition of grass clippings was significantly slower than the decomposition rate at higher temperatures. However, our results do not enable us to determine that the decomposition process continued at 0°C, or even by lower temperatures, because during that decomposition period there were days when the average temperature exceeded 5°C. It is possible that the majority of the decomposition process occurred during short periods when the temperatures were higher than 0°C. Due to variability in temperatures, average temperature cannot be considered a good indicator to characterize specified time periods for evaluating the influence of air temperature on decomposition. This assumption is true for late autumn, winter and early spring periods when average temperatures remain around 0°C but there are wide variations between night and day temperatures.

The influence of air temperature and humidity on easily decomposable compounds and cellulose

Our results indicated that during the first 2 weeks, when it was mainly the more easily decomposable compounds that were decomposing under conditions where the content of moisture was not the limiting factor, the rate of decomposition increased remarkably in response to increasing temperatures from 0 to 10°C. In such conditions the influence of increasing temperatures on the decomposition of easily decomposable compounds was especially noticeable at lower temperatures. In Period III, when the temperature was increasing from 0.1 to 3.2°C, the weight loss also increased remarkably as temperatures increased. Our results revealed that the decomposition of easily decomposable compounds depends on both air temperature and relative humidity. Andersen and Jensen (2001) investigated decomposition of previously dried plant residues at three different temperatures (3, 9 and 15°C) and constant air humidity and found that the decomposition of easily decomposable compounds (i.e. water-soluble compounds) was less dependent on temperature than was the decomposition of more slowly decomposable compounds (i.e. cellulose and lignin).

Several studies have concluded that the decomposition of slowly decomposable structural compounds is sensitive to changes in temperature, and the decomposition of such compounds by lower temperatures is more limited than by higher temperatures (De Neve et al. 1996, Nicolardot et al. 1994, Bol et al. 2003). According to chemical kinetic theory, decomposition of recalcitrant, slowly decomposing substrates has higher activation energy, and thus higher temperature sensitivity (Bosatta and Agren 1999). In our experiment we analysed the content of cellulose and lignin in the clippings but during the study period only the decomposition of cellulose took place, which does not confirm the earlier findings, that to decompose the structural compounds the higher temperature is needed. The decomposition of cellulose started when the weight loss of the initial sample was 33%. In decomposition Period I, during the first 2 weeks 43% of the sample had already decomposed due to the fact that

cellulose was also starting to decompose (assuming that clippings had the same chemical composition in both spring decomposition periods). The average temperature during the first 2 weeks of Period I was lower (10.4°C) than the average temperature in the first 2 weeks in Period IV (16.6°C). The decomposition of cellulose was more intensive if the decomposing material was more humid, because in Period I, due to lower air temperature, we can suppose that the material was not drying so fast as in Period IV. By continuing the decomposition process when the decaying material was already obtaining the characteristics of the surrounding environment (i.e. similar moisture content) the decomposition process is faster in conditions where the air temperature is lower and humidity higher. Donnelly et al. (1990) investigated the decomposition of cellulose and lignin at different temperature conditions (4, 12, 24°C) and soil moisture contents (20, 40, 60%). According to their results the highest rates of cellulose and lignin were decomposed at the highest of the investigated air temperatures and highest soil moisture content. The same research also revealed that, at lower soil moisture contents, the activity of microbes decomposing the cellulose and lignin was not increasing in response to increasing temperature. They concluded that if moisture is a limiting factor, then microbes did not respond to increasing temperatures and in this case the soil moisture content is a more important factor influencing the decomposition process than temperature. The results of our investigation for decomposition in weeks 5–8 are in accordance with the results of Donnelly et al. (1990). In decomposition Periods I and II until week 4, 53.6 and 53.1% was decomposed from the initial mass. The weight losses were similar, meaning that the material should contain similar amounts of different carbon compounds. The Periods I and II, weeks 5–8, the average air temperatures were 19.5 and 3.9°C and air humidity 73.0 and 97.7%, respectively. The decomposition in that time interval was faster in Period II, which was cooler and more humid than in Period I. Some other studies also support the finding that higher air temperatures do not increase the rate of the decomposition process, mainly due to lower moisture content in decomposing material (Giardina and Ryan 2000, Epstein et al. 2002, Aerts 2006).

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IV

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SPECIAL FEATURE: VEGETATION RESTORATION

Biomass accumulation during reed encroachment reduces efficiency of restoration of Baltic coastal grasslands

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Keywords
Boreal Baltic coastal meadows; Coastal salt marsh; Eutrophication; Grazing; Land-use change; Management; Nutrients; Restoration success

Abbreviations
DCA = Detrended Correspondence Analysis

Nomenclature
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Abstract

Question: To what extent is restoration of vegetation in coastal grasslands delayed by accumulation of nutrients after abandonment of traditional management and subsequent reed encroachment? How does nutrient flow in the plant–soil system react to reintroduction of grazing?

Location: Coast of Baltic Sea, western Estonia.

Methods: Abandoned, continuously managed and restored coastal meadows were selected in four different study regions and their vegetation composition sampled. Nitrogen, P, K, Na, Ca and Mg concentrations and C/N ratios were determined in both vegetation and soil. Differences between management groups were evaluated.

Results: Comparison among different management groups revealed several differences in both relative and total amount of nutrients in soil and vegetation. Most soil properties of restored sites were similar to those in abandoned sites. Carbon stock in the soil profile doubled after abandonment, total N concentration in the top soil layer increased while plant available P concentration decreased. The phytomass and chemical composition of phytomass rapidly changed back to a 'normal' level after restoration. Species composition remained different, but species typical of coastal grasslands were present in restored sites. There was a strong site specificity in the results.

Conclusions: Re-establishment of grazing had a rapid impact on plant biomass of coastal grasslands. Species composition responded more slowly, but target species returned relatively quickly. Slow recovery of soil properties, however, means that the results of restoration may be fragile and return of tall-growth vegetation is very probable if managent intensity declines. Long restoration periods should be planned to reach pre-abandonment environmental conditions when using non-destructive restoration methods.

Introduction

Boreal Baltic coastal meadows (Natura 2000 code 1630*; they also belong to the Northern European group of maritime salt marshes, Chapman 1977) are considered of very high nature conservation value (Adam 1990; Allen & Pye 1992; Rannap et al. 2004), serving as a habitat for many rare and threatened amphibian, bird (especially waders) and plant species (Rebassoo 1975; Puurmann & Ratas 1998; Kuresoo & Mägi 2004). In the northern part of the Baltic Sea area, they are located in an area of post-glacial isostatic land uplift. During the last several thousands of years, in regions where the shores are flat, sea is shallow and wave action is small, the land that has risen from the sea has been immediately taken into use by man to pasture livestock. This has prevented accumulation of nutrients in the soil, has kept the soils of coastal grasslands young and developing, and has created habitats with short-stature vegetation. As in all other semi-natural meadows in the hemiboreal zone of Europe, the quality of coastal meadows for nature conservation directly depends on human-induced management, mostly grazing (Gibson et al. 1987;

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Jerling 1999; Jutila 2001; Burnside et al. 2007). However, the intensification of agriculture in the 20th century has made pasturing in coastal areas economically unprofitable and most coastal meadows have now been abandoned.

On abandoned Baltic coastal grasslands, *Phragmites australis* becomes dominant and forms dense and high reedbeds. During this process, species that are adapted to the low statue of the vegetation of grazed coastal meadows lose their habitat and a considerable drop in biodiversity is observed (Dijkema 1990; Esselink et al. 2000; Burnside et al. 2007; Wanner 2009). Recognition of species decline has initiated several restoration projects in coastal grasslands (e.g. Kokovkin 2005). However, removal of *Phragmites* stands can be rather expensive and labour intensive and is not always successful (Burdick & Dionne 1994; Marks et al. 1994; Chambers et al. 1999). Therefore, conservation agencies in the Baltic region usually do not employ destructive methods of reed removal (e.g. sod cutting) and in most cases rely only on reintroduction of grazing as a restoration tool and allow the ecosystem to develop without extensive man-made transformations. However, in such modest and low-input restoration events, return of favoured bird and plant species to areas where grazing is reintroduced often takes longer than expected (Kuresoo & Mägi 2004).

We hypothesize that restoration success on abandoned coastal grasslands is reduced by natural eutrophication that takes place during reed encroachment (see also Chambers 1997; Chambers et al. 1999; Bart & Hartman 2000). Reed is highly productive and in natural reedbeds most of the biomass remains ungrazed and enters the detritus system (Polunin 1984; Hocking 1989). In coastal areas, where decomposition and mineralization are reduced due to moist conditions, reedbeds not only accumulate a large amount of biomass and nutrients above ground, but also in soil. Mineralization of litter in reedbeds is also suppressed by the high C/N ratio (Polunin 1984) of shoots of *Phragmites*.

Reintroduction of grazing should quickly reduce the amount of plant biomass and change the flow of nutrients in restored sites. Changes in vegetation composition can also be expected to occur relatively fast if there is still a seed bank present in the soil (although after a long time of abandonment this is often not the case; see e.g. Thompson et al. 1997; Wolters & Bakker 2002; Wanner 2009) or when there is a nearby source for species immigration (e.g. Bernhardt & Koch 2003). But the inertia of soil development processes is strong and could prevent re-establishment of small plant species characteristic of coastal grasslands due to increased productivity, even if their immigration is not limited by availability of propagules (Onaindia et al. 2001; Van Dijk et al. 2007).

The aim of the current work is to estimate the extent to which restoration effects are delayed by changes in nutrient availability on abandoned coastal grasslands and how nutrient flow in the plant–soil system reacts to reintroduction of grazing. We approach the issue using comparative analysis of vegetation and soil properties of abandoned coastal meadows (reedbeds or reed-dominated sites), restored coastal meadows and well-preserved (grazed) coastal meadows. We address the following specific questions: (1) What happens to the soils after management of coastal grassland ceases? (2) How do soil and vegetation properties (especially their macronutrient concentrations) respond to reintroduction of grazing? (3) To what extent is restoration of vegetation in coastal grasslands delayed by presumed accumulation of nutrients after abandonment of traditional management and subsequent reed encroachment?

Methods

Fourteen different coastal grasslands in four regions were selected along the western coast of Estonia (Fig. 1) based on information on their management history. In each region, continuously managed, abandoned (neither grazed nor mown for at least 30 yr before the study) and restored (by means of re-establishment of grazing about 3–5 yr before the study) coastal grassland sites were selected as close to each other as possible in order to minimize the effect of site specificity on soils (e.g. effects of parent material, stoniness, texture, weather). Restored sites were selected as close to abandoned sites as possible (in Haeska and in Piirumi separated only by a fence between the pastures) in order to assure the similarity of the vegetation and management history prior to the start of restoration. Managed sites were selected as having as similar geomorphology to the abandoned sites as possible. There was no restored site available in northernmost Silma region. Managed sites had been grazed primarily with cattle and occasionally with sheep and horses at ca. 0.5–1.5 livestock units per hectare per year.

All studied grasslands were relatively large and wide, with a distance from the shoreline to the landward edge of the grassland mostly exceeding 500 m. A relatively homogeneous upper part of the saline zone (middle to upper geolittoral) was selected for the study in all sites, and special care was taken to select areas without a clearly detectable elevation gradient in order to minimize differences in salinity, effects of waves, sedimentation, etc., between plots, and to also ensure comparability between sites. Plant associations dominating in managed grasslands were *Elytrigietum repentis*, *Junco-Glaucetum* and *Festucetum rubrae*. Abandoned grasslands were almost completely dominated

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Fig. 1. Locations of the study sites in four regions on the western coast of Estonia. Site numbers: 1 – Pürksi (abandoned; 58°59′51″ N, 23°34′03″ E); 2 – Tahu (managed; 58°59′38″ N, 23°33′56″ E); 3 – Põgari (managed; 58°48′13″ N, 23°31′10″ E); 4 – Saardu (abandoned; 58°47′20″ N, 23°36′0″ E); 5 – Haeska I (managed; 58°46′52″ N, 23°39′24″ E); 6 – Haeska II (restored; 58°46′58″ N, 23°41′44″ E); 7 – Haeska III (abandoned; 58°47′05″ N, 23°42′15″ E); 8 – Salmi (managed; 58°43′55″ N, 23°40′01″ E); 9 – Kastna (abandoned; 58°19′33″ N, 23°54′30″ E); 10 – Suti (managed; 58°18′55″ N, 23°58′30″ E); 11 – Kavaru (restored; 58°16′06″ N, 24°10′37″ E); 12 – Piirumi I (restored; 58°09′30″ N, 24°28′37″ E); 13 – Piirumi II (abandoned; 58°09′14″ N, 24°28′48″ E); 14 – Häädemeeste (managed; 58°5′27″ N; 24°29′08″ E).

by *Phragmitetum australis* while restored sites did not have fully developed plant associations (transitional versions of *Deschampsio-Caricetum nigrae*, *Elytrigietum repentis* and *Phragmitetum* prevailed).

In each site 20 0.5 m × 0.5 m relevés were investigated on two 90-m long transects (ten plots per transect, 10-m apart) located 30 m from each other and perpendicular to the coastline. In each relevé, plant species composition was determined, cover of each species estimated and vegetation height measured. From each second relevé (ten plots per site) the central 10 cm × 50 cm part was sampled for measurement of plant standing biomass (litter was excluded), and from the rest of the plot at least 15 g of living plant material was collected for chemical analyses. In the central part of the same plots the depth of a humus layer (A, AT and AO horizons, hereinafter referred to as top layer) of soil was measured and soil samples from the top layer were collected for chemical analyses. One soil pit (up to 1-m deep) was excavated between the transects for description of soil type, generic soil layers and collecting samples for estimation of bulk density and C content of the soil. Plant biomass samples were dried at 80 °C for 48 h and then weighed. Samples for chemical analyses were air-dried. Field analyses were undertaken in Jul and Aug 2005.

Acid digestion with sulphuric acid solution was used to determine total content of P, K, Na, Ca and Mg in the plant material. After digestion, the content of total P was determined colorimetrically. Total K and Na content were determined by flame photometry and total Ca and total Mg were measured using atomic absorption spectroscopy. Total N and total C content of oven-dried samples were determined by the dry combustion method on a varioMAX CNS elemental analyser (ELEMENTAR, Germany).

Soil samples were air-dried, sieved through a 2-mm sieve and analysed for total N (Kjeldahl method), plant available P, K, Ca and Mg (Mehlich-3), organic C (Tjurin) and pH (KCl). Organic matter content of the soil was determined by weight loss after heating for 4 h at 500 °C. All chemical analyses were performed in the Laboratory of Soil Science and Agrochemistry of the Estonian University of Life Sciences. Carbon stock in the soil profile was calculated by multiplying bulk density of each soil layer with the depth and C content of that layer and then correcting for the area.

We employed two-way analysis of variance (ANOVA) to estimate differences between management groups, effect of region and regional differences in management effects (as estimated by the interaction between the factors 'region' and 'management'). All variables were tested for normal distribution of residuals of model predictions with the Shapiro test. Number of species and vegetation height were log-transformed and plant canopy cover was arcsin-transformed prior to analyses to obtain a fit with normal distribution. We employed Dunnett's modified Tukey-Kramer pair-wise multiple comparison test (DTK test) for detection of homogeneous groups.

Correlation between concentration of nutrients in the soil and in plants was tested using linear correlation analysis, which was performed separately for different management types as well as for the pooled data.

We used detrended correspondence analysis (DCA) in order to describe variation of vegetation composition and passively fitted environmental vectors on the resulting ordination. Linear correlation analysis was employed to estimate the relationship between the axes values for each site and the non-categorical environmental variables. Differences in diversity of vegetation between management groups were assessed with Shannon diversity and evenness indices.

Twenty-nine species commonly found in ecologically well-preserved coastal grasslands (pers. obs.) were selected to serve as indicator or target species for evaluation of

restoration success (Appendix S1; hereinafter 'typical coastal grassland species'). Species selection was guided by the database of habitat preferences of Estonian species (Sammul et al. 2008). Relative frequency of presence in relevés (occurrence probability) of these species was calculated per each management group and arcsin-transformed to obtain normal distribution. Mean occurrence probabilities were compared between management groups with paired *t*-tests assuming unequal variance and with Welch adjustment to the degrees of freedom.

All statistical analyses were carried out with the R software version 2.10.1 (R Development Core Team, Vienna, Austria). DCA was carried out using the package VEGAN (version 1.17-0), and environmental vectors were fitted using the function 'envfit' with 999 permutations.

Results

Vegetation properties

Management had a strong influence on most vegetation properties (Table 1). Abandoned sites had larger plant canopy cover, taller vegetation and more plant biomass than managed and restored sites. Managed and restored sites had more diverse vegetation (more species per relevé, increase in diversity indices). Concentrations of N and Na were larger in plants of managed and restored sites than in plants of abandoned sites. Only the concentration of P and Ca, as well as the C/P and N/P ratios, in plants did not differ between sites with different management. Most of the vegetation properties in restored sites were similar to those of managed sites, while abandoned grasslands formed a separate homogeneous group. The exceptions to the above were mean Shannon diversity index and evenness of vegetation (restored sites were a separate homogeneous group with intermediate diversity values between low-diversity abandoned sites and high-diversity managed sites), concentration of Mg in plants (restored sites and abandoned sites were similar and both differed from managed sites), concentration of K in plants (plants in restored sites had a lower concentration of K than plants in managed or abandoned sites) and C/N ratio (managed sites did not differ from either restored or abandoned sites while the latter differed from each other).

There was also a strong difference between different regions in most vegetation properties. Only concentration of Mg, C/P ratio and N/P ratio in plants did not differ between different regions. Regional differences were also pronounced in the significance of the interaction between the effects of management and region, however, most interactions were ordinal and only N concentration in plants had a disordinal interaction between region and management.

Soil properties

The soils of the coastal grasslands studied are classified as Gleyic Fluvisols (Sodic), Histic Fluvisols (Sodic) and Eutric Histosols (Endofluvic features; WRB 2006). Soils have developed on sand or clay as parent material, they are moist and with slightly developed profiles. Abandoned areas with a fully developed reedbed had a very thick and tough top layer of soil with large quantities of roots and litter of *Phragmites*. Five of the grasslands had turf layers deeper than 10 cm. Two of these grasslands belong to group of abandoned grasslands (Pürksi: total range of turf layer 7–16 cm, share of plots with turf layer over 10 cm deep 60%; Kastna: 10–46 cm, 70%), two sites were managed (Häädemeeste: 8–23 cm, 80%; Suti: 10–19 cm; 90%) and one site was restored (Piirumi restored: 5–25 cm, 30%).

Management has a statistically significant influence on all soil properties studied except for C stock in the soil profile (Table 2). There was a two-fold difference in C soil stock between managed sites and either abandoned or restored sites; however, due to large variations and lack of replication (only one value could be estimated per each site) this difference is not statistically valid. Most soil parameters did not differ between restored and abandoned sites, whereas managed sites formed a separate homogeneous group. The exceptions were the depth of the top layer, N content and P content, for which managed and abandoned sites form separate homogeneous groups, while restored sites did not differ from sites of other types due to a large variation. Soils of managed sites had a shallower top layer, smaller organic matter content, smaller N, C and Mg concentration and C/N ratio, but higher P and Ca concentration, as well as higher pH, than abandoned sites.

The regional differences were important for the depth of top layer, pH and concentrations of K, Ca, Mg and C/N ratio of soils. The interaction of management and region was significant for all estimated soil parameters except P content in soils. Most interactions were ordinal, but K and Ca content in soil were disordinal.

The content of a particular mineral element in the soil was only infrequently correlated with its content in the plants (Table 3) and correlations were rare in several management types simultaneously. Content of P in plants was negatively correlated with its content in soil in restored sites, content of Mg in plants was negatively correlated with its content in soil in abandoned sites and when the data from different management groups were pooled. Content of Ca in plants was negatively correlated with Ca content in soil in managed sites and in pooled data. There was a positive correlation between C/N ratio in plants and in soil in managed sites and a negative correlation between C/N ratio in plants and soil in restored sites.

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Table 1. Mean values (with 95% confidence intervals) and homogeneous groups (a, b, or c; at $P < 0.05$) of vegetation characteristics and chemical properties of plant biomass from different management groups, with the results of two-way ANOVA for differences between management groups (df $F_{(effect,\ error)}$ of traits from no. species through evenness = 2, 269; for chemical properties df = 2, 129), between regions (df = 2, 269 and df = 2, 129, respectively) and their interaction (df = 5, 269 and df = 5, 129).

Dependent variable	Mean values and homogeneous groups							*F*-Value of a comparison		
	Management Group			Region						
	Managed N = 100[i]; 50[ii]; 20[iii]	Restored N = 120; 60; 24	Abandoned N = 60;30;12	Häädemeeste N = 60; 30; 12	Matsalu N = 120; 60; 24	Silma N = 40; 20; 8	Tõstamaa N = 60; 30; 12	Between management groups	Between regions	Interaction management * region
Number of species (0.25 m^{-2})[i]	10.2 ± 0.8[a]	9.27 ± 0.88[a]	7.78 ± 0.75[b]	11.6 ± 1.3[c]	9.44 ± 0.71[b]	8.03 ± 0.71[a,b]	6.72 ± 0.80[a]	F = 16.8***	F = 22***	F = 16.5***
Plant canopy cover (%)[i]	63 ± 3[a]	57 ± 4[a]	71 ± 4[b]	72 ± 4.3[c]	61.7 ± 3.1[b]	54.8 ± 5.8[a]	68.9 ± 4.8[c]	F = 19.9***	F = 18.8***	F = 4.32**
Vegetation height (cm)[i]	15.8 ± 1.5[a]	18.7 ± 2.5[a]	106 ± 17[b]	36 ± 11[a]	36.3 ± 8.3[a]	44.2 ± 9.3[a]	88.8 ± 28[b]	F = 242***	F = 15.7***	F = 15.4***
Plant above-ground biomass (g m^{-2})[ii]	305 ± 36[a]	292 ± 71[a]	652 ± 126[b]	21.3 ± 4.2[a,b]	18.1 ± 2.8[a]	17.9 ± 5.7[a]	30.1 ± 10[b]	F = 28***	F = 7.60***	F = 5.29***
Shannon diversity index[i]	1.77 ± 0.07[a]	1.56 ± 0.11[b]	1.35 ± 0.12[c]	1.75 ± 0.14[a]	1.65 ± 0.08[a]	1.54 ± 0.13[a]	1.26 ± 0.14[b]	F = 24***	F = 5.44**	F = 7.67***
Evenness[i]	0.79 ± 0.01[a]	0.72 ± 0.03[b]	0.66 ± 0.04[c]	0.74 ± 0.03[a]	0.76 ± 0.02[a]	0.75 ± 0.05[a]	0.66 ± 0.06[b]	F = 28***	F = 28***	F = 28***
N (%)[ii]	1.68 ± 0.12[a]	1.73 ± 0.21[a]	1.47 ± 0.11[b]	1.88 ± 0.23[b]	1.51 ± 0.12[a]	1.36 ± 0.07[a]	1.73 ± 0.09[b]	F = 8.58***	F = 13.8***	F = 28.8***, d
P (%)[ii]	0.054 ± 0.008[a]	0.046 ± 0.011[a]	0.05 ± 0.01[a]	0.06 ± 0.01[b]	0.046 ± 0.007[a]	0.04 ± 0.004[a]	0.05 ± 0.009[a,b]	F = 1.59 [n.s.]	F = 4.84**	F = 13.64***
K (%)[ii]	1.55 ± 0.10[a]	1.19 ± 0.09[b]	1.53 ± 0.15[a]	1.48 ± 0.19[b, c]	1.41 ± 0.11[a,b]	1.25 ± 0.07[a]	1.69 ± 0.15[c]	F = 9.41***	F = 10.1***	F = 3.08*
C (%)[iii]	42.5 ± 1.1[a]	41.5 ± 0.9[a]	42.3 ± 1.1[a]	41.7 ± 1.1[a]	42.2 ± 0.9[a]	43.8 ± 1.9[a]	41.7 ± 1.4[a]	F = 0.92 [n.s.]	F = 1.43 [n.s.]	F = 2.66*
Na (%)[ii]	1.32 ± 0.24[a]	0.98 ± 0.26[a]	0.40 ± 0.15[b]	0.45 ± 0.11[a]	1.20 ± 0.27[b]	0.73 ± 0.27[a,b]	0.94 ± 0.27[b]	F = 23***	F = 6.71***	F = 1.89 [n.s.]
Ca (%)[ii]	0.30 ± 0.05[a]	0.25 ± 0.04[a]	0.25 ± 0.02[a]	0.19 ± 0.03[a]	0.28 ± 0.04[b]	0.38 ± 0.07[c]	0.27 ± 0.03[b]	F = 2.62[n.s.]	F = 11.0***	F = 9.72***
Mg (%)[ii]	0.46 ± 0.03[a]	0.36 ± 0.03[b]	0.36 ± 0.03[b]	0.41 ± 0.04[a]	0.43 ± 0.03[a]	0.39 ± 0.05[a]	0.37 ± 0.04[a]	F = 14.6***	F = 1.50 [n.s.]	F = 2.43*
C/N ratio[iii]	27 ± 3[a,b]	25 ± 4[b]	31 ± 1[a]	24.4 ± 4.9[a]	30.5 ± 3.6[b]	30.6 ± 2.0[a,b]	25.0 ± 2.6[a]	F = 4.81*	F = 4.47**	F = 9.10***
C/P ratio[iii]	1333 ± 501[a]	1844 ± 958[a]	1157 ± 364[a]	1599 ± 899[a]	1471 ± 542[a]	1158 ± 193[a]	1125 ± 590[a]	F = 1.52 [n.s.]	F = 0.53 [n.s.]	F = 4.29**
N/P ratio[iii]	46 ± 14[a]	67 ± 31[a]	44 ± 19[a]	59 ± 27[a]	49.7 ± 18.6[a]	37.8 ± 5.8[a]	47.9 ± 28[a]	F = 1.52 [n.s.]	F = 0.26 [n.s.]	F = 3.23*

[i]Variables for which sample size is given as the first figure after N.

[ii]Variables for which sample size is given as the second figure after N.

[iii]Variables for which sample size is given as the third figure after N.

N = samples size; ***$P < 0.001$; **$P < 0.01$; *$P < 0.05$; [n.s.]not significant; d = disordinal interaction.

Table 2. Mean values (with 95% confidence intervals) and homogeneous groups (a, b, or c; at $P < 0.05$) of soil properties for different management groups with results of two-way ANOVA for differences between management groups (df of $F_{(effect,\ error)} = 2, 129$), between regions ($df = 2, 129$) and their interaction ($df = 5, 129$).

Dependent variable	Mean values and homogeneous groups							*F*-Value of a comparison		
	Management group			Region						
	Managed N = 50†; 5‡	Restored N = 60†; 6‡	Abandoned N = 30†; 3‡	Häädemeeste N = 30†; 3‡	Matsalu N = 60†; 6‡	Silma N = 20†; 2‡	Tõstamaa N = 30†; 3‡	Between management groups	Between regions	Interaction management* region
Depth of top layer (cm)	10.0 ± 1.2^{a}	11.6 ± 2.0^{a,b}	13.1 ± 2.5^{b}	13.9 ± 2.13^{b}	7.63 ± 0.65^{a}	9.45 ± 1.24^{a}	18.1 ± 3.40^{c}	$F = 4.95$**	$F = 30$***	$F = 3.19$**
Organic matter content (%)	35 ± 4^{a}	42 ± 6^{b}	47 ± 5^{b}	40.1 ± 6.74^{a}	40.0 ± 4.55^{a}	42.3 ± 4.24^{a}	41.9 ± 6.72^{a}	$F = 10.5$***	$F = 0.17$ n.s.	$F = 10.9$***
Bulk density of top layer (g cm^{-3})	0.42 ± 0.11^{a}	0.26 ± 0.15^{a}	0.23 ± 0.12^{a}	0.26 ± 0.16	0.36 ± 0.16	0.24 ± 0.20	0.36 ± 0.10	$F = 2.92$ n.s.	Could not be estimated	Could not be estimated
PH_{KCl}	6.10 ± 0.2^{a}	5.57 ± 0.2^{b}	5.68 ± 0.20^{b}	5.70 ± 0.30^{b}	5.99 ± 0.21$^{b,\ c}$	5.26 ± 0.19^{a}	6.04 ± 0.16^{c}	$F = 21$***	$F = 22$***	$F = 44$***
N (%)	1.26 ± 0.14^{a}	1.52 ± 0.22^{a,b}	1.52 ± 0.16^{b}	1.60 ± 0.23^{a}	1.30 ± 0.15^{a}	1.45 ± 0.13^{a}	1.41 ± 0.23^{a}	$F = 4.32$*	$F = 1.79$ n.s.	$F = 8.23$***
C (%)	11.4 ± 1.4^{a}	17.0 ± 2.9^{b}	17.0 ± 2.4^{b}	13.1 ± 2.45^{a}	14.9 ± 2.21^{a}	14.7 ± 1.48^{a}	15.4 ± 3.17^{a}	$F = 15.3$***	$F = 1.55$ n.s.	$F = 15.2$***
P (mg·kg^{-1})	32 ± 8^{a}	34 ± 20^{a,b}	21 ± 5^{b}	63.1 ± 19.2^{b}	24.0 ± 6.1^{a}	15.0 ± 5.5^{a}	12.0 ± 4.93^{a}	$F = 3.88$*	$F = 26$ n.s.	$F = 13.7$ n.s.
K (mg·kg^{-1})	441 ± 60^{a}	341 ± 66^{b}	414 ± 59^{a,b}	292 ± 75^{a}	461 ± 61^{b}	452 ± 44^{b}	397 ± 76^{a,b}	$F = 3.22$*	$F = 5.36$**	$F = 13.6$***, d
Ca (mg·kg^{-1})	2009 ± 532^{a}	928 ± 105^{b}	1025 ± 201^{b}	961 ± 302^{a}	2070 ± 519^{b}	599 ± 77^{a}	1153 ± 188^{a}	$F = 12.3$***	$F = 9.41$***	$F = 7.98$***, d
Mg (mg·kg^{-1})	1147 ± 165^{a}	1671 ± 177^{b}	1729 ± 173^{b}	1347 ± 202^{a}	1319 ± 193^{a}	1534 ± 102^{a,b}	1837 ± 226^{b}	$F = 22.7$***	$F = 6.83$***	$F = 13.2$***
C/N ratio	9.0 ± 1.4^{a}	11.3 ± 2.9^{b}	11.7 ± 2.4^{b}	8.03 ± 0.78^{a}	11.1 ± 0.84^{b}	10.1 ± 0.61^{a,b}	11.8 ± 2.96^{b}	$F = 7.83$***	$F = 6.87$***	$F = 10.2$***
C stock in soil profile (t·ha^{-1})	64 ± 15^{a}	122 ± 58^{b}	122 ± 120^{b}	81 ± 27	80 ± 41	50 ± 40	179 ± 194	$F = 0.7$ n.s.	Could not be estimated	Could not be estimated

†Applies to all dependent variables except bulk density and C stock in soil profile.

‡Applies to bulk density and C stock in soil profile.

N = samples size;

***$P < 0.001$; **$P < 0.01$; *$P < 0.05$; n.s. not significant; d = disordinal interaction.

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Table 3. Linear correlations between estimated chemical properties of soils and plants. Statistically significant correlations are printed in bold.

Correlation (Plants vs soil)	Managed		Restored		Abandoned		Pooled	
	r	*P*	*r*	*P*	*r*	*P*	*r*	*P*
N	0.018	0.89	−0.29	0.12	−0.0005	0.99	−0.09	0.29
P	0.18	0.17	**−0.39**	**0.031**	0.27	0.055	−0.026	0.76
K	−0.10	0.43	−0.36	0.050	0.096	0.51	0.014	0.87
Mg	0.002	0.99	0.33	0.071	**−0.41**	**0.003**	**−0.27**	**0.001**
Ca	**−0.44**	**0.0004**	−0.008	0.97	−0.050	0.73	**−0.30**	**0.0003**
C	**0.30**	**0.0007**	−0.018	0.89	0.009	0.93	0.059	0.33
C/N ratio	**0.53**	**<0.0001**	**−0.57**	**<0.0001**	−0.17	0.097	0.0003	0.99

r = Pearson correlation coefficient; *P* = probability level.

Species composition

The DCA on the species abundances produced four axes with eigenvalues of 0.76, 0.62, 0.43 and 0.32. The managed habitats and the restored habitats formed distinct groups in an ordination plane (Fig. 2), suggesting different vegetation composition. However, the abandoned habitats were very scattered, showing very large variations in vegetation composition of abandoned sites. The correlation of DCA axes with environmental parameters is given in Table 4. Axis 1 is primarily related to vegetation characteristics. The variation is led by the effect of management – abandoned sites are located on the positive side and managed sites on the negative side of DCA axis 1 (Fig. 2a). The axis is also negatively correlated with soil pH and positively correlated with vegetation height, which is effectively determined by management regime. DCA axis 2 is positively correlated with parameters indicating low mineralization rate and accumulation of biomass in soil (soil C content, organic matter content and C/N ratio, but also Mg content). The negative end of DCA axis 2 primarily indicates high species richness of a habitat.

The distribution of species in an ordination plane revealed a distinction between different ecological groups (Fig. 2b). The positive end of DCA axis 1 is characterized by tall species of productive habitats (e.g. *Urtica dioica* and *Anthriscus sylvestris*), while at the negative end typical coastal grassland species aggregate (e.g. *Plantago maritima*,

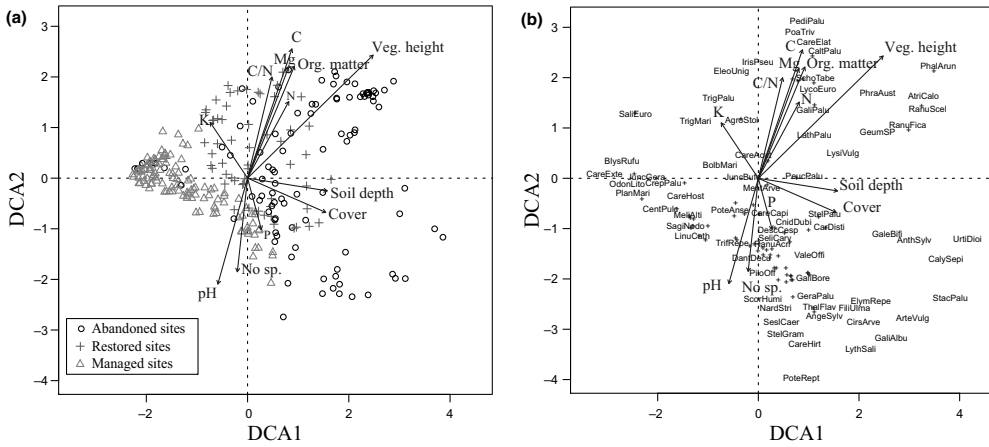

Fig. 2. Distribution of individual relevés (**a**), all 280 relevés included, and species (**b**) in the ordination diagram for the first two axes of the DCA. Environmental vectors are neutrally fitted into the ordination plane if their correlation with DCA axes is statistically significant at $P < 0.05$ (see also Table 4). In (**b**), where species names overlap, higher priority for plotting the name is given to the more abundant species: less abundant species are plotted as crosses (+). Abbreviations of environmental variables: pH – soil pH; No Sp – number of vascular plant species in relevé; Cover – plant canopy cover; Soil Depth – depth of the top layer of soil; Veg. Height – vegetation height; Org. Matter – organic matter content of soil; N – nitrogen content in soil; C – carbon content in soil; Mg – magnesium content in soil; C/N – C/N ratio in soil; K – potassium content in soil; P – phosphorus content in soil. Species names are given in Appendix S1.

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Table 4. Linear correlations between between site scores on DCA axes and non-categorical environmental variables.

Variable	DCA1	DCA2	DCA3	DCA4
Number of species	−0.36***	−0.35***	0.19**	0.18**
Plant canopy cover	0.17**	0.19**	0.29**	$0.10^{n.s.}$
Vegetation height	0.52***	0.36***	0.15*	−0.13*
Depth of top layer	0.32***	−0.12*	$0.10^{n.s.}$	−0.12*
Organic matter content in soil	0.47***	0.47***	$0.07^{n.s.}$	$-0.10^{n.s.}$
pH of soil	−0.41***	−0.22***	−0.15*	0.30***
N content in soil	0.32***	0.31***	$0.06^{n.s.}$	$-0.04^{n.s.}$
C content in soil	0.49***	0.54***	$0.06^{n.s.}$	−0.24***
C/N ratio in soil	0.33***	0.38***	$-0.04^{n.s.}$	−0.14*
P content in soil	$-0.06^{n.s.}$	−0.15*	0.19**	$0.04^{n.s.}$
K content in soil	0.20***	0.23***	$0.02^{n.s.}$	0.22***
Ca content in soil	$-0.07^{n.s.}$	$0.02^{n.s.}$	0.13*	$-0.10^{n.s.}$
Mg content in soil	0.46***	0.43***	$-0.01^{n.s.}$	$0.01^{n.s.}$

r = Pearson correlation coefficient; $^{***}P < 0.001$; $^{**}P < 0.01$; $^{*}P < 0.05$; $^{n.s.}$not significant.

Odontites spp., *Juncus gerardii*, *Carex extensa* and *Blysmus rufus*). The positive end of DCA axis 2 is characterized by species of moist habitats (e.g. *Pedicularis palustris*, *Carex elata*, *Eleocharis uniglumis*, *Iris pseudacorus* and *Caltha palustris*). The negative end of DCA axis 2 is characterized by species of alkaline and species-rich habitats (e.g. *Sesleria caerulea*, *Scorzonera humilis*, *Nardus stricta* and *Pilosella officinarum*).

Mean relative occurrence frequency of typical coastal grassland species was highest in relevés of managed meadows and lowest in abandoned meadows (Fig. 3). The difference was statistically significant at $P < 0.006$ (two-tailed test, $df = 28$, $t = 3.02$). Selected species were less common in restored sites, but this difference did not differ from that of managed sites ($t_{(28)} = 1.34$, $P = 0.19$), while being different from that of abandoned sites ($t_{(28)} = 2.39$, $P < 0.024$).

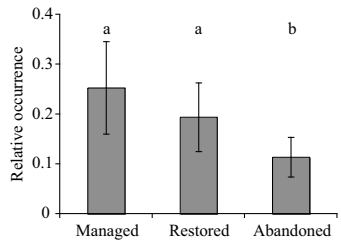

Fig. 3. Relative occurrence of typical coastal grassland species in relevés of the three studied management groups. Distribution of 29 species was analysed, bars denote 95% confidence intervals of the mean, and statistical differences between mean values are denoted with different letters.

Discussion

Grazing has been a traditional and regularly induced disturbance in Boreal Baltic coastal grasslands to which local species have adapted and which sustains local biodiversity, primarily by maintaing low-stature vegetation (Puurmann & Ratas 1998; Burnside et al. 2007; Wanner 2009). Depending on the differences between regions (e.g. abundance of reed or other habitats in a region), increasing height of vegetation or even a change from low-stature vegetation to a reedbed may or may not be considered favourable for nature conservation (see also Bakker et al. 1997). Such effects are especially debatable for bird species (Koivula & Rönkä 1998; Milsom et al. 2000; Bakker et al. 2003; Ottvall & Smith 2005). In the northern Baltic Sea area, following coastal grassland abandonment, bird diversity (especially waders) in most cases starts an immediate decline (e.g. Helle et al. 1988; Kuresoo & Mägi 2004). Plant diversity, however, often initially increases (e.g. Jutila 1997), only to decline in later stages of succession, especially after tall and dense reedbeds start to develop (see also Vestergaard 1998). After establishment of reed, distribution of typical small-stature seashore plants is restricted to the edges or occasional short-term openings within the reedbeds (pers. obs.). Our results indicate that in such abandoned sites with tall and *Phragmites*-dominated vegetation the different productivity components do not change in a uniform manner. Comparing abandoned sites to managed sites, N content in soil increases while P content decreases, even though the latter effect is not uniform across the regions studied. Nitrogen is typically the predominant limiting nutrient for salt marsh plants (Mendelssohn & Morris 2000; but see Van Wijnen & Bakker 1999 for more detailed analysis), in common with wetland plants in general (Van Duren & Pegtel 2000; Van de Riet et al. 2010), thus its increase could be interpreted as increased productivity. However, considering the large amount of organic C in soils of abandoned sites, one could assume that a large amount of N is actually bound with organic compounds in soil thus making it unavailable for plant growth. Carbon addition has even been used as a means to create N deficiency and reduce plant growth (Eschen et al. 2006; Reynolds & Haubensak 2009 and references therein). In our abandoned sites, the C/N ratio is higher than in managed sites, indicating a possible reduction in relative N availability. Moreover, in abandoned sites the availability of P is very low, making it the primary limiting factor for plant growth (see also Olff et al. 1997; Van Wijnen & Bakker 1997). Thus, the availability of nutrients in soil does not clearly increase with abandonment, as was initially hypothesized; yet plant biomass production increases considerably.

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The increase in plant biomass is due to the increased abundance of tall and competitive species with high relative growth rates – most notably *Phragmites australis*. *Phragmites* has a broad ecological amplitude and grows best in nutrient-rich habitats (Hocking et al. 1983; Güsewell & Koerselman 2002). It is also a species in which growth clearly benefits from increased N levels but is not as sensitive to variation in P levels (Romero et al. 1999). Eutrophication of the Baltic Sea (Rönnberg & Bonsdorff 2004) has probably contributed to the increased spread of *Phragmites*. Moreover, it has been shown that the disturbances from accumulation of wrack, to which large amounts of litter of *Phragmites* contribute, benefits its growth in some parts of coastal salt marshes (Minchinton 2002). Thus, there are several factors that simultaneously facilitate and even create a positive feedback for the development of reedbeds in abandoned coastal meadows.

The increased amounts of nutrients in soils of abandoned sites did not result in increased concentrations of mineral elements in plant biomass (Tables 1, 3). This suggests that mineralization of nutrients and their flow in the plant–soil system is reduced in such sites. The difference is again attributable to the dominance of tall grasses, mostly *Phragmites*. Importantly, low availability of N (in particular) and excess of C in plant litter decreases the speed of decomposition of litter and mineralization. This also affects mineralization of P, creating a strong deficiency of plant available P in soil and reducing P content in plants to extremely low levels. Considerable increases in the amounts of litter and poorer conditions for its decomposition lead to a large accumulation of organic matter in soils of abandoned sites, perhaps best illustrated as a two-fold increase in C stock in the soil profile. This change actually alters the whole structure of the soils – while soils of managed grasslands mostly belong to the class of mineral soils, soils of abandoned sites should mostly be classified as Histic Fluvisols and thereafter to Histosols (WRB 2006). This means that these soils have a considerable turf layer, reduced pH and provide completely different growth conditions for plants, as well as soil biota (e.g. Butt & Lowe 2004; Ivask et al. 2009). It is possible that reedbeds develop faster on sites where a turf layer is already present, in which case our results present not so much an effect of abandonment but rather regional differences in geomorphology of the coast, effect of elevation, etc. However, as we paid special attention to avoiding such effects when selecting the study sites, and also considering the number of sites and regions studied, we are certain that this is not the case. So far evidence of the impact of grazing on accumulation of biomass in soils is contradictory. Jeschke (1983) describes how on the German coast of the Baltic Sea trampling and soil compaction in grazed areas leads to reduced levels of decomposition and a build-up of organic matter (see also Cuttle 2008). Vestergaard (1998), on the other hand, describes accumulation of organic matter due to grassland abandonment on the southern Baltic coasts in SE Denmark, supporting our conclusions. Both studies also report the loss of a typical coastal grassland due to reed encroachment. There could be regional differences that are important to consider. While in the southern Baltic, the brackish coastal meadows are naturally on peaty soils, in the northern Baltic such accumulation of peat is considered an aberrance from the 'natural' state of coastal grasslands and, as such, a conservationally unfavourable process (see also Dijkema 1984) despite possible enhanced C sequestration (Chmura et al. 2003). Moreover, as our results also demonstrate differences between study areas, local conditions (such as differences in bedrock and geomorphology of the coast) could strongly affect the dynamics of coastal ecosystems. However, there could also be a discrepancy between the exact processes discussed. In the first case, the effect of trampling is discussed in areas that are still grasslands, and where soil compaction is an important factor. In the second case, the whole community (or even an ecosystem) is changing from grassland to reedbed and the importance of trampling is downgraded by the effect of increased production of plant biomass and increased litterfall.

Restoration of coastal grassland by means of simply reintroducing grazing, with or without initial cutting back of reed, but certainly without any application of intensive and destructive methods for reed reduction (e.g. top soil removal or herbicide application), does not succeed in changing the soil properties for at least the first 5 yr following restoration. This does not imply that such restorations are unsuccessful; as our results demonstrate, the vegetation of restored sites has reverted to a relatively similar state to traditionally managed (grazed) sites in terms of most properties. Thus, there is a considerable decrease in addition of organic matter to soil and the impact that plant litter (through both quantity and quality) has on soil formation has reverted to a state similar to managed (i.e. desired state) coastal meadows. Moreover, even though the soil N content and C/N ratio of restored sites is still similar to that of abandoned sites, the plant N content of restored sites is already similar to that of the managed sites, which indicates increased availability of N for plants and, hence, increased N mineralization in soil. If management of these areas continues, and new input of nutrients into the system can be avoided, the restoration of typical coastal grasslands should be possible. However, changes in soils take much longer than changes in above-ground properties of the plant community (see also Onaindia et al. 2001; Klimkowska et al. 2007; Van Dijk et al. 2007). This discrepancy should be considered when planning the duration of restoration projects as well as monitoring of restoration success.

The DCA axes show that species composition of the studied communities is primarily related to management regime and lushness of the vegetation. Obviously, these two aspects are negatively correlated, as grazing efficiently disturbs the vegetation and reduces the level of dominance by tall species. The second DCA axis is more complex and demonstrates the transition from species-rich pH-neutral (or even slightly alkaline) communities to moist, C- and organic matter-rich (peaty) communities. Ordination very clearly highlights the large variation in coastal communities (as also emphasized by differences between study regions in both vegetation and soil properties), especially variation in vegetation of abandoned sites. Not all reedbeds are species-poor; Vestergaard (1998) decribes multi-species *Phragmites*-dominated communities in the geolittoral of Danish coasts. In our study areas, reedbeds have more species if they are on drier or shallower soils (negative end of the second DCA axis) or have not yet fully developed. In such cases, the dominance of *Phragmites* is not as strong and several other competitive species with high growth rates dominate (e.g. *Elymus repens*, *Filipendula ulmaria*, *Lythrum salicaria* and *Angelica sylvestris*). The latter sites can be initially quite species-rich (see also Jutila 1997; Wanner 2009), but it is not certain whether their diversity can persist when the reedbed continues to develop. High small-scale species density (number of species per unit area) is not typical for coastal meadows and the richness of plant species alone is not usually the goal of conservation. In Estonia, these habitats are restored for protection of low-stature vegetation, suitable for small plants and specific groups of birds and amphibians. Surprisingly, even though the vegetation of restored sites seemed very different from the vegetation of continuously managed sites during the site selection process, and the distinction between these two groups in the DCA ordination is fairly clear, the typical coastal grassland species were relatively common (in terms of occurrence) in restored areas (Fig. 3). Thus, just 5 yr of restoration has been sufficient for developing early similarities in vegetation of restored and continuously managed coastal grasslands. We must point out, however, that this early success is fragile. First, the relatively common occurrence of typical coastal grassland species does not imply that they are also abundant. Second, *Phragmites australis* is still present in 52% of relevés of the restored sites (Appendix S1), and when present covers on average 10% of the relevé. Thus, whenever there is a drop in the grazing intensity, *Phragmites* and other highly competitive species that are still commonly present (e.g. *Deschampsia cespitosa* and *Filipendula ulmaria*) will flourish and the small species may be rapidly out-competed (pers. obs., see also Bakker 1989; Bakker et al. 1997). Therefore, it is essential to maintain efficient management of restored sites when evaluating the success of restoration, and to pay attention not only to the presence of favourable target species but also to the presence of species that are responsible for degradation of the habitat and the factors that benefit abundance of unfavourable species, such as accumulation of biomass and nutrients in soil.

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Supporting information

Additional supporting information may be found in the online version of this article:

Appendix S1. Relative occurrence of species in relevés of differently managed sites. Species with relative occurrence value at least 0.05 in at least one of the management groups are presented in the table. Species are ordered in a decreasing order of occurrence. Number of relevés per each group: managed sites – 120; restored sites – 60; abandoned sites – 100. *Denotes species which could be used as indicators of restoration success (typical species of Boreal Baltic coastal grasslands).

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The Effect of Cut Plant Residues Management and Fertilization on the Dry Matter Yield of Swards and on Carbon Content of Soil

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Abstract
The goal of this research was to study the impact of cut plant residues returned to or removed from the grassland sward, on the dry matter yield of swards and on the organic carbon (Corg) concentration of soil. The experiment was carried out during 2004-2008. The variables of the experiment were: (i) sward type: turfgrass sward (*Festuca rubra rubra* and *Poa pratensis*) and grass-clover sward (*Phleum pratense*, *Lolium perenne* and *Trifolium repens*); (ii) treatment of residues: the cut plant residues were returned (RRT) to the plots or removed (RRM) from the plots after the mowing. The fertilizer treatments were as follows: $N_0P_0K_0$, $N_{80}P_{11}K_{48}$, $N_{160}P_{22}K_{96}$ and $N_{400}P_{56}K_{240}$ kg ha^{-1} for the turfgrass sward and $N_0P_0K_0$ and $N_{80}P_{26}K_{50}$ kg ha^{-1} for the grass-clover sward. Corg and Ntot concentrations in the 20 cm soil layer were measured at the beginning and at the end of the experiment at depths of 0-5 cm and 5-20 cm.
Nitrogen was returned as plant residues to the grass-clover sward in treatment $N_0P_0K_0$ at 190 kg ha^{-1} and $N_{80}P_{26}K_{50}$ at 204 kg ha^{-1} and consequently the returned cut plant residues increased the yield by 31% and 22%, respectively. The amount of N returned as residues to turfgrass sward was 31-236 kg ha^{-1} but it had no significant influence on the sward dry matter yield.
During the five years of the experiment the Corg content in 0-5 cm soil layer of grass-clover sward in treatment RRT increased by 42.9% and in RRM by 32.0% as an average of both fertilization treatments. At the depth 5-20 cm the Corg concentration did not change in treatment RRT but in treatment RRM with fertilization, the Corg concentration decreased by 8.2%. In turfgrass soil the Corg concentration increased in RRT treatment by 21.6% and in treatment RRM by 7.2% during 5 years. In the lower soil layer the concentration of Corg decreased with removal and returning of plant residues. The fertilization did not influence the changes of Corg concentration in turfgrass swards soil.

Keywords turfgrass, grasses, white clover, sward dry matter yield, soil organic carbon stock, nitrogen

Introduction

Land use change can lead to changes in soil properties, including the soil carbon (Houghton, 1999) and nitrogen (Potter et al., 1996) cycles. The loss of soil organic carbon (Corg) by conversion of natural vegetation to cultivated use is well known (Schlesinger, 1985; Mann, 1986; Post and Mann, 1990; Davidson and Ackerman, 1993). Much of this loss in soil Corg

can be attributed to reduced inputs of organic matter, increased decomposability of crop residues, and tillage effects that decrease the amount of physical protection to decomposition. The reverse process of turning open field sites into grassland is an opportunity to increase carbon sequestration in the soil (Lal et al., 1999). The higher rates of organic carbon in grasslands soils compared with arable systems is explained partly by the greater supply of C to the soil under grassland (Jackson et al., 1996) and partly by the increased residence time of C resulting from the absence of disturbance by tilling. Carbon from plants enters the soil organic carbon pool in the form of either above-ground phytomass, litter or root material. The predictions of the magnitude of the increase in soil organic carbon are associated with great uncertainty (Vleeshouwers and Verhagen, 2002; Freibauer et al., 2004). Based on Janssens et al. (2005) the average change of C stock estimated for European grasslands is 0.6 t ha^{-1} y^{-1}. According to IPCC (2001) the value is about 0.8 Mg C ha^{-1} $year^{-1}$ during a 50 year period.

Recent experimental evidence demonstrates that the type and diversity of plant species in grasslands plays an important role for carbon transfer into the soil and is able to modify carbon stock under a given land use scheme (Tilman et al., 2006; Steinbeiss et al., 2008;). Also the grasslands' ability to sequester the carbon is influenced by plant management e.g. cutting regimes and fertilization (Johnston et al., 1994; Conant et al., 2001; Eriksen and Jensen, 2001). When used for cutting, most of the herbage is exported from the grasslands and carbon stock is increased by root biomass. An alternative option, when grasslands are used as set-aside land, the mown material will be left on the field to decompose. The decomposition of returned material depends on the chemical composition of plants (Gunnarsson and Marstorp, 2002), that influences the dry matter yield of sward, soil Ntot and Corg concentrations. There is lack of research concerning the decomposition of cut plant residues and its impact on grassland productivity, soil Ntot and Corg concentrations. The impact on plant productivity has only been studied with turfgrass swards which have resulted in fast decomposition on the plant surface (Starr and DeRoo, 1981; Kopp and Guillard, 2002; Qian et al., 2003; Kauer et al., 2008). The mowing frequency of natural, semi-natural and conservation grasslands is lower than in turfgrass swards. Therefore, the material left on the plant surface is significantly further in its growth stages and has larger biodiversity.

Our hypothesis was that different plant species have various impacts on the soil Corg concentration and consequently the carbon stock. The soil Corg concentration also varies if the plant residues after mowing are left to decompose on the field.

The objective of this work was to study the impact of cut plant residues of turfgrass and grass-white clover and fertilization on the dry matter yield of swards and soil Corg concentration.

Material and Methods

Background of experimental site

The field experiment was carried out at the Experimental Station Eerika of the Estonian University of Life Sciences (58°23'32" N latitude, 26°41'31" E longitude; elevation 60 m above sea level). The soil of the experimental field was a *Stagnic Luvisol* according to the WRB classification (FAO 2006). The field was previously under barley for three years. In autumn 2002 the area was ploughed. Soil samples were collected at depths of 0-20 cm in May 2003 before sowing. The soil Ntot and Corg concentration at depth of 0-20 cm was 1.49 g N kg^{-1} and 14.7 g Corg kg^{-1}, pH 1M KCl-solution was 5.5, plant available P content was 39.6 and K content 79.7 mg kg^{-1}. The contents of plant available elements (P and K) in the soil were determined by the AL-method (Egner et al., 1960).

In the spring of 2003 the site was cultivated and sown with two different seed mixtures: (i) a turfgrass mixture (*Festuca rubra rubra* 50% and *Poa pratensis* 50%) (hereafter turfgrass

sward); (ii) a grass-clover mixture (*Phleum pratense* 34%, *Lolium perenne* 38% and *Trifolium repens* 28%*)* (hereafter grass-clover sward). The percentages indicate the proportion of seeds of a given species in the seed mixture. The swards were unfertilized during the period between the sward establishment in 2003 and May 2004.

Fertilization of swards

The fertilizer treatments for turfgrass sward were as follows: $N_0P_0K_0$ (hereafter TGN0) as control, $N_{80}P_{11}K_{48}$ (TGN80), $N_{160}P_{22}K_{96}$ (TGN160) and $N_{400}P_{56}K_{240}$ (TGN400) kg ha^{-1}. The N as $(NH_4)NO_3$ and K as KCl fertilizer was applied by hand to the plots in 2 to 4 splits depending on the ratio of the N, P and K during the vegetation period. The P fertilizer as $Ca(H_2PO_4)_2$ was applied to the plots in 1 split at the beginning of May.

The experimental treatments for the grass-clover sward were divided based on fertilizer application as follows: $N_0P_0K_0$ (GCN0) as control and $N_{80}P_{26}K_{50}$ (GCN80). N fertilizer rates were applied in July after the second cutting. P and K fertilizers were applied in spring, at the end of April just after beginning of the growing season.

The experimental design was a randomized complete block with four replicates of each of four fertilization treatments, with a plot size of 1 x 7 m for each treatment plot.

Mowing of swards

The turfgrass sward was mown with a rotary lawn mower at 5 cm height when the height of grass was approximately 7-8 cm. The mowing frequency was 13-15 times per growing season and it depended on the rates of grass growth. The grass-clover sward was cut during the growing season 4-5 times with the sickle-bar mower at 4 cm. The first time the grass-clover sward cut was when the grasses were at the end of tillering and leaf tubes were beginning to form.

After mowing the cut plant residues were either returned to the plots (hereafter RRT) or were removed from them (hereafter RRM). In the RRT treatment, plots of turfgrass sward, immediately after weighing the fresh turfgrass clippings were returned and spread evenly over the plots from which they had been removed: Subsequently the whole plot was re-mowed ("mulched").

Dry matter yield of swards

The dry matter yield of swards was measured in 2004-2007. The material mown in each plot was weighed after every mowing. After the cutting and weighing procedure the plant sample was collected for the dry matter (DM) measurement. The dry matter content was determined by drying the sample in a forced-draught oven for 6 hours at 105°C.

N content in the plants and N uptake by the plant

In 2007 the N concentration of plant samples from the mown material, in RRT and RRM treatments with different fertilization regimes, was measured. The uptake of N by plants was calculated as follows:

Uptake of N (kg ha^{-1}) = (DM yield (kg ha^{-1}) * N content in residues (mg g^{-1}))/1000.

Soil and thatch sampling

In September 2008 soil samples from each plot at depths of 0-5 and 5-20 cm were collected. Soil was air dried and passed through a 2-mm sieve before analysis. In autumn of 2008 when the experiment ended, the thickness of thatch layer of the turfgrass was measured. For this purpose, ten samples (soil drill diameter 3.5 cm) were taken from each plot. The thickness of the thatch layer was measured with a plastic ruler.

Plant and soil analysis

The Ntot concentration in soil and plants was analyzed by a dry combustion method in a varioMAX CNS elemental analyzer (ELEMENTAR, Germany). Organic carbon in the soil was determined by wet oxidation with dichromate according to the method of Walkley and Black (1934) and modified by Tyurin (1951).

The stock of soil organic carbon

The soil Corg stock (t ha^{-1}) was calculated for two depths (0-5 and 5-20 cm) as follows:

$$\text{Corg stock} = BD \times Corg \times D / 10$$

where Corg is organic carbon content (mg g^{-1}); BD is soil bulk density (g cm^{-3}), and D is soil sampling depth (cm).

We estimated the bulk density (BD) using the Adams (1973) equation:

$$BD = 100 / ((OM / 10 / 0.244) + ((100 - (OM / 10)) / 1.64)),$$

where OM is the organic matter content of soil (mg g^{-1}): we assumed that OM contains 58% Corg (Mann, 1986).

Statistical analysis

Statistical analyses were carried out using the software program "STATISTICA" version 7.0 (StatSoft Inc.). Factorial ANOVA and one-way ANOVA were applied to test the effect of different management regimes (removal of plant residues or returning the residues, fertilization) on the dry matter yield of swards, soil Ntot and Corg concentration, Corg stock and Corg change per year. Fisher's LSD test for homogeneous groups was used for testing the significance of differences between treatments. The level of statistical significance was set at $P < 0.05$.

Results

Effect of plant residues on the dry matter yield of swards

Taking the average of 2004-2007 returning plant residues increased significantly only the dry matter yield of the grass-clover sward ($F(1, 254) = 39.983$, $P = 0.0000$). The effect was largest in the unfertilized treatment where the dry matter yield was increased by 30.8% (Table 1). With fertilization (GCN80) the value was 22.4%. Due to the fertilization effect only (RRM treatment) the yield was increased by 18%.

The clippings yield of turfgrass was not influenced by returning the clippings to the sward ($F(1, 2046) = 2.9294$, $P = 0.0871$). The only exception was treatment TGN160, where the yield was increased due to returning of grass clippings ($P < 0.05$). The yield of turfgrass sward was mainly influenced by fertilization ($F(3, 2044) = 269.67$, $P = 0.0000$).

The effect of plant residues on the plant N content and soil Ntot and Corg concentration

Nitrogen uptake when plant residues were returned ranged from 31-236 kg N ha^{-1} in the turfgrass sward and 190-204 kg N ha^{-1} in the grass-clover sward (Table 2). Returning the residues did not increase the Ntot content in plants occupying the turfgrass sward compared to removing. Ntot content was however affected by fertilization in turfgrass sward. In the grass-clover sward a significant effect of plant residues on the N content in plant was observed when considering the fertilized treatment. Returning the plant residues did not have a statistically significant effect on the N uptake by plants, also in the GCN80 treatment mentioned previously.

Considering the average of all treatments, the Ntot and Corg concentrations in the soil of grass-clover sward at depth of 0-5 cm increased over the five years of the study (Table 3).

The Corg concentration increased with returning of plant residues by 42.9% and with removal of plant residues by 32.0%. TotN concentrations increased in unfertilized treatment RRM by 23.5% and in RRT 40.3%. Fertilization further increased Ntot contents by 31.5% and 49.0%, respectively. The Ntot and Corg concentration in the lower soil layer (5-20 cm) of unfertilized treatments was not influenced by returning residues, but in fertilized treatments the Ntot and Corg concentrations were higher in the RRT treatment than in RRM one. In the turfgrass soil upper layer (0-5 cm) the Corg content was increased in RRM (by 21.6%) as well as in RRT (by 7.2%). The Ntot content at 0-5 cm did not change significantly when grass clippings were returned but decreased when they were removed (15.7%, $P < 0.05$). In the lower layer of turfgrass (5-20 cm) the Ntot content decreased in all treatments. Fertilization did not influence the Ntot or Corg content of turfgrass soil.
The C:N ratio of turfgrass soil was higher compared to grass-clover soil. The ratio did not vary significantly during the campaign (5 years) (Table 3).

The effect of plant residues on soil Corg stock of swards
Over the five years of the experiment, similarly to the Corg content, the Corg stock was increased more in the soil of grass-clover sward, where returning plant residues increased the value at 0-5 cm by 0.76 t ha^{-1} y^{-1} (Table 4). The increase was 0.58 t ha^{-1} y^{-1} when residues were removed. In the top soil (0-5 cm) of the turfgrass sward the Corg stock was increased by 0.33-0.45 t ha^{-1} y^{-1} when residues were returned and by 0.10-0.15 t ha^{-1} y^{-1} when they were removed. In the lower soil layer (5-20 cm) a decrease in Corg stock occurred. It was decreased by 0.20-0.44 t $ha^{-1}y^{-1}$ in grass-clover and by 0.74-0.96 t ha^{-1} in turfgrass sward (when grass clippings were removed). The reduction of Corg was lower when residues were returned.

Thatch layer formation
The thickness of the resulting thatch layer of the turfgrass sward was significantly higher in treatments in which grass clippings were returned and the fertilization rate was 160-400 kg N ha^{-1} ($F(3, 306) = 17.350$, $P = 0.0000$) (Figure 1). The thickness of the thatch layer in the TG400 treatment was 3.2 cm when residues were returned, but 2.5 cm when they were removed. There was no thatch layer in grass-clover sward.

Discussion
Our results indicate that the amount of N returned with plant residues had a significantly lower effect on the dry matter yield of turfgrass sward than expected from previous studies (Starr and DeRoo, 1981; Heckman et al., 2000; Kopp and Guillard, 2002). This may be due to the limited amount of N in the plant residues available for the growing plants. This is likely since, when returning plant residues, the amount of mineralized N (measured as N content in plants) did not change significantly during the experiment. In addition, the N contents of turfgrass swards in RRT as well as in RRM treatment were similar. If the amount of N added with returned residues had influenced the amount of mineralized N in the soil, the N content in RRT treatment would have been higher than in RRM treatments. The soil enrichment by mineralized N from turfgrass clippings may have been hindered by the thatch layer formed on the surface of turfgrass sward. Thatch accumulation occurs when turfgrass production of organic matter exceeds the decomposition rate (Beard, 1973). Turfgrass sward was mowed on average once a week. According to studies by Kauer et al. (2007) and Kopp and Guillard (2004) the decomposition rates of turfgrass residues in different fertilization treatments were the same and therefore the thatch layer was significantly thicker wherever large amounts of plant residues were returned. A thicker thatch layer could have reduced the contact surface

between turfgrass clippings and soil. The decomposition occurred mainly on the surface of the thatch layer on top of the herbage. The decomposition rate of plant residues could have been decreased due to their drying as was seen in a previous study (Kauer et al., 2011). Thus, the transfer of N into soil was hindered and a part of the N in plant residues could have been used for decomposition of the residues. Furthermore, part of the nitrogen, perhaps favoured by mulching the turfgrass clippings by lawnmower, may have been volatilized and lost to the atmosphere. It has been found that the amount of volatilized nitrogen could form 10-20% of total mineralized N (Janzen and McGinn, 1991). According to Whitehead et al. (1988) NH_3 volatilization may amount to 20-47% of herbage N in laboratory experiments. Overall the N content of turfgrass plant residues was high in our study. If we assume that plants contain 400 g C kg^{-1}, the C:N ratio of returned plant residues would be 9-13, depending on the fertilization rate. The optimum C:N ratio for rapid decomposition of organic matter is between 15:1 and 25:1 (McLeod, 1982). The lower than optimal C:N ratio indicates that there is a surplus of nitrogen for decomposers and nitrogen in plants may be easily decomposable (Ross et al., 2002). Consequently, a higher amount of nitrogen due to fertilization does not increase soil Ntot content, as the nitrogen will be used for decomposition or volatilized. In the turfgrass treatment more nitrogen was removed with residues than was provided with fertilizers, but this positive balance between input and output did not increase the soil Ntot content. The reason may be that, during decomposition of the thatch layer formed in the RRM treatment, an N deficit could occur and nitrogen become, immobilized from the soil decreasing the soil Ntot content. The higher C:N ratio in turfgrass sward also indicates reduced N mineralization in the soil. In grass-clover sward no thatch layer was formed. One of the explanations could be that the mowing frequency was less and the surface contact between returned residues and soil was better compared to the turfgrass sward. The C:N ratio (assuming that plants contain 400 mg C g^{-1}) in returned grass-clover residues was 15-16. This may be connected to the better decomposition conditions for plant residues and its impact on the nutrient cycle; hence the grass-clover plant residues had a significant impact on the sward dry matter yield. Better N mineralization conditions are also confirmed by the lower soil C:N ratio, compared to the turfgrass sward. The lower C:N ratio reflects the higher rate of N mineralization (Elgersma and Hassink, 1997).

Consequently, the soil C org content and stock was mostly influenced by the composition of the species of the sward. The composition of species in the sward had more impact because the below-ground carbon input of different species shows significant variation. Wedin et al. (1995) showed 25-fold differences in below-ground net primary productivity, whereas above-ground productivity only varied twofold. Fornara and Tilman (2008) found during their study that the presence of legumes significantly increased root biomass production compared to the grasses sward. While in the grass-clover sward the Corg and stock increased more, it can be assumed that the root biomass was higher than in turfgrass sward. The soil Corg content of both swards increased mainly in the top layer and decreased significantly in the lower layer. This indicates that the impact of different plant species and plant residues does not reach the deeper soil profiles. As 70 to 75 % of the root biomass in grasslands is located in the top 15 cm of the soil (Gill et al., 1999) organic carbon and nitrogen concentrations increase in the main rooting zone. According to Steinbeiss et al. (2008) the carbon concentration in deeper layers is reduced because the carbon below root zone will be leached to deeper layers. They conclude that plant derived carbon is preferentially mineralized and adsorbed to soil particles, while mobilized soil carbon is transported further down the soil profile.

The stronger effect of plant species in grass-clover sward (compared to turfgrass sward) on the soil Corg and Ntot content is caused by properties of white clover. Legume derived N is qualitatively important for building up soil organic matter and storing more C (Drinkwater et

al., 1998; Resh et al., 2002). Addition of legumes improves soil nutrient cycling which enhances the retention of newly and residing C and N in soil (De Deyn et al., 2011).
De Deyn et al (2011) point out in their research that changes in quality (e.g. C:N ratio) rather than quantity of plant community with legumes (Rochon et al., 2004) were primary drivers of rapid increase in soil C accumulation rates. Consequently in our experiment, the effect of the mixture of plant species on the soil Corg concentration is higher in grass-clover sward. In grass-clover sward the amount of nitrogen removed with plants exceeded the amount of nitrogen provided with fertilizers. Nevertheless, the soil Ntot content increased and therefore the nitrogen concentration increased mostly through clover's ability to bind nitrogen from the atmosphere. A legume that can derive N from symbiotic fixation would be expected to have a concomitant positive effect on both C and N added to the soil (Wu et al., 2006).
By returning the plant residues the Corg content and stock in the soil top layer was increased, compared to the treatment in which residues were removed. The effect of returning the residues on the soil carbon stock was smaller in grass-clover sward. By returning the residues in grass-clover sward the Corg in the soil top layer was increased by only 0.18 t $ha^{-1}y^{-1}$ (average of fertilization regimes) more than when residues were removed. In the lower layer (5-20 cm) the effect was larger (0.30 t ha^{-1} y^{-1}) and it occurred because in the lower soil layer of the RRM fertilized treatment the Corg content decreased, compared to unfertilized treatment. The decrease of Corg concentration may have caused reduction of the root biomass in the soil by fertilization (Ennik et al., 1980). The effect of treatment of plant residues is higher in turfgrass sward than in grass-clover sward. As a result of returning plant residues the difference in Corg as an average of fertilization regimes was 0.26 t $ha^{-1}y^{-1}$ in the upper layer of the turfgrass sward (in the lower layer 0.37 t $ha^{-1}y^{-1}$), compared to the treatment where residues were removed. If we sum Corg stock increase of both layers, the effect of turfgrass sward plant residues was slightly higher compared to the grass-clover sward (0.62 t $ha^{-1}y^{-1}$ and 0.48 t $ha^{-1}y^{-1}$, respectively), although more phytomass was returned to the grass-clover sward (on average 7.8 t DM ha^{-1} y^{-1}) compared to the turfgrass sward (1.3-5.9 t DM ha^{-1} y^{-1}). The effect of plant residues of the grass-clover sward could have been smaller because the biochemical composition of plant residues which became part of the soil organic matter was different. The thatch layer formed on the turfgrass sward surface remains a factor as discussed earlier. Based on the other studies, the thatch layer is recalcitrant to decompose because the high lignin content (Yao et al., 2009). Lignin degradation products contribute to an increase in Corg content in the soil (Takeda, 1998). The plant residues of grass-clover sward had better decomposition conditions and the thatch layer on surface did not form. These residues became part of soil organic matter and were of better quality than the thatch layer of turfgrass sward. High quality plant residues (high N, low lignin concentrations) mineralize rapidly, but may not contribute much to the maintenance of soil organic matter (Handayanto et al., 1997). Thus, the larger amount of recalcitrant material on turfgrass sward became part of soil organic matter compared to the grass-clover sward and the increase of Corg content was higher in turfgrass sward soil compared to grass-clover sward.
It is often stated that the productivity of grassland and Corg content in soil are positively linked (Karlen and Cambardella, 1996), hence increasing the sward productivity may be one of the main reasons to upgrade soil C sequestration. Among obvious benefits for phytomass production, fertilization has been proposed as a technique for enhancing soil C storage (Rasmussen and Rohde, 1988; Conant et al., 2001; Fornara and Tilman, 2008). At the same time, several authors have concluded that, by adding large amount of phytomass into the soil, the C concentration will not be increased (Gill et al., 2002; Fontaine et al., 2004a; De Deyn et al., 2011), because Corg is more influenced by the quality of the material returned to the soil (De Deyn et al., 2011). In our experiment the Corg content was not affected by returning

either different (turfgrass sward) nor the same amount (grass-clover sward) of phytomass to the soil. The Corg content of turfgrass sward at a soil depth of 0-5 cm increased equally in every fertilization treatment as well as in the control. According to Steinbeiss et al. (2008), the higher the input, the more soil organic material was decomposed. However, the input did not increase the soil organic matter content. They concluded that the litter input may have induced a priming of microbial decomposition in soil that resulted in faster turnover rates and a mobilization of organic carbon already present in the soil, as found in other investigations (Fontaine et al., 2004b). As a result of the 'priming effect' the increased C input into the soil may cause higher decomposition of soil organic matter resulting in no differences on C org content due to the varying amounts of residues returned. Soil organic matter contains 95% of the total nitrogen in the soil (Stevenson, 1994). Therefore, the soil Corg and N are closely related and hence the change in Corg content would cause a similar change in nitrogen content. Piovanelli et al. (2006) found that soil carbon and nitrogen concentrations are highly correlated under different tillage systems. Our results on C:N ratios also showed rather small variations between different fertilization regimes. Further, different treatments of plant residues did not have a significant effect on the soil C:N ratio.

Conclusions

The change of carbon and N content in the grassland soil depends to a great extent on the species of the sward, their properties and the treatment they undergo. The carbon stock increases more in soil of grass-clover than in soil of turfgrass sward. Leaving the plant residues on the grassland after mowing increases the soil Corg stock, mostly in the soil upper layer. The effect of plant residues on the C stock is higher in turfgrass sward.

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Table 1. The effect of returning plant residues on the dry matter (DM) yield (kg ha^{-1}) of swards during 2004-2007.

Fertilization rate, kg N ha^{-1}	RRT[1]	RRM[2]
Turfgrass sward		
TGN0	1330[a3A4]	1496[aA]
TGN80	2780[aB]	2581[aB]
TGN160	3888[bC]	3290[aC]
TGN400	5924[aD]	5647[aD]
Grass-clover sward		
GCN0	7465[bA]	5168[aA]
GCN80	8162[bA]	6336[aB]

[1]RRT – plant residues were returned to the plots.
[2]RRM – plant residues were removed to the plots.
[3]Different small letters within each row indicate a significant effect ($P < 0.05$) of returning plant residues on the DM yield of sward within a fertilization regime.
[4]Different capital letters within column indicate a significant effect ($P < 0.05$) of fertilization on the DM yield of sward.

Table 2. The dry matter (DM) yield of swards, Ntot concentration in the plants and the N uptake by plants in 2007.

Fertilization rate, kg N ha^{-1}	RRT[1]	RRM[2]	RRT	RRM	RRT	RRM
	DM yield of sward, kg ha^{-1}		Ntot content in plant, mg g^{-1}		N uptake by plant, kg N ha^{-1}	
Turfgrass sward						
TGN0	990^{a3A4}	1140^{aA}	31^{aA}	30^{aA}	31^{aA}	34^{aA}
TGN80	2510^{aB}	2320^{aB}	34^{aB}	34^{aB}	85^{aB}	79^{aB}
TGN160	3540^{bC}	2900^{aC}	38^{aC}	37^{aC}	135^{bC}	99^{aC}
TGN400	5480^{aD}	5150^{aD}	43^{aD}	43^{aD}	236^{aD}	221^{aD}
Grass-clover sward						
GCN0	7787^{bA}	4469^{aA}	24^{aA}	24^{aA}	190^{bA}	107^{aA}
GCN80	7723^{bA}	6210^{aB}	26^{bB}	24^{aA}	204^{bA}	149^{aB}

[1]RRT – plant residues were returned to the plots.
[2]RRM – plant residues were removed to the plots.
[3]Different small letters within each row indicate a significant effect ($P < 0.05$) of returning plant residues on the DM yield of swards, Ntot concentration in plants and N uptake by plants.
[4]Different capital letters within column indicate a significant effect ($P < 0.05$) of fertilization on the DM yield of swards, Ntot content in plants and N uptake by plants.

Table 3. Dependence of the soil Ntot and Corg concentrations and C:N ratio in the soil of turfgrass and grass-clover sward on plant residue management and fertilization.

Fertilization rate, kg N ha^{-1}	Arable[1]	RRT[2]	RRM[3]	Arable	RRT	RRM	Arable	RRT	RRM
	Ntot, mg g^{-1}			Corg, mg g^{-1}			C:N		
Turfgrass 0-5 cm									
TGN0	1.49^{b4A5}	1.52^{bA}	1.08^{aA}	14.7^{a1A2}	18.0^{cA}	15.9^{bA}	9.9^{aA}	11.9^{bA}	14.7^{cB}
TGN80	1.49^{bA}	1.54^{bA}	1.27^{aB}	14.7^{aA}	17.6^{cA}	15.7^{bA}	9.9^{aA}	10.8^{bA}	12.0^{cA}
TGN160	1.49^{bA}	1.58^{cA}	1.39^{aB}	14.7^{aA}	17.4^{cA}	15.8^{bA}	9.9^{aA}	11.0^{bA}	11.3^{bA}
TGN400	1.49^{bA}	1.56^{cA}	1.28^{aB}	14.7^{aA}	18.4^{cA}	15.5^{bA}	9.9^{aA}	11.8^{bA}	12.1^{bA}
Grass-clover 0-5 cm									
GCN0	1.49^{aA}	2.09^{cA}	1.84^{bA}	14.7^{aA}	20.8^{cA}	19.3^{bA}	9.9^{aA}	10.0^{aB}	10.5^{bB}
GCN80	1.49^{aA}	2.22^{cB}	1.96^{bB}	14.7^{aA}	21.2^{cA}	19.5^{bA}	9.9^{bA}	9.5^{aA}	9.9^{bA}
Turfgrass 5-20 cm									
TGN0	1.49^{bA}	1.18^{aA}	1.16^{aA}	14.7^{cA}	13.4^{bA}	12.2^{aA}	9.9^{aA}	11.4^{cB}	10.5^{bB}
TGN80	1.49^{bA}	1.17^{aA}	1.18^{aA}	14.7^{cA}	13.2^{bA}	12.5^{aA}	9.9^{aA}	11.3^{cB}	10.6^{bB}
TGN160	1.49^{bA}	1.22^{aA}	1.10^{aA}	14.7^{cA}	13.6^{bA}	12.3^{aA}	9.9^{aA}	11.2^{bB}	11.2^{bB}
TGN400	1.49^{bA}	1.23^{aA}	1.17^{aA}	14.7^{cA}	13.1^{bA}	12.7^{aA}	9.9^{aA}	10.7^{bB}	10.9^{bB}
Grass-clover 5-20 cm									
GCN0	1.49^{aA}	1.49^{aA}	1.48^{aA}	14.7^{aA}	14.5^{aB}	14.1^{aB}	9.8^{bA}	9.7^{bA}	9.5^{aA}
GCN80	1.49^{aA}	1.55^{bA}	1.42^{aA}	14.7^{bA}	14.7^{bB}	13.5^{aC}	9.8^{bA}	9.5^{aA}	9.5^{aA}

[1]Arable – arable land before the conversion to grassland in 2003.
[2]RRT – plant residues were returned to the plots.
[3]RRM – plant residues were removed to the plots.
[4]Different small letters within each row indicate significant influence ($P < 0.05$) of returning plant residues on soil Ntot and Corg concentrations and C:N ratio at the given soil depth.
[5]Different capital letters within each column indicate significant influence ($P < 0.05$) of fertilization on soil Ntot and Corg concentrations and C:N ratio at the given soil depth.

Table 4. Soil Corg stock and its changes in the soil of turfgrass and grass-clover sward, as a function of plant residues management and fertilization.

Fertilization rate, kg N ha^{-1}	Arable[1]	RRT[2]	RRM[3]	RRT	RRM
	Corg stock, t ha^{-1}			Change of Corg stock, t ha^{-1} y^{-1}	
Turfgrass 0-5 cm					
TGN0	10.5^{a4A5}	12.6^{cA}	11.3^{bA}	0.41^{bA}	0.15^{aA}
TGN80	10.5^{aA}	12.3^{cA}	11.1^{bA}	0.35^{bA}	0.13^{aA}
TGN160	10.5^{aA}	12.2^{cA}	11.2^{bA}	0.33^{bA}	0.14^{aA}
TGN400	10.5^{aA}	12.8^{cA}	11.0^{bA}	0.45^{bA}	0.10^{aA}
Grass-clover 0-5 cm					
GCN0	10.5^{aA}	14.1^{cB}	13.3^{bB}	0.73^{bB}	0.56^{aB}
GCN80	10.5^{aA}	14.4^{cB}	13.5^{bB}	0.78^{bB}	0.59^{aB}
Turfgrass 5-20 cm					
TGN0	31.5^{cA}	29.1^{bA}	26.7^{aA}	-0.48^{aA}	-0.96^{bC}
TGN80	31.5^{cA}	28.8^{bA}	27.3^{aA}	-0.54^{aA}	-0.84^{bC}
TGN160	31.5^{cA}	29.5^{bA}	27.0^{aA}	-0.40^{aA}	-0.90^{bC}
TGN400	31.5^{cA}	28.6^{bA}	27.8^{aA}	-0.59^{aA}	-0.74^{bC}
Grass-clover 5-20 cm					
GCN0	31.5^{bA}	31.2^{bB}	30.5^{aB}	-0.51^{aB}	-0.20^{bA}
GCN80	31.5^{bA}	31.5^{bB}	29.3^{aB}	0.13^{aB}	-0.44^{bB}

[1]Arable – arable land before the conversion to grassland in 2003.
[2]RRT – plant residues were returned to the plots.
[3]RRM – plant residues were removed to the plots.
[4]Small letters within each row indicate a significant influence ($P < 0.05$) of returning plant residues on soil Corg content and change of Corg stock in soil at given depth.
[5]Capital letters within each column indicate a significant influence ($P < 0.05$) of fertilization on soil Corg content and change of Corg stock in soil at given depth.

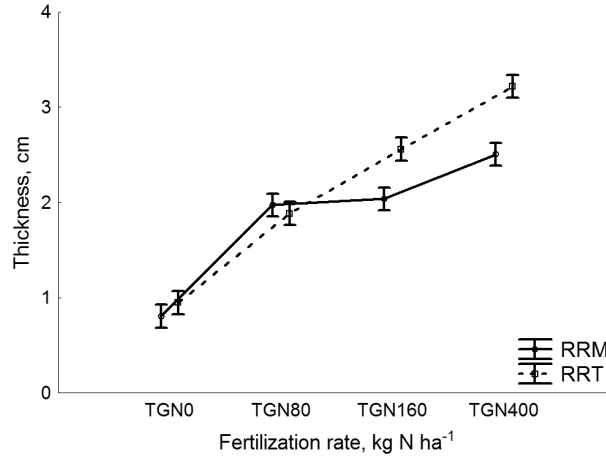

Figure 1. The thickness of thatch layer of turfgrass sward with different fertilization regimes in RRM and RRT treatments in 2008 at the end of the experiment. Vertical bars denote 0.95 confidence intervals.

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Impact of Returned Clippings on Turfgrass Growth as Affected by Nitrogen Fertilizer Rate, Time of Return, and Weather Conditions

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Abbreviations: ANOVA, analysis of variance; C, carbon; CRM, clippings removed; CRT, clippings returned; DM, dry matter; DMY, dry matter yield; $DMY_{CRT,}$ amount of returned clippings, or dry matter yield of treatment where clippings are returned; K, potassium; LSD, least significant difference; N, nitrogen; NREC, apparent N recovery; $NREC_{CL}$, apparent N recovery of clippings; $NREP_{CL,}$ N amount replaced by clippings; NS, not significant; NUE, N-use efficiency; NUE_{CL}, N-use efficiency applied with clippings; NUP, total N uptake; P, phosphorous; RCLE, returned clippings efficiency; RCLI, returned clippings impact, or effect of returned clippings; SOC, soil organic carbon.

ABSTRACT

The impact of returned clippings (RCLI) on mown turf, in terms of the value of clippings as a nitrogen (N) source as well as management interactions on grass growth, has not been well examined. The objectives of the present research were to study: (i) the influence of returning grass clippings (CRT) on turfgrass growth, N-use efficiency (NUE), and apparent N recovery (NREC); and (ii) the impact of N rate and weather during the growing season on RCLI. Over a five-year period, a field experiment was carried out on a sward of *Festuca rubra rubra* and *Poa pratensis*. Measured experimental factors included two clippings treatments, with clippings removed (CRM) or CRT, as well as seven fertilizer rates (kg ha^{-1}): $N_0P_0K_0$, $N_0P_{22}K_{96}$, $N_{80}P_{11}K_{48}$, $N_{160}P_{22}K_{96}$, $N_{240}P_{34}K_{144}$, $N_{320}P_{45}K_{192}$, and $N_{400}P_{56}K_{240}$. RCLI was significant for turfgrass growth only on treatments $N_{160}P_{22}K_{96}$ and $N_{240}P_{34}K_{144}$; NUE and NREC also were higher with CRT treatment than with CRM treatment. The annual RCLI depended mainly on the amount of returned clippings (DMY_{CRT}) in May and June, as well as on precipitation levels during May. Furthermore, effect of returning grass clippings on subsequent turfgrass growth depended on the N rate, time of return, and the amount of precipitation in May.

Lawn mowing is one of the most important lawn management methods. In Estonia, mowing takes place on average once every 7–10 days, which amounts to 15–20 mowings over the growing season. Mulching lawn mowers often are used at homes and parks and leave behind finely shredded grass clippings. It is well known that exposed turfgrass clippings decompose rapidly and release nitrogen (N) (Kopp and Guillard, 2004, 2009; Shi et al., 2006b; Kauer et al., 2007). Less information is available for how much N is recycled by plants, as well as factors that influence the efficiency of returned clippings (RCLE) as an N source. Few studies have examined the impact of returned clippings (RCLI) on turfgrass growth, and the results reported have been variable.

In a three-year trial, where RCLI was studied on sward fertilized with either 195 kg N ha^{-1} (years 1 and 2) or 180 kg N ha^{-1} (year 3), Starr and DeRoo (1981) found that although returned clippings (CRT) increased the yield 15–55%, the effect on dry matter yield (DMY) was different in all years. Heckman et al. (2000) demonstrated that the amount of applied N could be cut in half (from 195 kg N ha^{-1} y^{-1} to 98 kg N ha^{-1} y^{-1}) when clippings were returned. Kopp and Guillard (2002) studied RCLI at two different experimental stations and on two different soil types; they found that the DMY, total N uptake (NUP), apparent N recovery (NREC), and N-use efficiency (NUE) were markedly higher as clippings were returned, but with dissimilar results between the two test sites, with the increase in DMY varying between 79% and 254% depending on the site. Qian et al. (2003) used the CENTURY ecosystem model to predict RCLI and found that returning clippings to the turf-soil ecosystem increased both the N and carbon (C) pools, while reducing N fertilization requirements by 25–60% (depending on turf age).

From the above studies, it can be concluded that leaving clippings onsite is beneficial for turfgrass, and that RCLI varies greatly, even though the reasons are not completely understood. Factors that may influence RCLI include soil moisture holding capacity (Kopp

and Guillard, 2002); soil organic matter content (Kopp and Guillard, 2002; Qian et al., 2003); the age of the sward (Qian et al., 2003); and the thickness of the thatch layer (Kopp and Guillard, 2009).

Besides lawn mowing, fertilization is a second important component of turfgrass management. N fertilization increases turf vegetative growth, the yield of turfgrass clippings, and the total N amount returned with clippings. By increasing N rates, the N concentration in turfgrass clippings is increased, N mineralization is accelerated, and the N amount released from clippings per unit weight is increased (Kauer et al., 2008). On the other hand, it is known that plant residue decomposition also results in high N losses through N volatilization as NH_3 and N_2O (Janzen and McGinn, 1991; Quemada and Cabrera, 1997; Larsson et al., 1998). According to Whitehead et al. (1988), NH_3 volatilization in laboratory experiments amounted to between 20% and 47% of N herbage. Plants compete for N with soil microorganisms (Kuzyakov et al., 2000) and use only that N not otherwise immobilized. From these results, it can be hypothesized that RCLI is different with various N rates and that the effect increases with higher N fertilization rates.

On turf surfaces, decomposition of clippings and subsequent nutrient release are significantly influenced by weather conditions. Microbial degradation and N mineralization are delayed by dry conditions (Kalburtji et al., 1998; Kauer et al., 2011) and decrease at low temperatures (Andersen and Jensen, 2001). Therefore, the amount of recycled nutrients from returning comparable amounts of clippings can be different depending on weather. As a consequence, it can be hypothesized that growing season climate is one of the main factors influencing RCLI.

The objectives of the present research were to study: (i) the influence of CRT on turfgrass growth, NUE, and NREC; and (ii) the impact of N rate and weather during the growing season on RCLI.

MATERIALS AND METHODS

The experiment was carried out between 2004 and 2008 at the Eerika Experimental Station of the Estonian University of Life Sciences (58°23'32″ N, 26°41'31″E; elevation 60 m above sea level). The experimental field soil was *Stagnic Luvisol* according to the World Reference Base for Soil Resources classification system (FAO, 1998), and the soil texture was a sandy loam. The field was previously under barley for three years. Each year, straw was collected from the field and the land was ploughed afterwards. In the spring of 2003, the site was cultivated and sown with a turfgrass mixture of *Festuca rubra rubra* c 'Kauni' and *Poa pratensis* c 'Esto'. Both cultivars are bred in Estonia. Turfgrass sowing rate was 20 g m^{-2} (germinating seed), and each species contributed 50% in terms of seed number. At the beginning of the experiment, the 0–10 cm humus horizon contained 14.7 mg organic C g^{-1}, total N content was 1.5 mg g^{-1}, plant available phosphorous (P) was 34.4 mg kg^{-1}, and potassium (K) was 95.4 mg kg^{-1}. The C:N ratio was approximately 9.8:1, and pH_{KCl} was 5.25. The specific soil surface area was 26.2 m^2 g^{-1}.

The experiment was arranged as a 2 x 7 factorial and set out in a randomized complete block design with four replicates. The factors were as follows: (i) two clippings treatments, with clippings removed (CRM) or CRT; and (ii) seven applied fertilizer rates (kg ha^{-1}): $N_0P_0K_0$ (hereafter, N_0, for unfertilized), $N_0P_{22}K_{96}$ (PK), $N_{80}P_{11}K_{48}$ (N_{80}), $N_{160}P_{22}K_{96}$ (N_{160}), $N_{240}P_{34}K_{144}$ (N_{240}), $N_{320}P_{45}K_{192}$ (N_{320}), and $N_{400}P_{56}K_{240}$ (N_{400}). NH_4NO_3, $Ca(H_2PO_4)_2$, and KCl were used as the sources of N, P, and K fertilizers, respectively. Depending on the ratio of N, P, and K, the fertilizers were applied by hand to the plots in two to four splits: N fertilizer was applied in two to four splits: (i) one week after the start of the growing season (at the end of April, or the beginning of May), and during (ii) the first 10 days (i.e., decade) of June, (iii) the first decade of July, and (iv) the first decade of August. P fertilizer was applied in two splits: (i) at

the end of April, and (ii) at the end of September. K fertilizer was applied in three splits: (i) at the end of April, (ii) during June, and (iii) at the end of September.

Both clippings treatments (CRM and CRT) occurred in two adjacent blocks separated by a 1 m band of *Festuca rubra commutata.* Individual treatment plots were 10 m^2. No watering was applied.

The swards were cut 15–20 times at a height of 5 cm during the growing season by using traditional Estonian lawn mowing techniques. Cutting took place when the sward height was approximately 7.5 cm, and carried out on average once a week, except during periods of drought and in autumn, when cutting occurred less frequently (i.e., about once every two weeks). A rotary mulching lawn mower (Partner 5553 CMDEW) with a bag attachment (for clipping collection) was used to cut the plots. After each cutting, the harvested material was removed from the bag and weighed. Subsamples (100 g) were taken for determination of dry matter (DM) and N content (but only during the fourth year, 2007). For CRT treatment plots, fresh turfgrass clippings were returned immediately after weighing, spread evenly by hand over the area where they had been removed, and then mulched with a lawn mower. The end results were intended to resemble how the clippings would have looked had they been mulched throughout mowing.

Subsamples were dried immediately after harvesting, at 50–60 °C to a DM content of 85–90%, and then weighed. For absolute DM content determination, the samples were ground in a Knife Mill (Grindomix, Retsch GmbH) to a particle size of 0.5 mm and then dried at 105 °C for 6 hours. All yield calculations were made on the basis of absolute DM content. Total N was determined by using a dry combustion method in a varioMAX CNS elemental analyzer (Elementar Analysensysteme GmbH).

Equations for calculation

Returned clippings impact, or effect of returned clippings (RCLI), g m^{-2} y^{-1} indicates the DMY (g) difference between the CRT and CRM treatments per m^2 (m^{-2}) and per year (y^{-1}):

$RCLI = DMY_{CRT} - DMY_{CRM}$,

where the indices $_{CRT}$ and $_{CRM}$ indicate the specific clippings treatment.

Returned clippings efficiency (RCLE), g m^{-2} y^{-1} indicates how much the DMY (g) increased due to 100 g (DM) clippings returned per m^2 (m^{-2}) and per year (y^{-1}):

$RCLE = RCLI * 100 / DMY_{CRT}$,

where DMY_{CRT} = amount of returned clippings (g) per m^2 (m^{-2}).

Total N uptake (NUP), g m^{-2} y^{-1} indicates the total N amount (g) taken up by plants per m^2 (m^{-2}) and per year (y^{-1}):

$NUP_{CRT, CRM}$ = DMY * N concentration in clippings,

where NUP_{CRT} = N amount (g m^{-2} y^{-1}) returned with clippings, and NUP_{CRM} = N amount (g m^{-2} y^{-1}) removed with clippings.

Apparent N recovery NREC, % indicates the percentage of N applied with fertilizer that was recovered in the yield:

$NREC = \{(NUP \text{ at } N_x - NUP \text{ at } N_0) / N_x\} * 100$,

where N_x = applied N rate (g m^{-2}), and N_0 = unfertilized.

N-use efficiency NUE, g g^{-1} indicates how much DMY (g) was produced per gram (g^{-1}) N and per year (y^{-1}):

$NUE = (DMY \text{ at } N_x - DMY \text{ at } N_0) / N_x$

N-use efficiency applied with clippings (NUE_{CL}), g g^{-1} indicates the yield (g) produced per gram (g^{-1}) of returned clippings N:

NUE_{CL} = ((DMY_{CRT} at N_x – DMY_{CRT} at N_0) – (DMY_{CRM} at N_x – DMY_{CRM} at N_0)) / NUP_{CRT} at N_x

Apparent N recovery of clippings ($NREC_{CL}$), % indicates the percentage of N applied with clippings that was recovered in the yield:

$NREC_{CL}$ = {(NUP_{CRT} at N_x – NUP_{CRT} at N_0) – (NUP_{CRM} at N_x – NUP_{CRM} at N_0) / NUP_{CRT} at N_x} * 100, and

$NREC_{CL}$ (%) at control treatment = {(NUP_{CRT} – NUP_{CRM}) / NUP at N_0} * 100

N amount replaced by clippings ($NREP_{CL}$), kg ha^{-1} equal to the mineral N amount needed during CRM treatment to obtain a DMY yield equal to CRT treatment:

$NREP_{CL}$ = (N_x * DMY_{CRT} at N_x / DMY_{CRM} at N_x) – N_x

Weather Conditions

The climate of Estonia is moderated by maritime influences in the west, but is more continental in the east. The winter period (when mean air temperature is permanently below 0 °C) lasts for an average of 115 days; the mean temperature of the coldest months is -5.5 °C. The average duration of the growing season (air temperature permanently above 5 °C) is 175–190 days. The average period without night frosts is four months, during which time the average midsummer (July) temperature is 16–17 °C. Mean annual precipitation is 550–700 mm; the average rainfall during the warm months (i.e., April through the end of October) is 350–500 mm (Keppart and Loodla, 2006). Throughout the experimental period, we monitored the field micrometeorological conditions with Metos Model MCR300 weather stations (Pessl Instruments GmbH, Weiz, Austria); the sensors were positioned 2 m above ground.

Weather conditions (i.e., average air temperature, relative air humidity, and precipitation) from April through October during the experimental years 2004–2008 are presented in Table 1.

Statistical Analysis

The statistical package Statistica version 9.1 (StatSoft Inc., 2010) was used for all statistical analyses. Factorial analysis of variance (ANOVA) and one-way ANOVA were applied to test the effect of both treatments for RCLI and RCLE. Correlation analysis was used for testing the impact of weather conditions on RCLI and N fertilization for NUE, NUE_{CL}, NREC, and $NREC_{CL}$. Fisher's least significant difference (LSD) test for homogeneous groups was used for testing the significance of differences between fertilizaton and clippings treatments and years.

Multiple regression analysis was used for studying the relationship between applied N rate, DMY_{CRT}, and the impact and efficiency of returned clippings (RCLI and RCLE), as well as the impact of N fertilization on the N value of returned clippings, NUE_{CL}, and $NREC_{CL}$. The probability level was set at $P<0.05$.

RESULTS

The influence of CRT on DMY

Based on the average annual yield over the five-year experimental period, the influence of CRT on DMY was significant only for the fertilization treatments N_{160} ($F_{1,38}$)=12.0, P <0.01 and N_{240} ($F_{1,38}$)=6.36, P<0.05, where the yield increased by 17.7% and 18.5%, respectively (Fig. 1). Except for treatment N_0, the other fertilization treatments in combination with CRT caused slight but non-significant (i.e., 2.2–7.9%) increases on DMY. For treatment N_0, the DMY of the CRT plots was slightly lower (-7.3%) than on CRM plots (P >0.05).

Relationships between applied N rate, DMY_{CRT}, RCLI, and RCLE

DMY_{CRT} was in positive correlation (R^2=0.84, P<0.0001) with applied N rate and varied between 141 g DM m^{-2} y^{-1} and 596 g DM m^{-2} y^{-1} in average over the five-year period (Fig. 2). RCLI (R^2=0.74, P<0.01) and RCLE (R^2=0.56, P<0.01) at first increased with applied N rate and peaked at the rate of 240 kg N ha^{-1} where DMY_{CRT} was on average 550 g DM m^{-2} y^{-1} (Fig. 2). Future increase N rate (240-400 kg ha^{-1}) RCLI slightly (R^2=0.21, P>0.05) and RCLE significantly decreased (R^2=0.30, P<0.05).

NUP, NREC, and NUE

For both clippings treatments, NUP, NREC, and NUE were studied only in the fourth year (i.e., 2007) of the experiment. According to the results, NUP (-0.3–7.2 g m^{-2} y^{-1}; difference between CRT and CRM treatments); NREC (4.5–31.3%); and NUE (1.2–6.5 g g^{-1}) all were higher in CRT treatments (Table 2).

Depending on the N applied rate as mineral fertilizer, NUP_{CRT} was between 3.1 g m^{-2} y^{-1} and 23.6 g m^{-2} y^{-1} (Table 2). Compared to NUE_{CRM} and $NREC_{CRM}$, respective indices for clippings NUE_{CL} (-4.9–7.0 g g^{-1}) and $NREC_{CL}$ (-9.7–33.8%) were lower (Table 3), and these response

patterns against the mineral N rate were different. Both NUE_{CRM} and $NREC_{CRM}$ did not exhibit significant change (Table 2) in response to an increase of applied mineral fertilizer N rate. The NUE_{CL} (r=0.90, P=0.1) and $NREC_{CL}$ (r=0.99, P<0.01) increased at first up to the N rate of 240 kg ha^{-1} and started to decrease thereafter (Table 3). For the N_0 treatment, where the amount of N in the returned clippings was 3.1 g m^2 y^{-1} (Table 2), the $NREC_{CL}$ and NUE_{CL} both were negative (Table 3).

Relationships between weather conditions, time of CRT, RCLI, and RCLE

To investigate the effect of weather conditions and the time of CRT for RCLI and RCLE, only data from the N_{240} treatment was used, as this was the treatment for which CRT had the most significant effect.

DMY_{CRT} and its effect varied during the experiment (P<0.05) (Table 4). There was no correlation between DMY_{CRT} and RCLI (P>0.05) and RCLE (P>0.05). In fact, returning approximately the same amount of clippings gave different results in the years 2004, 2005, and 2008 (P<0.05).

RCLI and RCLE depended mainly on the amount returned in May and June (Table 5). The correlation between DMY_{CRT} and RCLI (r=0.56, P<0.01) and DMY_{CRT} and RCLE (r=0.48, P<0.01) was the highest with the total amount of clippings returned in May and June. The correlations were weaker when the amounts returned in May and June were considered separately. The effect of DMY_{CRT} during the second part of the growing season (July to October) between RCLI and RCLE either was not statistically significant, or negative (P<0.05).

In terms of the effect of weather conditions, RCLI (r=0.85, P<0.01) and RCLE (r=0.83, P<0.01) were mainly affected by the amount of precipitation in May (Table 6). RCLI was largest during May, with high levels of precipitation in 2005 and 2007 (Table 1). In 2004,

2006, and 2008, when precipitation was lower, RCLI (Table 4) was smaller, and the shortage of precipitation in May was not compensated by higher precipitation in the following months. In 2004 and 2008, a dry May was followed by a wet June, but RCLI was smaller than in 2005 and 2007. The effect of precipitation and temperatures during other months, and growing season average air temperature and sum of precipitation on RCLI and RCLE were weaker, or not statistically significant (P>0.05).

RCLI and RCLE are best described by an equation using the time of CRT and weather conditions during a growing season. In constructing this equation, the average air temperatures, precipitation, DMY_{CRT} in single months (May, June, etc.) and in different periods (May–June, May–July, May–August, May–September, May–October, June–July, June–August, etc.) were used as variables. Using multiple regression and stepwise procedure, variables were found that best described RCLI and RCLE changes. The value of RCLI (R^2=0.85, P<0.001) was mostly affected by the sum of DMY_{CRT} during the period from May–July and precipitation from May–June. The value of RCLE (R^2=0.86, P<0.001) was mostly affected by precipitation in May–June and the average air temperatures in July (Table 7).

$NREP_{CL}$

$NREP_{CL}$ is affected by N fertilization and varied in wide limits (Table 8). By increasing the N rate, $NREP_{CL}$ increased at first and reached a maximum at the N_{240} treatment (i.e., 47.5 kg ha^{-1} on average, with the highest at 90 kg ha^{-1}). By further increasing the N rate, $NREP_{CL}$ decreased, but its variation also increased. In 2005, for treatments N_{320} and N_{400}, $NREP_{CL}$ was 88 kg ha^{-1} and 53 kg ha^{-1}, respectively; in 2007, 86 kg ha^{-1} and 26 kg ha^{-1}, respectively; but in 2004, 2006, and 2008, $NREP_{CL}$ was negative in both treatments. In treatments with N rates between N_{80} and N_{240}, the $NREP_{CL}$ variation was smaller. In treatment

293 N_0, the only positive result from returning clippings was obtained in the final year (when
294 $NREP_{CL}$ was 3.9 kg ha^{-1}).
295
296 **DISCUSSION**
297 **RCLI on turfgrass growth**
298 According to our results, RCLI on sward yield was less pronounced than in several previous
299 studies (Heckman et al., 2000; Kopp and Guillard, 2002; Qian et al., 2003). Even though in
300 some experimental years (depending on the fertilization treatments), the DMY of plots
301 receiving CRT treatment surpassed the yield of CRM treatment by as much as 51%, the
302 average DMY increase from 2004 through 2008 did not exceed 18.5% in any fertilization
303 treatment. This is less than the increases reported by Starr and DeRoo (1981) and Kopp and
304 Guillard (2002). Our method of study was different, however, as mulched grass clippings
305 were returned on the sward surface; this may have affected the results, as plant cells are
306 broken during mulching and cell fluid flows out. When plant remains decay on the soil
307 surface, some N is volatilized (Whitehead et al., 1988; Janzen and McGinn, 1991; Larsson et
308 al., 1998). We assume that mulching can increase N loss from clippings, but are unaware of
309 any previous study on this topic. We performed three N content measurements of clippings
310 before and after mulching (results not presented); one measurement showed that N content
311 was smaller after mulching, while the other two results were not statistically different. This
312 issue should be investigated in more detail.
313 Sward height may be a second factor contributing to reduced RCLI and RCLE. In previous
314 studies (Starr and DeRoo, 1981; Heckman et al., 2000; Kopp and Guillard, 2002), the cutting
315 height (3.8–4.4 cm) was lower than in our experiment. The combined effect of taller grass in
316 the sward after cutting, as well as longer grass clippings, may have hindered clippings from
317 reaching the soil surface. In treatments with N rates higher than 160 kg ha^{-1}, where the yield

of clippings was higher, we observed grass clippings sticking together during mulching and forming clumps. This made it difficult for the clippings to reach the soil surface; as a result, a large amount remained near sward surface at the time of the next cutting, which became even more obvious in the second part of the growing season when the sward was denser. Several studies have shown that early-phase decomposition of plant residues is positively influenced by the degree of contact between plant residues and soil (Jensen, 1994; Sørensen et al., 1996). Close contact with soil usually increases the microbial decomposition of organic matter (Douglas et al., 1980; Cogle et al., 1989) due primarily to higher moisture content in the residues (Parr and Papendick, 1978). Our parallel experimental investigation of grass residue decomposition showed that the decay rate was influenced by how long the grass residues stayed moist after cutting (Kauer et al., 2011). Grass clippings left on turfgrass dry quickly under the sun and wind, which slows down their rate of decomposition. Consequently, RCLI and RCLE may be influenced by the length and density of the turfgrass sward; in general, RCLI and RCLE are greater in swards that are shorter and less dense.

The influence of mineral N fertilization on RCLI

Our results showed that NUP, NREC, and NUE generally increase when clippings are returned (Table 2), which is consistent with findings reported by Starr and DeRoo (1981) and Kopp and Guillard (2002). One exception occurred with N_0 treatment, when the NUP value was higher when grass clippings were removed (Table 2). Using isotopes, Starr and DeRoo (1981) showed that N derived from clippings was about 9.1% of the total N applied in a current year, but 20% of total N returned in the previous two years. Our study method did not allow us to study N uptake so precisely. Nevertheless, by assuming that all additional yield from CRT treatment may be attributed to N released from grass clippings, we estimate that $NREC_{CL}$ was between -9.7% and 33.8% in one experimental year (i.e., year four, in Table 3).

This result supports our hypothesis that RCLI is not only significantly influenced by N fertilization, but also may vary to a large extent depending on the applied rate.

Returning clippings in an unfertilized treatment does not increase DMY; in fact, the effect is opposite to that of fertilized treatments, although this effect is not significant at lower N applied rates (e.g., N_{80} treatment). This result differs from that obtained by Kopp and Guillard (2002), where CRT showed DMY increasing significantly even for N_0 treatment.

In our parallel investigation, where we studied RCLI on soil organic C (SOC) and total N content, it was evident that the total N concentration in 0–5 cm soil layer was significantly higher (P<0.05) at the end of the experiment for N_0 treatments combined with CRT (Kauer et al., 2012). This result suggests that most N from clippings was incorporated into soil organic matter and not available for plant use. Thus, appropriate N fertilization increases N concentration and decreases the C:N ratio in clippings, which then results in increased net N mineralization per unit weight of residues (Kauer et al., 2008). Newly established grasslands have a great potential to sequester C and N (Tyson et al., 1990; Shi et al., 2006a; Qian et al., 2010). Our test fields had soils with low SOC content, causing wide C and N sequestration; therefore, RCLI and RCLE depended significantly on applied N rate. Shi et al. (2006a) has shown that N mineralization increases in soils with higher SOC and total N content.

Kopp and Guillard (2002) showed that RCLI increases linearly with higher N rates (i.e., 0…392 kg ha^{-1}). In our experiment, RCLI and RCLE at first increased with increasing N rates, but then started to decline from N_{240} kg ha^{-1} (Fig. 2). The effect of this decline above N_{240} weakened the total N amount returned even at higher N rates (Fig. 2). Our swards were established with a seed mixture of *Festuca rubra rubra* and *Poa pratensis*, which have a basic requirement of 160 kg N ha^{-1} and 400 kg N ha^{-1}, respectively, under Estonian climate conditions (Raave and Hein, 1989). In the N_{320} and N_{400} treatments, the N amount probably

368 exceeded plant requirements because even with CRM treatment, the DMY of the clippings
369 still increased significantly (Fig. 1) at those N rates. This suggests that CRT is efficient only
370 until the point where the total N input (CRT and fertilizer) does not exceed plant
371 requirements.
372
373 **The influence of time of CRT and weather conditions on RCLI and RCLE**
374 Our results indicate that RCLI and RCLE are significantly affected by time of CRT. Positive
375 RCLI was exhibited mainly when clippings were returned within the first part of the growing
376 season (May–July); returning clippings during the second part of the season (August–
377 September) had no effect on RCLI. During the second part of the growing season, it was
378 typical for residues to remain on the sward, where they had a negative impact on grass growth
379 and caused gaps and bare patches in the sward. During the spring months, however, the sward
380 was less dense, and an accumulation of grass residues on the sward was not observed. Thus,
381 RCLI is related to sward density, being more efficient early in the growing season when lower
382 densities allowed clippings to move freely downward to the soil surface.
383 RCLI and RCLE were most affected by precipitation in May (Table 6). This result is
384 indirectly consistent with the results of Kopp and Guillard (2002), who found RCLI
385 dependent on soil moisture content. The month of May was specific in our experimental site
386 conditions because DMY_{CRT} was the highest ($P<0.05$), contributing 24% of the annual total
387 clippings returned to the sward. Furthermore, the decomposition of CRT from the previous
388 autumn continues into May (Kauer et al., 2011). Therefore, the N amount released in May had
389 a bigger impact on the yield of clippings across the growing season than N released in any
390 other month.
391 We expected that precipitation would have a positive effect on N recycling speed by
392 accelerating the movement of clippings through the sward to the soil surface, while helping to

maintain the moisture needed for grass residue decomposition. Thus, N recycling would happen faster in years where precipitation in May was greater. We did not, however, find a significant relationship between precipitation and DMY_{CRT} (P>0.05). DMY_{CRT} in May was quite stable during the entire experimental period, compared to other months where greater differences were observed among years. Therefore, it may be assumed that RCLI and RCLE could be higher if the lawn is mowed immediately before rainfall, or watered shortly after mowing.

CONCLUSIONS

The literature has suggested that leaving clippings onsite is beneficial for turfgrass, and that RCLI varies greatly, although the reasons for the variation are not well understood. Our results showed that RCLI and RCLE may be influenced by sward properties, fertilization rates, and amount of precipitation. RCLI is greater in swards that are shorter and less dense. In addition, NUP, NREC, and NUE generally increase when clippings are returned.

We hypothesized that RCLI is different with various N rates and that the effect increases with higher N fertilization rates. Our study showed that returning turfgrass clippings provides significant positive effects only when plants are fertilized with N; without fertilization, the effect is negative. While applying higher N rates does increase the RCLI effect, this increase is evident until the N supplied from fertilization and CRT exceeds basic plant requirements, at which point adding extra N has no discernible effect. In our study, RCLI was significant for turfgrass growth only with N_{160} and N_{240} treatments.

We investigated the influence on timing of CRT and found that RCLI depends mostly on DMY_{CRT} during the first part of the growing season (i.e., May and June), when swards are sparse and clippings can reach the soil more easily. During the second part of the growing season, DMY_{CRT} impact on RCLI was not statistically significant, as the clippings remained on top of the sward and sometimes damaged the sward by forming gaps and empty spots.

We also hypothesized that growing season climate is one of the main factors influencing RCLI and RCLE. In general, RCLI and RCLE were most affected by precipitation in May; average air temperature and sum of precipitation during the growing season, however, did not affect either variable. We assume precipitation has a positive effect on N recycling speed by accelerating the movement of clippings through the sward to the soil surface, while helping to maintain the moisture needed for grass residue decomposition. Later swards grow tighter, thus causing RCLI to decrease.

For unfertilized swards, RCLI requires further research because the results from different regions are contradictory. The question of how N is released from grass residues and to what extent recycling is affected by mulching also need further investigation.

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FIGURE CAPTIONS

Fig, 1. Influence of clippings treatment on dry matter yield (DMY) at different fertilization treatments from 2004 through 2008 (mean±SE, $P<0.05$).

Fig. 2. The relationships between applied N rate, amount of returned clippings (DMY_{CRT}) (Y_1), returned clippings impact (RCLI) (Y_2), and returned clippings efficiency (RCLE) (Y_3).

$DMY_{CRT} = 130.68+2.29*x-0.0028*x^2$, $R^2=0.84$, $P<0.01$

$RCLI = -19.69+0.80*x-0.0018*x^2$, $R^2=0.29$, $P<0.01$

$RCLE = -7.709+0.208*x-0.0005*x^2$, $R^2=0.40$, $P<0.01$

592 Table 1. Monthly average air temperature (°C), relative air humidity (%), and precipitation
593 (mm) during each growing season.

Year	Month						
	April	May	June	July	August	September	October
			Average air temperature, °C				
2004	7.7	10.3	13.3	16.6	16.7	11.8	7.2
2005	4.8	10.9	14.6	19.4	16.6	12.7	10.7
2006	8.1	11.9	16.2	18.8	16.9	13.5	9.2
2007	6.8	11.9	16.0	16.8	17.8	10.8	11.0
2008	7.1	10.7	14.5	16.1	15.7	9.8	8.6
			Relative air humidity, %				
2004	68.9	72.6	82.0	84.0	88.0	93.4	94.1
2005	78.4	81.2	80.6	73.2	85.1	90.2	92.9
2006	57.0	61.9	79.0	71.0	88.0	92.0	98.2
2007	79.9	82.0	78.0	88.0	85.0	93.0	97.3
2008	82.0	76.0	80.0	90.0	93.0	94.1	96.3
			Precipitation, mm				
2004	4.2	37.8	184.0	76.2	104.8	86.2	10.2
2005	3.2	114.8	54.2	21.8	92.4	59.4	0.4
2006	3.8	33.8	47.0	13.4	84.4	37.0	48.0
2007	18.2	101.4	42.0	94.8	36.0	40.2	23.6
2008	26.8	27.4	110.6	53.8	117.8	45.6	24.4

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606 Table 2. Dry matter yield (DMY), N content in the yield, total N uptake (NUP), apparent N
607 recovery (NREC), and N-use efficiency (NUE) in year four (2007) for harvested clippings.

N rate	DMY		N content in the yield		NUP		NREC		NUE	
	CRM†	CRT‡	CRM	CRT	CRM	CRT	CRM	CRT	CRM	CRT
kg ha^{-1}	g m^{-2} y^{-1}		%		g m^{-2} y^{-1}		%		g g^{-1}	
0	114	99	3.0	3.1	3.4	3.1	-	-	-	-
80	232	251	3.4	3.4	7.9	8.5	56.2	67.5	14.8	19.0
160	290	354	3.7	3.8	10.7	13.5	45.6	65.0	11.0	15.9
240	375	516	4.0	4.3	15.0	22.2	48.3	79.6	10.9	17.4
320	437	509	4.3	4.5	18.8	22.9	48.1	61.9	10.1	12.8
400	515	548	4.3	4.3	22.1	23.6	46.7	51.2	10.0	11.2

608 † CRM = clippings removed

609 ‡ CRT= clippings returned

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Table 3. Apparent N recovery of clippings ($NREC_{CL)}$) and N-use efficiency applied with clippings ($NUE_{CL)}$) in year four (2007).

N rate	$NREC_{CL}$	NUE_{CL}
kg ha^{-1}	%	g g^{-1}
0	-9.7	-4.9
80	10.5	4.0
160	23.0	5.9
240	33.8	7.0
320	19.2	3.8
400	7.6	2.0

Table 4. Amount of returned clippings (DMY_{CRT}), returned clippings impact (RCLI), and returned clippings efficiency (RCLE) throughout the experimental period for the N_{240} treatment.

Year	DMY_{CRT}	SE†	RCLI	SE	RCLE	SE
	g m^{-2} y^{-1}		g m^{-2} y^{-1}		g m^{-2} y^{-1}	
2004	603[a‡]	11.2	48.0[ab]	12.9	7.8[a]	19.8
2005	610[a]	6.0	131.5[c]	8.8	21.5[b]	12.9
2006	413[c]	10.5	35.4[a]	8.5	8.5[a]	20.4
2007	516[b]	8.9	141.3[c]	8.6	27.3[c]	14.3
2008	608[a]	10.9	75.3[b]	11.4	12.4[a]	20.1
Mean	550		86.3		15.5	

†SE = standard error

‡Within the same column, values with different letters are significantly different (P <0.05).

Table 5. Correlation coefficients between returned clippings impact (RCLI), returned clippings efficiency (RCLE), and amount of returned clippings (DMY_{CRT}) during different months of the growing season.

Variable	Month					
	May	June	July	August	September	October
RCLI	0.38	0.42	NS†	NS†	NS†	NS†
RCLE	0.44	0.32	NS†	NS†	-0.18	NS†

†NS = not significant

Table 6. Correlation coefficients between returned clippings impact (RCLI), returned clippings efficiency (RCLE), and weather conditions during different months of the growing season.

	Month						May – October
	May	June	July	August	September	October	
				Average air temperature			
RCLI	NS†	0.19	NS	0.36	-0.26	NS	NS
RCLE	0.34	0.36	NS	0.51	-0.22	NS	-0.26
				Precipitation			
RCLI	0.85	-0.47	0.28	NS	-0.19	-0.48	0.30
RCLE	0.83	-0.47	0.31	NS	-0.32	-0.32	0.41

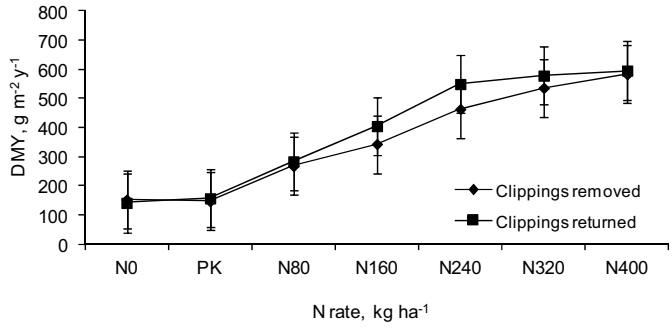
†NS = not significant

Table 7. Multiple regression models showing the relationship between time of CRT, weather conditions, returned clippings impact (RCLI), and returned clippings efficiency (RCLE). Results reflect the best fit obtained from a stepwise procedure.

	Correlation coefficients	Std. Beta Error	Regression coefficients	Std. Beta Error	$t_{(117)}$	P-level
			RCLI			
Intercept			-10.64	6.13	1.73	P=0.085
Sum of CRT, May–July	0.50	0.048	0.49	0.048	10.29	P<0.001
May precipitation, mm	0.70	0.044	0.88	0.056	15.70	P<0.001
June precipitation, mm	-0.41	0.057	-0.35	0.049	-7.049	P<0.001
			RCLE			
Intercept			634.1	49.92	12.7	P=0.001
May precipitation, mm	0.83	0.042	1.84	0.093	19.83	P<0.001
June precipitation, mm	-0.39	0.046	-0.60	0.070	-8.60	P<0.001
July average air temperature, ^{o}C	-0.49	0.042	-30.91	2.69	-11.50	P<0.001

Table 8. Mineral N amount replaced by clippings ($NREP_{CL}$).

N rate	$NREP_{CL}$,		
	Experimental average	Minimum	Maximum
kg ha^{-1}		kg ha^{-1}	
80	5.1	-1.2	11.5
160	31.3	2.3	82.2
240	47.5	20.3	90.2
320	27.0	-15.1	88.0
400	8.5	-30.4	86.2



Fig, 1. Influence of clippings treatment on dry matter yield (DMY) at different fertilization treatments from 2004 through 2008 (mean±SE, $P<0.05$).

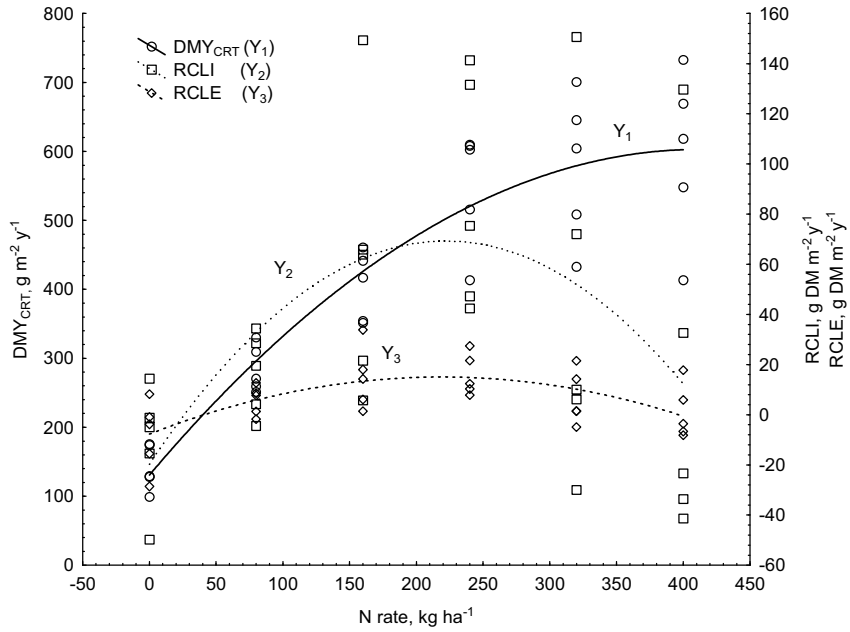


Fig. 2. The relationships between applied N rate, amount of returned clippings (DMY_{CRT}) (Y_1), returned clippings impact (RCLI) (Y_2), and returned clippings efficiency (RCLE) (Y_3).

$DMY_{CRT} = 130.68+2.29*x-0.0028*x^2$, $R^2=0.84$, $P<0.01$

$RCLI = -19.69+0.80*x-0.0018*x^2$, $R^2=0.29$, $P<0.01$

$RCLE = -7.709+0.208*x-0.0005*x^2$, $R^2=0.40$, $P<0.01$

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2009 – …	Estonian Soil Science Society
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2001 – …	European Society for Soil Conservation

Research Interests:

Nutrient cycling in plant-soil system
Decomposition of organic matter

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2008–2013	Target financed project No. SF0170052s08: „Diversity, integrity and sustainability of agroecosystems", principal executor since 2011
2010–2012	Ministry of the Agriculture of Estonia project No. 8-2/10038PKPK: „Alternative fertilizers environment-saving utilization opportunities and efficiency in conventional and organic farming in comparison with traditional organic and mineral fertilizers", principal executor
2011–2012	Ministry of the Agriculture of Estonia project No. 8-2/T11028PKTM: „Effective use of local fertilizers and economic analysis of grassland utilization in farm-based feed production", principal executor
2009–2012	Central Baltic INTERREG IV project No. 8-2/T9132PKPK: „Energy Positive Farm", principal executor
2008–2010	Ministry of the Agriculture of Estonia project No. 8-2/T8015PKPK: „The use of liquid manure (slurry) as fertilizer for grassland and field crops and its impact on the environment and yield", principal executor
2004–2007	ETF grant No. 5751: „The relationships between nutrient cycling and grassland phytoproductivity depending on stand composition, defoliation frequency and fertilizer application", principal executor

Professional training:

18 Nov–3 Dec 2004	The course „Package of STATISTICA“
13-23 May 2008	The course „Soil and Water Relationship“, organized by Estonian University of Life Sciences, 6 ECTS
20-21 Jan 2010	The course „Energy analysis of agricultural production“, organized by Helsinki University, 5 ECTS
1-4 March 2010	The course „Scientific Writing“, organized by Estonian University of Life Sciences and program PRIMUS, 1 ECTS
23-28 Aug 2010	The course „ An introduction to uses in ecology and plant physiology“, organized by University of Munchen, 5 ETCS
17-21 Jan 2011	The course „Statistics: experiences, training, and supervision in the field of agricultural and environmental sciences“, organized by Estonian University of Life Sciences and program PRIMUS, 2 ECTS
22-25 March 2011	The course „Use of isotopic tracers in N dynamics studies in agricultural ecosystems“, organized by Estonian University of Life Sciences and program PRIMUS, 2 ECTS
8-11 Aug 2011	The course „Spoken English“, organized by Estonian University of Life Sciences and program PRIMUS, 2 ECTS

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Teenistuskäik:

2010–….	Eesti Maaülikool, Põllumajandus- ja keskkonnainstituut, taimekasvatuse ja rohumaaviljeluse osakonna teadur
2000–2009	Eesti Maaülikool, Põllumajandus- ja keskkonnainstituut, mullateaduse ja agrokeemia osakonna vanemlaborant

Teaduskraad: MSc (mullateadus)

Teaduskraadi väljaandnud asutus, aasta:
Eesti Põllumajandusülikool, 2005

Teadusorganisatsiooniline ja – admistratiivne tegevus:

2009–…	Eesti Mullateaduse Selts
2007–…	Akadeemiline Põllumajanduse Selts
2001–…	Euroopa Mullakaitse Selts

Teadustöö põhisuunad:

Toitainete aineringed taim-muld süsteemis

Orgaanilise aine lagunemine

Osalemine uurimusprojektides:

2008–2013	Sihtfinantseeritav teema nr. SF0170052s08: „Agroökosüsteemide mitmekesisus, terviklikkus ja jätkusuutlikkus“, põhitäitja alates 2011
2010–2012	EV Põllumajandusministeerium grant nr. 8-2/10038PKPK: „Alternatiivsete väetusainete keskkonnahoidliku kasutuse võimalused ja efektiivsus tava- ja maheviljeluses võrdlevalt traditsiooniliste orgaaniliste ja mineraalväetistega“, põhitäitja
2011–2012	EV Põllumajandusministeeriumi grant nr. 8-2/T11028PKTM: „Kohalike väetiste efektiivsem kasutamine ja rohusöötade tootmise majanduslik hinnang kohapealse söödatootmise arendamisel“, põhitäitja
2009–2012	INTERREG IV A Programm 2007-2010 projekt 8-2/T9132PKPK: „Energiapositiivne farm“, põhitäitja
2009–2010	Siseriiklik leping nr. 8-2/10004: „Setomaa ja Ape regiooni loodus- ja inimressursside alane uuring Eesti -Läti koostööprojekti BY LOCAL raames“, põhitäitja
2008–2010	KIK projekt nr. 8-2/T9023PKPK: „Eesti Maaülikooli mullateaduse ja agrokeemia laboratooriumi võimekuse täiustamine“, põhitäitja

2008–2010	EV Põllumajandusministeeriumi grant nr. 8-2/T8015PKPK: „Vedelsõnniku (läga) kasutamine rohumaade ja põllukultuuride väetisena ning mõju keskkonnale ja saagi kvaliteedile“, põhitäitja
2004–2007	ETF grant nr. 5751: „Rohumaa aineringete seosed fütoproduktiivsusega olenevalt taimiku koosseisust, kasutusviisist ja väetamisest“, põhitäitja

Erialane enesetäiendamine:

18. nov.–3. dets. 2004	Kursus „Statistikapakett STATISTICA kasutamine“
13-23. mai 2008	Eesti Maaülikooli poolt korraldatud kursus „Vee ja mulla seosed“, 6 ECTS
20-21. jaan. 2010	Helsingi Ülikooli poolt korraldatud kursus „Energiaanalüüs põllumajanduses“ 5 ECTS
1-4. märts 2010	Eesti Maaülikooli ja PRIMUS programmi poolt korraldatud kursus „Teaduslik kirjutamine“, 1 ECTS
23-28. aug. 2010	Müncheni Ülikooli poolt korraldatud kursus: „Sissejuhatus kasutamaks stabiilseid isotoope ökoloogias ja taimefüsioloogias“, 5 ETCS
17-21. jaan. 2011	Eesti Maaülikooli ja PRIMUS programmi poolt korraldatud kursus „Statistika: kogemused, praktika ja juhendamine põllumajandus- ja keskkonnateadustes“, 2 ECTS
22-25. märts 2011	Eesti Maaülikooli ja programmi PRIMUS poolt korraldatud kursus „Isotoopide kasutamine N aineringe uurimisel põllumajanduslikes ökosüsteemides“, 2 ECTS
8-11. aug. 2011	Eesti Maaülikooli ja programmi PRIMUS poolt korraldatud kursus „Rääkimise inglise keel“, 2 ECTS

LIST OF PUBLICATIONS

1.1. Articles indexed by Thomson Reuters Web of Science

Kauer, K., Kõlli, R., Viiralt, R., Köster, T., Noormets, M., Laidna, T., Keres, I., Parol, A., Varul, T., Selge, A., Raave, H. 2012. The effect of cut plant residues management and fertilization on the dry matter yield of swards and on carbon content in soil. *Communications in Soil Science and Plant Analysis* (accepted for publication).

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Reintam, E., Trükmann, K., Kuht, J., Nugis, E., Edesi, L., Astover, A., Noormets, M., **Kauer, K.,** Krebstein, K., Rannik, K. 2009. Soil compaction effects on soil bulk density and penetration resistance and growth of spring barley (Hordeum vulgare L.). *Acta Agriculturae Scandinavica: Section B, Soil and Plant Science*, 59, 265–272.

Kõlli, R., Ellermäe, O., Köster, T., Lemetti, I., Asi, E., **Kauer, K.** 2009. Stocks of organic carbon in Estonian soils. *Estonian Journal of Earth Sciences*, 58, 95–108.

1.2. Peer-reviewed articles in other International research journals with an ISSN code and International editorial board

Kõlli, R., Köster, T., **Kauer, K.,** Lemetti, I. 2010. Pedoecological regularities of organic carbon retention in Estonian mineral soils. *International Journal of Geosciences*, 1, 139–148.

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2.1. High-level peer-reviewed research monographs

Noormets, M., Köster, T., Tõnutare, T., **Kauer, K.,** Kõlli, R., Paal, T., Oder, M. 2006. The pedoecological conditions of natural and opencast peat field in Estonia. In: Martin-Duque, J.F., Brebbia, C.A., Emmanouloudis, D.E., Mander, U. (Eds.). *Geo-Environment and Landscape Evolution II*. WIT Press, Southampton, UK, pp. 413–422.

3.1. Articles in collections indexed by the Thomson Reuters ISI proceedings

Noormets, M., Köster, T., Tõnutare, T., **Kauer, K.,** Kõlli, R., Karp, K. 2004. Determination of microbial activity and nitrogen and carbon forms in a peat soils in Estonia. In: Martin-Duque, J.F., Brebbia, C.A., Godfrey, A.E., Diaz De Teran, J.R., (Eds.). *Geo-Environment 2004. International conference on Monitoring, Simulation and Remediation of the Geological Environment.* WIT Press, Southampton, UK, pp. 59–68.

Köster, T., **Kauer, K.,** Tõnutare, T., Kõlli, R. 2004. The management of the coastal grasslands in Estonia. In: Brebbia, C. A., Saval Perez J. M., Garcia Andion, L. (Eds.). *Coastal Environment V incorporating Oil Spill Studies*. WIT Press, Southampton, UK, pp. 45–54.

3.2. Articles/chapters in books published by the publishers not listed in Annex

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3.4. Papers published in the proceedings of International conferences

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Raave, H., **Kauer, K.,** Viiralt, R., Köster, T., Noormets, M., Laidna, T., Keres, I., Parol, A., Varul, T., Selge, A. 2011. Surface decomposition of mown residues of grasses and white clover. In: *Proceedings of the 16th Symposium of the European Grassland Federation: Grassland farming and land management systems in mountainous regions.* Irdning, Austria, 28-31 August 2011, 449–451.

Raave, H., **Kauer, K.,** Noormets, M., Selge, A., Viiralt, R. 2010. Nitrogen and potassium leaching from grassland soil depending on applied fertilizer type and rate and sward botanical composition. In: *Proceedings of the 23th General meeting of the European Grassland Federation: Grassland in a changing world.* Kiel, Germany August 29th-2th September 2010, 1058–1060.

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Kauer, K., Köster, T., Raave, H., Viiralt, R., Selge, A. 2007. Decomposition of cut and returned grass in turfgrass and grass-clover sward. In: *Proceedings of the 14th Symposium of the European Grassland Federation: Permanent and temporary grassland: plant, environment and economy*. Ghent, Belgium. 3-5 September 2007, 339–342.

Kauer, K., Köster, T., Sammul, M., Viiralt, R. 2006. Grazing influence on nutrient cycle and hay quality of coastal grassland plant associations. In: *Proceedings of the 21st General Meeting of the European Grassland Federation: Sustainable Grassland Productivity*. Badajoz, Spain. 3-6 April 2006, 342–344.

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Noormets, M., **Kauer, K.,** Köster, T., Kriipsalu, M., Jänes, A., Merivee, A. 2005. The Testing of Model Plant Raphanus Sativus Germination and Phytomass Production on Oil-Composts. In: *Proceedings of Kalmar ECO-TECH' 05 conference on Waste to Energy Bioremediation and Lechate treatment: The Second Baltic Symposium on Environmental Chemistry. The First Kalmar Nanotechnology Workshop. ECO-TECH' 05 conference on Waste to Energy Bioremediation and Lechate treatment.* Kalmar, Sweden. 28-30 Nov 2005, 579–588.

3.5. Articles/presentations published in local conference proceedings

Tampere, M., **Kauer, K.,** Noormets, M., Parol, A., Raave, H., Selge, A., Viiralt, R. 2011. Väetamise mõju lämmastiku ja kaaliumi leostumisele rohumaal. *Agronoomia 2010/2011,* Saku, 139–146.

Viiralt, R., Raave, H., **Kauer, K.,** Selge, A., Parol, A. 2009. Vedelsõnniku kasutamine rohumaade väetisena. *Agronoomia 2009.* Jõgeva, 146–155.

Kauer, K., Köster, T., Kõlli, R. 2005. Taimetoitainete ringe ekstensiivselt karjatatavatel rannarohumaadel. EPMÜ Teadustööde kogumik *Agronoomia 2005*, 220, 135–137.

Kõlli, R., **Kauer, K.,** Köster, T., Noormets, M. 2005. Looduslikku mulda kaitseb taimkate, põllumulda- ökoloogiliselt otstarbekas tehnoloogia. EPMÜ Teadustööde kogumik *Agronoomia 2005*, 220, 9–11.

5.2. Published meeting abstracts, not indexed by Thomson Reuters Web of Science

Kauer, K., Köster, T., Raave, H., Viiralt, R., Selge, A. 2008. Nitrogen release from turfgrass clippings with different fertilization rates . In: *XXI International Grassland Congress, VIII International Rangeland Congress: multifunctional grasslands in changing world. Vol II: XXI International Grassland Congress, VIII International Rangeland Congress*. Hohhot, China. 29 June – 5 July 2008, 302.

Köster, T., Kauer, K., Sammul, M., Kõlli, R., Selge, A., Noormets, M., Raave, H. 2008. Processes in plant-soil system of coastal grasslands induced by management practice. In: *XXI International Grassland Congress, VIII International Rangeland Congress: multifunctional grasslands in changing world. Vol II: XXI International Grassland Congress, VIII International Rangeland Congress*. Hohhot, China. 29 June – 5 July 2008, 508.

Köster, T., Penu, P., Kikas, T., Astover, A., Kauer, K., Kõlli, R. 2008. Soil water and wind erosion estimation by using the test areas, GIS land use and soil map. Eurosoil 2008. Book of Abstracts, 25.

Sammul, M., Köster, T., Kauer, K. 2008. Nutrient accumulation during reed encroachment reduces efficiency of restoration of Baltic coastal grasslands. In: *Towards a sustainable future for European ecosystems. Providing restoration guidelines for Natura 2000 habitats and species: 6th European Conference on Ecological Restoration*. Ghent, Belgium. 8-12 September 2008.

Köster, T., Kauer, K., Viiralt, R., Selge, A. 2005. Changes in grass quality of coastal meadows in Estonia. In: *XX International Grassland Congress: Offered papers: XX International Grassland Congress*. Dublin, Ireland, UK, pp. 284–284.

VIIS VIIMAST KAITSMIST

PRIIT PÄKK

ALTERATIONS OF EPIDERMAL CELLS` FUNCTIONAL ACTIVITY IN FISH DUE TO INFECTION
NAKKUSTE POOLT PÕHJUSTATUD EPIDERMISE RAKKUDE TALITUSLIKU AKTIIVSUSE MUUTUSED KALADEL
Prof. **Tiit Paaver**, dr. **Piret Hussar**
14. veebruar 2012

SIRJE VÄRV

MARKER-BASED GENETIC CHARACTERIZATION OF THE ESTONIAN DAIRY CATTLE BREEDS.
EESTI PIIMAVEISETÕUGUDE ISELOOMUSTAMINE GENEETILISTE MARKERITE ALUSTEL.
Prof. **Haldja Viinalass**, prof. **Juha Kantanen**
20. märts 2012

KÄTLIN BLANK

DYNAMICS AND INTERACTIONS OF PHYTO- AND ZOOPLANKTON AS INDICATORS OF THE STATUS OF LAKE PEIPSI
FÜTO- JA ZOOPLANKTONI DÜNAAMIKA JA NENDE OMAVAHELISED SUHTED KUI PEIPSI JÄRVE SEISUNDI INDIKAATORID
Dr. **Ellen-Juta Haberman**, Dr. **Reet Laugaste**, Dr. **Külli Kangur**
23. aprill 2012

ANTS VAIN

CORRECTING AND CALIBRATING AIRBORNE LASER SCANNING INTENSITY DATA USING NATURALLY AVAILABLE TARGETS
AEROLASERSKANEERIMISE INTENSIIVSUSE PARANDAMINE JA KALIBREERIMINE LOODUSLIKKE PINDASID KASUTADES.
Dr. **Natalja Liba**, Prof. **Kalev Sepp**
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