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**Effect of manure application on carbon budget
in managed grassland and corn field in southern
Hokkaido, Japan**

〔 北海道南部の草地と飼料畑における炭素収
支への堆肥施与効果 〕

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PH.D. Dissertation

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Chapter 1. General Introduction

Recently, research in natural resources management, greenhouse gasses emission and agro-ecosystem sustainability has attracted more and more attention. Carbon dioxide (CO_2) transfer in agricultural ecosystems is important not only for understanding greenhouse gasses emission from the agricultural ecosystem but also provide the mechanism for photosynthesis, respiration and crop yield formation. These transfers determine whether an ecosystem is a source or sink of carbon (C). Between 1970 and 2004, global emissions of greenhouse gasses increased by 70%, from close to 29–49 Gt CO_2 -eq, with CO_2 as the largest source, representing an increase of approximately 80% (Barkert *et al.*, 2007). The rate of increase of atmospheric CO_2 concentration has been about 1.5 ppm (0.4%) per year over the past two decades (IPCC, 2001). Moreover, CO_2 is the most abundant trace gas and is radioactively active in the atmosphere responsible for 50% of the greenhouse warming effect (Bouwman, 1990). Therefore, quantifying the CO_2 exchange provides valuable knowledge into the dynamics of the long-term carbon ecosystem.

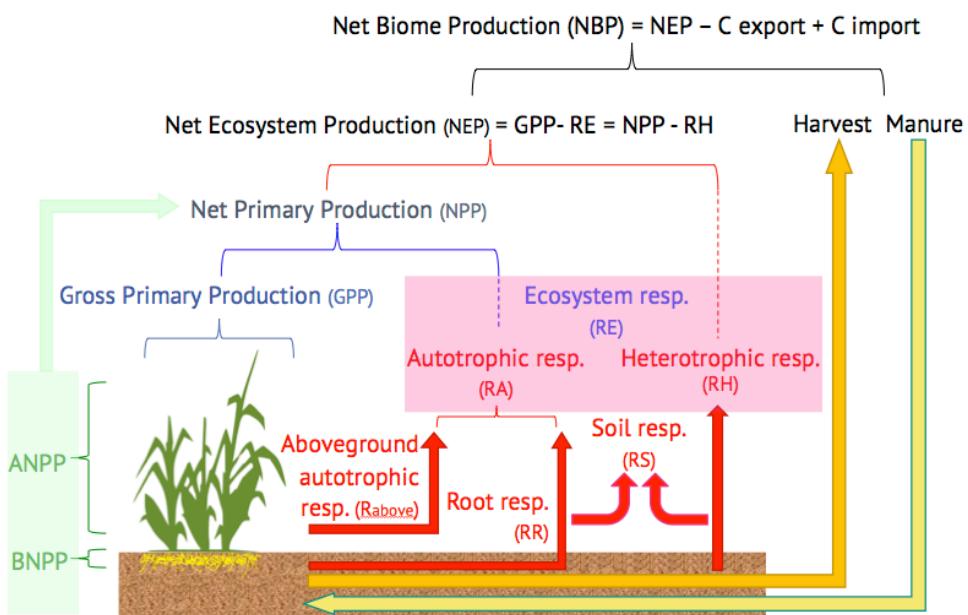


Fig. 1.1 Schematic diagram of C balance.

The C moves into and out of the terrestrial ecosystem through several processes. The balance of C can be calculated by the differences between the rates of C input and output. The vast majority of the input C into an ecosystem is through photosynthesis. The C is then partitioned among various plant parts, such as leaves, stems and roots. The next C transfers are from the decay of organic matter in litter and soil. C in the ecosystem is then released back into the atmosphere through respiration. These internal C processes are universal although their rates vary with ecosystems and environments. Fig. 1.1 shows the movement of C in terrestrial ecosystem in this study. The main C import into the ecosystem occurs via photosynthesis as assimilation of atmospheric CO₂ (gross primary production; GPP). This uptake is counterbalanced to a large part by respiration of plants and soil micro-organisms (ecosystem respiration; RE). The difference between these two processes (GPP - RE) is denoted as net ecosystem production (NEP). The NEP represents mainly the amount of C used to produce new biomass. Therefore, NEP is also described as the difference between net primary production (NPP) and soil heterotrophic respiration (RH). Management practices such as harvest and manure application can affect the C balance in the ecosystem. When aboveground biomass is harvested, the contained C is exported from the ecosystem. In the case of manure applications, C is imported into the ecosystem and has to be considered as well in the C budget. The sum of all C imports and exports equals the change of the C stored in the grassland soil (net biome production; NBP)

Grassland ecosystems, which cover approximately 40% of the earth's terrestrial area, are important ecological and socio-economic resources through their contribution to biodiversity and their use for feed, forage, and livestock production

(White *et al.*, 2000; Soussana *et al.*, 2007). Many studies have reported that grassland ecosystems function as potential C sinks or are near equilibrium with respect to C exchange. For example, a southern Great Plains mixed-grass prairie has been identified as carbon sink (Sims and Bradford, 2001); however a native tallgrass prairie in Texas (Dugas *et al.*, 1999), non-grazed mixed-grass prairie in North Dakota (Frank and Dugas, 2001), and tallgrass in Oklahoma (Suyker *et al.*, 2003) were found to be near equilibrium in terms of C. Although these studies suggested that grassland might be C sinks or near equilibrium, alternation between acting as C sink and C source frequently occurs. For example, a switch from sink to source was observed in a pasture in the southern Great Plains (Meyers, 2001), in a Canadian temperate mixed prairie during drought (Flanagan *et al.*, 2002), and in a warm temperate grassland in southeastern U.S after harvesting (Novick *et al.*, 2004).

Studies on C storage suggest that most of the C in grassland originates from belowground biomass (Hungate *et al.*, 1997; Jackson *et al.*, 2002). Grasslands have greater root biomass compared to annual crops, and grass roots play an important role in the C cycle and in C storage (Dupont *et al.*, 2010). McNaughton *et al.* (1998) and Luo *et al.* (2009) measured the belowground net primary production (BNPP) by using root soil cores and estimation of BNPP from root biomass and root turnover rates and found that BNPP could represent more than half of the total net primary production (NPP), and represents the major input of organic matter into soil. However, knowledge of BNPP is limited compared to aboveground NPP (ANPP) due to the methodological difficulties associated with observing and measuring root biomass (Milchunas and Lauenroth, 2001; Wu *et al.*, 2011). The NEP is defined as the difference between NPP and RH. In this study, NEP was provided by eddy covariance measurement directly, while ANPP and RH were provided by biometric measurement

to assess the C budget and dynamics. Therefore, BNPP was estimated by using the combination of eddy covariance and biometric methods.

Land use change is regarded as the second largest anthropogenic source of greenhouse gas emission (IPCC, 2000). Most changes in land use affect the amount of C held in vegetation and soil by either releasing CO₂ to, or removing it from the atmosphere. Organic C storage in the soil is estimated to be greater than that in the atmosphere or biosphere (Batjes, 1996). Consequently, a small change in the soil C pool may affect the atmospheric CO₂ concentration (Wang *et al.*, 1999). Soil surface CO₂ efflux is the main pathway of C emission from the soil to the atmosphere (Peng *et al.*, 2008). Between 1750 and 2011, land use change activity (mainly from deforestation) has released 180 ± 80 Pg C to the atmosphere (Ciais *et al.*, 2013). Several studies have reported that the land use change could increase the soil CO₂ fluxes of RS. Grover *et al.* (2012) measured the soil CO₂ flux of RS by using automated chambers and found that the conversion from savanna to pasture in north Australia increased the soil CO₂ flux of RS by 30%. However, land use change also can decrease or have no effect on soil CO₂ flux of RS. Sheng *et al.* (2010) measured soil CO₂ flux of RS using automated soil CO₂ flux system (LI-8100, Li-Cor Inc) in the subtropical region of China and found that the annual soil CO₂ flux of RS was reduced by 32% following the conversion of natural forest to secondary forest, 46-48% to plantations, 63% to citrus orchard and 50% to sloping tillage land, with the average reduction of 48%. Liu *et al.* (2011) measured soil CO₂ flux of RS using static closed chamber method in subtropical region of China and found that the conversion of the natural subtropical broadleaf evergreen forest to Moso bamboo does not increase soil CO₂ flux of RS.

Management practices and climatic conditions could affect the ecosystem's capacity as either C source or sink (Peng *et al.*, 2009; Sheng *et al.*, 2009; Shimizu *et al.*, 2009). Some management practices that increase the amount of organic C added to the soil such as continuous crop cover, elimination of fallow, increasing yield or manure application will potentially aid in sequestering atmospheric C (Janzen *et al.*, 1998, Shimizu *et al.*, 2009). Practices such as tillage and crop residue removal decrease the amount of organic C in the soil.

Several studies (Zenone *et al.*, 2011; Zenone *et al.*, 2013) have pointed out that the conversion of natural habitats to cropland would release CO₂. Zenone *et al.* (2013) found that the conversion from perennial grassland into no-tillage cropland induced a large C emission and caused the ecosystem to become a C source. However, the knowledge of effect conversion of perennial grassland into annual cropland with continuous manure application treatment on the C budget is limited, as manure application could enhance the C sequestration (Shimizu *et al.*, 2009). In Shin-Hidaka city located in southern Hokkaido, large-scale arable land farming is practiced under a crop rotation system between grassland and corn field with manure application treatment.

The effect of land use conversion on the C budget with the manure application treatment is continuously measured without disturb the study area. Eddy covariance and chamber method are two main methods used for measuring ecosystem CO₂ fluxes. The main advantages of eddy covariance method are the absence soil disturbance during measurements and the ability to measure fluxes continuously for long period of time in relatively large area. Chamber method is used to measure the spatial and temporal change of CO₂ flux in a relatively small area. Its main advantages are the easy of use, low cost, and ability to measure the specific C

component. Therefore, in this study, the C budget was examined by using both eddy covariance method and chamber method.

The aim of this study was to understand the effect of management practices (i.e. manure application) on the different components of C cycle and their influence on the C budget in managed grassland and corn field. Thus, the objectives of this study are: (1) to determine a reasonable method to estimate BNPP, especially in managed grassland, (2) to clarify the effect of management practices such as harvest and continuous manure application on the C cycle in grassland, (3) to find out the effect of land use conversion from managed grassland to corn field on the various components [C input (manure application) and output (RH, RS, NPP, harvest)] of the C budget.

This dissertation is divided into six chapters. The current chapter (Chapter 1) is a general introduction to the study. Chapter 2 provides a review of the relevant literature on the research topic. Chapter 3 describes the materials and methods used during the research. Chapter 4 reports on the evaluation of manure application on the C balance of managed grassland. Chapter 5 mentions about the land use change effect on C balance from a managed grassland to corn field. Finally, a general discussion on the result of this study is presented in Chapter 6

Chapter 2. Literature review

2.1 Carbon balance of an ecosystem

The C cycle is one of the most studied topics in the natural sciences (Fenn *et al.*, 2015; Xu *et al.*, 2014; Zanotelli *et al.*, 2015). The increasing concentration of CO₂ in the atmosphere since the industrial revolution is the most significant human influence on the global environment (Malhi *et al.*, 2002), driving much of the research on the global C cycle. The global C cycle involves the C cycling among the atmosphere, the oceans, and the vegetation and soils of the earth's terrestrial ecosystems. Atmospheric CO₂ increased by 40% from 278 ppm in 1750 to 390.5 ppm in 2011 (Ciais *et al.*, 2013) with anthropogenic CO₂ emissions to the atmosphere of about 555 ± 85 Pg C over the same period (IPCC, 2013; Ciais *et al.*, 2013). Of this amount land use change (including deforestation, afforestation and reforestation) contributed 180 ± 80 Pg C. The increase in CO₂ emissions from fossil fuel burning and from land use change is the dominant cause of the observed increase in atmospheric CO₂ concentration (Ciais *et al.*, 2013).

The C balance of an ecosystem at any point in time is the difference between its C gains and losses (Heimann and Reichstein, 2008). Carbon enters the ecosystem through photosynthesis, the processes by which autotrophs (plants and photosynthetic bacteria) produce their own organic compounds; most of this C returns to the atmosphere by autotrophic and heterotrophic (animals, fungi and micro-organisms) respiration (others losses of C could be as volatile organic compounds, methane or dissolved C).

The metabolic processes of organisms hence constitute the engine that drives the global C cycle on time scales of seconds (e.g., photosynthetic rates,) to centuries (large-scale climatic changes) (Chapin *et al.*, 2011). The major C pools are: the

atmosphere, oceans, vegetation and soil, sediments and rocks. Within these, while the atmosphere is the smallest but most dynamic C pool, the terrestrial biosphere is the largest biological C reservoir (Chapin *et al.*, 2011).

2.2 Environmental control on C balance

Components of the C balance such as photosynthesis and respiration often depend upon more than one factor. Net ecosystem exchange (NEE), the balance between photosynthetic uptake and release of CO₂ by ecosystem respiration from autotrophic and heterotrophic, represent the carbon sequestration between terrestrial ecosystem and the atmosphere during a given period. A number of environmental factors can play important roles in governing the rate of NEE. Several studies have shown that micrometeorology parameters such as air temperature, precipitation and photosynthetically active radiaton (PAR) controlled the dynamics of NEE. Wang *et al.* (2008) studied the environmental effects on NEE for three years in a typical *Stipa krylovii* steppe in northern China using eddy covariance method and found that: (1) magnitude of NEE was strongly regulated by annual precipitation, (2) soil water content was important environmental factor regulating the variation of NEE, and (3) air temperature influenced the NEE-PAR relationship. The NEE between an ecosystem and the atmosphere is the net balance of GPP and RE. Chen *et al.* (2013) reviewed the environmental effects on the GPP and RE. The NEE of different terrestrial ecosystems in the Asian region was quantified based on long-term observation data of ChinaFlux (19 sites) and published data from AsiaFlux (37 sites) and 32 other sites in Asia. The studies found that the spatial variation of GPP and RE were mainly controlled by climatic factors such as mean annual temperature and mean

annual precipitation. The GPP is also affected by abiotic factors such as PAR, temperature and soil moisture (Lambers *et al.*, 1998)

Soil respiration (RS) is the sum of respiration from ground vegetation, roots, rhizosphere, mycorrhizia and microbes. There are many different factors that control RS, but temperature and moisture are the dominant factors (Lloyd and Taylor, 1994; Davidson *et al.*, 2000; Swanson and Flanagan, 2001). The temperature sensitivity (Q_{10}) of RS varies under different temperature ranges (Kirschbaum, 1995) and with the different RS components such as root and microbes (Boone *et al.*, 1998; Janssens *et al.*, 2003). Temperature and respiration from the different components fluctuate seasonally and the temperature sensitivity differs accordingly (Rayment and Jarvis, 2000; Widén, 2000).

2.3 Eddy covariance technique

Eddy covariance measurements were first made about 40 years ago, but became more widely used in the late 1980s (Baldocchi *et al.*, 1988). More recently, eddy covariance technique is used to evaluate the exchange of C between the vegetation and atmosphere over a variety of ecosystems (Barr *et al.*, 2007; Dunn *et al.*, 2007).

Eddy covariance method provides a direct measurement of the net exchange of CO₂, water vapor and sensible heat between a vegetated surface and the atmosphere (Baldocchi *et al.*, 1988). The technique relies on atmospheric turbulence generated from the movement of air immediately above the earth's surface. Atmospheric turbulence causes the formation of eddies which transport heat and mass between the surface and free atmosphere. The basic principle of this technique is that the vertical flux can be calculated as covariance between concentration of the entity of interest (e.g., CO₂) and vertical wind speed in the eddies (Burba and Anderson, 2007).

Therefore, during periods of C uptake by the ecosystem, CO₂ rich air moves down into the ecosystem, while CO₂ poor air is correlated with upward vertical wind velocity. The covariance of the atmospheric CO₂ concentration and the vertical wind gives an estimate of the net amount of C absorbed or released by the ecosystem, over a given area at half-hourly timescales with minimal disturbance to the underlying vegetation, providing a reliable measure of NEE (Balocchi *et al.*, 1988). This technique provides accurate and continuous measurements of NEE at an ecosystem spatial scale. From these measurements of NEE, GPP and RE can be derived (Reichstein *et al.*, 2005; Wohlfahrt *et al.*, 2005a; Lasslop *et al.*, 2010). This flux partitioning was classified in those that use only night time data for the estimation of RE and those that exploit day time data or both, day and night time data using light response curve. For the estimation of the temperature sensitivity, the RE is related to temperature using the exponential regression model (Lloyd and Taylor, 1994). Then for the day and night time data, RE and GPP are simultaneously modeled as parts of one model equation with state dependent parameters, where RE is y-intercept from light response curve of GPP (Falge *et al.*, 2001).

In mathematical terms, eddy flux is computed as the covariance between the instantaneous deviation in vertical wind speed (w') from its mean value (w) and the instantaneous deviation in gas mixing ratio (c'), from its mean value (c) and multiplied by mean air density; the vertical flux can be expressed by the following general equation (Balocchi, 2003):

$$F = \overline{\rho_a} \overline{w'c'}$$

where overbar denotes temporal averaging (e.g. half-hour or hour), the prime denotes the deviation from the mean, ρ_a , the air density, w and c vertical wind speed and gas concentration, respectively. This equation of flux is derived from vertical flux

equation by using Reynold decomposition (i.e. break equation into means and deviations). Air density fluctuations and divergence or convergence of turbulence flux are considered to be negligible (Burba and Anderson 2010). The example of eddy covariance at a single point was shown in Fig. 2.1.

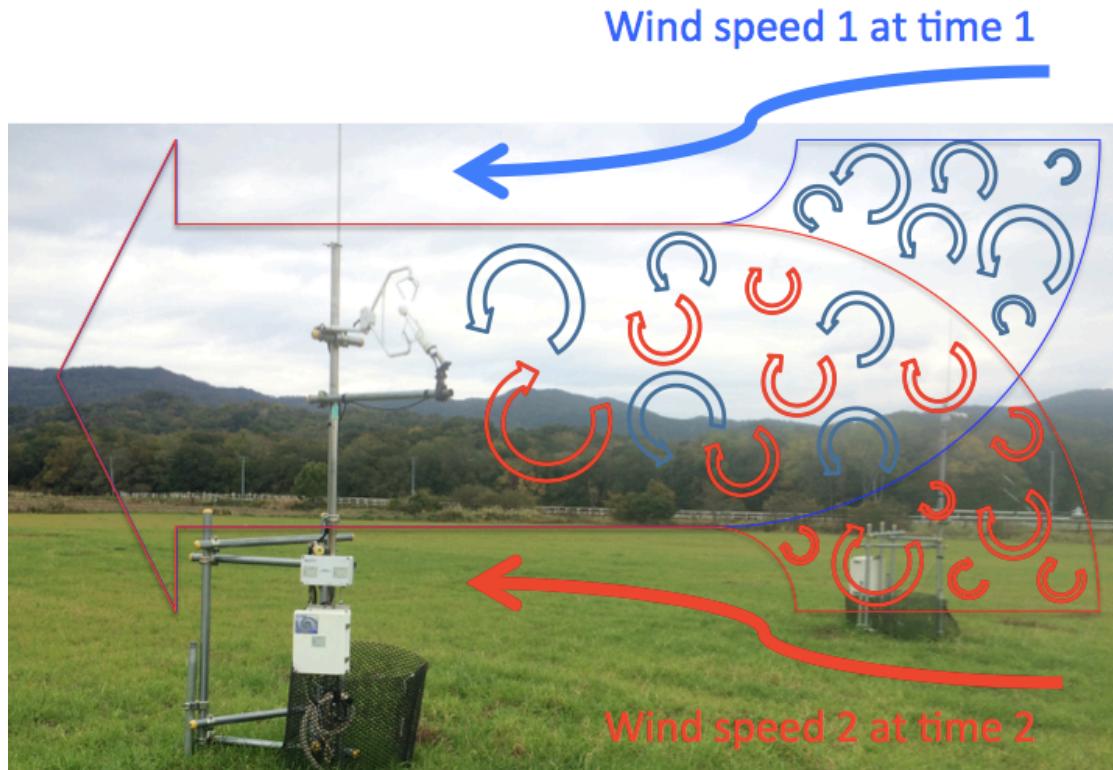


Fig. 2.1 Example of eddy covariance at a single point (modified from Burba and Anderson, 2010). The amount of gas that observed by eddy covariance at single point can be determined by the difference in amount of molecules gas that moves upward (red color curve) and downward (blue color curve) at almost same time (time 1 ~ time 2) and with a certain wind speed (wind speed 1 and 2)

As a convention, when the net flux is towards the vegetation, indicating net uptake of CO₂ by ecosystem, NEE has a negative sign, and when the net flux is away from the surface, indicating release of CO₂ from ecosystem to the atmosphere, NEE has a positive sign.

The NEE can also be expressed as:

$$-NEE = GPP - RE; \text{ as eddy covariance system computes the net flux of CO}_2.$$

The other components of the carbon cycle such as GPP and RE could be estimated by using modeling techniques (Llyod and Taylor equation (RE); light response curve (GPP)).

Common eddy covariance instrumentation is composed of a 3-dimensional (3-D) sonic anemometer for the measurement of wind speed in the three components (w , u , v) and an infrared gas analyzer (IRGA) to evaluate atmospheric mixing ratios of CO₂ and H₂O (or another gas of interest). Instruments are mounted on a tower above the vegetation, at a height that depends on site characteristics (height of the vegetation, extent of the fetch, range of wind velocity and frequency response of the instruments). The turbulent up and down motions occur in part at very small temporal scales and in part at larger ones, for this reason eddy covariance requires sophisticated instrumentation to capture these fluctuations. Several instrument models to perform measurements with very high frequencies such as 10 or 20 Hz have been developed, and are employed worldwide. An example of eddy covariance typical configuration is presented in Fig. 2.2.

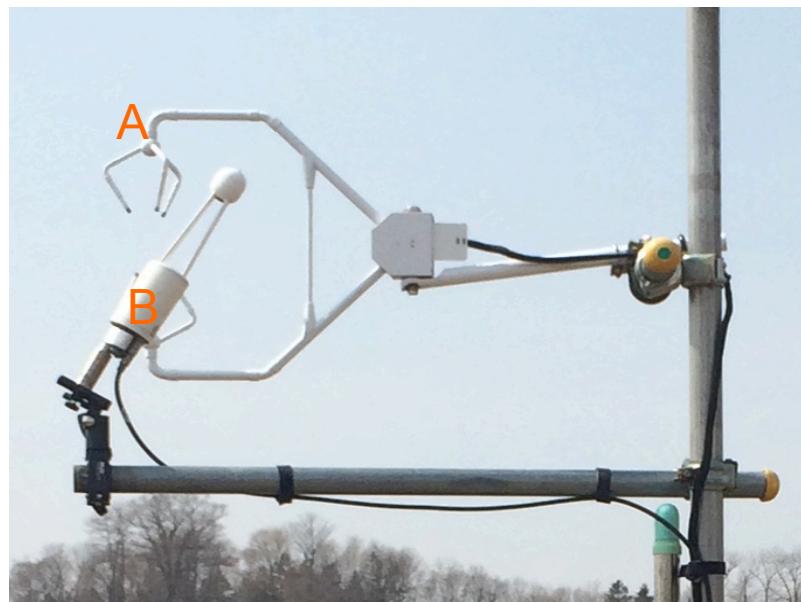


Fig. 2.2 Example of an eddy covariance installation, composed by (A) a 3-D sonic anemometer (CSAT3, Campbell Scientific, Inc) and (B) open path IRGA (LI-7500, LI-COR, Inc., Lincoln, NE, USA).

Two different functional modalities exist for IRGA: closed-path system and open-path system. In closed-path systems air is actively pumped to the sample cell through a tube, while open-path gas analyzer assesses “*in situ*” free air. Both have advantages and disadvantages, and are generally more or less suitable to different environmental conditions. Open-path sensors require low power supply, don’t have pump and tube and are suitable for harsh environments, on the other hand, they suffer of loss of data when some environmental conditions such as precipitation and snow cover or make dirty the sensor window. Open-path sensors do not directly measure CO₂ and H₂O mixing ratios but their densities. As atmospheric gas densities could be influenced by fluctuations in pressure, temperature and humidity that generate expansion and contraction of air volume, corrections (Webb *et al.* 1980, WPL) to compensate these effect on measured fluxes are required. Closed-path analyzers can output directly gas mixing ratios, making density corrections (WPL) less important, they don’t suffer of data loss due to precipitation and icing but have a high power requirement and are affected by CO₂/H₂O signal tubing attenuation.

Removing bad data is an important part of the data quality control processes. It ensures that results do not have a bias or errors due to some obvious or common reasons. Bad data are usually removed for one of the following causes: instrument malfunctions, processing or mathematical artifacts, ambient condition not satisfying eddy covariance method, winds are not from the footprint of interest, and heavy precipitation (Burba and Anderson 2010).

2.4 Effect of land use change on carbon balance

Land use and land-use change directly affect the exchange of greenhouse gases between terrestrial ecosystems and the atmosphere (IPCC, 2000). Changes such as the clearing of forests for use in agriculture or as settlements are associated with clear changes in land cover and carbon stocks. Much of the world's land area continues to be managed for food and wood production, human habitation, recreation, and ecosystem preservation without a change in land use (Briassoulis, 2000). Management of these land uses affects sources and sinks of greenhouse gases (IPCC, 2000). Furthermore, the resulting agricultural and wood products contain carbon. The carbon stocks held in these products are eventually released back to the atmosphere, after the products have served their use. Biomass carbon stocks are also used to produce energy that serves as a substitute for, and as complement to, fossil fuels.

Different factors and mechanisms drive land use and land cover transformation. In many cases, local climate and weather, topography, technology, and economics appear to be determinants of land-use change at different spatial and temporal scales (Briassoulis, 2000). Land-use change is often associated with a change in land cover and an associated change in carbon stocks (Guo and Gifford, 2002; Houghton, 2004). Houghton (1991) assessed seven types of land-use change for carbon stock changes: (1) conversion of natural ecosystems to permanent croplands, (2) conversion of natural ecosystems for shifting of cultivation, (3) conversion of natural ecosystems to pasture, (4) abandonment of croplands, (5) abandonment of pastures, (6) harvest of timber, and (7) establishment of tree plantations.

Studies on grassland CO₂ exchange have shown that they may act as either a source or sink of CO₂ (Leahy, 2004). Novick *et al.* (2004) collected information on annual grassland NEE estimates based on eddy-covariance measurements and Bowen

Ratio Energy Balance techniques and reported values varying from a net source of 400 g C m⁻² to a net sink of -88 g C m⁻². A review of available data has shown that large uncertainties remain in resolving whether grassland ecosystems function as CO₂ sources or sinks (Janssens *et al.*, 2003). This uncertainty is primarily attributable to the sensitivity of grasslands to inter-annual variability in climate and associated biomass dynamics (Meyers, 2001; Flanagan *et al.*, 2002) and incomplete understanding of the regulation of grassland assimilation and respiration (Wever *et al.*, 2002).

Land-use change has been estimated to be a major global carbon source, adding about 124 PgC yr⁻¹ to the atmosphere, with the majority being lost due to conversion of forestry to agriculture as well as changes from grasslands to croplands (Houghton, 1999; Smith, 2008). Alberti *et al* (2010), by using eddy covariance and automated chamber, investigated short-term effects (2 years) of land conversion from continuous maize (*Zea mays* L.) to alfalfa (*Medicago sativa* L.) on C balance and reported that NEP increased two years after conversion from corn to alfalfa (281 g C m⁻² yr⁻¹). However, continuous maize was a lower source of C in terms of NBP (-96 g C m⁻² yr⁻¹) than maize-alfalfa land use conversion (-354 g C m⁻² yr⁻¹). From the result, it appears that this type of land conversion is not an effective measure of C sequestration in the short term (1–3 years), but it could be in the long term.

Zenone *et al* (2011) studied the effects of land use change from perennial grassland to an annual crop (soybean) in southwestern Michigan, US. The study examined the effect of land conversion on carbon (C) fluxes using the eddy covariance technique at seven sites and quantified the effect of major management activities on C fluxes during the conversion year. Four sites had been managed as grasslands under the Conservation Reserve Program of the USDA. Three fields had

previously been cultivated in a corn/soybean rotation with corn until the year before this study. The effects of land use change were studied during 2009 when six of the seven sites were converted to soybean cultivation, with the seventh site kept as grassland. In winter (December to March), the corn fields were C neutral while the grasslands were C sources, with average emissions of $15 \text{ g C m}^{-2} \text{ month}^{-1}$. In April, first year of conversion, while the corn fields continued to be a C source to the atmosphere, the grasslands switched to C sinks. The conversion of grassland to soybean induced net C emissions with NEE ranging from 155.7 ± 25 to $128.1 \pm 27 \text{ g C m}^{-2} \text{ yr}^{-1}$. The annual NEE at the unchanged plot (grassland) was $-81.6 \pm 26.5 \text{ g C m}^{-2} \text{ yr}^{-1}$ while at the plot converted from corn/soybean rotation was remarkably different with two sites being sinks of -91 ± 26 and $-56.0 \pm 20.7 \text{ g m}^{-2} \text{ yr}^{-1}$ whereas one site was a source of $31.0 \pm 10.2 \text{ g C m}^{-2} \text{ yr}^{-1}$. This study concluded that the conversion of perennial grasslands back to an annual cropping system induced a large emission of CO₂ during the first year of cultivation, mainly due to the dead aboveground biomass present after the herbicide application and decomposition of roots. On the other hand, the undisturbed reference grassland continued to be a C sink.

2.5 Effect of management practices

Management of croplands, and grassland affects sources and sinks of greenhouse gases (CO₂, CH₄, and N₂O) (Brandyopadhyay and Lal, 2014; Linguist *et al.*, 2012; Shimizu *et al.*, 2009). Cropland soils can lose C as a consequence of soil disturbance (e.g., tillage). Tillage increases aeration and soil temperatures (Tisdall and Oades, 1982; Elliott, 1986), making soil aggregates more susceptible to breakdown and physically protected organic material more available for decomposition (Elliott, 1986; Beare *et al.*, 1994). In addition, erosion can significantly affect soil C stocks

through the removal or deposition of soil particles and associated organic matter. Erosion and redistribution of soil may not result in a net loss of carbon at the landscape level because carbon may be re-deposited on the landscape instead of being released to the atmosphere (van Noordwijk *et al.*, 1997; Lal *et al.*, 1998; Stallard, 1998). Although some organic matter may be re-deposited, the productivity of the eroded soil is decreases due to the reduced ability to support C fixation and storage. Losses through leaching of soluble organic C occur in many soils; although this leaching is seldom a dominant C flux in soils, it is a contributor to the transport of C from the terrestrial environment to the marine environment via runoff (Meybeck, 1982; Sarmiento and Sundquist, 1992). Soil C content can be protected and even increased through alteration of tillage practices, crop rotations, residue management, reduction of soil erosion, improvement of irrigation and nutrient management, and other changes in forestland and cropland management (Kern and Johnson, 1993; Cole *et al.*, 1996).

Livestock grazing on grasslands, converted cropland, savannas, and permanent pastures is the largest areal extent of land use (FAO, 1993). Grazing alters ground cover and can leads to soil compaction and erosion, as well as alteration of nutrient cycles and runoff. Soil C, in turn, is affected by these changes. Avoiding overgrazing can reduce these effects (Mscherry and Ritchie, 2013). Ecosystem conservation may also influence carbon sinks. Many forests, savannas, and wetlands, if managed as natural reserves or/and recreation areas, can preserve significant stocks of C (IPCC, 2000), although these stocks might be affected negatively by climate change (IPCC, 2000). Some wetlands and old-growth forests exhibit particularly high C densities (Erwin, 2009; Marton *et al.*, 2014) while other semi-natural ecosystems (e.g.,

savannas) may conserve C simply because of their large areal extent (Alam *et al.*, 2013).

Chapter 3. Materials and methods

3.1 Site description and treatments

This study was conducted from 2008 to 2014. The study site is located at Niikapu station, National Livestock Breeding Center, Shin-Hidaka city, Southern Hokkaido (42°24' N, 142°28' E). This region belongs to the cool-temperate zone and is characterized by cold winters and warm summers without apparent wet or dry season. The mean annual precipitation is 1365mm and the mean annual temperature is 7.9°C, with the mean monthly temperature ranging from 20.6°C in August to -5.7°C in January (Shimizu *et al.*, 2010). The site is covered with snow from the end of December to the beginning of March.

The grassland was established more than 100 years ago, and renovated in 2004 from grazed grassland. Dominant grass species in this site were orchard grass (*Dactylis glomerata* L.) and meadow fescue (*Festuca pratensis* Huds.). In late August of 2010, over-seeding of orchard grass ($10 \text{ kg}\cdot\text{ha}^{-1}$) and red clover ($5 \text{ kg}\cdot\text{ha}^{-1}$) was conducted at the study site, by using a rotary-tilling-type grassland renovator with narrow tilling device, to maintain grass productivity. Herbicide was applied to the field in mid-October 2012 to convert the grassland to corn field in 2013.

The soil is derived from Tarumae-b volcanic ash (Jin *et al.*, 2010; Shimizu *et al.*, 2010), and is classified as Andosol (IUSS Working Group WRB, 2006). The soil profile survey was conducted in September 2007 before the treatment. The soil physical and chemical properties are shown in Table 3.1. The thickness of the Ap-horizon was 26 cm and total C and total N contents of the Ap-horizon were 86 and 6.3 $\text{g}\cdot\text{kg}^{-1}$, respectively with a dry bulk density of $0.63 \text{ Mg}\cdot\text{m}^{-3}$.

Table 3.1 Detailed description of the soil characteristics.

Horizon	Depth (cm)	pH	Dry bulk density (Mg·m ⁻³)	Total C (g·kg ⁻¹)	Total N (g·kg ⁻¹)	Sand (g·g ⁻¹)	Silt (g·g ⁻¹)	Clay (g·g ⁻¹)
Ap1	0-6	5.7	0.620	85.4	5.8	0.70	0.21	0.09
Ap2	6-26	5.4	0.639	87.3	6.6	0.71	0.21	0.08
2A	26-40	5.6	0.458	180.5	10.2	0.49	0.32	0.19
3A	40-55	5.5	0.491	107.2	6.7	0.44	0.37	0.18
3C	55-100+	5.6	0.538	35.0	2.5	0.56	0.32	0.12

The survey was conducted in September 2007 before the treatment.

3.2 Management practices

3.2.1 Managed grassland

Two adjacent experimental plots (170 m × 150 m) were set up; in one plot only chemical fertilizer was applied (fertilizer plot; F plot) and the other plot was applied with manure and supplementary chemical fertilizer (manure plot; M plot). In order to measure the soil organic matter decomposition and manure decomposition, small chamber plots were set up (10 m × 10 m) (Fig. 3.1).

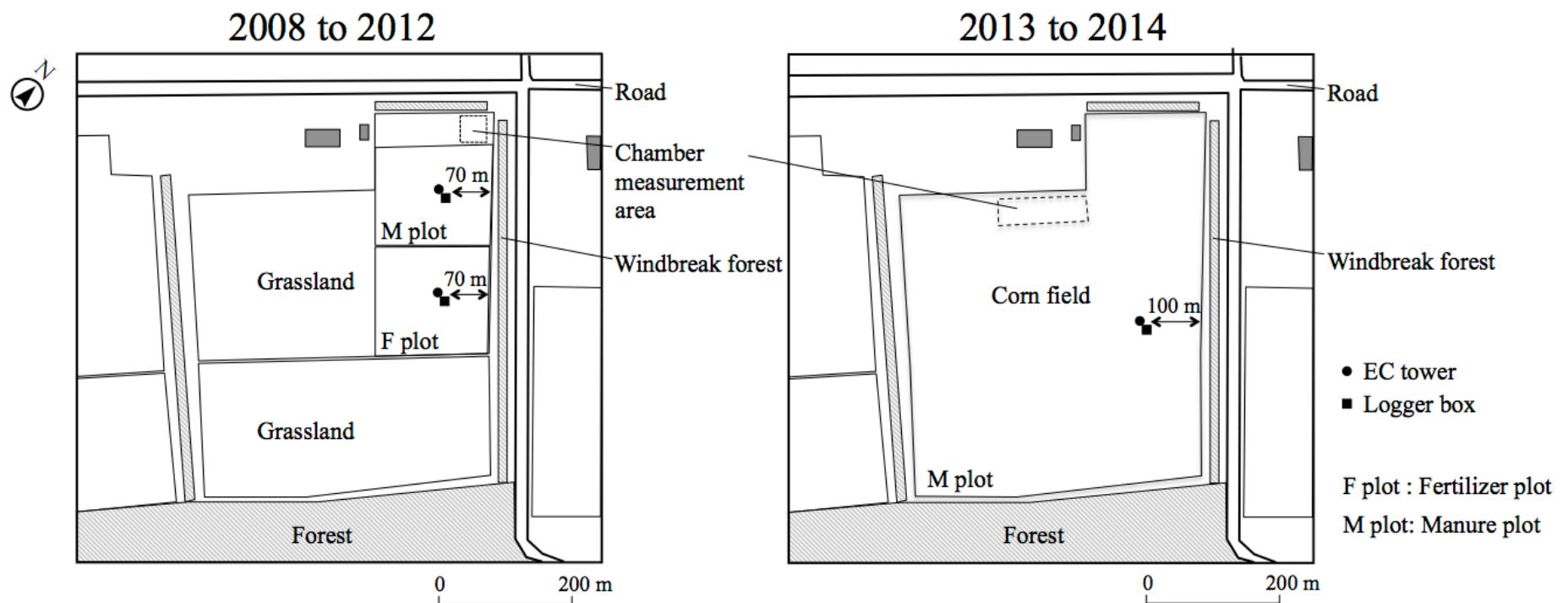


Fig. 3.1 The layout of the experimental plots.

Table 3.2 The date of chemical fertilizer and manure applications, and the amount applied in the fertilizer (F) and manure (M) plots.

Year	Date	Application rates ($\text{kg}\cdot\text{ha}^{-1}$)									
		F plot					M plot				
		Chemical fertilizer			Manure		Chemical fertilizer				
		T-N	P ₂ O ₅	K ₂ O	T-C	T-N	P ₂ O ₅	K ₂ O	T-N	P ₂ O ₅	K ₂ O
2008	Apr. 30	50	125	50	2142	205	177	367	31	81	0
	Jun. 17	20	40	40	0	0	0	0	20	54	0
	Aug. 22	20	40	40	0	0	0	0	20	54	0
2009	Apr. 24	40	80	80	2348	241	223	471	24	63	0
	Jun. 15	30	60	60	0	0	0	0	17	45	0
	Aug. 11	30	60	60	0	0	0	0	17	45	0
2010	Apr. 26	67	33	0	1821	152	133	256	33	17	0
	Jun. 18	22	22	0	0	0	0	0	17	17	0
2011	May 5	28	75	100	2025	136	75	165	17	17	0
	Jun. 21	19	50	0	0	0	0	0	17	17	0
	Aug. 22	19	50	0	0	0	0	0	0	0	0
2012	May 14	33	67	67	1280	63	31	39	17	83	0
	Jun. 14	17	33	33	0	0	0	0	0	0	0
2013	Jan. 21	150	354	0	5446	344	204	421	103	313	0
2013	Oct. 16	136	0	0	4089	174	101	181	111	51	0

The harvesting of grass was carried out two or three times per year. In 2010, the 3rd crop was cut but was not removed from the field because of low volume of production. The 3rd crop of 2012 was also not harvested due to the herbicide application for renovation. The information on the date of application and the application rates of fertilizer and manure are given in Table 3.2. The fertilizer application rates in the F plot ranged from 50 to 100 kg N ha⁻¹·yr⁻¹. The amount of manure applied to the M plot was 10 Mg fresh matter ha⁻¹·yr⁻¹ (1923 ± 407 kg C ha⁻¹·yr⁻¹, 159 ± 68 kg N ha⁻¹·yr⁻¹). In the M plot, the nutrient supply rate from manure was estimated by multiplying the application rate by the mineralization rate, and the difference between the supply rate from manure and the application rate in F plot was supplied with chemical fertilizer. The N mineralization rate was estimated based on Uchida's model, which was developed in Japan (Shiga et al., 1985). According to this model, the cumulative N mineralization rate ($1 - y_t$) was estimated to be 0.132 in the applied year ($t = 1$), 0.202 in two years ($t = 2$), and 0.257 in three years ($t = 3$), respectively. The equation is as follows:

$$y_t = a \times 0.01t + c \times 0.63t + f \times 0.955t \quad [3.1]$$

where y_t is the rate of N remaining in the manure t years after application, t is the years after manure application (inclusive of the applied year), a , c , and f are the rates of organic matter fractions with different decomposition rates ($a + c + f = 1$). In the case of cattle manure, a , c , and f were estimated to be 0.04, 0.15, and 0.81, respectively. Therefore, the annual N mineralization rates were 13.2, 7.0, 5.5, 4.6, and 3.9% of the total manure N in the first to fifth years after manure application, respectively. The rate of phosphorus and potassium mineralization from the manure was estimated based on a handbook on animal waste management and utilization in

Hokkaido published in 2004 (Hokkaido Prefectural Experiment Stations and Hokkaido Animal Research Center, 2004), namely 20% of phosphorus and 70% of potassium are available in the applied year, 10% of phosphorus and 10% of potassium are available the second year, respectively.

We defined the crop-growing season as a 7-day moving average of daily air temperature above 5 °C and the non-growing (NG) season as the rest. The length of the growing season ranged from 202 to 228 days. The growing season was divided into three or four periods; from the beginning of the growing season to the first crop harvest (G1), from the first harvest to the second harvest (G2), from the second harvest to the third harvest (G3), and from the third harvest to the end of the growing season (G4) (Table 3.2). In 2010 and 2011, the 3rd crop was cut towards the end of the growing season, and was not calculated for the G4 period. We did not estimate the G4 period in 2010, 2011, and 2012 due to the late cutting or herbicide use in those years.

3.2.2 Corn field

Corn field was established from 2013 to 2014. There was only one experimental plot; applied by manure and supplementary chemical fertilizer. The study site was ploughed in the end of April. In the beginning of May, corn was planted and was harvested at the end of October. The amount of manure applied in the corn field was 50 Mg Fresh matter ha⁻¹·yr⁻¹ (4864 ± 823 kg C ha⁻¹·yr⁻¹, 268 ± 108 kg N ha⁻¹·yr⁻¹). The nutrient supply rate from manure and N mineralization rate was estimated the same way as mentioned previously for the managed grassland.

3.3 Eddy covariance system and meteorological station

All measurements were performed according to those carried out in a previous study conducted at Shizunai experimental farm (Hirata *et al.*, 2013). An eddy covariance system was set up in both F and M plots in the managed grassland and M plot in the corn field to measure net ecosystem exchange (NEE) of CO₂. We employed an open-path eddy covariance system, which consisted of a three dimensional sonic anemometer (CSAT3; Campbell Scientific, Logan, UT, USA), open-path infrared gas analyzer (IRGA) (LI-7500; Li-Cor, Lincoln, NE, USA), and a data logger (CR23X, Campbell Scientific) equipped with a compact flash recorder. The sensor head of the sonic anemometer and IRGA were set at 2.45 m and 5.4 m above the ground for managed grassland and corn field, respectively. The IRGA was calibrated in the laboratory at least once a year using dry zero gas (pure air, CO₂ < 0.1 ppm), CO₂ span gas (500 ppm CO₂ in air), and H₂O span gas supplied from a dew-point generator (LI-610, Li-Cor).

Air temperature and relative humidity were measured at the same height that the flux measurement was taken using a temperature-humidity sensor (HMP-45A, Vaisala, Helsinki, Finland) equipped with a homemade ventilator. Incoming photosynthetically active radiation (PAR) was measured above the soil surface (1.6 m in managed grassland and 5.4 m in corn field), using a quantum sensor (LI-190SL, Li-Cor). All meteorological measurements were sampled at 1 s intervals and averaged over 30 min. Gaps in the meteorological data due to power failure were filled with data from the nearest Automated Meteorological Data Acquisition System (AMeDAS) station of the Japan Meteorological Agency. Precipitation data were obtained from the nearest AMeDAS station, which is located about 20 km from the study site.

Half-hourly CO₂ fluxes were calculated from covariance between vertical wind speed and CO₂ density. We used standard procedures (Mano *et al.*, 2007) to correct the coordinate rotation of the wind field, the influence of water vapor flux on sonic thermometry, high frequency of cospectral losses due to sensor separation and path-length averaging, and the influence of air density fluctuations due to variation of air temperature and water vapor. Non-stationarity and integral turbulence tests (Foken and Wichura, 1996) were applied to discard any erroneous half-hourly flux data obtained under unfavorable atmospheric conditions or by malfunctioning instruments. Because the measurement height was 2.5 m in managed grassland and 5.4 m in corn field, the changes in CO₂ storage below this height were neglected, and the CO₂ flux was therefore regarded as NEE. Fluxes were also discarded for periods when friction velocity (u^*) was below the critical threshold of 0.05 m·s⁻¹ (Hirata *et al.*, 2013). In total, data gaps caused by sensor malfunctioning, exclusion of data from the afforested sector, and data filtering during post-processing steps required gap-filling of 32 ± 2 and $34 \pm 5\%$ of the dataset during the observation period for F and M plots, respectively. In corn field, total of $45 \pm 10\%$ of data was gap-filled. NEP has the same magnitude as NEE but is represented by the opposite mathematical sign, i.e., positive NEP represents C uptake by the ecosystem.

We used the following equation according to Lloyd and Taylor (1994) to fill gaps in the nighttime NEE ($F_{\text{NEE,night}}$):

$$F_{\text{NEE,night}} = F_{\text{RE,T}_{\text{ref}}} \exp \left\{ \frac{E_0}{R} \left(\frac{1}{T_k + T_{\text{ref}} - T_0} - \frac{1}{T_K + T_a - T_0} \right) \right\} \quad [3.2]$$

where $F_{\text{RE,T}_{\text{ref}}}$ is the $F_{\text{NEE,night}}$ at the reference temperature (T_{ref}), which was fixed at 10°C, E_0 is the activation energy (J·mol⁻¹), R is the ideal gas constant (8.31 J·mol⁻¹·K⁻¹) and T_a is the air temperature (°C). The values of T_K and T_0 are 273.15 and 227.13 K,

respectively. The fitted parameters, E_0 and $F_{\text{RE,Tref}}$ were determined each day for a 15-day moving window by means of the least-squares method.

Data gaps in daytime NEE ($F_{\text{NEE,day}}$) were filled by the following non-rectangular hyperbola equation as a function of PAR (Q) (Prioul and Chartier, 1977).

$$F_{\text{NEE,day}} = \frac{\phi Q - P_{\max} + \sqrt{(\phi Q + P_{\max})^2 - 4\phi Q \theta P_{\max}}}{2\theta} + R_d$$

[3.3]

In this equation, P_{\max} , ϕ , θ (= 0.9) and R_d are the maximum GPP at light saturation ($\mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$), the initial slope ($\text{mol}\cdot\text{mol}^{-1}$), the convexity of the light response curve and daytime respiration ($\mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$), respectively. The fitted parameters, P_{\max} , ϕ and R_d were determined each day for a 7-day moving window by means of the least-squares method.

NEP is the difference between photosynthetic CO₂ assimilation by plants (gross primary productivity; GPP) and respiratory CO₂ released from plants and soil (ecosystem respiration; RE). We divided NEE into GPP and RE using a nonlinear empirical model. Nighttime RE equals nighttime NEE, because nighttime GPP is regarded as zero. Daytime RE was estimated from air temperature by extrapolation using Eq. 3. Daytime GPP was estimated by subtracting the observed NEE from daytime RE.

3.4 Measurement of soil respiration (RS) and heterotrophic respiration (RH)

Soil respiration (RS) was estimated as the soil surface CO₂ flux, which includes respiration of plant roots, rhizosphere, microbes and fauna, while RH in this study was estimated as the soil surface CO₂ fluxes from the root exclusion plot (Shimizu *et al.*, 2009). In managed grassland, four root exclusion plots (1 m × 1 m) were established only in 2011 and 2012 in the chamber area (Fig. 3.1). In the root exclusion

plots in managed grassland, chemical fertilizer was applied at the same application rate and timing as in F plot, and manure was not applied. The soil surface CO₂ flux from the root exclusion plot with chemical fertilizer treatment is denoted as RHs. In corn field, the root exclusion plots (1 m × 1 m) were established in 2013 and 2014 in the chamber area (Fig. 3.1). In the root exclusion plots in corn field, manure was applied at the same amount and timing as M plot. In order to obtain heterotrophic respiration of manure in corn field (RHm), root exclusion plots with manure application (RH) and with only fertilizer application (RHs) were installed in the chamber area and RHm was obtained from the difference between RH and RHs. In root exclusion plot, the aboveground plants and roots were removed with a shovel and by hand, and a root-proofing permeable sheet (BKS9812; TOYOBO, Osaka, Japan) was vertically inserted to a depth of 20 cm below the ground surface to inhibit regrowth of roots. The surface of the root exclusion plots was covered by a 1-mm-mesh nylon net to prevent the soil surface from aggregate degradation by rain, because this may enhance organic matter decomposition (Lundquist *et al.*, 1999; Casals *et al.*, 2009). The net could also prevent an increase of daily maximum soil temperature in the root exclusion plot.

Soil respiration and RH were measured using the static closed chamber method (Shimizu *et al.*, 2009) (Fig. 3.2). The flux measurements were conducted bi-weekly during the crop-growing season, but the frequency was increased to twice per week after fertilization. The measurements were conducted monthly during the non-growing season. In managed grassland, two types of chamber were used. Small chambers which were made from stainless steel chambers (20 cm diameter and 25 cm high) were used to measure RH and big chambers (40 cm diameter and 30 cm high) were used to measure RS. In corn field, RH and RS was measured using the small

chambers only. The small chambers were placed on chamber-bases made of stainless steel with a diameter of 20 cm, which were kept on the ground except during harvesting. The big chamber was installed at a depth of 3 cm into the soil one day before the measurement day. The headspace gas samples from each chamber were collected and placed into a Tedlar bag before the chamber was closed and after 6 min under a closed-chamber condition. The CO₂ gas concentrations were determined in the laboratory using a CO₂ infrared gas analyzer (ZFP9GC11, Fuji Electric, Tokyo, Japan). Soil respiration and heterotrophic respiration were calculated using a two-point regression of CO₂ concentration in the chamber (Nakano *et al.*, 2004).

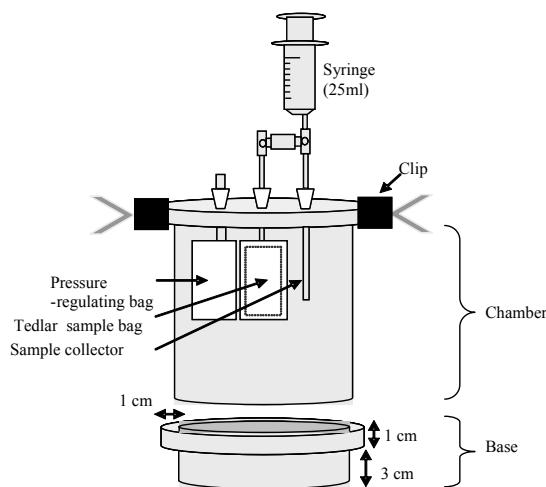


Fig. 3.2 Outline of the chamber and base used in the closed chamber method (Jin, 2010; Toma and Hatano, 2007).

Gas fluxes were calculated from the change in gases concentration in the chamber using following equation

$$F = \rho \times h \times (\Delta c / \Delta t) \times [273 / (273 + T)] \quad [3.4]$$

where F is the gas flux ($\text{mg C m}^{-2} \text{ h}^{-1}$), ρ is the gas density ($\text{CO}_2\text{-C} = 0.538 \times 10^6 \text{ mg m}^{-3}$), h is the height of the chamber from soil surface (m), $\Delta c / \Delta t$ is the change in gas concentration inside the chamber during the sampling period ($\text{m}^3 \text{ m}^{-3} \text{ h}^{-1}$), T is the air

temperature inside the chamber (°C). A positive flux denotes the emission from the soil, whereas a negative flux denotes the uptake from the atmosphere.

Air and soil temperature were measured concurrently at depths of 5 cm with the CO₂ flux measurements using a thermistor thermometer (CT220, CUSTOM, Tokyo, Japan). Soil temperatures at 5 cm depth were also monitored every 30 min with a thermocouple thermometer (TR-52, T&D, Nagano, Japan). Relationship between the soil surface CO₂ flux from the root exclusion plot (Grassland: F plot; Corn field: F and M plots) and soil temperature at 5 cm depth was fitted by Lloyd and Taylor equation (Eq. 3.2). The RHs in managed grassland (F and M plots) and cornfield (F plot) and RH in M plot in corn field were estimated using the Lloyd and Taylor equation driven by the 30-minute mean soil temperature in the F and M plots. The RH in M plot in managed grassland was estimated by the sum of RHs in M plot and heterotrophic respiration of manure (RHm).

Q_{10} is the temperature sensitivity of respiration and is calculated by the following

$$Q_{10} = \exp 10 \left\{ \frac{E_0}{R} \left(\frac{1}{T_k + T_{ref} - T_0} - \frac{1}{T_K + T_a - T_0} \right) \right\} \quad [3.5]$$

In managed grassland, the RH of manure (RHm) was estimated as the difference of RE between M and F plots as follows:

$$RHm = RE_{Mplot} - RE_{Fplot} \quad [3.6]$$

And heterotrophic respiration in M plot of grassland was estimated as,

$$RH = RHs + RHm \quad [3.7]$$

In F plot of managed grassland RHm is zero, heterotrophic respiration is written as:

$$RH = RHs \quad [3.8]$$

In corn field, RHm was obtained from RH in M plot and RHs in F plot as follows:

$$RH_m = RH - RH_s \quad [3.9]$$

Ecosystem respiration can be partitioned into autotrophic respiration (RA) and RH contributions. Autotrophic respiration was calculated as the difference between RE and RH

$$RA = RE - RH \quad [3.10]$$

Autotrophic respiration can be further subdivided into aboveground autotrophic respiration and belowground autotrophic respiration (root respiration: RR). Root respiration is part of RS and is estimated as;

$$RR = RS - RH \quad [3.11]$$

3.5 Plant biomass measurement

We divided plant biomass into two parts as the above ground biomass and the below ground biomass. In managed grassland, aboveground biomass was measured four or five times a year: in April, June (before 1st crop harvest), August (before 2nd crop harvest), October (before 3rd crop harvest), and at the end of the growing season. The aboveground biomass at the time of harvest was estimated as the sum of the harvests and the stubbles and leaf litters. The harvest was measured by taking clippings 10 cm above the ground using the 100 cm × 100 cm quadrants, and stubble biomass and leaf litters at the time of crop harvest and the aboveground biomass in April and the end of the growing season were measured by clipping using the 50 cm × 50 cm quadrants with eight replications. The samples were oven-dried at 70°C for 72 hours and then weighed. The belowground biomass was measured by taking four soil blocks (50 cm × 25 cm × 30 cm deep) with four replications. The belowground biomass was collected four times a year: in April and at the end of G1, G2 and G3 periods. From 2008 to 2011, the belowground biomass in M plot was collected only at

the end of G3. In 2012, the belowground biomass was collected three times (in April, at the end of G1 and G2 periods) in both plots. The belowground samples were washed using a 0.5 mm sieve, over-dried at 70°C for 72 hours, and weighed. In corn field, aboveground biomass was measured at harvesting time (end of October) and divided into two parts as the harvest and residues (i.e. root and stubble). Eight plant samples were taken and dried at 70°C for 72 h and then weighed. Each dried sample of managed grassland and corn field was analyzed for total C and N contents with an N/C analyzer (SUMIGRAPH NC-1000, Sumika Chemical Analysis Service, Ltd., Osaka, Japan).

The ANPP was estimated as an increment in biomass during each period in the growing season (Shimizu *et al.*, 2009). The annual ANPP in managed grassland was estimated by the sum of biomass increase in each period, while for corn field it was estimated from the harvested biomass. BNPP_{est} was estimated using the following equations,

$$\text{BNPP}_{\text{est}} = \text{NEP} + \text{RH} - \text{ANPP} \quad [3.12]$$

where NEP is provided by eddy covariance measurement, ANPP, RHs and RHm were provided by biometric measurement. NPP_{est} was calculated as the sum of ANPP and BNPP_{est}. BNPP from harvest method (BNNP_{bio}) was estimated from the root biomass. NPP was calculated as the sum of ANPP and BNPP_{bio}.

3.6 Soil sampling

Soil core samples were collected from four depths (0-5, 5-10, 10-20 and 20-30 cm) at 10 and 20 m distance from the EC tower in four directions from the tower (south, west, north and east). Total C content of soil was analyzed with an N/C

analyzer (SUMIGRAPH NC-1000, Sumika Chemical Analysis Service, Ltd., Osaka, Japan).

3.7 Net biome production

The annual net biome production (NBP) is net gain or loss of carbon in the field, which approximates change in soil organic C (Shimizu *et al.*, 2009), and was calculated as following equation,

$$\text{NBP} = \text{NEP} + \text{Manure application} - \text{Harvest} \quad [3.13]$$

3.8 Statistical analysis

Statistical analyses were performed with SPSS Statistic (version 20; IBM Corp, Armonk, NY, USA) and R software (R Development Core Team, 2012; Ver. 2.15.1). The various components of the seasonal C budget among the treatments and seasons were compared using two-way repeated measures analysis of variance (ANOVA) and one-way repeated measures ANOVA for annual data. The differences between treatments and seasons were analyzed using the Tukey–Kramer (honest significant difference; HSD) test. The relationship between the C budget and each environmental variable was subjected to stepwise multiple regression analysis.

The two-sided 95% confidence interval of ANPP and BNPP were calculated (Shimizu *et al.*, 2009), and uncertainties were calculated using the following equation:

$$\text{uncertainty} = \frac{(\text{two-sided 95\% confidence interval})/2}{\text{means}} \times 100 \quad [3.14]$$

The uncertainties of annual ANPP and BNPP were estimated as the sum of uncertainties in ANPP and BNPP for each period using an error propagation.

Chapter 4. Manure application has an effect on the carbon budget of a managed grassland in southern Hokkaido, Japan.

4.1 Introduction

The use of manure is expected not only to reduce the dependence on chemical fertilizer but also to sequester the applied manure C in soils (Janzen *et al.*, 1999). Cattle manure application on dry land and irrigated soils over 25 years in Canada increased the soil organic C through the addition of organic C in the manure (Whalen and Chang, 2002). Organic C that remains after application of manure is assumed to be a part of the soil organic C, and will be decomposed gradually over time, releasing plant nutrients in a way that resembles a slow release fertilizer (Brandjes *et al.*, 1996). Understanding how the C dynamics are affected by long-term manure application is imperative to evaluate the effect of manure application on C sequestration and grass production.

In grassland, belowground net primary production (BNPP) could represent more than half of the total net primary production (NPP), and represents the major input of organic matter into soil (Briggs and Knapp, 1995; McNaughton *et al.*, 1998; Luo *et al.*, 2009). However, knowledge of BNPP remains limited in comparison to aboveground NPP (ANPP), which is better understood, largely due to the methodological difficulties associated with observing and measuring root biomass. Therefore, by using the combination of eddy covariance and biometric method we have developed a reasonable method to estimate BNPP in grassland.

Understanding BNPP dynamics is fundamentally important to improve our knowledge of C allocation and storage in grasslands. In this study, we have developed

a method to estimate BNPP by the combination of the eddy covariance and biometric methods. NEP is defined as the difference between NPP and heterotrophic respiration (RH).

The NEP of intensively managed grassland showed a net C sink (Gilmanov *et al.*, 2010). However, these apparent sinks accumulate in crops and forage, which are C pools that are harvested, transported, and decomposed off site. Therefore, the C budget in managed grasslands includes C output through crop harvest and grazing and C input through manure application as well as NEP. This budget is defined as the net biome production (NBP) (Schulze *et al.*, 2000)

The objectives of this Chapter were to determine a reasonable method to estimate BNPP in grassland and to clarify the effect of management practices such as harvest and continuous manure application on the C cycle in grassland.

4.2 Results

4.2.1 Environmental variables

The daily precipitation is shown in Fig. 4.1, and the cumulative precipitation for each period is shown in Table 4.1. There were no significant differences in cumulative precipitation in each period. The range of period cumulative precipitations was 175–313 mm in the G1 period, 257–522 mm in the G2 period, and 112–565 mm in the G3 period. Mean daily precipitation in the G1, G2, and G3 periods ranged from 1.9 to 8.3 $\text{mm}\cdot\text{d}^{-1}$, and there was no significant difference among the periods. The annual precipitation ranged from 883 to 1526 mm.

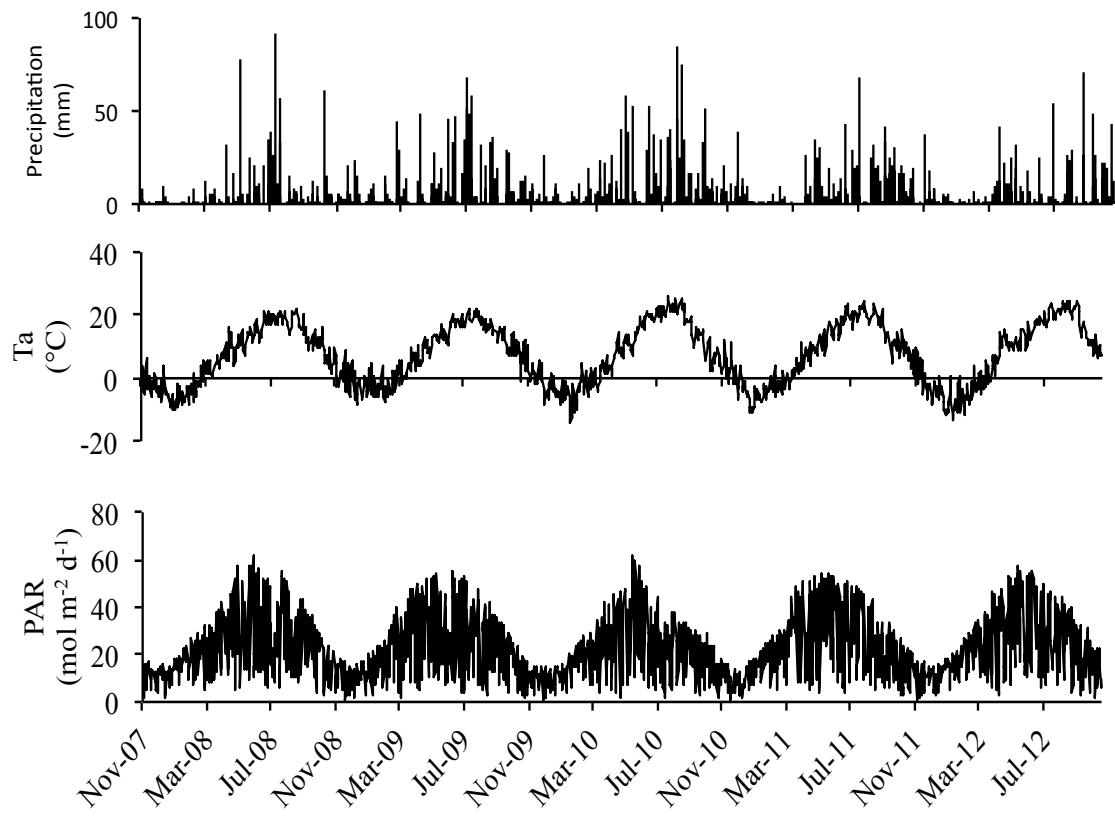


Fig. 4.1 Seasonal variation in meteorological variables in the fertilizer (F) and manure (M) plots; precipitation(a), daily air temperature (Ta) (b), incident of photosynthetically active radiation (PAR) (c)

The daily mean air temperature increased from February and reached the maximum temperature in August before decreasing gradually thereafter (Fig. 4.1). The patterns of daily air temperature were similar across all years. There were significant differences in mean air temperature among the periods ($p < 0.01$) (Table 4.1). Although there was no significant difference in mean air temperature between the G2 and G3 periods (17.1–19.9°C in G2 and 13.8–18.9°C in G3, respectively), air temperatures in the G2 and G3 periods were significantly higher than that in the G1 period (8.4–0.6°C). The annual mean air temperature ranged from 7.2 to 7.7 °C.

Table 4.1 Mean air temperature (Ta), cumulative precipitation (P) and cumulative photosynthetically active radiation (PAR). NG is non-growing season. G1 is the period from the beginning of the growing season to the first crop harvest, G2 is the period from the first harvest to the second harvest and G3 is the period from the second harvest to third harvest, and G4 is the period from the third harvest to the end of the growing season.

Year	Period	Start date	End date	Days	Ta (°C)	P (mm)	PAR (mol·m ⁻²)
2008	NG	Nov. 17, 2007	Apr. 4, 2008	139	-2.6	114	2216
	G1	Apr. 5, 2008	Jun. 9, 2008	65	9.3	175	2198
	G2	Jun. 10, 2008	Aug. 16, 2008	67	17.4	386	1987
	G3	Aug. 17, 2008	Oct. 14, 2008	58	15.8	112	1564
	G4	Oct. 15, 2008	Nov. 18, 2008	34	7.7	97	591
2009	NG	Nov. 19, 2008	Apr. 6, 2009	138	-1.2	311	2273
	G1	Apr. 7, 2009	Jun. 8, 2009	62	9.6	182	2126
	G2	Jun. 9, 2009	Aug. 8, 2009	60	17.1	498	1740
	G3	Aug. 9, 2009	Oct. 5, 2009	57	16.5	286	1477
	G4	Oct. 6, 2009	Nov. 14, 2009	39	8.4	92	611
2010	NG	Nov. 15, 2009	Apr. 24, 2010	160	-1.7	362	2533
	G1	Apr. 25, 2010	Jun. 7, 2010	43	9.8	251	1411
	G2	Jun. 8, 2010	Aug. 17, 2010	70	19.9	522	1845
	G3	Aug. 18, 2010	Nov. 15, 2010	89	13.8	392	1782
2011	NG	Nov. 16, 2010	Apr. 12, 2011	147	-1.7	255	2625
	G1	Apr. 13, 2011	Jun. 2, 2011	50	8.4	313	1593
	G2	Jun. 3, 2011	Aug. 2, 2011	60	17.7	257	1923
	G3	Aug. 3, 2011	Nov. 2, 2011	91	15.8	565	2043
2012	NG	Nov. 3, 2011	Apr. 15, 2012	164	-2.7	329	2858
	G1	Apr. 16, 2012	May 31, 2012	45	10.6	215	1525
	G2	Jun. 1, 2012	Aug. 18, 2012	78	17.5	274	2337
	G3	Aug. 19, 2012	Oct. 16, 2012	58	18.9	246	1477
2008		Nov. 17, 2007	Nov. 18, 2008	368	7.2	883	8557
2009		Nov. 19, 2008	Nov. 14, 2009	361	7.7	1367	8226
2010		Nov. 15, 2009	Nov. 15, 2010	366	7.7	1526	7570
2011		Nov. 16, 2010	Nov. 2, 2011	352	7.7	1389	8183
2012		Nov. 3, 2011	Nov. 3, 2012	367	7.3	1180	8473

Result of one way repeated measures ANOVA during growing season (G1, G2 and G3 periods)

	d.f.	Ta		P		PAR	
		F	p	F	p	F	p
Season	2	49.373	<0.01	2.579	0.166	0.213	0.706

The daily PAR increased from January to May, and then decreased gradually (Fig. 4.1). Similar patterns of daily PAR were recorded during the 5-year study period. The cumulative PAR in each growing season period ranged from 1411 to 2198 mol·m⁻² in the G1 period, from 1740 to 2337 mol·m⁻² in the G2 period, and from 1477 to 2043 mol·m⁻² in the G3 period. There was no significant difference in the cumulative PAR between the periods (Table 4.1). However, there was significant difference in mean PAR ($p < 0.01$) among periods, and the seasonal mean PAR in the G1 season was highest (31.9–34.3 mol·m⁻²·d⁻¹), followed by that in the G2 period (26.4–32.0 mol·m⁻²·d⁻¹) and G3 period (20.0–27.0 mol·m⁻²·d⁻¹). The annual cumulative PAR ranged from 7570 to 8557 mol·m⁻²·yr⁻¹.

4.2.2 Seasonal change of NEP, GPP and RE

The daily NEP showed a similar seasonal pattern throughout the year (Fig. 4.2). Daily NEP increased rapidly from April when grass started growing, and reached the maximum NEP just before the first crop harvest. The maximum daily NEP during the G1 period ranged from 7.7 to 12.8 g C m⁻²·d⁻¹ in the F plot, and from 7.8 to 11.7 g C m⁻²·d⁻¹ in the M plot. The daily NEP just after the first crop harvest was negative, which indicates that CO₂ was emitted from the ecosystem. However, CO₂ uptake began about 2 weeks after the first crop harvest, and the maximum daily NEP during the G2 period ranged from 8.4 to 10.7 g C m⁻²·d⁻¹ in the F plot, and from 6.7 to 8.5 g C m⁻²·d⁻¹ in the M plot. The seasonal change of daily NEP after the second crop harvest was similar to that after the first crop harvest, but the maximum daily NEP during the G3 period was lower than during the G1 and G2 periods. The daily GPP and RE also increased from April (Fig. 4.2). The daily GPP dropped sharply after harvest, while daily RE did not change significantly after harvest.

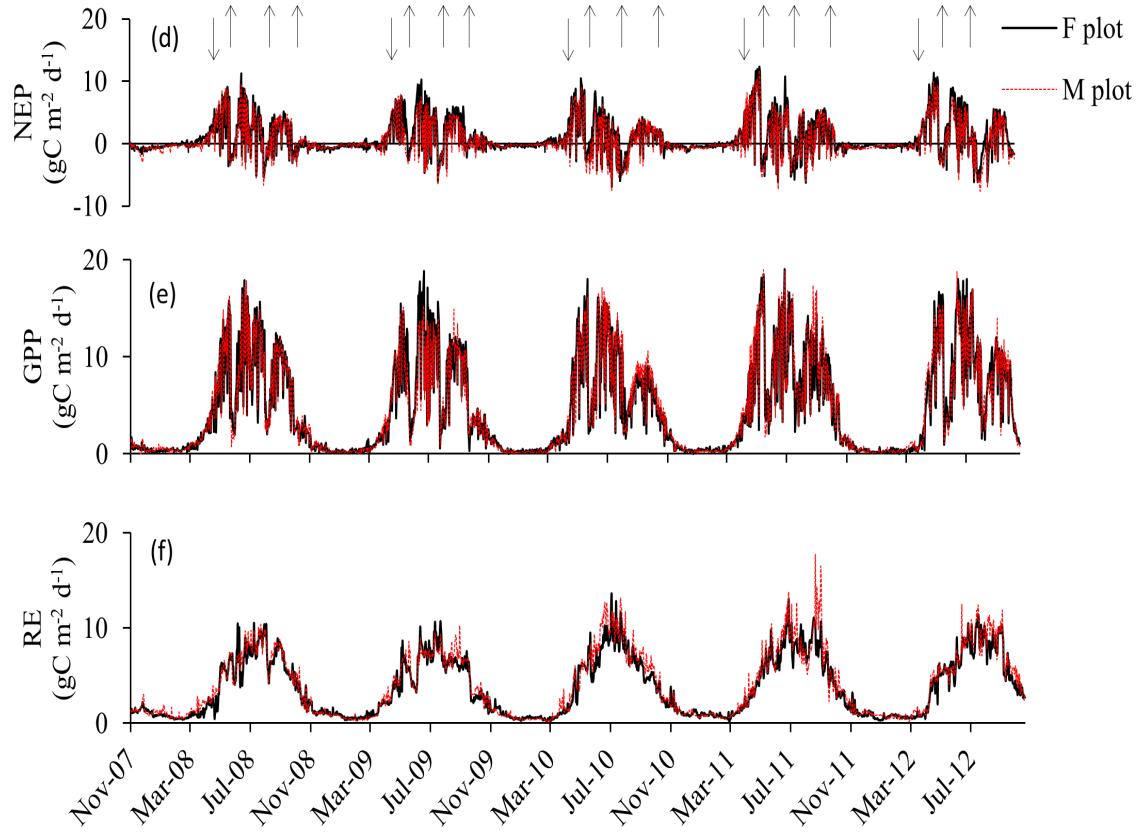


Fig. 4.2 Seasonal variation in net ecosystem production (NEP) (a), gross primary production (GPP) (b) and ecosystem respiration (RE) (c). The up arrows indicate the timing of harvest and the down arrows indicate the timing of manure application.

Table 4.2 Seasonal net ecosystem production (NEP), gross primary production (GPP), ecosystem respiration (RE) and harvest in the fertilizer (F) and manure (M) plots (Mg C ha^{-1} per period). NG is non-growing season. G1 is the period from the beginning of the growing season to the first crop harvest, G2 is the period from the first harvest to the second harvest, G3 is the period from the second harvest to third harvest, and G4 is the period from the third harvest to the end of the growing season. The start and end date for each season was shown in Table 4.1

Year	Period	NEP		GPP		RE		Harvest	
		F	M	F	M	F	M	F	M
2008	NG	-0.5	-0.8	0.7	0.8	1.3	1.6	0.0	0.0
	G1	1.7	1.7	3.9	4.3	2.2	2.5	1.5	1.5
	G2	1.6	1.2	6.7	6.4	5.1	5.2	1.6	1.7
	G3	0.7	0.6	4.5	4.5	3.8	3.9	1.3	1.1
	G4	-0.2	-0.2	0.8	0.9	1.0	1.1	0.0	0.0
2009	NG	-0.6	-0.5	0.7	0.6	1.2	1.1	0.0	0.0
	G1	1.6	1.3	4.1	4.1	2.5	2.8	1.2	1.3
	G2	1.8	1.2	6.2	5.3	4.4	4.2	1.3	1.2
	G3	1.1	0.7	4.7	4.7	3.6	4.0	0.7	0.7
	G4	0.1	0.0	1.1	1.2	1.0	1.2	0.0	0.0
2010	NG	-0.2	-0.3	1.1	1.0	1.3	1.3	0.0	0.0
	G1	1.8	1.6	4.0	3.9	2.2	2.2	1.3	1.2
	G2	0.7	-0.1	6.6	6.9	5.9	6.9	1.3	1.1
	G3	0.5	0.5	4.9	5.4	4.4	4.9	0.0	0.0
2011	NG	-0.6	-0.6	1.0	1.0	1.5	1.5	0.0	0.0
	G1	2.3	2.2	4.3	4.5	2.1	2.3	1.8	2.0
	G2	1.1	0.4	6.0	5.9	4.9	5.5	1.5	1.6
	G3	0.9	1.0	7.0	7.5	6.1	6.6	0.9	1.1
2012	NG	-0.6	-0.8	0.8	0.9	1.5	1.7	0.0	0.0
	G1	2.0	1.9	3.9	4.0	1.9	2.1	1.8	1.6
	G2	1.6	1.0	7.9	7.5	6.2	6.5	1.3	1.2
	G3	0.3	-0.2	4.5	4.6	4.1	4.7	0.0	0.0

Continued

Table 4.2 continue

Year	Period	NEP		GPP		RE		Harvest	
		F	M	F	M	F	M	F	M
Average	NG	-0.5 ± 0.2	-0.6 ± 0.2	0.9 ± 0.2	0.8 ± 0.1	1.4 ± 0.1	1.4 ± 0.2	0.0 ± 0.0	0.0 ± 0.0
± SD	G1	1.9 ± 0.3	1.8 ± 0.4	4.0 ± 0.2	4.2 ± 0.3	2.2 ± 0.2	2.4 ± 0.3	1.5 ± 0.3	1.7 ± 0.6
	G2	1.3 ± 0.5	0.8 ± 0.6	6.7 ± 0.7	6.4 ± 0.8	5.3 ± 0.8	5.7 ± 1.1	1.4 ± 0.1	1.2 ± 0.1
	G3	0.7 ± 0.3	0.5 ± 0.4	5.1 ± 1.1	5.3 ± 1.3	4.4 ± 1.0	4.8 ± 1.1	0.6 ± 0.6	0.6 ± 0.6

Result of two way repeated measures ANOVA during growing season (G1, G2 and G3 periods)

	d.f.	NEP		GPP		RE		Harvest	
		F	p	F	p	F	p	F	p
Treatment	1	39.1	<0.01	0.0	0.847	22.7	<0.01	0.02	0.91
Season	2	10.1	<0.01	9.8	<0.05	19.7	<0.01	14.1	<0.01
Treatment × Season	2	11.0	<0.01	4.5	<0.05	0.7	0.54	0.0	0.97

The cumulative NEP, GPP, and RE across the study period are shown in Table 4.3. During the growing season, there was a significant difference in the cumulative NEP among the treatment plots and periods ($p < 0.01$), and there was a significant treatment-by-season interaction ($p < 0.01$). The NEP in the G1 period ($1.9 \pm 0.3 \text{ Mg C ha}^{-1}$ per period in the F plot, and $1.8 \pm 0.4 \text{ Mg C ha}^{-1}$ per period in the M plot) was the highest. This was followed by G2 period ($1.3 \pm 0.5 \text{ Mg C ha}^{-1}$ per period in the F plot, and $0.8 \pm 0.6 \text{ Mg C ha}^{-1}$ per period in the M plot) and G3 period ($0.7 \pm 0.3 \text{ Mg C ha}^{-1}$ per period in the F plot, and $0.5 \pm 0.4 \text{ Mg C ha}^{-1}$ per period in the M plot). The cumulative NEP was significantly greater in F plots than in M plots in the G2 period, and did not differ significantly between the F and M plots in the G1 and G3 periods. Annual NEP was 3.4 ± 0.4 and $2.4 \pm 0.6 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ in the F and M plots, respectively, and there was a significant difference between the treatment plots ($p < 0.01$). The result of multiple regression analysis showed that the mean NEP was significantly correlated with mean air temperature, daily precipitation, and daily PAR throughout the study period, following the equation: $\text{NEP} = -0.080 - 0.087 \times \text{air temperature} + 0.098 \times \text{precipitation} + 0.068 \times \text{PAR}$ ($R^2 = 0.69$, $p < 0.01$). The annual NEP was 3.2 ± 0.5 and $2.2 \pm 0.5 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ in the F and M plots, respectively, and there was a significant difference between the treatment plots ($p < 0.01$) (Table 4.3).

There was a significant difference in the cumulative GPP among the periods ($p < 0.01$), as well as significant treatment-by-period interactions ($p < 0.05$). However, there was no significant difference among the treatments. The cumulative GPP was significantly greater in the G2 period ($6.7 \pm 0.7 \text{ Mg C ha}^{-1}$ per period in the F plot, and $6.4 \pm 0.8 \text{ Mg C ha}^{-1}$ per period in the M plot) than in the G1 ($4.0 \pm 0.2 \text{ Mg C ha}^{-1}$

per period in the F plot, and $4.2 \pm 0.3 \text{ Mg C ha}^{-1}$ per period in the M plot) and G3 ($5.1 \pm 1.1 \text{ Mg C ha}^{-1}$ per period in the F plot, and $5.3 \pm 1.3 \text{ Mg C ha}^{-1}$ per period in the M plot) periods in both plots. Multiple regression analysis showed that the mean GPP was correlated with the mean air temperature and daily precipitation across the periods, following the equation; $\text{GPP} = 1.61 + 0.21 \times \text{air temperature} + 0.14 \times \text{precipitation}$ ($R^2 = 0.46$, $p < 0.01$). There was no significant difference in annual GPP (17.1 ± 0.7 and $17.2 \pm 1.1 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ in the F and M plots, respectively) between the treatments (Table 4.3).

There was a significant difference in the cumulative RE among the treatment plots and periods ($p < 0.01$), and there was no significant treatment-by-period interaction. The cumulative RE was significantly greater in the M plot than in the F plot. The RE in the G1 period ($2.2 \pm 0.2 \text{ Mg C ha}^{-1}$ per period in the F plot, and $2.4 \pm 0.3 \text{ Mg C ha}^{-1}$ per period in the M plot) was significantly lower than those in the G2 ($5.3 \pm 0.8 \text{ Mg C ha}^{-1}$ per period in the F plot, and $5.7 \pm 1.1 \text{ Mg C ha}^{-1}$ per period in the M plot) and G3 ($4.4 \pm 1.0 \text{ Mg C ha}^{-1}$ per period in the F plot and $4.8 \pm 1.1 \text{ Mg C ha}^{-1}$ per period in the M plot) periods. There was no significant difference between the G2 and G3 periods. Multiple regression analysis showed that the mean RE was significantly correlated with the mean air temperature and daily PAR throughout the study period, by using the following equation: $\text{RE} = 1.83 + 0.30 \times \text{air temperature} - 0.0709 \times \text{PAR}$ ($R^2 = 0.72$, $p < 0.01$). The mean annual RE in the M plot ($14.8 \pm 1.0 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) was significantly greater than that in the F plot ($13.7 \pm 0.7 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) (Table 4.3).

4.2.3 Heterotrophic respiration

The seasonal change of soil surface CO₂ flux of RHs from the root exclusion plot is presented in Fig. 4.3. The RHs increased rapidly from April with the increase of temperature, and reached the maximum CO₂ flux in July, then decreased with the decrease of temperature. There was a significant exponential correlation between the RHs and soil temperature at 5 cm depth (Fig. 4.4).

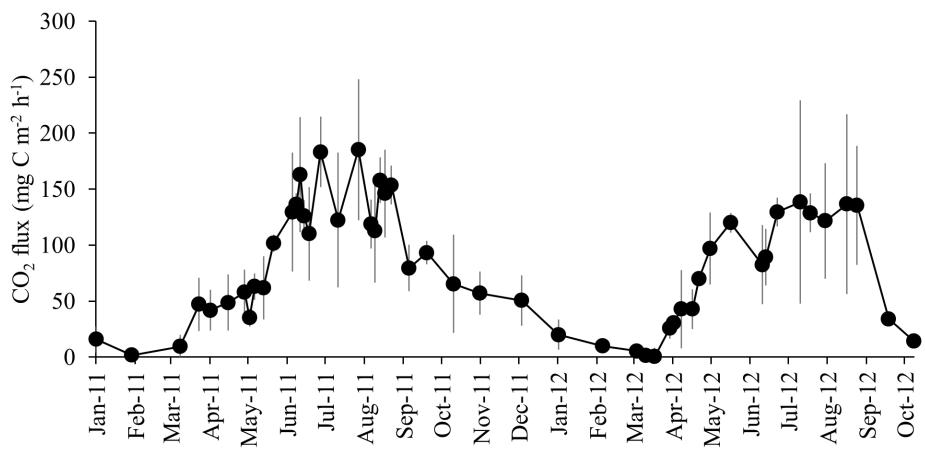


Fig. 4.3 Seasonal variation in soil surface CO₂ flux in the root exclusion plots (RHs) established in fertilizer plot (F) from 2011 to 2012. Data are represented as mean \pm SD.

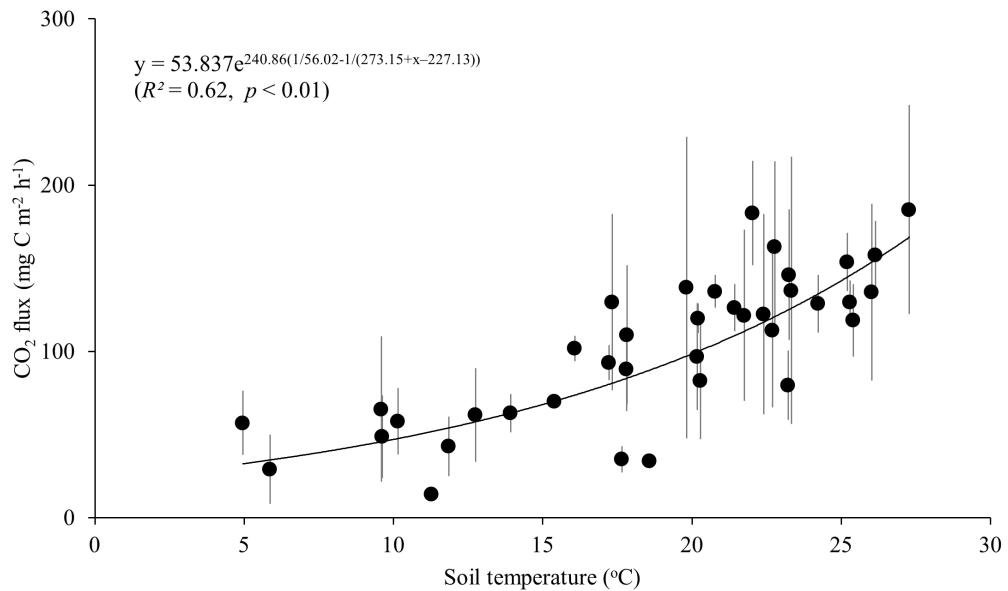


Fig. 4.4 Relationship between soil surface CO₂ flux (RHs) and soil temperature in the fertilizer (F) plot from 2011 to 2012. The line indicates an exponential regression. Data are represented as mean \pm SD.

There was no significant difference in the cumulative RHs among the periods (Table 4.4). However, the RHs in the G1 period tended to be significantly smaller than those in the G2 and G3 periods. The annual RHs in the F and M plots ranged from 4.5 to 5.1 Mg C ha⁻¹·yr⁻¹. RHm in managed grassland was estimated as the difference between the value of RE in M and F plot (Eq. 3.6). Annual RHm was 1.3 and 1.2 Mg C ha⁻¹·yr⁻¹ in 2011 and 2012, respectively (Table 4.4).

The relationships between RH (RH = RHs + RHm) and RE in both F and M plots are shown in Fig. 4.5. There was a significant positive correlation between the RH and RE in both plots ($p < 0.01$). The contribution of the annual RH to RE in 2011 and 2012 was 31% and 33% in the F plot, and 40% and 42% in the M plot, respectively.

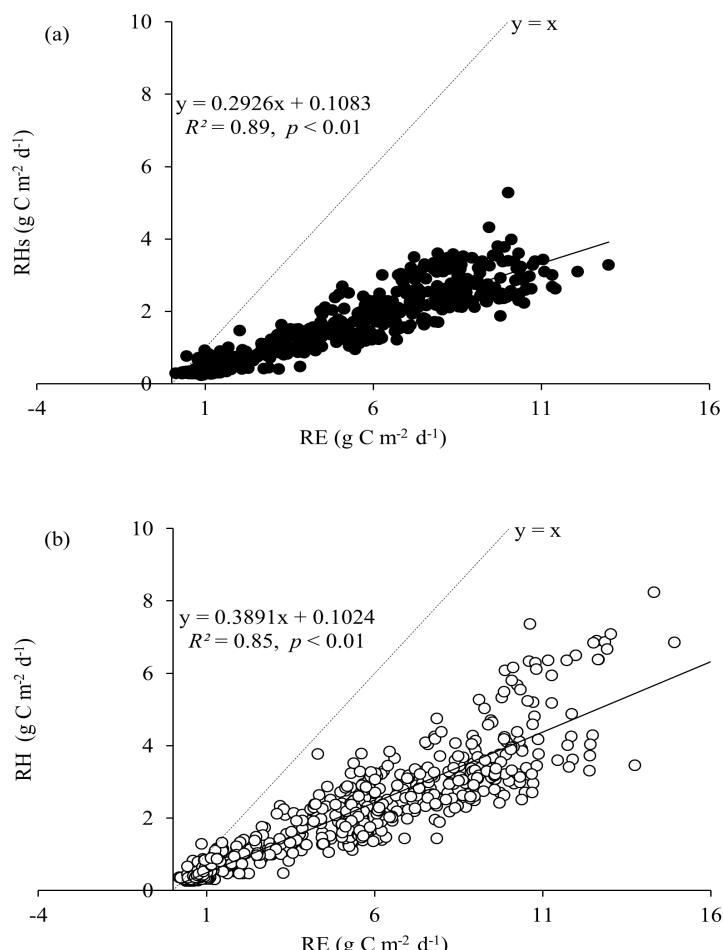


Fig. 4.5 The relationship between ecosystem respiration (RE) and heterotrophic respiration in fertilizer plot (RHs) (a) and manure plot (RH) (b) from 2011 to 2012.

Table 4.3 Annual net ecosystem production (NEP), gross primary production (GPP), ecosystem respiration (RE), ANPP, BNPP, heterotrophic respiration (RH), harvest, net biome production (NBP) and heterotrophic respiration of manure (RHm) in fertilizer plot (F), and manure plot (M) in the managed grassland ($\text{Mg C ha}^{-1} \cdot \text{yr}^{-1}$) and the statistical result of one way ANOVA with repeated measures.

Year	NEP		GPP		RE		Harvest		NBP		RHm
	F	M	F	M	F	M	F	M	F	M	M
2008	3.2	2.6	16.7	16.9	13.4	14.3	4.3	4.3	-1.1	0.4	0.9
2009	3.9	2.7	16.7	15.9	12.8	13.3	3.2	3.2	0.7	1.8	0.5
2010	2.8	1.7	16.6	17.1	13.8	15.3	2.6	2.3	-0.2	1.3	1.5
2011	3.7	3.1	18.2	19.0	14.6	15.9	4.2	4.7	-0.5	0.4	1.3
2012	3.3	1.8	17.1	17.0	13.8	15.0	3.1	2.8	0.2	0.3	1.2
Average \pm SD	3.4 \pm 0.4	2.4 \pm 0.6	17.1 \pm 0.7	17.2 \pm 1.1	13.7 \pm 0.7	14.8 \pm 1.0	3.3 \pm 0.7	3.5 \pm 1.0	-0.1 \pm 0.8	0.8 \pm 0.7	1.1 \pm 0.4

Result of one way repeated measures ANOVA

	df.	NEP		GPP		RE		Harvest		NBP	
		F	p	F	p	F	p	F	p	F	p
Treatment	1	32.3	<0.01	0.2	0.685	38.4	<0.01	0.2	0.90	16.5	<0.05

Table 4.4 Heterotrophic respiration of soil (RHs) and manure (RHm), aboveground net primary production (ANPP), belowground net primary production (BNPP) estimated BNPP (BNPP_{est}) and estimated NPP (NPP_{est}) in the fertilizer (F) and manure (M) plots (Mg C ha^{-1} per period). NG is non-growing season. G1 is the period from the beginning of the growing season to the first crop harvest, G2 is the period from the first harvest to the second harvest and G3 is the period from the second harvest to third harvest. The start and end date for each season was shown in Table 4.1.

Year	Period	RHs		RHm M	ANPP		BNPP_{bio}		BNPP _{est}		NPP _{est}	
		F	M		F	M	F	M	F	M	F	M
2011	NG	0.5	0.7	0.0	-0.2 (0.0)	-0.2 (0.2)	-0.2 (0.8)	-	0.1	0.4	-0.1	0.2
	G1	0.5	0.6	0.2	2.4 (0.1)	2.6 (0.5)	2.0 (2.4)	-	0.4	0.4	2.8	3.0
	G2	1.5	1.5	0.6	1.1 (0.1)	1.1 (0.4)	-1.1 (2.4)	-	1.4	1.4	2.5	2.5
	G3	2.0	2.1	0.5	1.0 (0.1)	1.1 (0.3)	0.3 (2.5)	-	1.9	2.8	2.9	3.8
2012	NG	0.6	0.8	0.2	0.0 (0.0)	0.1 (0.2)-	-1.8 (2.5)	-2.1 (0.6)	0.0	0.1	0.0	0.2
	G1	0.6	0.7	0.2	1.9 (0.1)	1.5 (0.2)	0.0 (0.6)	0.0 (0.2)	0.7	1.3	2.6	2.8
	G2	1.7	2.0	0.2	1.4 (0.0)	1.3 (0.2)	-0.1 (0.5)	0.2 (0.2)	1.9	2.0	3.3	3.3
	G3	1.6	1.6	0.6	0.8 (0.0)	0.7 (0.2)	-	-	1.2	1.4	2.0	2.1
2011	Annual	4.5	4.9	1.3	4.3 (0.1)	4.5 (0.8)	1.1 (4.3)	-	3.9	5.0	8.2	9.5
2012	Annual	4.5	5.1	1.2	4.0 (0.1)	3.5 (0.4)	-1.8 (2.6)*	-1.8 (0.6)*	3.7	4.7	7.7	8.2

ANPP and BNPP represent means \pm (uncertainties/100 \times means).

* Annual BNPP in 2012 showed the sum of BNPP in NG, G1 and G2 periods.

4.2.4 Net primary production

Aboveground biomass significantly increased from the beginning to the end of each period (Fig. 4.6). The belowground biomass ranged from 0.5 to 3.5 Mg C ha⁻¹ in the F plot and from 0.3 to 2.7 Mg C ha⁻¹ in the M plot (Fig. 4.6). Belowground biomass was not significantly different between periods.

There was a significant difference in the harvest among periods ($p < 0.05$), but there was no significant difference between the treatments, or treatments-by-period interactions. Harvest in the G3 period (0.7–1.3 and 0.7–1.1 Mg C ha⁻¹ per period in the F and M plots, respectively) tended to be smaller than those in the G1 and G2 periods (Table 4.2). The annual harvest was 3.3 ± 0.6 and 3.2 ± 0.8 Mg C ha⁻¹·yr⁻¹ in the F and M plots, respectively, and there was no significant difference in the annual harvest between treatments (Table 4.3).

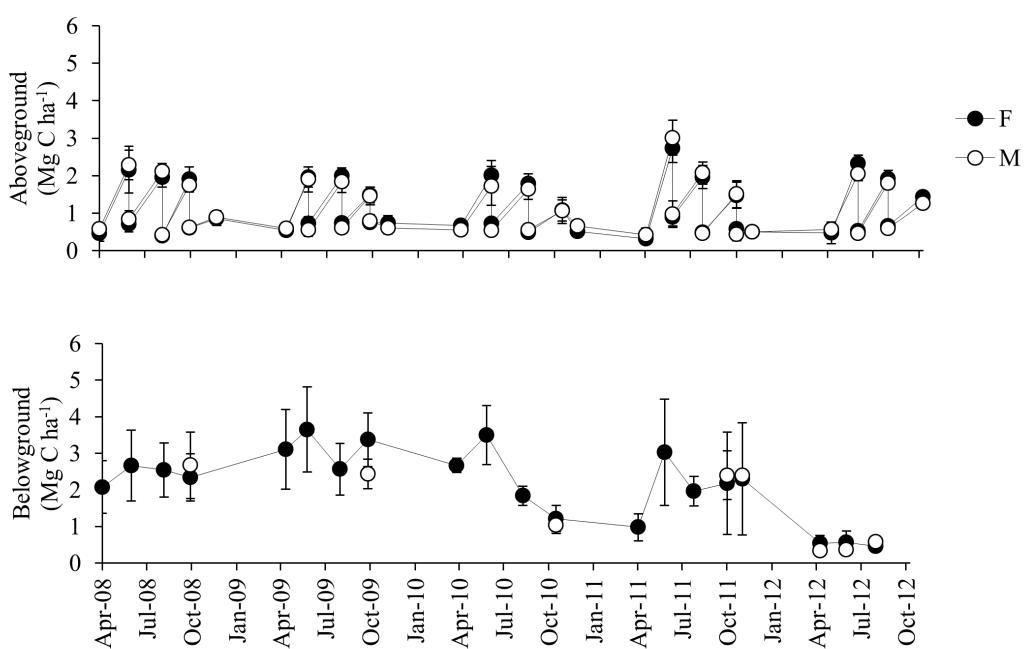


Fig. 4.6 Seasonal change of aboveground and belowground biomass in fertilizer plot (F) and manure plot (M). Data are represented as mean \pm SD.

The ANPP, BNPP_{bio} and BNPP_{est} in 2011 and 2012 are shown in Table 4.4. Annual ANPP for the F and M plots ranged from 4.0 to 4.5 Mg C ha⁻¹·yr⁻¹, and annual BNPP_{bio} ranged from -1.8 to 1.1 Mg C ha⁻¹·yr⁻¹. Uncertainties ranged from 2% to 18% for annual ANPP, and from 33% to 391% for annual BNPP_{bio}. There was no significant difference in BNPP_{est} between the treatments and periods. However, BNPP_{est} in the G2 period tended to be greater than BNPP_{est} in the G1 period. The values of annual BNPP_{est} in 2011 and 2012 were 3.9 and 3.7 Mg C ha⁻¹·yr⁻¹ for the F plot, and 5.0 and 4.7 Mg C ha⁻¹·yr⁻¹ for the M plot, respectively, and these were significantly larger than annual BNPP_{bio} ($p < 0.05$).

The NPP_{est} in G1, G2 and G3 period ranged from 2.0 to 3.3 Mg C ha⁻¹ per period in the F plot, and from 2.1 to 3.8 Mg C ha⁻¹ per period in the M plot. There was no significant difference in the contribution of BNPP_{est} to NPP_{est} (BNPP_{est}/NPP_{est}) among periods and treatment plots. However, BNPP_{est}/NPP_{est} in the G2 (0.56–0.58 and 0.56–0.61 in the F and M plots, respectively) and G3 period (0.60–0.66 and 0.67–0.74 in the F and M plots, respectively) tended to be higher than that in the G1 period (0.14–0.27 and 0.13–0.46 in the F and M plots, respectively). The annual BNPP_{est}/NPP_{est} in 2011 and 2012 was 0.47 and 0.48 for the F plot, and 0.52 and 0.57 for the M plot, respectively.

4.2.5 NBP

Annual NBP is shown in Table 4.5. The annual NBP was -0.1 ± 0.7 and 0.8 ± 0.7 Mg C ha⁻¹·yr⁻¹ in the F and M plots, respectively. There was a significant difference between the treatment plots ($p < 0.05$).

4.3 Discussion

4.3.1 Effect of management practice on C dynamics and budgets

There was no significant difference in annual GPP and harvest between the F and M plots (Table 4.2). This indicates that application of manure alongside chemical fertilizer did not reduce grass production compared with application of chemical fertilizer alone. However, the RE was significantly higher in the M plot than in the F plot (Table 4.2). This would be due to RH of manure (RHm) in the M plot, and the RHm can be estimated as the difference of RE between the M and F plots. The annual RHm from 2010 to 2012 was larger than that in 2008 and 2009 (Table 4.4). This indicates that manure applied in the grassland was still being decomposed more than two years after application. Bhogal *et al.* (2011) have reported that improvements in soil quality and functioning after repeated additions of livestock manures can persist for more than two years after the cessation of applications in England.

The annual RHm of $1.3 \pm 0.1 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ accounted for 40% of manure C applied (Table 4.4). Lee *et al.* (2007) estimated that approximately 45% of added C with cattle manures application for four years in Switch grass grassland of South Dakota was respired. Matsuura *et al.* (2014) reported similar values of RHm ranging from 0.5 to $1.3 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ with an average of $0.8 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ from grassland following application of cattle manure for three-years in Tochigi, Japan. The total amount of C in applied manure reported by Matsuura *et al.* (2014) ranged from 2061 to 4207 $\text{kg C ha}^{-1} \cdot \text{yr}^{-1}$, and the rate of decomposition was about 23% during the three years. The different decomposition rate reported by Matsuura *et al.* (2014) compared with the present study might be due to differences in the C/N ratio of manure. The C/N ratio of manure used by Matsuura *et al.* (2014) was 24.4 ± 4.8 , while that used by us was 13.5 ± 4.3 . It is well known that the decomposition rate of organic matter with

a high C/N ratio is generally low (Eiland *et al.*, 2001; Toma and Hatano; 2007; Vigil and Kissel, 1991).

NBP in the M plot was significantly larger than that in the F plot (Table 4.3). The managed grassland in the M plot acted as C sink, which was confirmed by the positive value of NBP during the 5-years of this study, while the NBP in the F plot was negative, indicating that the ecosystem was acting as a carbon source. A similar result was obtained by Ammann *et al.* (2007), who reported that the NBP (NBP = – NEE – harvest biomass export + carbon input through organic fertilization) was 1.5 and -0.6 Mg C ha⁻¹·yr⁻¹ in intensively managed (manure and fertilizer application) and extensively managed (no application of manure and fertilizer) agricultural grassland in Switzerland respectively. These results indicate that manure application can enhance the annual NBP.

4.3.2 Estimates of BNPP

In this study, RHs was estimated using root exclusion technique, and RH was calculated as the sum of RHs and RHm. Byrne and Kiely (2006) measured RE in eight plots with grass cover and RH in five plots with grass removed using chamber method in an intensively managed grassland in Ireland with slurry was applied, and reported RH/RE ratio of 0.56. This is higher than the ratio of this study. However, Ciais *et al.* (2010) reported RH/RE ratio of 0.40 in European grasslands, in which RE was measured using eddy covariance measurement, and RH was estimated from mass balance calculation. Kosugi *et al.* (2010) found RH/RE ratio of 0.28 at 25 °C using the chamber method in a managed C₃ turfgrass field in Japan. These values are comparable to the values obtained in this study.

In this study, RH of leaf litter and root litter were not measured. However, the decreased amount of litter from the aboveground and belowground parts was included in ANPP and BNPP_{bio} in this study, because the biomass was measured together with the dead parts of the previous period. The annual BNPP_{bio} in 2012 showed negative values, even though the uncertainties were large (Table 4.4). Shimizu *et al.* (2009) also reported that the belowground biomass ranged from 8 to 11 Mg C ha⁻¹ throughout the year without distinct seasonal change in a managed grassland of reed canary grass in Hokkaido, and the annual BNPP was -1.2 ± 8.4 Mg C ha⁻¹ yr⁻¹. This negative value would indicate that decreased amount of root litter is faster than the production amount of root. In this study, uncertainties in BNPP_{est} were not able to be directly calculated. However, Dragoni *et al.* (2007) reported that random uncertainties in annual NEP estimated using eddy covariance flux measurement over a mixed deciduous forest varied from 3-4%. Even though there is potential for the effect of systematic bias in the measured eddy covariance flux, annual BNPP_{est} showed positive values (Table 4.4), and was significantly larger than annual BNPP_{bio} ($p < 0.05$). This indicates that the root litter might still remain in soil, even though some of them were decomposed as CO₂. Carbon leaching loss was not estimated in this study. However, Kindler *et al.* (2011) reported that leaching of biogenic dissolved inorganic C was 0.24 ± 0.07 Mg C ha⁻¹ yr⁻¹, and dissolved organic C leaching was 0.05 ± 0.02 Mg C ha⁻¹ yr⁻¹ for grasslands, which are extremely lower than BNPP_{est} in this study.

BNPP_{est}/NPP_{est} in the G2 and G3 period tended to be higher than that in the G1 period in this study. This indicates that the allocation rate of photosynthetic products to aboveground biomass was larger in the G1 period than in other periods. The peak of aboveground biomass was found in G1 period when the flush of rapid growth was happening in spring. This is thought to be because the remobilization from the

belowground biomass (Xiong *et al.*, 2009). The higher BNPP_{est}/NPP_{est} in the G2 and G3 periods indicates that the allocation of photosynthetic products to belowground biomass was larger. This is attributed to the enhanced proportion of photosynthetic products allocated to belowground biomass for overwintering. In an early review on belowground structures of grasses, Weinmann (1948) suggested that the roots or rhizomes of grasses act as a storage organ during winter. Xiong *et al.* (2009) reported that reed canary grass in northern Sweden exhibited low rhizome growth in June when the re-shooting had just started, after which growth increased steadily during the growing season and reached a peak sometime in late autumn and remained high until the next spring. Miscanthus (Himken *et al.*, 1997) and common reed (Granéli *et al.*, 1992) showed similar seasonal dynamics in their roots and rhizomes. These seasonal patterns may be attributed to the mobilization of rhizome carbohydrates and mineral nutrient stores to support re-growth in the spring and translocation of carbohydrates and mineral nutrients at the end of the growing season.

The annual BNPP_{est}/NPP_{est} ranged from 0.47 to 0.57 in this study. Cahill *et al.* (2009) estimated BNPP by using five methods, in which three of them are based on models predicting root-turnover proposed by Gill *et al.* (2002), which are direct function of ANPP method (DIR), constant function method (CON), exponential function of mean annual temperature method (MAT), and two of them are based on direct observations of root turnover, which are minirhizotron method (MRH) (Brye *et al.*, 2002) and root window method (RW) (Hayes and Seastedt, 1987) in prairie restoration (*Bromus inermis* Leyss) in southwestern Wisconsin. The ratio of BNPP/NPP estimated by these models were 0.27 for DIR, 0.2 for CON and 0.14 for MAT, and those from direct observations were 0.59 for MRH and 0.17 for RW. Among the values of BNPP/NPP ratio from different methods, the value of 0.59 from

MRH method was closest to the value in this study, which was obtained indirectly based on the ecosystem carbon budget. Cahill *et al.* (2009) suggested that turnover estimate by MRH method might be the outlier, but could represent the best estimate of prairie root dynamics, because MRH method used the video image with high resolution taken by high frequency which can detect rapid growth and death of fine roots.

4.4 Conclusion

The C budget was estimated for five years in two experimental plots established in a managed grassland: one receiving only chemical fertilizer and the other receiving composted cattle manure and supplementary chemical fertilizer. There was a significant difference in seasonal NEP and RE between the treatment plots, while there was no significant difference in GPP and harvest. This indicates that combined application of manure and chemical fertilizer did not reduce grass production compared with application of chemical fertilizer only, but manure application would enhance RE through manure decomposition. However, long-term manure application can enhance the annual NBP. In both treatments, root litters may remain in the soils, and become significant contributor to soil C stock in the grassland.

Chapter 5. Land use change effect on carbon budget: from managed grassland to corn field

5.1 Introduction

Current and past land use practices are critical in determining the distribution and size of global terrestrial C sources and sinks (Canadel JG., 2002). Land use type is an important factor to control C storage, and shifts from one type to another are responsible for large C fluxes in and out of the terrestrial ecosystem. Land use change (LUC) in agriculture is generally associated with changes in vegetation in an ecosystem, and thus affects the cycling and storage of C (Guo and Gifford, 2002), which results in C emissions to the atmosphere (IPCC, 2000; McGuire *et al.*, 2001; Houghton *et al.*, 2012). The total emission of C associated with deforestation and fossil-fuel burning in the period 1985-1990 was about 450 PgC, of which 180-200 PgC was from the land use change, and CO₂ emission contribute about 10-30% of the total anthropogenic C (Houghton *et al.*, 1999, Defries *et al.*, 1999)

Most changes in land uses affect the vegetation and soil of an ecosystem and thus change the amount of C. The changes may be large, for example, with the conversion of forest to cropland. A meta-analysis of 74 studies revealed that conversion from grassland to cropland resulted in the largest loss of soil C (59%) followed by the conversion of native forest to crops (42%) (Guo and Gifford, 2002). Land use changes inevitably influence the structure and species composition of plant communities that alter the nature of litter, induce variation in the soil physical properties and cause photosynthesis changes, all of which ultimately affect root respiration (RR) and soil respiration (RS) (Hopkins *et al.*, 2013; Emran *et al.*, 2012).

Management practices and climatic conditions could affect the ecosystem's ability to be either a source or sink of C (Peng *et al.*, 2009; Sheng *et al.*, 2009; Shimizu *et al.*, 2009). Some management practices could increase the organic C in the soil (e.g., manure application) while others could remove or decrease organic C (e.g. tillage). The knowledge about effect of land use change with continuous manure application treatment on the C budget is still limited. In this study, after five consecutive years of managed grassland, the research site was converted to a corn field with continuous manure application.

The objectives of this study was to find out the effect of land use conversion from managed grassland to corn field on the various components [C input (manure application) and output (RH, RS, NPP, harvest)] of the C budget.

5.2 Results

5.2.1 Environmental variables

The daily precipitation is shown in Fig. 5.1, and the cumulative precipitation for each year is shown in Table 5.1. The cumulative annual precipitation ranged from 883 to 1526 mm during the five years in managed grassland and from 1240 to 1355 mm during two years in corn field. The daily mean air temperature increased from February and reached a maximum temperature in August before decreasing gradually thereafter (Fig. 5.1). The patterns of daily air temperature were similar across all years. The daily PAR increased from January to May, and then decreased gradually (Fig. 5.1). Similar patterns of daily PAR were recorded during the 7-year study period. The annual cumulative PAR ranged from 7570 to 8720 mol·m⁻²·yr⁻¹.

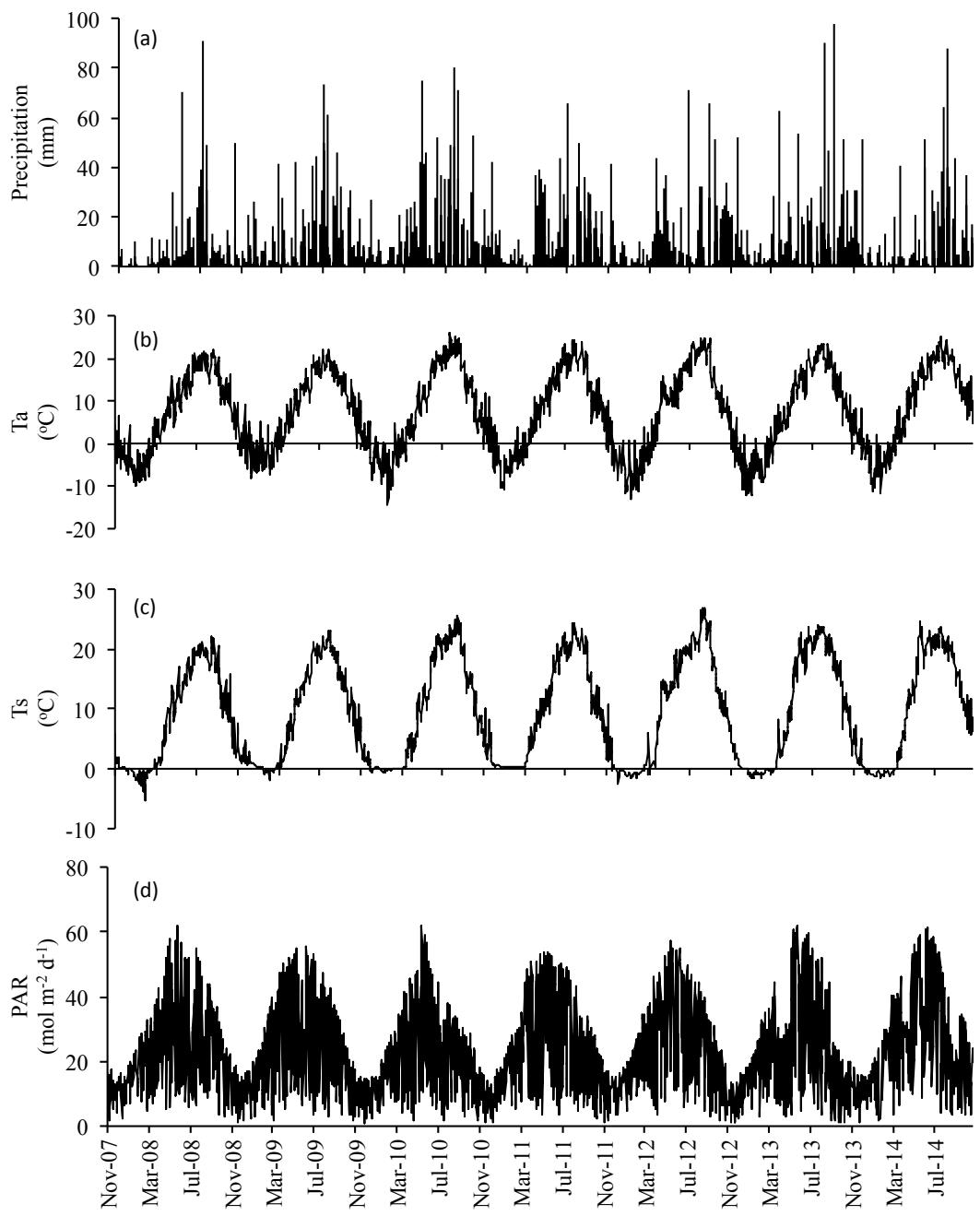


Fig. 5.1 Seasonal variation in precipitation (a), daily air temperature (T_a) (b), daily soil temperature at 5 cm depth (c) and photosynthetically active radiation (PAR) (d) in managed grassland (2008 to 2012) and corn field (2013 to 2014).

Table 5.1 Annual mean air temperature (Ta), cumulative precipitation (P), cumulative photosynthetically active radiation (PAR) and mean soil temperature at 5 cm depth (Ts).

Year	Star date	End date	Days	Ta (°C)	P (mm)	PAR (mol·m ⁻² ·yr ⁻¹)	Ts (°C)
2008	Nov. 17, 2007	Nov. 18, 2008	368	7.2	883	8557	8.3
2009	Nov. 19, 2008	Nov. 14, 2009	361	7.7	1367	8226	9.1
2010	Nov. 15, 2009	Nov. 15, 2010	366	7.7	1526	7570	9.4
2011	Nov. 16, 2010	Nov. 2, 2011	352	7.7	1389	8183	9.6
2012	Nov. 3, 2011	Nov. 11, 2012	367	7.3	1180	8473	9.9
2013	Nov. 4, 2012	Nov. 4, 2013	366	6.9	1355	7730	9.6
2014	Nov. 5, 2013	Nov. 7, 2014	368	8.2	1240	8720	9.7

5.2.2 NEP, GPP and RE

The general pattern of NEP in both managed grassland and corn field followed a similar seasonal cycle with most photosynthesis occurring in spring and summer when incoming solar radiation was greatest (Fig. 5.2). In the managed grassland, the daily NEP had a similar pattern throughout the year with the highest rate of net C uptake occurring from the end of May to September in all the five years from 2008 to 2012 (7.6 to 11.7 g C m⁻²·d⁻¹). The daily NEP in corn field also showed a similar pattern in both years of the study, with the highest C uptake occurring in the middle of July (16.2 to 21.1 g C m⁻²·d⁻¹). The daily NEP quickly decreased after harvest following the reduction of GPP, while RE was not affected at all (Fig. 5.4). Herbicide application, similar to harvest, also led to decreased GPP and NEP. However the reduction of NEP and GPP after herbicide application was slower, taking a few days to reach the same level as that following harvesting (Fig. 5.3). The annual NEP in managed grassland ranged from 1.6 to 2.7 . In corn field, the annual NEP in 2014 was greater than that in 2013. The annual NEP in corn field was 1.2 and 4.3 Mg C ha⁻¹·yr⁻¹ in 2013 and 2014, respectively. Herbicide application during the conversion from managed grassland to corn field in 2012 led to a decrease of the NEP as the C source (Fig. 5.3).

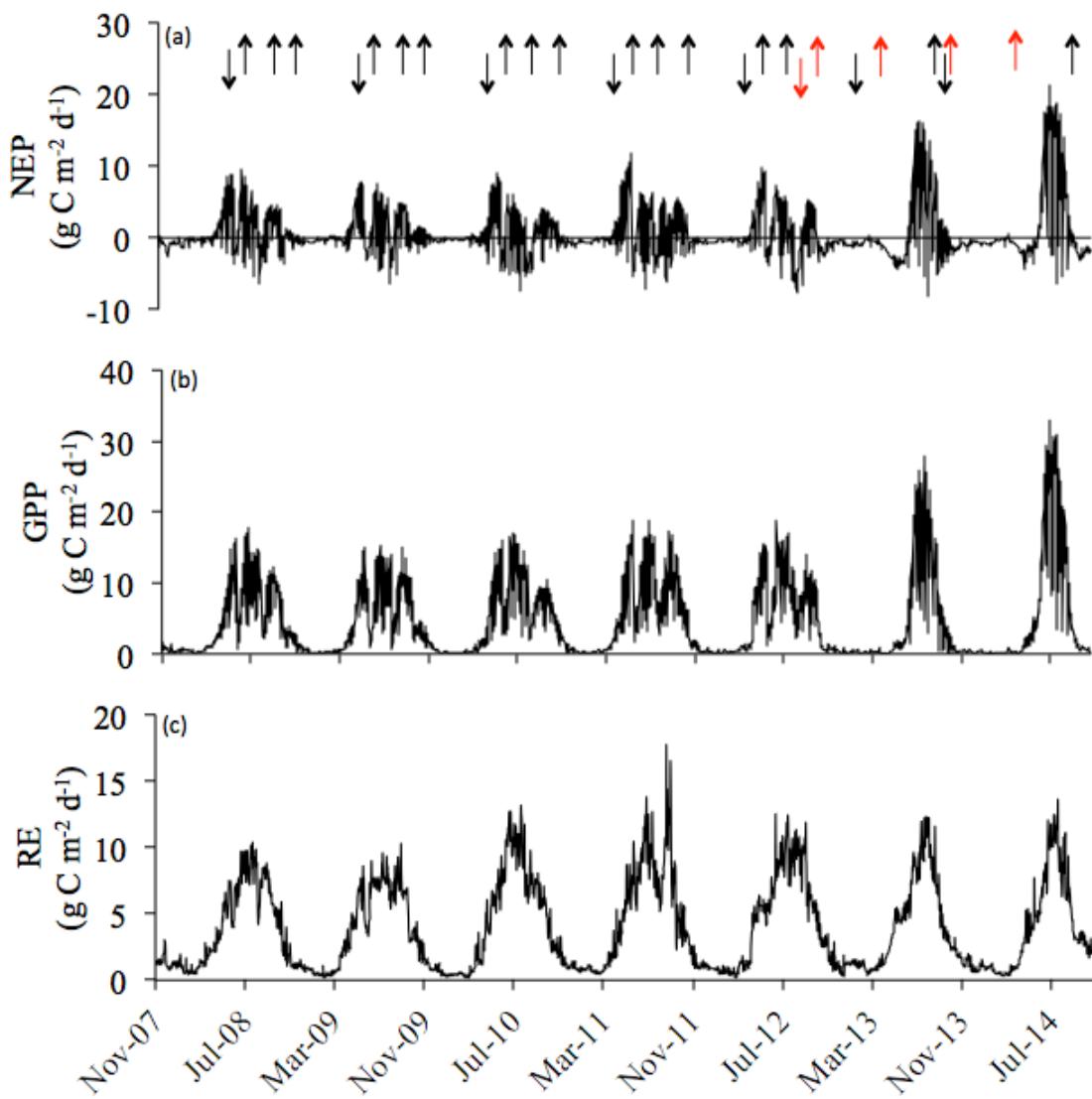


Fig. 5.2 Seasonal variation in net ecosystem production (NEP) (a), gross primary production (GPP) (b) and ecosystem respiration (RE) (c) in managed grassland (2008 to 2012) and corn field (2013 to 2014). The up black arrows indicate the timing of harvest, up red arrows indicate the timing of tillage (Disc and rotary harrow), down black arrows indicate the timing of manure application, and down red arrows indicate the timing of herbicide application.

The peak of daily GPP in corn field occurred in July and August from 2013 to 2014 (27.9 to 32.9 $\text{g C m}^{-2} \cdot \text{d}^{-1}$) and it was higher than in managed grassland which ranged from 15.3 to 19.0 $\text{g C m}^{-2} \cdot \text{d}^{-1}$ and occurred in June and July from 2008 to 2012. Although peak of daily GPP was greater in corn field than in managed grassland due to canopy architecture and C4-type photosynthesis, the annual average

GPP was lower in the corn field than in the managed grassland due to shorter growing period in the corn field (160 days in corn field and 198 days in managed grassland). The annual average GPP was 17.2 ± 1.1 and $16.1 \pm 1.8 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ in managed grassland and corn field, respectively. The increase of RE occurred when air temperature was higher than 5°C . The daily RE had a similar pattern in both managed grassland to corn field. The daily RE in 2013 and 2014 rapidly increased after tillage practices following the reduction of NEP (Fig. 5.2) The annual average RE in managed grassland ($14.9 \pm 1.1 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) was slightly higher than in corn field ($13.3 \pm 0.5 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$).

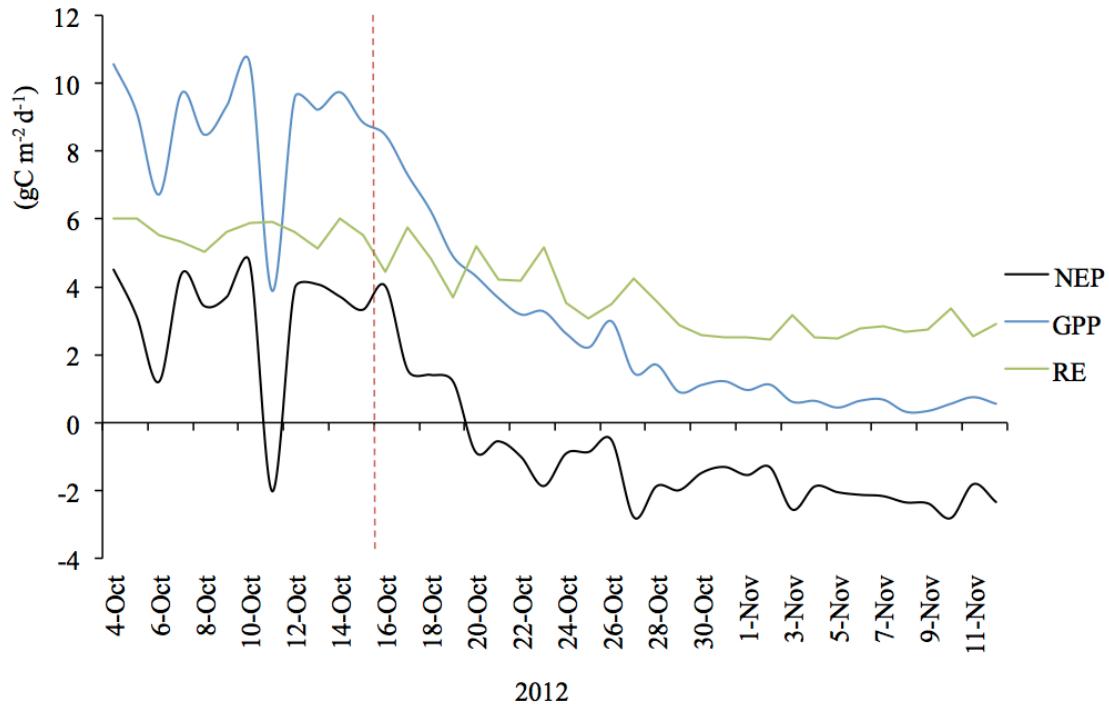


Fig. 5.3 Daily variation in net ecosystem production (NEP) (black color line), gross primary production (GPP) (blue color line) and ecosystem respiration (RE) (green color line) from October 2012 to November 2012. The dashed red line indicated the timing of herbicide application during the conversion from managed grassland to corn field.

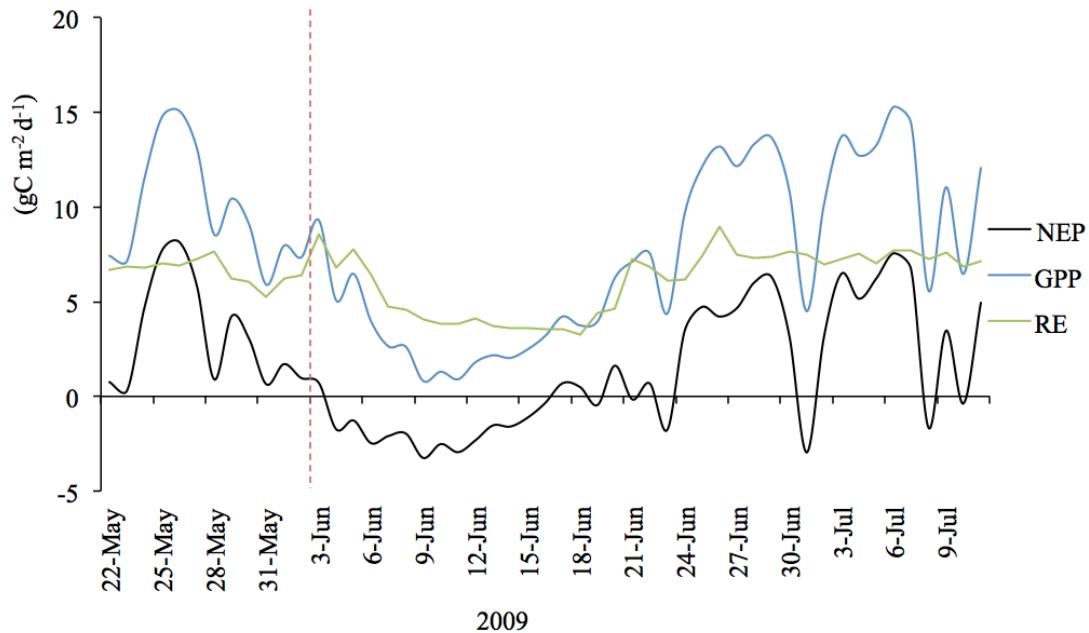


Fig. 5.4 Examples of the effects of harvest on daily net ecosystem production (NEP) (black color line), gross primary production (GPP) (blue color line) and ecosystem respiration (RE) (green color line) from end of May 2009 to beginning of July 2009. The dashed red line indicated the timing of first harvest in 2009.

5.2.3 Heterotrophic and soil respiration

The seasonal change of soil surface CO₂ flux from the root exclusion plot (Heterotrophic respiration, RH) is presented in Fig. 5.5. The RH in managed grassland and corn field had a similar pattern, whereby it increased rapidly from April with the increase of temperature, and reached the maximum CO₂ flux in July, then decreased with the decrease of temperature. There was a significant positive correlation between RH and soil temperature at 5 cm depth (Fig. 5.6). There was no difference of Q₁₀ value of RHs in managed grassland and corn field (Table 5.2). The Q₁₀ of RH in the corn field was 1.6, while there was no Q₁₀ value of RH in managed grassland. The RE increased with the increase in RH. The annual average RH was 6.4 ± 0.0 and 8.1 ± 0.0 Mg C ha⁻¹·yr⁻¹ in managed grassland and corn field respectively (Table 5.2). The contribution of the annual RH to RE in corn field (61%) was greater

than in managed grassland (42%) (Table 5.3). The RE decreased with the decrease in RA. The annual average RA was 9.1 ± 0.6 and $5.2 \pm 0.4 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ in managed grassland and corn field, respectively. The contribution of the annual RA to RE in managed grassland (58%) was greater than in corn field (39%) (Table. 5.2).

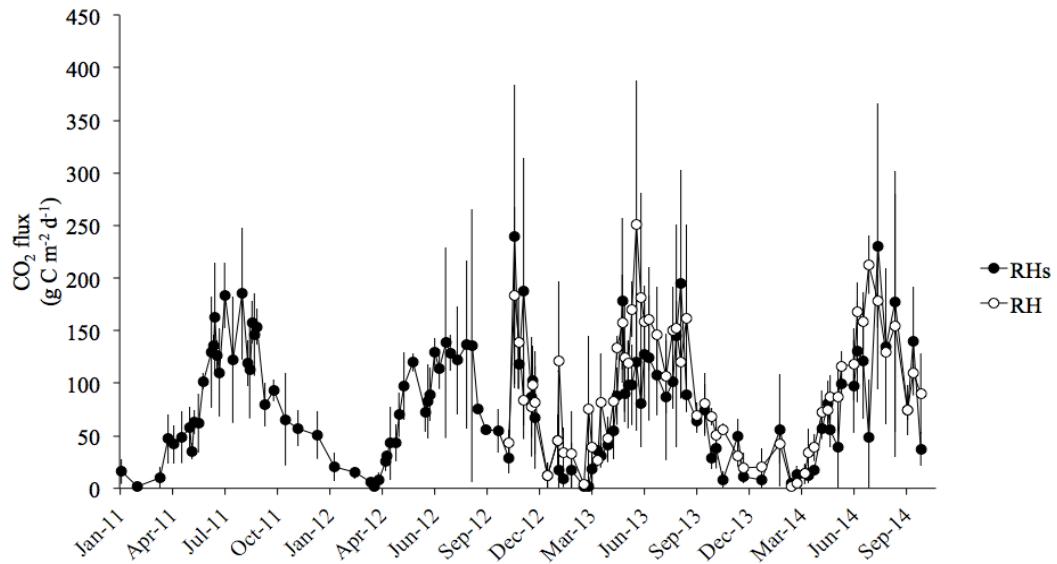


Fig. 5.5 Seasonal variation of soil surface CO_2 flux (RHs and RH) in the root exclusion plots established in managed grassland and corn field. Data are represented as mean \pm SD.

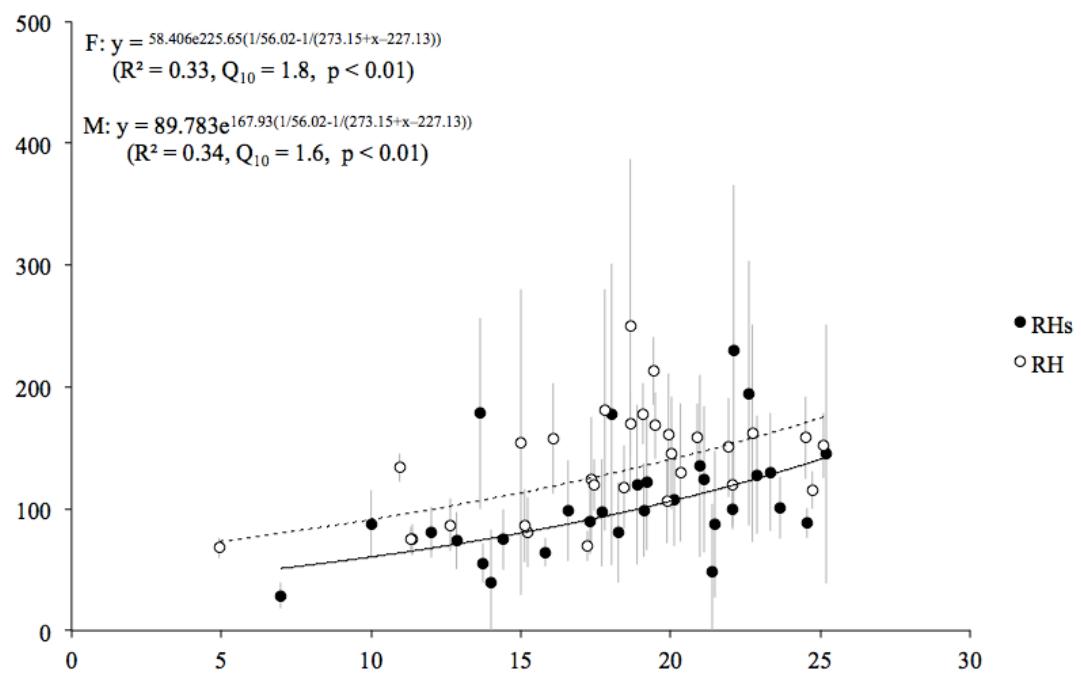


Fig. 5.6 Relationship between soil surface CO_2 flux (RHs and RH) and soil temperature from 2013 to 2014 (Corn). The line indicates an exponential regression of RHs and dot-dash line indicates an exponential regression of RH. Data are represented as mean \pm SD.

Seasonal variation of soil respiration (RS) in managed grassland (2008-2012) and corn field (2013-2014) is shown in Fig. 5.7. The RS had a similar pattern with RH with the soil CO₂ flux increasing with the increase of soil temperature. The seasonal dynamics of RS were positively related to soil temperature in both managed grassland and corn field (Fig. 5.8). The Q₁₀ of RS was higher under managed grassland (3.0) than under corn field (1.9). In contrast with the result of RH, where annual RH increased by about 28% after the conversion to corn field, the annual average RS decreased by 20% two years after conversion to corn field. The annual average RS was $10.8 \pm 1.0 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ in managed grassland and $8.6 \pm 0.0 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ in corn field (Table 5.2). The contribution of annual RR to RS was 44% in managed grassland and 6% in the corn field (Table 5.3).

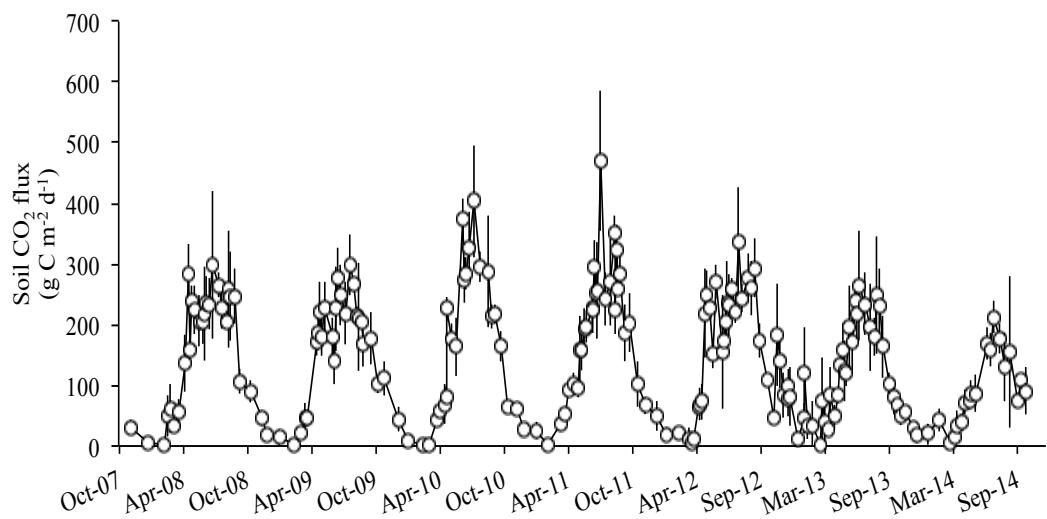


Fig. 5.7 Seasonal variation in soil surface CO₂ flux (RS) in managed grassland (2008 to 2012) and corn field (2013 to 2014). Data are represented as mean \pm SD.

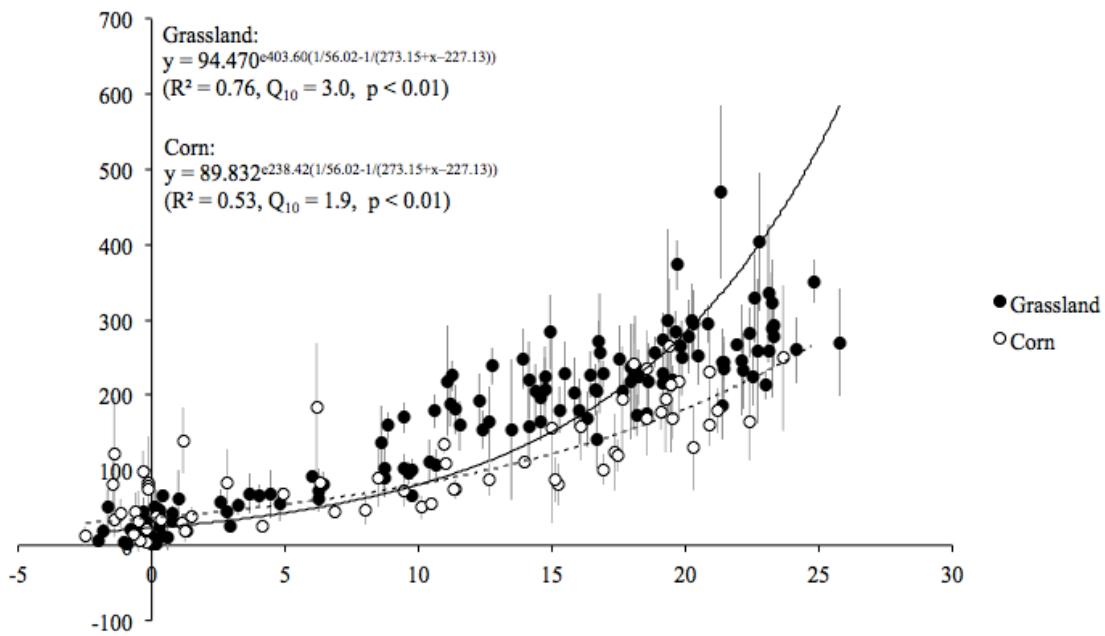


Fig. 5.8 Relationship between soil surface CO₂ flux of RS and soil temperature in the managed grassland (2008 to 2012) and corn field (2013 to 2014). The line indicates an exponential regression in managed grassland and dot-dash line indicates an exponential regression in corn field. Data are represented as mean \pm SD.

Table 5.2 Parameters of the exponential relationship between heterotrophic respiration (RHs and RH), heterotrophic respiration of manure (RHm) and soil respiration (RS) with soil temperature and annual average.

		R ²	p value	Q ₁₀	Average \pm SD
Grassland	RHs	0.62	<0.01	1.9	4.5 \pm 0.0
	RH	-	-	-	6.4 \pm 0.0
	RHm	-	-	-	1.3 \pm 0.1
	RS	0.76	<0.01	3.0	10.8 \pm 1.0
Corn	RHs	0.33	<0.01	1.8	5.6 \pm 0.0
	RH	0.34	<0.01	1.6	8.1 \pm 0.0
	RHm	-	-	-	2.5 \pm 0.0
	RS	0.53	<0.01	1.9	8.6 \pm 0.0

RH in managed grassland is the sum between RHs and RHm (Table 4.4).

Table 5.3 The ratio of heterotrophic respiration (RH) to ecosystem respiration (RE), root respiration (RR) to soil respiration (RR), autotrophic respiration (RA) to ecosystem respiration (RE), heterotrophic respiration (RH) to soil respiration (RS) and heterotrophic respiration of manure (RHm) to heterotrophic respiration (RH) in managed grassland and corn field.

	RH:RE	RA:RE	RR:RS	RH:RS	RHm:RH
Grassland	0.42	0.58	0.44	0.56	0.20
Corn	0.61	0.39	0.06	0.94	0.31

5.2.4 Plant biomass

The aboveground and belowground biomass were significantly influenced by land use. The annual harvest in corn field was significantly greater than in managed grassland. The annual harvest was 3.5 ± 1.0 and 7.7 ± 0.9 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$ in the managed grassland and corn field, respectively (Table 5.4). The annual ANPP in corn field was twice greater than in managed grassland. The annual ANPP ranged from 2.3 to 4.7 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$ in the managed grassland and 8.9 to 11.5 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$ in corn field (Table 5.4). The annual average of BNPP_{est} in managed grassland was greater than in corn field. The annual BNPP_{est} in managed grassland ranged from 5.0 and 4.7 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$ in 2011 and 2012, respectively. In corn field, there was no significant difference between BNPP_{est} (0.7 ± 0.4 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$) and BNPP by biometric method (1.0 ± 0.0 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$). The annual NPP_{est} values in managed grassland were 9.5 and 8.2 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$ in 2011 and 2012, respectively (Table 5.4). In corn field, the annual NPP_{est} in 2014 was greater than that NPP_{est} in 2013. The annual NPP_{est} was 9.3 and 12.5 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$ in 2013 and 2014, respectively (Table 5.4). During the conversion of managed grassland to corn field in 2012, there was no harvest in G3 period. Therefore, NPP_{est} of G3 in 2012 is regarded as the residue (2.1 Mg C ha^{-1}) of managed grassland that will decompose in the next year. In corn field, the residue is the sum of stubble and root biomass. The residue of corn field was 1.0 and 1.6 Mg C $\text{ha}^{-1} \cdot \text{yr}^{-1}$ in 2013 and 2014, respectively.

Table 5.4 Annual ANPP, BNPP estimate (BNPP_{est}), BNPP biometric (BNPP_{bio}), and net primary production (NPP_{est}) in the managed grassland (2011 to 2012) and corn field (2013 to 2014) ($\text{Mg C ha}^{-1} \cdot \text{yr}^{-1}$).

	Year	ANPP	BNPP_{est}	BNPP_{bio}	NPP_{est}
Grassland	2011	4.5 (0.8)	5.0	-	9.5
	2012	3.5 (0.4)	4.7	-1.8 (2.6) *	8.2
Corn field	2013	8.9 (1.9)	1.2	1.0 (0.2)	9.3
	2014	11.5 (1.6)	0.6	1.0 (0.4)	12.5

ANPP and BNPP represent means \pm (uncertainties/100 \times means).

* Annual BNPP in 2012 showed the sum of BNPP in NG, G1 and G2 periods.

5.2.5 Soil organic carbon

Soil carbon storage in managed grassland (2008 to 2012) and corn field (2013 to 2014) at 0–5, 5–10, 10–20 and 20–30 cm depth of soil is shown in table 5.5. At 0–30 cm depth interval the average total carbon soil storage in managed grassland was equal $155.4 \pm 8.1 \text{ Mg C ha}^{-1}$, with values varying from 151.1 to $169.4 \text{ Mg C ha}^{-1}$ (Fig. 5.9). The average total carbon soil stock at the 0–30 cm depth interval in the corn field was equal $166.3 \pm 11.1 \text{ Mg C ha}^{-1}$, with values varying from 158.5 to $174.2 \text{ Mg C ha}^{-1}$.

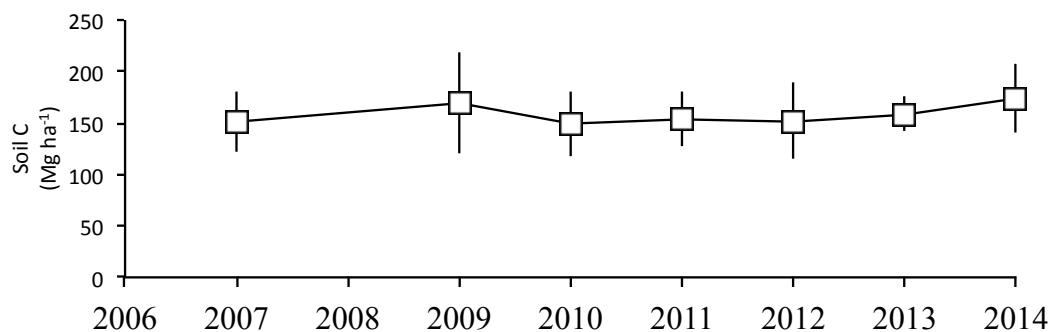


Fig. 5.9 Annual soil carbon storage at 0 to 30 cm depth of soil from 2007 to 2014. Data are represented as mean \pm SD.

Table 5.5 Carbon soil storage in managed grassland (2008 to 2012) and corn field (2013 to 2014) at 0–5, 5–10, 10–20 and 20–30 cm depth of soil in manure plot at 10 and 20 m distance from the EC tower at four direction (south, west, north and east).

Soil depth (cm)	C content (Mg C ha^{-1})						
	2008	2009	2010	2011	2012	2013	2014
0–5	28.3	30.1	27.6	30.3	31.1	26.2	31.6
5–10	25.0	26.4	25.7	25.1	24.1	25.9	29.6
10–20	50.0	56.2	47.0	44.6	49.9	52.8	57.4
20–30	48.2	56.6	49.1	54.3	47.0	53.6	53.8
Total	151.5	169.4	149.4	154.4	152.1	158.5	172.5

5.2.6 NBP

Annual NBP values are shown in Table 5.6. The annual NBP in managed grassland was $0.7 \pm 0.7 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$, with values ranging from 0.3 to 1.8 $\text{Mg C ha}^{-1} \cdot \text{yr}^{-1}$. The annual NBP in corn field was $-2.0 \pm 0.4 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$, with values ranging from -1.7 to -2.3 $\text{Mg C ha}^{-1} \cdot \text{yr}^{-1}$.

Table 5.6 Annual net ecosystem production (NEP), gross primary production (GPP), ecosystem respiration (RE), heterotrophic respiration (RH), soil respiration (RS), autotrophic respiration (RA), root respiration (RR), harvest and net biome production (NBP) in managed grassland (2008-2012) and corn field (2013-2014) in manure (M) plot ($\text{Mg C ha}^{-1} \cdot \text{yr}^{-1}$).

Plot	Year	NEP	GPP	RE	RH	RS	RA	RR	Harvest	NBP
Grassland	2008	2.6	16.9	14.3	-	9.6	-	-	4.3	0.4
	2009	2.7	15.9	13.3	-	10.1	-	-	3.2	1.8
	2010	1.7	17.1	15.3	-	11.5	-	-	2.3	1.3
	2011	3.1	19.0	15.9	6.2	10.8	9.5	4.5	4.7	0.4
	2012	1.8	17.0	15.0	6.3	12.0	8.7	5.7	2.8	0.3
Corn field	2013	1.2	14.8	13.6	8.1	8.6	5.5	0.5	8.3	-1.7
	2014	4.4	17.3	12.9	8.1	8.7	4.9	0.6	10.9	-2.3

Result of ANOVA

	NEP			GPP			RE			RH			RS			RA			RR			Harvest			NBP		
	df.	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p
Plot	1	0.2	0.7	1.1	0.3	3.8	0.1	1369	<0.01	8.5	<0.05	60.8	<0.05	57.1	<0.05	36.9	<0.01	28.9	<0.01								

5.3 Discussion

5.3.1 Effect of land use on RH and RS

The accumulation of soil organic C is the result of a delicate balance between C fixation and microbial decay of senescent vegetation (mainly root mass and residues). Conversion to corn field changed the dynamic equilibrium between inputs and outputs established in the previous five years of managed grassland. The conversion from managed grassland to corn field caused an increase in soil C emission. The RH in corn field ($8.1 \pm 0.0 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) was significantly greater than in managed grassland ($6.4 \pm 0.0 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) (Table 5.2). The contribution of RH to RS in corn field (94%) was greater than that in managed grassland (56%) (Table 5.3) The increase of RH after the conversion from managed grassland to corn field was mainly because of tillage-induced aeration and stronger soil aggregation (Paustian *et al.*, 1997). Tillage associated with soil mixing and inversion, which was carried out in corn field, promotes breakdown of soil aggregates and structure and expose physically protected soil organic carbon to microbial decomposition (Schjonning and Rsmussen, 2000). Tillage also exposes surface soils to the freeze–thaw and wet–dry cycles which accelerates breakdown of macro aggregates (Six *et al.*, 2004). No tillage in managed grassland enhances soil aggregate stability which slows the decomposition of soil organic matter by providing protection within soil aggregates (Ussiri and Lal, 2009).

The temperature dependency of heterotrophic and soil respiration, Q_{10} , is an important parameter for predicting RH and RS, at a given temperature and it changes with land use, micro-environment and ecosystem type (Pavelka *et al.*, 2007; Peng *et al.*, 2009). Wang *et al.* (2008) reported Q_{10} values ranging from 1.2 to 3.2 for a forest, from 1.3 to 3.5 for a meadow, and from 1.2 to 3.7 for a farm. Many studies

have shown that Q_{10} changes in response to change in soil temperature, the spatial and temporal scales of the system, and substrate availability (Wu and Qi, 2001; Fierer *et al.*, 2005; Gaumont-Guay *et al.*, 2006; Wang *et al.*, 2006; Almagro *et al.*, 2009). When Q_{10} is close to or less than 1, it means that soil respiration is not sensitive to temperature, whereas it is extremely sensitive at values greater than 2 (Xu and Qi, 2001; Pavelka *et al.*, 2007). In our study, Q_{10} of RS in managed grassland was 3.0 greater than 2, while in corn field it was 1.9, slightly lower than 2. The Q_{10} of RH (RHs: 1.8 and RH: 1.6) in corn field was slightly lower than Q_{10} of RH (RHs: 1.9) in managed grassland (Table 5.2). The Q_{10} of RHs, RH and RS in both managed grassland and corn field was either slightly lower or greater than 2, which means RHs, RH and RS was sensitive to soil temperature in both land uses..

Land use change and management practices strongly influence global C dynamics, especially in terms of their effects on RS (Zhang *et al.*, 2012). The vegetation structure and species composition, both of which change in response to changes in land use, also strongly influence the C allocation pattern (Wang *et al.*, 2006). In this study, RS differed among the two land use types. RS under corn field was significantly lower than under managed grassland after the conversion. Wang *et al.* (2014) found that RS averaged $8.8 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ under 15 years of restored grassland which was higher compared to $4.7 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ under cropland (wheat and maize), because the grassland had a significantly greater root biomass and return of plant biomass. Soil respiration originates mainly from root and microbial activity, and this makes RR an important component of RS on the global carbon balance (Hogberg *et al.*, 2001; Jia *et al.*, 2006). The contribution of RR to RS varies among vegetation types, vegetation ages, species composition, soil types, season of the year, and environmental conditions (Hanson *et al.*, 2000; Raich *et al.*, 2002; Lee *et al.*,

2003; Tang *et al.* 2005; Gong *et al.*, 2012). Yazaki *et al.* (2004) reported that contribution of RR to RS ranged from 22-53% in temperate grassland (*M. sinesis*); Shimizu *et al.* (2009) reported a range of 49-64% in managed grassland; and Jans *et al.* (2010) reported 38% in corn field. In our study, the contribution of RR to RS in managed grassland was seven times greater than in corn field (Table 5.3). Although the soil organic matter decomposition (RH) in corn field was greater than managed grassland due to management practices such as tillage event and manure application, the conversion of managed grassland to corn field would significantly decrease total CO₂ fluxes (RS) from the soil to the atmosphere due to the reduction of root biomass.

5.3.2 Impact of land use change on C sequestration

The NEP of managed grassland exhibited variability over time. Inter-annual variability in NEP is common in grassland ecosystems (Gilmanov *et al.*, 2010; Meyers, 2001, Aires *et al.*, 2008, Zenone *et al.*, 2013). Overall, grasslands tend to be C sinks during years with favorable precipitation (Zhang *et al.*, 2010). The result of this study showed that the managed grassland sequestered C, and the C dynamics were driven by environmental factors such as precipitation, air temperature and PAR.

The NEP showed a response after herbicide application when the field was converted from grassland to corn field (Fig. 5.3). Large difference between annual NEP in 2013 and 2014 was potentially driven by NPP. The NPP_{est} in 2014 was 34% greater than in 2013 (Table 5.4), which was affected by the amount of GPP. The GPP was positively correlated with NPP. Another factor that potentially drove NEP was RE. The annual RE in 2013 was slightly greater than RE in 2014 (Table 5.3). Thus higher RE in 2013 was probably due to the large CO₂ flux from the decomposition of grass biomass killed during the conversion in the previous year. Zenone *et al.* (2011)

reported that conversion of perennial grassland and soybean to an annual cropping system invoked a large emission of CO₂ during the first year of conversion, mainly due to the decomposition of dead aboveground biomass present after the herbicide application and decomposition of root.

Manure application rate in the corn field was five times more than in managed grassland. As a consequence of the large amount of fresh manure in the corn field, the average annual manure decomposition in the corn field ($2.5 \pm 0.0 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) was two times greater than in managed grassland ($1.3 \pm 0.1 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) (Table 5.2). The contribution of RHm to RH in corn field (31%) was greater than that in managed grassland (20%) (Table 5.3). The RHm accounted for 40 and 52% of the total manure C applied in managed grassland and in corn field, respectively. The conversion from managed grassland to corn field increased soil C by 7%. These results were probably due to higher amount of C input into the soil from the cattle manure in the corn field. Liu *et al.* (2008) measure long-term (1979 to 2008) effect of manure application on soil organic C pools in dryland farming in Northwest China and found that the soil organic C from 0 to 60 cm depth of soil was increased by 41.3% in farmyard manure with supplementary chemical fertilizer treatments.

Accounting for C import and C export, the NBP of managed grassland during five years remained positive. The average NBP of managed grassland was $0.8 \pm 0.7 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$, which was similar to values reported in other grassland ecosystem. Ammann *et al.* (2007) found positive value for NBP ($1.5 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) in grasslands in Switzerland with manure and fertilizer treatment. However, the conversion from managed grassland to corn field decreased the NBP into negative ($-2.0 \pm 0.4 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$), although NEP was positive during two years study period and C import was five times more than in managed grassland. The net C loss from corn field was mainly

caused by the export from biomass. These result are similar to those reported by Jans *et al.* (2010) who measured the NBP of corn field in the Netherlands with manure treatment ($0.51 \text{ Mg C ha}^{-1}$) and found that corn field is a moderate source of C.

5.4 Conclusion

The C budget after conversion from managed grassland (2008 to 2012) to corn field (2013 to 2014) with composed cattle manure and supplementary chemical fertilizer treatment was estimated. The conversion of managed grassland to corn field significantly decreased RS due to the reduction of RR. Annual RH after the conversion from managed grassland to corn field significantly increased due to increase of RHm, which may be ascribed to tillage in corn field. Although the amount of manure C applied in corn field was 2.5 times greater than that in managed grassland, the conversion from managed grassland to corn field caused the ecosystem to be C source.

Chapter 6. General discussion

6.1 Effect of management practices on C budget

Management practices affected C budget in the ecosystems. Daily?? GPP quickly decreased after the harvesting whereas the herbicide application took several days to decrease GPP to the same level as that after harvest (Fig. 5.2 and Fig. 5.3). Reduction in NEP was caused by decrease of GPP, which was affected by herbicide application, or increase of RE due to the effect of tillage events or a combination of both. In this study, decrease in living vegetation decreased GPP which contributed to the large reductions in NEP shortly after herbicide application and harvest. The RE decreased with decrease of RA, but RE also increased with increase of RH due to addition of new labile C sources. In this study, herbicide application and harvest did not change RE greatly (Fig. 5.3 and Fig. 5.4). Eugster *et al.* (2010) observed the effect of management practices on ecosystem respiration by comparing the respiration fluxes during 7, 14 and 28 days after the management event and found that the greatest changes in RE was during the 7-days following tillage and irrigation events. However during 28 days after management practices, the effect of herbicide applications on RE was lower than the effect of irrigation. Managed grassland began to uptake CO₂ (positive NEP) about two weeks after the first harvest (Fig. 5.4). However, the CO₂ uptake after the second harvest was lower than after first harvest, and third harvest was further lower than first and second harvest. This was due to decrease of NEP from G1 to G3. Similar results were found by Schmitt *et al.* (2010) who reported that the grassland took an average of 14 days after the first harvest to recover and become a net sink for CO₂. Hussein *et al.* (2011) measured NEP of extensively managed grassland in Germany and found that the increase of annual frequency of harvests decreased CO₂ sink potential of grassland.

Fresh manure was added to managed grassland and corn field. The applications of manure into the field were not followed by any immediate noticeable changes to NEP, GPP and RE. Eugster *et al.* (2010) also found that respiration did not generally increase following organic fertilizer application. The continuous manure application from managed grassland (2008 to 2012) to corn field (2013 to 2014) was slightly increased soil C storage by 7%. It was due to the amount of manure C applied in corn field ($4.9 \pm 0.8 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) was 2.5 times greater than that in managed grassland ($1.9 \pm 0.4 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$).

Usually increase of soil CO₂ emission following tillage events is expected. There are several reports that CO₂ emissions were proportional to the intensity of the tillage events; more specifically tillage depth and post tillage surface roughness (Reicosky, 1997; Reicosky and Lindstrom, 1993; Reicosky *et al.*, 2005). Tillage, similar to the effect of harvest events, led to a rapid decrease in NEP. The decrease in NEP could be due to either reduction in GPP or an increase in RE. Since there was very little photosynthesis vegetation in the field at the time of tillage, the cause of decrease of NEP can be attributed to increase of RE. As winter approached, air temperature decreased causing photosynthesis to slow down and eventually stop, marked by reduction in NEP, GPP and RE. Therefore, tillage at low temperature is recommend to reduce soil CO₂ emission caused by tillage.

6.2 Effect of land use conversion on C budget

In managed grassland the BNPP_{est} value was more reasonable than BNPP_{bio}. There was no significant difference between annual BNPP_{est} and BNPP_{bio} in corn field. The BNPP_{est} in managed grassland was greater than in corn field due to the vegetation type (Table 5.4). Although the RH was higher in corn field than in

managed grassland due to the higher RHm,, there was no significant difference in RE between managed grassland and corn field (Table 5.3). This result is probably due to the decrease of RR after the conversion to corn field, which is supported by the decreasing of RS. The overall effect of conversion seems to be most strongly related to management practices and the type of the vegetation.

6.3 Conclusion

The results of this study have shown the effect of management practices on the different components of C cycle and their influence on the C budget in managed grassland and corn field. Management practices such as harvest, pesticide addition, tillage and manure application significantly influence the C budget. Manure application with supplementary chemical fertilizer increased biomass production. However, manure application increased annual RE due to increase of RHm. The RE was also influenced by tillage event, rapidly increasing following the reduction of NEP. Harvest and herbicide applications decreased GPP while RE remained constant. In case of managed grassland, long-term manure application enhanced annual NBP, but not for the corn field where NBP was shown as C source due to high C export from the ecosystem.

Summary

Carbon dioxide (CO_2) exchange between terrestrial ecosystem and the atmosphere results from photosynthesis and autotrophic and heterotrophic respirations. Photosynthesis assimilates atmospheric CO_2 to the plant body, which is the main natural carbon (C) import into the ecosystem, called as gross primary production (GPP). Autotrophic and heterotrophic respirations emit CO_2 from plant body and dead organic matter including soil organic matter to the atmosphere, respectively, which are the main natural C export from the ecosystem, called as ecosystem respiration (RE). The difference between GPP and RE is net ecosystem production (NEP), and is directly measured by eddy covariance method. The NEP represents the amount of C used to produce new biomass in the ecosystem. Therefore, NEP is also described as the difference between net primary production (NPP) and heterotrophic respiration (RH), which are measured by biometric method and chamber method, respectively. The RH is often measured in the root exclusion plot. Root respiration (RR) is estimated as the difference between soil respiration (RS) measured in root intact plot and RH. In agricultural ecosystem, management practices influence the ecosystem C budget. Manure application is the main anthropogenic import of organic C into the ecosystem, and harvest is the main anthropogenic export of organic C from the ecosystem. Therefore, C budget in agricultural ecosystem is estimated as $\text{NEP} + \text{manure application} - \text{harvest}$, which is called as net biome production (NBP). In this study, the combination of eddy covariance, biometric and chamber methods was used to evaluate and compare the effect of manure application on the C budget in managed grassland and corn field.

1. Effect of manure application on the C budget of managed grassland

The C budget of a managed grassland in Shin-Hidaka, Hokkaido, Japan was estimated for five years (2008-2012). Chemical fertilizer was applied to fertilizer (F) plot at a rate of $79 \pm 20 \text{ kg N ha}^{-1} \cdot \text{yr}^{-1}$. In the manure (M) plot, dairy cattle manure was applied at a rate of $10 \text{ Mg fresh matter ha}^{-1} \text{ yr}^{-1}$ ($1923 \pm 407 \text{ kg C ha}^{-1} \cdot \text{yr}^{-1}$, $159 \pm 68 \text{ kg N ha}^{-1} \cdot \text{yr}^{-1}$). There was no significant difference in seasonal GPP and harvest between the treatment plots, indicating that both fertilizer and manure can increase the biomass production. Annual NEP and RE were significantly different between the treatment plots. The difference in RE, and between M and F plots is probably due heterotrophic respiration of manure (RHm), which ranged from 1.2 to 1.3 Mg C ha $^{-1} \cdot \text{yr}^{-1}$. Average annual RHm was $1.3 \pm 0.1 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$, and accounted for 40 % of the total amount of applied manure C. The annual NBP in the M plot (from 0.5 to 1.5 Mg C ha $^{-1} \cdot \text{yr}^{-1}$) was significantly higher than in the F plot (-1.1 to 0.7 Mg C ha $^{-1} \cdot \text{yr}^{-1}$). The long-term effect of manure application combined with chemical fertilizer did not reduce grass production compared with chemical fertilizer only; however, manure application decreased the NEP through manure decomposition, and long-term manure application enhanced the NBP.

2. Land use change effect on C budget

The knowledge on the effect of land use change on C budget is limited. This study compared the effect of manure application in managed grassland and corn field. The manure applied managed grassland was converted into corn field in 2013 and C budget was measured for two years. $50 \text{ Mg fresh matter ha}^{-1} \cdot \text{yr}^{-1}$ ($4864 \pm 823 \text{ kg C ha}^{-1} \cdot \text{yr}^{-1}$, $268 \pm 108 \text{ kg N ha}^{-1} \cdot \text{yr}^{-1}$) of manure was applied to the corn field. Annual GPP

and RE slightly decreased after the conversion from managed grassland to corn field, while NEP slightly increased. Annual harvest in corn field was three times greater than that in managed grassland. The conversion to corn field significantly decreased RS due to the reduction of RR. Annual RH significantly increased due to increase of RHm after the conversion to corn field, which may be attributed to tillage in corn field. Annual RHm in corn field was $2.5 \pm 0.0 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$ which accounted for 52% of manure C application rate. Consequently, annual NBP in corn field decreased to $-2.0 \pm 0.4 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$. Although the amount of manure C applied in corn field ($4.9 \pm 0.8 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$) was 2.5 times greater than that in managed grassland ($1.9 \pm 0.4 \text{ Mg C ha}^{-1} \cdot \text{yr}^{-1}$), the conversion from managed grassland to corn field led the ecosystem to be C source.

References

- Aires LMI, Pio CA, Pereira JS 2008: Carbon dioxide exchange above a Mediterranean C3/C4 grassland during two climatologically contrasting years. *Glob. Change Biol.*, 14, 539-555.
- Alam SA, Starr M, Clark BJF 2013: Tree biomass and soil organic carbon densities across the Sudanese woodland savannah: a regional carbon sequestration study. *J. Arid. Environ.*, 89, 67-76.
- Alberti G, Vedove GD, Zuliani M, Peressotti A, Castaldi S, Zerbi G 2010: Changes in CO₂ emission after crop conversion from continuous maize to alfalfa. *Agric. Ecosys. Environ.*, 136, 139-147.
- Almagro M, López J, Querejeta JI, Martínez-Mena M 2009: Temperature dependence of soil CO₂ efflux is strongly modulated by seasonal patterns of moisture availability in a Mediterranean ecosystem. *Soil Biol. Biochem.*, 41, 594-605.
- Ammann C, Flechard CR, Leifeld J, Neftel A, Fuhrer J 2007: The carbon budget of newly established temperate grassland depends on management intensity. *Agric. Ecosyst. Environ.*, 121, 5-20.
- Baldocchi D 2003: Assessing ecosystem carbon balance: problems and prospects of the eddy covariance technique. *Glob. Change Biol.*, 9, 478-492.
- Baldocchi D, Hincks B, Meyers T 1988: Measuring biosphere-atmosphere exchanges of biologically related gases with micrometeorological methods. *Ecology*, 69, 1331-1340.
- Barker T, Bashmakov I, Bernstein L, Bogner JE, Bosch P, Dave R, Davidson O, Fisher BS, Gupta S, Halsnaes K, Heij BJ, Ribeiro SK, Kobayashi S, Levine MD, Martino DL, Masera O, Metz B, Meyer L, Nabuurs GJ, Najam A, Nakicenovic N, Rogner HH, Roy J, Sathaye J, Shock R, Shukla P, Sims REH, Smith P, Tirpak DA, Urge-Corsatz D, Zhou D 2007: Technical Summary.

In: Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Metz B, Davidson OR, Bosch PR, Dave R, Meyer LA (eds)], pp. 25-93. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Barr AG, Black TA, Hogg EH, Griffis TJ, Morgenster K, Kljun N, Theede A, Nesic Z 2007: Climatic controls on the carbon and water balances of a boreal aspen forest, 1994-2003. *Glob. Change Biol.*, 13, 561-576.

Batjes NH 1996: Total carbon and nitrogen in soils of the world. *Eur. J. Soil Sci.*, 47, 151-163.

Beare MH, Hendrix PF, Coleman DC. 1994. Water-stable aggregates and organic matter fractions in conventional and no-tillage soils. *Soil Sci. Soc. Am. J.* 58, 777-786.

Boone RD, Nadelhoffer KJ, Canary JD, Kaye JP 1998: Roots exert a strong influence on the sensitivity of soil respiration. *Nature*, 396, 570-572.

Brandjes PJ, Wit JD, Meer HGVD, Keulen HV 1996: Livestock and the environment finding a balance - environmental impact of animal manure management. International Agriculture Centre Wageningen, The Netherlands.
<http://www.fao.org/ag/againfo/programmes/en/lead/toolbox/Refer/IACman.pdf>.
(January, 1996).

Brandyopadhyay KK, Lal R 2014: Effect of land use management on greenhouse gas emissions from water stable aggregates. *Geoderma*, 232, 363-372.

Briassoulis 2000: Analysis of land use change: theoretical and modeling approaches.
<http://www.rri.wvu.edu/Webbook/Briassoulis/contents.htm>.

- Briggs JM, Knapp AK 1995: Interannual variability in primary production in tallgrass prairie: climate, soil moisture, topographic position, and fire as determinants of aboveground biomass. *Am. J. Bot.*, 82, 1024-1030.
- Brye KR, Gower ST, Norman JM, Bundy LG 2002: Carbon budgets for a prairie and agroecosystems: effects of land use and interannual variability. *Ecol. Appl.*, 12, 962-979.
- Bouwman AF 1990: Soils and greenhouse effect. John Wiley and Sons, Chichester.
- Burba GG, Anderson DJ 2010: A Brief Practical Guide to Eddy Covariance Flux Measurements. Principles and Workflow Examples for Scientific and Industrial Applications, LI-COR Biosciences, Lincoln, USA.
- Cahill KN, Kucharik CJ, Foley JA 2009: Prairie restoration and carbon sequestration: difficulties quantifying C sources and sinks using a biometric approach. *Ecol. Appl.*, 19, 2185-2201.
- Canadel GS 2002: Land use effects on terrestrial carbon sources and sinks. *Sci. China*, 45, 1-9.
- Casals P, Gimeno C, Carrara A, Lopez - Sangil L, Sanz MJ 2009: Soil CO₂ efflux and extractable organic carbon fractions under simulated precipitation events in a Mediterranean Dehesa. *Soil Biol. Biochem.*, 41, 1915-1922.
- Chapin III FS, Matson PA, Vitousek P 2011: Principles of terrestrial ecosystem ecology. Springer Science and Business Media.
- Chen Z, Yu G, Ge J, Sun X, Hirano T, Saigusa N, Wang Q, Zhu W, Zhang Y, Zhang J, Yan J, Wang H, Zhao L, Wang Y, Shi P, Zhao F 2013: Temperature and precipitation control of the spatial variation of terrestrial ecosystem carbon exchange in the Asian region. *Agr. Forest Meteorol.*, 182, 266-276.

Ciais P, Sabine C, Bala G, Bopp L, Brovkin V, Canadell J, Chhabra A, Defries R, Galloway J, Heimann M, Jones C, Quéré CL, Myneni RB, Piao S, Thornton P 2013: Carbon and Other Biogeochemical Cycles. In Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker TF, Qin D, Plattner GK, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM (eds)], pp. 465-570. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Cole V 1996: Agricultural options for mitigation of greenhouse gas emissions. In: Impact, adaptation and mitigation of climate change: scientific-technical analyses, pp. 745-771. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Davidson EA, Verchot LV, Cattánio JH, Ackerman IL, Carvalho JEM 2000: Effects of soil water content on soil respiration in forest and cattle pastures of eastern Amazonia. *Biogeochemistry*, 48, 53-69.

DeFries RS, Houghton RA, Hansen MC, Field CN, Skole D, Townshend J 2002: Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 1990s. *Proc. Natl. Acad. Sci. USA*, 99, 14256-14261.

Dugas WA, Heuer ML, Mayeux HS 1999: Carbon dioxide fluxes over Bermuda grass, native prairie and sorghum. *Agr. Forest Meteorol.*, 93, 121-139.

Dunn AL, Barford CC, Wofsy SC, Goulden ML, Daube BC 2007: A long-term record of carbon exchange in a boreal black spruce forest: means, responses to interannual variability, and decadal trends. *Glob. Change Biol.*, 13, 577-590.

DuPont ST, Culman SW, Ferris H, Buckley DH, Glover JD 2010: No-tillage conversion of harvested perennial grassland to annual cropland reduce root biomass, decreases active carbon stocks, and impacts soil biota. *Agric. Ecosyst. Environ.*, 137, 25-32.

Eiland F, Klamer M, Lind AM, Leth M Bååth E 2001: Influence of initial C/N ratio on chemical and microbial composition during long term composting of straw. *Microb. Ecol.*, 41, 272-280.

Elliott ET 1986: Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* 50, 627-633.

Emran M, Gispert M, Pardini G 2012: Comparing measurements methods of carbon dioxide fluxes in a soil sequence under land use and cover change in North Eastern Spain. *Geoderma*, 170, 176-185.

Erwin KL 2009: Wetlands and global climate change: the role of wetland restoration in a changing world. *Wetl. Ecol. Manage.*, 17, 71-84.

Eugster W, Moffat AM, Ceschia E, Aubinet M, Ammann C, Osborne B, Davis PA, Smith P, Jacobs C, Moors E, Dantec VL, Béziat P, Saunders M, Jans W, Grünwald T, Rebmann C, Kutsch WL, Czerny R, Janous D, Moureaux C, Dufranne D, Carrara A, Magliulo V, Tommasi PD, Olesen JE, Schelde K, Olioso A, Bernhofer C, Cellier P, Larmanou E, Loubet B 2010: Management effects on European cropland respiration. *Agric. Ecosys. Environ.*, 139, 346-362.

Falge E, Baldocchi D, Olson R, Anthoni P, Aubinet M, Christian B, Burba G, Ceulemans R, Clemet R, Dolman H, Granier A, Gross P, Grünwald T, Hollinger D, Jensen NO, Katul G, Keronen P, Kowalski A, Lai CT, Law BE, Meyers T, Moncrieff J, Moors E, Munger JW, Pilegaard K, Rannik Ü, Rebmann C, Suyker

A, Tenhunen J, Tu K, Verma S, Vesala T 2001: Gap filling strategies for defensible annual sums of net ecosystem exchange. *Agr. Forest Meteorol.*, 107, 43-69.

Food and Agriculture Organization (FAO) 1993: Guidelines for land use planning. FAO development series 1, FAO/AGLS, Rome.
<http://www.fao.org/docrep/t0715e/t0715e00.htm>.

Fenn J, Malhi Y, Morecroft M, Lloyd C, Thomas M 2015: The carbon cycle of maritime ancient temperate broadleaved woodland at seasonal and annual scales. *Ecosystems*, 18, 1-15.

Fierer N, Craine JM, McLauchlan K, Schimel JP 2005: Litter quality and the temperature sensitivity of decomposition. *Ecology*, 86, 320-326.

Flanagan LB, Wever LA, Carlson J 2002: Seasonal and interannual variation in carbon dioxide exchange and carbon balance in a northern temperate grassland. *Glob. Change Biol.*, 8, 599-615.

Foken TH, Wichura B 1996: Tools for quality assessment of surface-based flux measurements. *Agr. Forest Meteorol.*, 78, 83-105.

Frank AB, Dugas WA 2001: Carbon dioxide fluxes over a northern semiarid, mixed grass prairie. *Agr. Forest Meteorol.*, 108, 317-326.

Gaumont-Guay D, Black TA, Griffis TJ, Barr AG, Jassal RS, Nesic Z 2006: Interpreting the dependence of soil respiration on soil temperature and water content in a boreal aspen stand. *Agr. Forest Meteorol.*, 140, 220-235.

Gill RA, Kelly RH, Parton WJ, Day KA, Jackson RB, Morgan JA, Scurlock JMO, Tieszen LL, Castle JV, Ojima DS, Zhang XS 2002: Using simple environmental variables to estimate below-ground productivity in grassland. *Glob. Ecol. Biogeogr.*, 11, 79-86.

Gilmanov TG, Aires L, Barcza Z, Baron VS, Belelli L, Beringer J, Billesbach D, Bonal D, Bradford J, Ceschia E, Cook D, Corradi C, Frank A, Gianelle D, Gimeno C, Gruenwald T, Guo H, Hanan N, Haszpra L, Heilman J, Jacobs A, Jones MB, Johnson DA, Kiely G, Li S, Magliulo V, Moors E, Nagy Z, Nasyrov M, Owensby C, Pinter K, Pio C, Reichstein M, Sanz MJ, Scott R, Soussana JF, Stoy PC, Svejcar T, Tuba A, Zhou G 2010: Productivity, respiration, and light-response parameters of world grassland and agroecosystems derived from flux-tower measurements. *Rangeland Ecol. Manag.*, 63, 16-39.

Gong JR, Ge ZW, An R, Duan QW, You X, Huang YM 2012: Soil respiration in poplar plantations in northern China at different forest ages. *Plant Soil*, 360, 109-122.

Grover SPP, Livesley SJ, Hutley LB, Jamali H, Fest B, Beringer J, Butterbach-Bahl K, Arndt SK 2012: Land use change and the impact on greenhouse gas exchange in north Australian savanna soils. *Biogeosciences*, 9, 423-437.

Guo LB, Gifford RM 2002: Soil carbon stocks and land use change: a meta analysis, *Glob. Change Biol.*, 8, 345-360.

Hanson PJ, Edwards NT, Garten CT, Andrews JA 2000: Separating root and soil microbial contributions to soil respiration: a review of methods and observations. *Biogeochemistry*, 48, 115-146.

Heimann M, Reichstein M 2008: Terrestrial ecosystem carbon dynamics and climate feedbacks. *Nature*, 451, 289-292.

Himken M, Lammel J, Neukirchen D, Czypionka-Krause U, Olfs HW 1997: Cultivation of *Miscanthus* under west European conditions: seasonal changes in dry matter production, nutrient uptake and remobilization. *Plant Soil*, 189, 117-126.

- Hirata R, Miyata A, Mano M, Shimizu M, Arita T, Kouda Y, Matsuura S, Niimi M, Saigusa T, Mori A, Hojito M, Kawamura O, Hatano R 2013: Carbon dioxide exchange at four intensively managed grassland sites across different climate zones of Japan and the influence of manure application on ecosystem carbon and greenhouse gas budgets. *Agr. Forest Meteorol.*, 177, 57-68.
- Högberg P, Nordgren A, Buchmann N, Taylor AFS, Ekblad A, Högberg MN, Nyber G, Ottosson-Löfvenius M, Read DJ 2001: Large-scale forest girdling shows that current photosynthesis drives soil respiration. *Nature*, 411, 789-792.
- Hokkaido Prefectural Experiment Stations and Hokkaido Animal Research Center. 2004: *Handbook of Animal Waste Management and Utilization in Hokkaido.* (In Japanese.), Sapporo 64-67.
- Hopkins F, Gonzales-Meler MA, Flower CE, Lynch DJ, Czimczik C, Tang J, Subke JA 2013: Ecosystem-level controls on root-rhizosphere respiration. *New Phytol.*, 199, 339-351.
- Houghton RA 1991: Tropical deforestation and atmospheric carbon dioxide. *Climate Change*, 19, 99-118.
- Houghton RA 1999: The annual net flux of carbon to the atmosphere from changes in land use 1850-1990. *Chem. Phys. Meteorol.*, 51, 298-313.
- Houghton RA 2004: Effect of land use change on carbon balance of terrestrial ecosystem. *Ecosys. Land Use Change*, 153, 85-98.
- Houghton RA, House JI, Pongratz J, van der Werf GR, DeFries RS, Hansen MC, Quéré CL, Ramankutty N 2012: Carbon emissions from land use and land-cover change. *Biogeosciences*, 9, 5125-5142.

Hungate BA, Holland EA, Jackson RB, Chapin III FS, Mooney HA, Field CB 1997: The fate of carbon in grassland under carbon dioxide enrichment. *Nature*, 338, 576-579.

Intergovernmental Panel on Climate Change (IPCC) 2000: Land use, land-use change, and forestry. Special report of the Intergovernmental Panel on Climate Change [Watson RT, Noble IR, Bolin B, Ravindranath NH, Verardo DJ, Dokken D (eds)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Intergovernmental Panel on Climate Change (IPCC) 2001: Climate change 2001: The scientific basis. Contribution of working group I to the Third assessment report of the Intergovernmental Panel on Climate Change [Houghton JT, Ding Y, Griggs DJ, Noguer M, van der Linden PJ, Dai X, Maskell K, Johnson CA (eds)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Intergovernmental Panel on Climate Change (IPCC) 2013: Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker TF, Qin D, Plattner GK, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM (eds.)], pp. 3-23. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

International Union of Soil Sciences (IUSS) Working Group World Reference Base (WRB) 2006: World Reference Base on Soil Resources 2006: World Soil Resources Reports 103, FAO, Rome.

Jackson RB, Banner JL, Jobbey EG, Pockman WT, Wall DH 2002: Ecosystem carbon loss with woody plant invasion of grassland. *Nature*, 418, 623-626.

- Jans WWP, Jacobs CMJ, Kruijt B, Elbers JA, Barendse S, Moors EJ 2010: Carbon exchange of a maize (*Zea mays* L.) crop: influence of phenology. *Agric. Ecosys. Environ.*, 139, 316-324.
- Janssens IA, Freibauer A, Ciais P, Smith P, Nabuurs GJ, Folberth G, Schlamadinger B, Hutz RWA, Ceulemans R, Schulze ED, Valentini R, Dolman AJ 2003: Europe's terrestrial biosphere absorbs 7 to 12% of European anthropogenic CO₂ emissions. *Science*, 300, 1538-1542.
- Janssens IA, Pilegaard K 2003: Large seasonal changes in Q₁₀ of soil respiration in a beech forest. *Glob. Change Biol.*, 9, 911-918.
- Janzen RA, McGill WB, Leonard JJ, Jeffrey SR 1999: Manure as a resource – Ecological and economic considerations in balance. *T. ASABE*, 42, 1261-1273.
- Jia BR, Zhou GS, Wang FY, Wang YH, Yuan WP, Zhou L 2006: Partitioning root and microbial contributions to soil respiration in *Leymus chinensis* populations. *Soil Biol. Biochem.*, 38, 653-660.
- Jin T, Shimizu M, Marutani S, Desyatkin AR, Iizuka N, Hata H, Hatano R 2010: Effect of chemical fertilizer and manure application on N₂O Emission from reed canary grassland in Hokkaido, Japan. *Soil Sci. Plant Nutr.*, 56, 53-65.
- Kern, Johnson 1993: Conservation tillage impacts on national soil and atmospheric carbon levels. *Soil Sci. Soc. Am. J.*, 57, 200-210.
- Kindler R, Siemens J, Kaiser K, Walmsley DC, Bernhofer C, Buchmann N, Cellier P, Eugster W, Gleixner G, Grünwald T, Heim A, Ibrom A, Jones SK, Jones M, Klumpp K, Kutsch W, Larsen KS, Lehuger S, Loubet B, McKenzie R, Moors E, Osborne B, Pilegaard K, Rebmann C, Saunders M, Schmidt MWI, Schrumpf M, Seyfferth J, Skiba U, Soussana JF, Sutton MA, Tefs C, Vowinkel B, Zeeman MJ, Kaupenjohann M 2011: Dissolved carbon leaching from soil is a crucial

component of the net ecosystem carbon balance. *Glob. Change Biol.*, 17,1167-1185.

Kirschbaum MUF 1995: The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. *Soil Biol. Biochem.*, 27, 753-760.

Lal R, Kimble JM, Follett RF, Cole CV 1998: The potential of U.S. cropland to sequester carbon and mitigate the greenhouse effect. CRC Press, Chelsea, MI, USA.

Lasslop G, Reichstein M, Papale D, Richardson AD, Arneth A, Barr A, Stoy P, Wohlfahrt G 2010: Separation of net ecosystem exchange into assimilation and respiration using a light response curve approach: critical issues and global evaluation. *Glob. Change Biol.*, 16, 187-208.

Leahy P, Kiely G, Scanlon TM 2004: Managed grasslands: a greenhouse gas sink or source. *Geophys. Res. Lett.* doi:10.1029/2004GL021161.

Lee DK, Owens VN, Doolittle JJ 2007: Switchgrass and soil carbon sequestration response to ammonium nitrate, manure, and harvest frequency on conservation reserve program land. *Agron. J.*, 99, 462-468.

Lee MS, Nakane K, Nakatsubo T, Koizumi H 2003: Seasonal changes in the contribution of root respiration to total soil respiration in a cool-temperate deciduous forest. *Plant Soil*, 255, 311-318.

Linguist B, van Groenigen KJ, Adviento-Borbe MA, Pittelkow C, van Kessel C 2012: An agronomic assessment of greenhouse gas emissions from major cereal crops. *Glob. Change Biol.*, 18, 194-209.

- Liu J, Jiang P, Wang H, Zhou G, Wu J, Yang F, Qian X 2011: Seasonal soil CO₂ efflux dynamics after land use change from a natural forest to Moso bamboo plantations in subtropical China. *Forest Ecol. Manag.*, 262, 1131-1137.
- Lloyd J, Taylor JA 1994: On the temperature dependence of soil respiration. *Funct. Ecol.*, 8, 315-323.
- Lundquist EJ, Jackson LE, Scow KM 1999: Wet-dry cycles affect dissolved organic carbon in two California agricultural soils. *Soil Biol. Biochem.*, 31, 1031-1038.
- Luo Y, Sherry R, Zhou X, Wan S 2009: Terrestrial carbon-cycle feedback to climate warming: experimental evidence on plant regulation and impacts of biofuel feedstock harvest. *Glob. Change Biol. Bioenergy*, 1, 62-74.
- Malhi Y, Meir P, Brown S 2002: Forest, carbon and global climate. *Phil. Trans. R. Soc. Lond. A.*, 360, 1567-1591.
- Mano M, Miyata A, Yasuda Y, Nagai H, Yamada T, Ono K 2007: Quality control for the open-path eddy covariance data. *Agric. Meteorol.*, 63, 125-138.
- Marton JM, Fennesy MS, Craft CB 2014: Functional differences between natural and restored wetlands in the glaciated interior plains. *J. Environ. Qual.*, 43, 409-417.
- Matsuura S, Miyata A, Mano M, Hojito M, Mori A, Kano S, Sasaki H, Kohyama K, Hatano R 2014: Seasonal carbon dynamics and the effects of manure application on carbon budget of a managed grassland in a temperate, humid region in Japan. *Jpn. Grassl. Sci.*, 60, 76-91.
- McGuire AD, Sitch S, Clein JS, Dargaville R, Esser G, Foley J, Heimann M, Joos F, Kaplan J, Kicklighter DW, Meier RA, Melillo M, Moore B, Prentice C, Ramankutty N, Reichenau T, Schloss A, Tian H, Williams LJ, Wittenberg U 2001: Carbon balance of the terrestrial biosphere in the twentieth century:

- Analyses of CO₂, climate and land use effects with four process-based ecosystem models. *Glob. Biogeochem. Cy.*, 15, 183-206.
- McNaughton SJ, Banyikwa FF, McNaughton MM 1998: Root biomass and productivity in grazing ecosystem: the Serengeti. *Ecology*, 79, 587-592.
- Meybeck M 1982: Carbon, nitrogen, and phosphorus transport by world rivers. *Am. J. Sci.*, 282, 401-450.
- Meyers TP 2001: A comparison of summertime water and CO₂ fluxes over rangeland for well watered and drought conditions. *Agr. Forest Meteorol.*, 106, 205-214.
- Milchunas DG, Lauenroth WK 2001: Belowground primary production by carbon isotope decay and long-term root biomass dynamics. *Ecosystems*, 4, 139-150.
- Nakano T, Sawamoto T, Morishita T, Inoue G, Hatano R 2004: A comparison of regression methods for estimating soil atmosphere diffusion gas fluxes by a closed-chamber technique. *Soil Biol. Biochem.*, 36, 107-113.
- Novick K, Stoy P, Katul GG, Ellsworth DS, Siqueira MB, Juang J, Oren R 2004: Carbon dioxide and water vapor exchange in a warm temperate grassland. *Ecosys. Ecol.*, 138: 259-274.
- Paustian K, Andren O, Janzen HH, Lal R, Smith P, Tian G, Tiessen H, Noordwijk MV, Woomer PL 1997: Agricultural soils as a sink to mitigate CO₂ emissions. *Soil Use Manag.*, 13, 230-244.
- Pavelka M, Acosta M, Marek MV, Kutsch W, Janous D 2007: Dependence of he Q₁₀ values on the depth of soil temperature measuring point. *Plant Soil*, 292, 171-179.
- Peng SS, Piao SL, Wang T, Sun JY, Shen ZH 2009: Temperature sensitivity of soil respiration in different ecosystems in China. *Soil Biol. Biochem.*, 41, 1008-1014.

- Prioul JL, Chartier P 1977: Partitioning of transfer and carboxylation components of intracellular resistance to photosynthetic CO₂ fixation: a critical analysis of the methods used. *Ann. Bot.*, 41, 789-800.
- Raich JW, Potter CS, Bhagawati D 2002: Interannual variability in global soil respiration 1980-94. *Glob. Change Biol.*, 8, 800-812.
- Rayment MB, Jarvis PG 2000: Temporal and spatial variation of soil CO₂ efflux in Canadian boreal forest. *Soil Biol. Biochem.*, 32, 35-45.
- Reichstein M, Falge E, Baldocchi D, Papale D, Aubinet M, Berbigier P, Bernhofer C, Buchmann N, Gilmanov T, Granier A, Grünwald T, Havráneková K, Ilvesniemi H, Janous D, Knobička A, Laurila T, Lohila A, Loustau D, Matteucci G, Meyers T, Miglietta F, Ourcival JM, Pumpanen J, Rambal S, Rotenberg E, Sanz M, Tenhunen J, Seufert G, Vaccari F, Vesala T, Yakir D, Valentini R 2005: On the separation of net ecosystem exchange into assimilation and ecosystem respiration: review and improved algorithm. *Glob. Change Biol.*, 11, 1424-1429.
- Reicosky DC 1997: Tillage-induced CO₂ emission from soil. *Nutr. Cy. Agroecosys.*, 49, 273-285.
- Reicosky DC, Lindstrom MJ 1993: Fall tillage method: effect on short-term carbon dioxide flux from soil. *Agro. J.*, 85, 1237-1243.
- Reicosky DC, Lindstrom MJ, Schumacher TE, Lobb DE, Malo DD 2005: Tillage-induced CO₂ loss across an eroded landscape. *Soil Till. Res.*, 81, 183-194.
- Sarmiento JL, Sundquist ET 1992: Revised budget for the oceanic uptake and anthropogenic carbon dioxide. *Nature*, 356, 589-593.

- Schmitt M, Bahn M, Wohlfahrs G, Tappeiner U, Cernusca A 2010: Land use affects the net ecosystem CO₂ exchange and its components in mountain grasslands. *Biogeosciences*, 7, 2297-2309.
- Schulze ED, Wirth C, Heimann M 2000: Managing forests after Kyoto. *Science*, 289, 2058-2059.
- Sheng H, Yang YS, Yang ZJ, Xhen G, Xie J, Guo J, Zou S 2010: The dynamic response of soil respiration to land-use changed in subtropical China. *Glob. Change Biol.*, 16, 1107-1121.
- Shiga H, Ohyama N, Maeda K, Suzuki M 1985: An evaluation of different organic materials based on their decomposition pattern in paddy soils. *Bull. Natl. Agric. Res. Cetr.*, 5, 1-19.
- Shimizu M, Hatano R, Arita T, Kouda Y, Mori A, Matsuura S, Niimi M, Jin T, Desyatkin AR, Kawamura O, Masayuki H, Miyata A 2013: The effect of fertilizer and manure application on CH₄ and N₂O emission from managed grassland in Japan. *Soil Sci. Plant Nutr.*, 59, 69-86.
- Shimizu M, Marutani S, Desyatkin AR, Jin T, Hata H, Hatano R 2009: The effect of manure application on carbon dynamics and budgets in a managed grassland of Southern Hokkaido, Japan. *Agric. Ecosyst. Environ.*, 130, 31-40.
- Shimizu M, Marutani S, Desyatkin AR, Jin T, Nakano K, Hata H, Hatano R 2010: Nitrous oxide emissions and nitrogen cycling in managed grassland in Southern Hokkaido, Japan. *Soil Sci. Plant Nutr.*, 56, 676-688.
- Sims PL, Bradford JA 2001: Carbon dioxide fluxes in a southern plains prairie. *Agr. Forest Meteorol.*, 109, 117-134.
- Smith P 2008: Land use change and soil organic carbon dynamics. *Nutr. Cy. Agroecosys.*, 81, 169-178.

Soil Survey Staff 2006: Keys to Soil Taxonomy, 10th ed. USDA-Natural Resources Conservation Service, Washington, DC.

Soussana JF, Fuhrer J, Jones M, Amstel A 2007: The greenhouse gas balance of grasslands in Europe. *Agric. Ecosyst. Environ.*, 121, 1-4.

Stallard RF 1998: Terrestrial sedimentation and carbon cycle: coupling weathering and erosion to carbon burial. *Glob. Biogeochem. Cy.*, 12, 231-257.

Suyker AE, Verma SB, Burba GG 2003: Interannual variability in net CO₂ exchange of native tallgrass prairie. *Glob. Change Biol.*, 9, 1-11.

Swanson RV, Flanagan LB 2001: Environmental regulation of carbon dioxide exchange at the forest floor in a boreal black spruce ecosystem. *Agr. Forest Meteorol.*, 108, 165-181.

Tang JW, Misson L, Gershenson A, Cheng WX, Goldstein AH 2005: Continuous measurements of soil respiration with and without roots in a ponderosa pine plantation in the Sierra Nevada Mountains. *Agr. Forest Meteorol.*, 132, 212-227.

Tisdall JM, Oades JM 1982: Organic matter and water stable aggregates in soil. *J. Soil Sci.*, 33, 141-161.

Toma Y, Hatano R 2007: Effect of crop residue C:N ratio on N₂O emissions from Gray Lowland soil in Mikasa, Hokkaido, Japan. *Soil Sci. Plant Nutr.*, 53, 198-205.

Ussiri DAN, Lal R 2009: Long-term tillage effects on soil carbon storage and carbon dioxide emissions in continuous corn cropping system from an alfisol in Ohio. *Soil Till. Res.*, 104, 39-47.

Van Noordwijk M, CerriC, Woomer PL, Nugroho K, Bernoux 1997: Soil carbon dynamics in the humid tropical forest zone. In: The management of carbon in

- tropical soils under global change science, practice and policy (eds. Elliot ET, Kimble J, Swift MJ), *Geoderma*, 79, 187-225.
- Vigil MF, Kissel DE 1991: Equations for estimating the amount of nitrogen mineralized from crop residues. *Soil Sci. Soc. Am. J.*, 55, 757-761.
- Wang CK, Jiang JY, Zhang QZ 2006: Soil respiration in six temperate forests in China. *Glob. Change Biol.*, 12, 2103-2114.
- Wang Y, Amundson R, Trumbore S 1999: The impact of land use change on C turnover in soils. *Glob. Biogeochem. Cy.*, 13, 47-57.
- Wang Y, Zhou G, Wang Y 2008: Environmental effects on net ecosystem CO₂ exchange at half-hour and moth scales over *Stipa krylovii* steppe in northern China. *Agr. Forest Meteorol.*, 148, 714-722.
- Webb E, Pearman G, Leuning R 1980: Correction of flux measurements for density effects due to heat and water vapour transfer. *Q. J. R. Meteorol. Soc.*, 106, 85-100.
- Weinmann H 1948: Underground development and reserves of grasses: a review. *J. Br. Grassl. Soc.*, 3, 115-140.
- Wever LA, Flanagan LB, Carlson PJ 2002: Seasonal and inter-annual variation in evapotranspiration, energy balance, and surface conductance in northern temperate grassland. *Agr. Forest Meteorol.*, 112, 31-49.
- Whalen JK, Chang C 2002: Macro aggregate characteristics in cultivated soils after 25 annual manure applications. *Soil Sci. Soc. Am. J.*, 66, 1637-1647.
- White R, Murray S, Rohweder M 2000: Pilot analysis of global ecosystems: grassland ecosystems. World Resources Institute, Washington, DC, USA.
http://pdf.wri.org/page_grasslands.pdf.

- Widén B 2002: Seasonal variation in a forest-floor CO₂ exchange in a Swedish coniferous forest. *Agr. Forest Meteorol.*, 111, 283-297.
- Wohlfahrt G, Bahn M, Haslwanter A, Newesely C, Cernusca A 2005: Estimation of daytime ecosystem respiration to determine gross primary production of a mountain meadow. *Agr. Forest Meteorol.*, 130, 13-25.
- Wu Y, Wu J, Deng Y, Tan H, Du Y, Gu S, Tang Y, Cui X 2011: Comprehensive assessments of root biomass and production in a *Kobresia humilis* meadow on the Qinghai-Tibetan Plateau. *Plant Soil*, 338, 497-510
- Xiong S, Landstrom S, Olsson R 2009: Delayed harvest of reed canary grass translocates more nutrients in rhizomes. *Acta Agr. Scand. B-S. P.*, 59, 306-316.
- Xu M, Qi Y 2001: Spatial and seasonal variations of Q₁₀ determined by soil respiration measurements at a Sierra Nevadan forest. *Glob. Biogeochem. Cy.*, 15, 687-696.
- Yazaki Y, Mariko S, Koizumi H 2004: Carbon dynamics and budget in a *Miscanthus sinensis* grassland in Japan. *Ecol. Res.*, 19 511-520
- Zanotelli D, Montagnani L, Manca G, Scandellari F, Tagliavini M 2015: Net ecosystem carbon balance of an apple orchard. *Eur. J. Agron.*, 53, 97-104.
- Zenone T, Chen J, Deal MW, Wilske B, Jasrotia P, Xu J, Bhardwaj AK, Hamilton SK, Robertson GP 2011: CO₂ fluxes of transitional bioenergy crops: effect of land conversion during the first year of cultivation. *Glob. Change. Biol. Bioenergy*, 3, 410-412.
- Zenone T, Gelfand I, Chen J, Hamilton SK, Robertson GP 2013: From set-aside grassland to annual and perennial cellulosic biofuel crops: effects of land use change on carbon balance. *Agr. Forest Meteorol.*, 182, 1-12.

Zhang ZH, Duan JH, Wang SP, Luo C, Chang X, Zhu X, Xu B, Wang W 2012:
Effects of land use and management on ecosystem respiration in alpine meadow
on the Tibetan plateau. *Soil Till. Res.*, 124, 161-169.

Zhang L, Wylie BK, Ji L, Gilmanov TG, Tieszen LL 2010: Climate-driven
interannual variability in net ecosystem exchange in the Northern Great Plains
grasslands. *Rangeland Ecol. Manag.*, 63, 40-50.