Ecosystem carbon storage following different approaches to grassland restoration in south-eastern Horqin Sandy Land, northern China

ji yuan[1], Zhiyun Ouyang[2], Hua Zheng[2], and Yirong Su[3]

[1]Yunnan University
[2]Chinese Academy of Sciences
[3]Chinese Acad Sci, Inst Subtrop Agr, Key Lab Agroecol Proc Subtrop Reg, Changsha 410125, Hunan, Peoples R China

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Abstract

Global climate change and extensive socio-economic development together decrease ground cover in the semi-arid sandy grasslands of Horqin district in northern China and thereby increase the direct exposure of surface soil to erosion by strong winds—a process that ultimately converts the grassland into a sandy desert. Three ways to restore such degraded lands through afforestation were evaluated in terms of total carbon stored in the restored ecosystems compared to that in the control. Total carbon comprised that stored in the biomass of trees, herbs, and standing litter and in soil (up to a depth of 100 cm). The three restoration treatments were (1) enclosing the grassland within a shelter belt of Populus × beijingensis, (2) afforesting small but well-distributed patches within the grassland using Pinus sylvestris var. mongolica, and (3) similar afforestation using Ulmus pumila. Total ecosystem carbon storage increased significantly in all the three treatments over more than 20 years; at the end of that period, total ecosystem carbon was maximum (104.29 t/ha) in the grassland enclosed by the forest belt, followed, in that order, by afforestation with P. sylvestris (102.96 t/ha), that with U. pumila (92.24 t/ha), and the control (24.48 t/ha). The structure of the plant community created by these treatments is different from that found in natural stands of forest and in grasslands without trees or shrubs, and all the three treatments are suitable for restoring the moderately desertified sandy grasslands in south-eastern Horqin, northern China, depending on the availability of water and soil nutrients.

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Jiyou Yuan [*1], Zhiyun Ouyang [2], Hua Zheng [2], Yirong Su [3]

[1]School of Ecology and Environmental Sciences & Yunnan Key Laboratory for Plateau Mountain Ecology and Restoration of Degraded Environments, Yunnan University, Kunming, Yunnan, 650091, China;

[2] State Key Lab of Urban and Regional Ecology, Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences, Beijing 100085, China

[3]Key Laboratory of Agro-ecological Processes in Subtropical Region, Institute of Subtropical Agriculture, Chinese Academy of Sciences, Changsha, 410125, China

[*]Corresponding author: E-mail address, yuanjy@ynu.edu.cn

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Abstract: Global climate change and extensive socio-economic development together decrease ground cover in the semi-arid sandy grasslands of Horqin district in northern China and thereby increase the direct exposure of surface soil to erosion by strong winds—a process that ultimately converts the grassland into a sandy desert. Three ways to restore such degraded lands through afforestation were evaluated in terms of total carbon stored in the restored ecosystems compared to that in the control. Total carbon comprised that stored in the biomass of trees, herbs, and standing litter and in soil (up to a depth of 100 cm). The three restoration treatments were (1) enclosing the grassland within a shelter belt of *Populus* × *beijingensis* , (2) afforesting small but well-distributed patches within the grassland using*Pinus sylvestris* var. *mongolica* , and (3) similar afforestation using *Ulmus pumila* . Total ecosystem carbon storage increased significantly in all the three treatments over more than 20 years; at the end of that period, total ecosystem carbon was maximum (104.29 t/ha) in the grassland enclosed by the forest belt, followed, in that order, by afforestation with *P. sylvestris* (102.96 t/ha), that with *U. pumila* (92.24 t/ha), and the control (24.48 t/ha). The structure of the plant community created by these treatments is different from that found in natural stands of forest and in grasslands without trees or shrubs, and all the three treatments are suitable for restoring the moderately desertified sandy grasslands in south-eastern Horqin, northern China, depending on the availability of water and soil nutrients.

Keywords: Desertification; sparsely scattered afforestation; enclosure by tree belts; Ecosystem carbon storage; grassland restoration

1 INTRODUCTION

Grassland is one the most widespread types of vegetation worldwide, covering one-fifths of the earth's land surface and among the world's most widespread biomes (Parton et al. 1995, Scurlock and Hall 1998, He et al. 2009, Corona et al. 2016). Grasslands in China occupy vast contiguous areas of the country's northern temperate regions with continental arid climates, the cold and arid climate of the Tibetan Plateau at very high elevations, and small, scattered patches with a warm climate (Hou 1982, Ni 2002). These vast and widely distributed grasslands are responsible for many important ecosystem functions and offer such services as livestock production, biodiversity maintenance, and soil and water conservation in addition to carbon (C) sequestration and mitigation of the adverse effects of climate change. The grasslands store about 10% of the organic C (Scurlock et al. 2002) and 34% of the total C stored in global terrestrial ecosystems and are also the largest source of uncertainty in estimating the quantities of terrestrial C from biomass accumulation (Cheng et al. 2011), It has been estimated that C storage in Chinese grasslands accounts for 8% of C storage in global grasslands and 16.7% of C storage in Chinese terrestrial ecosystems (Jian 2001, Ni 2002, Ma et al. 2016). Hence, C storage by grassland ecosystems is valuable in maintaining a stable climate. However, global climate change and extensive socio-economic development have rapidly degraded grasslands, and the desertification of grasslands in particular has had significant impacts on soil degradation and the global C cycle (Zhou et al. 2008). The loss of even relatively small patches of grassland through either climate change or human activity represents a significant loss of terrestrial C stocks, which, in turn can affect plant diversity and ecosystem functions. Lal (2001) estimated that global desertification led to a total loss of 19–29 Pg of C from the plant–soil continuum, and Zhou et al. (2008) estimated that the total losses from the plant–soil system of the Horqin sandy lands due to desertification in the last century amounted to 107.53 Mt, as compared to the total C present in the grasslands before. Restoring these degraded ecosystems through appropriate land use could increase the uptake and storage of C in the ecosystems. Therefore, it is important to the sustainability of regional ecosystems to study the effects of restoration strategies on C storage in grassland ecosystems as part of the efforts to mitigate the adverse effects of climate change.

The sandy grassland in Horqin, which is part of the semi-arid agro-pastoral transition zone between the Inner Mongolian Plateau and the North-East China Plain, is considered one of the four great sandy lands.

Historically, the Horqin sandy land was a grassland with many lakes and lush vegetation dominated by palatable grass species, along with sparsely scattered woody species. However, almost 80% of this region has suffered desertification since the 1950s (Li et al. 2013, Li et al. 2017). Desertification is driven by many factors including overexploitation by continued cultivation, over-grazing, unsustainable collection of fuelwood, irresponsible use of water resources, and extreme climatic conditions (Zhang et al. 2004, Li et al. 2013). To mitigate the degradation of the Horqin sandy grasslands, various restoration measures have been implemented in the area, including the establishment of grazing enclosures to protect the surviving vegetation from livestock and afforestation using indigenous and introduced tree and shrub species (Yan et al. 2011). The efforts to restore degraded and overexploited grasslands, which aim to recover biodiversity and ecosystem functions services and to strike a balance between ecological protection and socio-economic development, also increase C sequestration in soil and improve soil structure. Most of the studies so far related to the Horqin sandy lands have focused on restoration of plant diversity, benefits in the form of soil and water conservation, productivity, soil fertility, and climatic conditions (Su and Zhao 2003, Shirato et al. 2004, Su et al. 2004, Zhang et al. 2005, Zhao et al. 2007, Zuo et al. 2009, Hu et al. 2015). However, the effects of afforestation on ecosystem C storage in this area remain poorly understood, and only a few studies have examined the impacts of different approaches to grassland restoration on C storage and the C cycle in south-eastern Horqin sandy lands in northern China; even fewer have attempted to quantify C pools of the total ecosystem—a knowledge gap the present study seeks to bridge.

Accordingly, we explored in quantitative terms the impacts of three typical restoration protocols (and a control) on C storage in the ecosystem. More specifically, the main objective of the present study was to estimate the amounts of C stored in different components of the ecosystem, namely trees, herbs, standing litter, and soil (to a depth of 100 cm) and thus to arrive at the total C. Such estimations will provide suitable benchmarks in devising appropriate strategies to restore vegetation and to mitigate the adverse effects of climate change in the south-eastern Horqin sandy lands in northern China.

2 MATERIALS AND METHODS

2.1 Study area

The sandy land in Horqin is part of north-eastern China's Inner Mongolia desert zone, from the North-East China Plain to the transition zone of the Inner Mongolia plateau (42°41'–45deg15' N,118deg35'–123deg30' E, 178.5–631.9 m above the sea level), and is spread over approximately 51.75 km2. The climate ranges from temperate semi-arid to semi-humid continental monsoon. The study site was at the Daqinggou Ecological Station, Institute of Applied Ecology, Chinese Academy of Science. Average annual temperature in the area is 3–7 degC (ranging from -12.6 degC to -16.2 degC, in January, to 20.3–23.9 degC, in July), with 140–160 frost-free days a year. Average annual precipitation is 343–500 mm, about 70% of which is received between June and August. Spring is fairly dry with hardly any rainfall in most places. Average annual evaporation is 1500–2500 mm; average annual wettability is 0.3–0.5; the drying coefficient ranges from 1 to 1.8; cumulative active temperature ([?]10 degC) is 3032–3168 degC; and annual daily mean sunshine hours are 2837–2892, with 120 kcal/cm2 of global radiation. Annual average wind speed is 3.4–4.4 m/s (4.2–5.9 m/s in spring); for 210–310 days a year, and even up to 330 d in some years, the wind is laden with sand and its speed is more than 5 m/s. Each year also sees 25–40 gale days (wind speeds exceeding 17.2 m/s), including 10–15 days, mainly in the spring, of sand storms. The annual sunshine hours are 2900–3100, maximum in May and minimum in December. The soils are mainly aeolian sandy soils and meadow soils. Substratum mobility varies from fixed sand dunes, semi-fixed sand dunes, and mobile dunes. The zonal soils in the Horqin sandy lands are dark brown, chestnut, and loess soils whereas the non-zonal soils are mainly sandy, meadow, and saline. Due to desertification, the zonal soils are gradually turning into sandy soils. Sandy plains are widely distributed, consisting mainly of sandy soils, and, following the standard soil classification, can be divided into raw grass sandy soils, flowing sand soils, and aeolian sandy chestnut soils. The area is a typical agro-pastoral transition zone in northern China and comprises many different ecosystems including natural deciduous broad-leaved forests, forest plantations, grasslands, wetlands, and farmlands. The vegetation consists of mainly psammophytes of the Mongolian and Huabei floras.

2.2 Experimental design

The experiment was initiated at the Daqinggou Ecological Station in 1992. The single-factor completely randomized design was used for comparing the three treatments or approaches aimed at restoring degraded grasslands, namely (1) enclosing the grassland within a shelter belt of *Populus* x *beijingensis* , referred to as FG, in which the dominant species were *Artemisia scoparia* , *Setaria viride* , *Agropyron cristatum* , and *Cleistogenes chinensis* ; (2) afforesting small but well-distributed patches within the grassland using *Pinus sylvestris* var. *mongolica* , referred to as MG, in which the dominant species were *Cleistogenes chinensis* , *Setaria virides* , *Artemisia scoparia* , and *Calamagrostis arundinacea* ; and (3) similar afforestation using *Ulmus pumila* , referred to as UG, in which the dominant species were *Setaria virides* , *Cleistogenes chinensis* , *Temisia scoparia* , and *Artemisia lavandulaefolia* . All the restoration sites were established in 1992. After tree planting, the land was left without any intervention or disturbance. A plot of desertified sandy grassland in which the dominant species were *Agriophyllum squarrosum* (Linn.) Moq. and *Cenchrus echinatus* Linn. served as a control or a check, referred to as CK, which was also established in 1992 and has remained unmanaged since then.

2.3 Biomass determination

For each treatment, three 20 x 20 m plots were set aside. The diameter at breast height (DBH), height of the tree, and basal diameter were recorded for all the trees in each plot. Also, a total of 8 trees of *Populus* x *beijingensis* , 11 of *P. sylvestris* var. *mongolica* , and 13 of *U. pumila* were harvested from the respective plots, including roots to a depth of 100 cm. The trees were washed and cut into 2 m sections, and each section was separated into stems, branches, and leaves. Portions of stems, branches, leaves, and roots were sampled from each tree. All samples were weighed in the field, packed into polythene bags, transported to the laboratory, and dried to constant weight at 80 degC for estimating their C content. Biomass of individual trees was estimated using allometric regressions (Table 1).

Biomass of the herbs component of the vegetation was measured by destructive sampling in five randomly chosen 1 x 1 m subplots within each main plot. The above-ground plant parts and roots were harvested in each subplot, and their fresh weights recorded in the field. Subsamples of above-ground plant parts and roots were packed into polythene bags, transported to the laboratory, and dried to constant weight at 80 degC for estimating their C content. The biomass of the herbs component was determined from the ratio of the dry weight to the fresh weight.

Biomass of the litter was estimated the same way as that for the herbs component.

2.4 Soil carbon pool

In each treatment, three soil profiles were dug to a depth of 100 cm. Soil samples were taken at

intervals of 10 cm to determine their C content and bulk density, which was measured for three replicates from each depth. A pooled sample from three individual sampling points at each depth yielded a homogeneous sample, which was ground fine enough to pass through a 2 mm sieve after removing roots and other debris, air-dried in the laboratory, and then stored in sealed polythene bags for determining soil C content.

2.5 Carbon analysis

Samples of biomass were ground in liquid nitrogen in a mortar and pestle to a particle size of 500 µm. The oil-bath $K_2Cr_2O_7$ titration method was used for estimating organic C in the plant tissues (Dong 1997), and organic C content of the soil samples was estimated using the $K_2Cr_2O_7$ volumetric dilution heating method (Nelson and Sommers 1982).

Calculating total carbon storage and statistical analysis

Carbon content of trees, herbs, and litter was determined by multiplying the respective biomass by its C content. The total stock of soil organic C (SOC) to a depth of 100 cm was calculated based on its organic C content, sample depth, and bulk density (Guo and Gifford 2002) as follows: SOC = BD $\times$ C_c / 100 $\times$ D, where BD is soil bulk density (g/cm^3), C_c is soil C concentration (%), and D is soil sampling depth (cm). Ecosystem carbon storage was calculated as the total of each component (trees, herbs, litter, and soil):

$$C_T = C_t + C_h + C_l + C_S$$

where C_T is total ecosystem C, C_t is the total C stored in trees (t/ha); C_h, that in herbs (t/ha); C_l, that in litter; and C_s, that in soil (t/ha in each case).

2.6 Statistical analyses

All the analyses were performed using SPSS ver. 21.0 for Windows. Means and standard deviations were calculated for all the data. One-way analysis of variance (ANOVA), Student–Newman–Keuls (SNK) tests, and least significant difference (LSD) tests were performed to compare mean values of biomass and C storage in the plant and soil components among all the treatments. Significance levels were set at $P < 0.05$ for all statistical analyses.

3 RESULTS

3.1 Carbon storage components

3.1.1 Carbon storage in trees

Concentrations of C did not vary significantly between species (S-N-K:$P>0.05$,MS=2.141,F=1.739,df=2)but the differences between different organs were significant (S-N-K:$P>0.05$,MS=32.778,F=13.457,df=3): the maximum concentrations were in leaves, followed, in that order, by woody stems, branches, and roots (Fig. 1). In the case of biomass, the differences were significant between species (SNK post hoc tests, $P>0.05$, MS=150.885, F=75.787, df = 2) and between organs (SNK post hoc tests, $P>0.05$, MS=3.191, F=2.747, df = 3), with one exception, namely that the species did not differ significantly in terms of their leaf biomass(SNK post hoc tests, $P>0.05$, MS=0.166, F=1.005, df = 2). The biomass of woody stems was maximum, followed, in that order, by that of branches, roots, and leaves (Fig. 2).

The conversion of biomass to carbon indicated that woody stems of all the tree species formed the largest pool of stored C, although the size of the pool differed significantly among the species (SNK post hoc tests, $P>0.05$, MS=20.464, F=32.922, df = 2). The size of the pool in the remaining organs was maximum in branches, followed, in that order, by roots and leaves (Table 2).

3.1.2 Carbon storage in herbs and litter

Carbon contents of above- and below-ground herb organs were significantly higher in all the three treatments than in those in the control (above, SNK post hoc tests: $P>0.05$, MS=19.390, F=6.455, df=3; below, SNK post hoc tests: $P>0.05$, MS=34.679, F=3.190, df=3), although the treatments did not differ significantly among themselves (Fig. 3). The carbon content of standing litter did not differ significantly among the three treatments (SNK post hoc tests, $P>0.05$, MS=6.777, F=0.467, df=2) and was comparable to that in the control (SNK post hoc tests, $P>0.05$, MS=4.858, F=0.397, df=3).

As with the C concentrations, herb biomass was significantly higher in all the three treatments than that in the control (Fig. 4) (SNK post hoc tests, $P>0.05$, MS=1.950, F=8.871, df=3); the biomass of standing litter too did not differ significantly among the three treatments(SNK post hoc tests, $P>0.05$, MS=0.003, F=0.083, df=2) and was comparable to that in the control (SNK post hoc tests, $P>0.05$, MS=0.002, F=0.058, df=3).

Carbon concentrations of herbs and of standing litter are presented in Table 3. Above-ground C storage in the herb layer was significantly lower in the control than that in any of the three treatments (SNK post hoc tests, $P>0.05$, MS=0.213, F=3.908, df=3). Moreover, this C pool was significantly higher in FG than in UG and MG, although the latter two did not differ significantly between themselves (SNK post hoc tests, $P>0.05$, MS=0.155, F=2.248, df=2). The below-ground component was also significantly lower in the control than in that in any of the treatments (SNK post hoc tests, $P>0.05$, MS=0.202, F=12.723, df = 3), although the treatments did not differ significantly among themselves (SNK post hoc tests, P>0.05, MS=0.0.031, F=1.513, df=2).

3.1.3 Carbon storage in soil

Bulk density of soil from the control was significantly higher than that from any of the treatments at all depths up to 80 cm (Fig. 5) (SNK post hoc tests, P>0.05, MS=0.255, F=15.488, df = 3), beyond which the difference ceased to be significant, although the bulk density increased with depth.

Concentrations of SOC decreased with depth (Fig. 6) and differed significantly between the control and any of the three treatments at all depths (other than 90–100 cm) (SNK post hoc tests, p>0.05, MS=0.775, F=14.794, df=3). The values also differed significantly between the treatments at some depths.

The contents of stored C (as calculated from the soil bulk density and the concentration of C) at each depth are presented in Table 4. Although bulk density was the highest in the control, it had significantly lower stored C than that in any of the other treatments at all depths (SNK post hoc tests, p>0.05, MS=201.857, F=12.969, df = 3). The treatments differed significantly among themselves at all depths (SNK post hoc tests: p>0.05, MS = 10.301, F = 0.509, df = 2). The contents of stored C decreased with depth, with the highest values in the 10–20 cm layer. The highest value was recorded in FG (18.82 t/ha, followed, in that order, by MG (14.91t/ha), UG (12.36 t/ha), and CK (5.93 t/ha). The lowest values were recorded in the lowest or the deepest layer (90–100 cm); at that depth, the rank order was somewhat different: the maximum value was in MG (3.18 t/ha), followed, in that order, by CK (1.64 t/ha), UG (1.45 t/ha), and FG (0.98 t/ha).

3.2 Total carbon storage pools and their distribution

Ecosystem C storage is summarized in Table 5. Different components of the ecosystem differed considerably among themselves in terms of the amount of C stored, the soil pool being the largest (72.08%–96.53% of the total ecosystem C pool), followed, in that order, by the tree pool (0%–25.99%), the herb pool (1.37%–2.70%), and the standing litter pool (0.18–0.78%). Carbon storage in all the pools was significantly lower in the control than that in any other treatment (SNK post hoc tests: P>0.05, MS=4324.15, F=3077.85, df = 3). The treatments differed significantly among themselves in the amounts of stored C in all the pools except that the standing litter pool (SNK post hoc tests: P>0.05, MS=0.0004, F=0.706, df = 2), in which the values were comparable across the three treatments.

The total ecosystem C pool was 104.29 t/ha in FG, 92.24 t/ha in UG, 102.96 t/ha in MG, and 24.48 t/ha in CK (Fig. 7). Compared to that in CK, ecosystem C storage increased significantly over time in all the treatments (SNK post hoc tests: P>0.05, MS=4324.15, F=3077.85, df=3), the pool in FG being the largest.

4 DISSCUSSION

4.1 Influence of planting trees on carbon storage

Afforestation and reforestation are regarded as potentially effective strategies for mitigating the adverse effects of global climate change (Canadell and Raupach 2008, Kaul et al. 2010). The restoration of degraded lands in arid and semi-arid tracts by planting appropriate tree species is a method followed worldwide because it protects the soil, arrests desertification by increasing vegetation cover, increases the amount of C sequestered, and supplies natural resources (Kumar et al. 2001, Maestre and Cortina 2004, Nosetto et al. 2006, Grünzweig et al. 2010). The capacity of trees species to store C can be exploited to increase regional C budgets, the choice of tree species for afforestation programmes aimed at greater C sequestration being guided by that capacity (Schulp et al. 2008, Chen et al. 2016). The high potential for C storage by afforestation of desertified grasslands has been established by several researchers. For example, Hu et al. (2008) found a distinct increase in the above-ground C stocks after the former grasslands in the Keerqin sand lands had been planted with Mongolian pines and poplars, and Li et al. (2013) noted an increase in the C sink in the form of the ecosystem's biomass component.

In the present study, we found that 20 years after planting, the amount of stored C had increased by 27.11 t/ha in the stands of *P. × beijingensis* , by 22.38 t/ha in those of *U. pumila* , and by 21.34 t/ha in those of *P.*

sylvestris var. *mongolica* .*Populus* thus significantly outperformed the other species. These results support the view that living wood in regrown forests is a dominant sink for atmospheric CO_2. We do not have data on the growth rates of the above tree species: those data would have been useful in determining the rates of C sequestration. Nevertheless, the contributions of tree species in restoring desertified sandy grassland by reducing wind erosion and by greater stabilization of surface sand are clearly established by the present study. These benefits accrue while trees sequester C and enhance local biodiversity (Kirby and Potvin 2007). The differences in biomass C among the treatments were probably closely linked to the differences in the tree species, in management intensity, and in soil quality (Giese et al. 2003, Lal 2005, Mills and Cowling 2010).

The grasslands that were enclosed by shelter belts of forest trees and those in which scattered pockets had been planted with trees had greater amounts of biomass C than that in the control (Table 3). Average C stored in biomass amounted to 19.77% of the ecosystem total and was significantly higher in FG than in UG and MG. Vegetation is a very important pool of C in sandy grasslands, where wood acts as a substantial reservoir of C (Dixon et al. 1994). Approximately 50% of C assimilated by young plants may be transferred below the ground, where it is used for building and maintaining the root system and for respiration; some organic matter from roots is lost to the soil through exudation and root turnover (Rees et al. 2005). Trees, shrubs, and perennial grasses can be grown on arid and semi-arid lands to reduce soil erosion; re-establishing vegetation is particularly important for arresting the desertification of grasslands.

4.2 Sequestration of soil organic carbon by vegetation

Sequestration of soil C implies the transfer of atmospheric CO_2 into the soil of a land unit through plants grown on that land (Lal et al. 2015). Although the effects of afforestation on soil C storage in arid and semi-arid areas have been examined in the past, the results continue to be uncertain: some found that afforestation of grasslands had no effect on SOC from 5 to 30 years after planting (Davis and Condron 2002,Perez-Quezada et al. 2010, Cunningham et al. 2012) whereas Paul et al. (2002) found that following such afforestation, soil C decreased during the first 5 years but recovered later so that after about 30 years, the difference was either minimal or that soil C was slightly more and continued to increase gradually. Hu et al. (2008) suggested that the loss of C in the mineral component of soil was partly compensated for by the increased stocks of C in roots. Jackson et al. (2002) found a clear negative relationship between precipitation and changes in soil organic matter in grasslands that had been invaded by woody vegetation, with drier sites gaining SOC and wetter sites losing it. In contrast with these studies, our results suggest that planting trees in grasslands could increase SOC significantly, a finding supported by Li et al. (2013), who maintain that the accumulation of C in soil may be due to annual inputs through net primary production that exceeds the amount lost by decomposition. However, these effects were found to vary with the region, because they are influenced by soil texture, tree species, age of the stand, and management practices. Therefore, more empirical data are needed to identify the point in time after which soil C begins to be sequestered as a result of afforesting a desertified grassland, such as the one examined in the present study.

Storage of soil C decreased with depth, an observation consistent with that made by Grandy and Robertson (2007), the storage being maximum in the 10–20 cm layer. The concentration of soil C across all depths (except the 90–100 cm layer) differed significantly between the three treatments and the control. Among the three treatments, the concentration in the 0–40 cm layer was significantly higher in FG than in UG and MG. Total C in the uppermost layer (0–5 cm) was found to increase substantially after tree planting, but that in the 0–30 cm layer remained the same (Cunningham et al. 2012). Noble et al. (1999) also found higher SOC in the 0–5 cm layer of a plantation than that of a pasture, but not at greater depths (5–50 cm). Similarly, at most of the 28 sites studied by Davis and Condron (2002), afforestation showed no pronounced influence on soil C at depths greater than 10 cm.

The total estimated pool of SOC up to a depth of 100 cm was 23.63–79.98 t/ha, which amounted to 80.12% of the total pool in the ecosystem, showing that terrestrial C is sequestered mainly in soil. This finding is consistent with the findings of Fang et al. (2001) and Li et al. (2007). Dixon et al. (1994) reported that more than a third of the C pool in terrestrial ecosystems is contributed by soil organic matter. Detwiler (1986) reported that 35%–80% of the C stored up to a depth of 100 cm is concentrated in the 0–40 cm layer in tropical and subtropical soils; we obtained similar results, with 41.81%–60.13% stored in the 0–30 cm layer and 59.42%–80.80% in the 0–50 cm layer. The accumulation of SOC is controlled by the rates of biomass formation and decomposition (Lal et al. 1995). The turnover of fine roots is another major component of the dynamics of SOC; the size of this turnover matches that of leaf litter (Rasse et al. 2005). In the present study, the lowest amount of soil C was in the control, probably because of much lower input of litter, greater abundance of woody and herbaceous vegetation, different patterns of distribution of roots at different depths, and the differences in the availability of soil moisture and soil temperature, all of which accelerated the decomposition of organic matter (Covington 1981, Jackson et al. 2000). The storage of soil C was significantly greater in MG than in FG and UG, an observation consistent with earlier observations on the enrichment of soil organic matter as a result of greater vegetation cover ensured by appropriate intervention (Fearnehough et al. 1998, Wezel et al. 2000).

4.3 Carbon sequestration potential of grassland restoration in Horqin sandy land

Desertification expanded from near-zero levels during the 19th century to reach 50.198 km^2 by 2000, which amounts to 47.6% of the total land area of Horqin (Zhao et al. 2009). Desertification affects the storage of C greatly; for example, Zhou et al. (2008) calculated that a total of 107.53 Mt of C was lost because of the desertification of the Horqin sandy lands during the 20th century. As a result of various efforts at restoring these lands, the desertified area decreased from 48.5% of total land area in 1975 to 47.5% (50.147 km^2) in 2000. These efforts include afforestation of active sand dunes and banning grazing.

Li et al. (2013) found that SOC to a depth of 100 cm increased by 205 kg ha^{-1} every year over 28 years of afforestation with Mongolian pine, and Hu et al. (2008) found that C stock of the total ecosystem increased following the afforestation of grassland, above-ground stocks by 2.75 Mg ha^{-1} a year in the poplar plantations and by 1.06 Mg ha^{-1} a year in the Mongolian pine plantations. Li et al. (2013) estimated that if 30% of the severely desertified land (290.61 ha) were to be afforested with Mongolian pine, the C sequestered to a depth of 100 cm could reach 3.01 Mt after 38 years and that the ecosystem biomass could reach 4.48 Mt.

As to the present study, a conservative estimate puts the ecosystem C pools, after more than 20 years, at 104.29 t/ha in FG, 102.96 t/ha in MG, 92.24 t/ha in UG, and 24.48 t/ha in CK. Compared to CK, the amounts increased significantly over time, the total pools in any of the three treatments being, on average, four times those in the untreated control, particularly in soil and in the trees. The differences between the three treatments were probably due to the treatments themselves, and also due to the species, the extent of disturbance, and management practices in the past (zheng et al. 2008, Bisbing et al. 2010). These grasslands thus have a considerable potential to sequester C, as was also shown by Jones and Donnelly (2004) and He et al. (2008).

Controlling desertification requires re-establishment of vegetation cover, conservation of soil and water, enhancement of soil quality, and increased biomass production (Lal 2001). Different management practices that enhance or weaken C storage in the Horqin grasslands have significant implications for the global C budget, because the grasslands of northern China form a significant proportion of the Eurasian continent (Ojima et al. 1993).

4.4 Challenges for afforestation in Horqin sandy land

Degradation of grasslands reduces ground cover, leaving bare surfaces exposed directly to erosion by strong winds. In these damaged landscapes where soils are sandy and winds speeds are high, large quantities of C are lost to wind and to water (Gomes et al. 2003, Li et al. 2004, Su et al. 2004, Hoffmann et al. 2011). Afforestation, either by planting trees or by allowing natural forests to regenerate, is effective in restoring degraded soils and ecosystems, one of the principal advantages of these measures being that they deploy natural processes instead of potentially expensive artificial processes (Li et al. 2013). Dryland forests play an important role in the stabilization of land, reduction of erosion, protection of watersheds, and the control of desertification. However, tree planting in drylands remains controversial because the trees compete with other vegetation for water. Not all trees can be grown everywhere, and trees should be planted only when and where the water available in a dryland ecosystem to sustain their growth and survival in the long term, while permitting the survival and growth of other vegetation. Trees have also been dying in recent years, and Wang et al. (2010) and Yan et al. (2011) attribute the deaths partly to the warmer and drier climate associated with climate change, because the warmer climate is expected not only to decrease the availability of water to trees greatly but also to increase transpiration. Therefore, the choice of suitable tree species is particularly important for the afforestation of degraded lands in arid and semi-arid regions (Reubens et al. 2011). Further research will be needed to determine the minimum rainfall required to support each species used in afforestation, because many species cannot survive under these conditions (Li et al. 2017). This work will help managers to choose species that are most likely to survive in a given area. On the other hand, the need for a more sustainable trade-off between protecting and improving the local environment and developing the local economy is becoming increasingly acute in the Horqin sandy lands.

In the present study, comprehensive analysis shows that grasslands enclosed by a shelter belt of *Populus* x *beijingensis* or planted with *Pinus sylvestris* var. *mongolica* or with *Ulmus pumila* in widely distributed but small pockets result in a mosaic of pure forests across the landscape, which is dotted with large trees—a landscape that is different not only from the complex structure of natural forests but also from the simple structure of grassland, with different levels of reflection, a unique community structure that offers protection from strong winds, effectively reducing wind speeds that affect the herbaceous layer. *Populus* x*beijingensis, Pinus sylvestris* var. *mongolica* , and *Ulmus pumila* are better adapted to the moderately desertified grasslands in the south-eastern Horqin sandy land in northern China**.** Therefore, afforesting grasslands through establishing small but widely distributed pockets of trees or by enclosing the grasslands with a shelter belt is a suitable approach in semi-arid areas that may be lacking in adequate levels of soil moisture and available nutrients.

5 CONCLUSION

In the south-eastern sandy land in Horqin in northern China, enclosing grasslands within a shelter belt of *Populus* x *beijingensis*(FG) or raising in them *Pinus sylvestris* var. *mongolica* or *Ulmus pumila* in widely distributed but small pockets increased the accumulation of C in the ecosystem (in plants and in soil) significantly. Mainly because of the differences in the tree species, the C pool was the largest in FG. Biomass C in all the three treatments was also significantly higher than that in the control. The greatest increase in soil C was in the upper layer of the soil profile (to a depth of 50 cm). All the three approaches will contribute greatly to C sequestration globally and are suitable for arresting desertification in semi-arid areas that may be lacking in adequate levels of soil moisture and available nutrients.

CONFLICT OF INTEREST STATEMENT

The authors declare that they have no conflicts of interest.

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Table captions

Table 1. Diameter at breast height-based regression equations for computing biomass in tree organs.

Table 2. Carbon storage in tree organs

Table 3. Carbon storage in herbs and in standing litter.

Table 4. Soil carbon content at varying depths up to 100 cm.

Table 5. Carbon storage in different components (pools) of the ecosystem.

Table 1. Regression equations for computing biomass of different organs of trees based on DBH, or diameter at breast height.

Tree	Organ	Model	R^2
Populus × beijingensis	Stem	$W_s = 0.1790(D^2H)^{1.8444}$	0.96

Tree	Organ	Model	R^2
	Branch	$W_b = 0.0331(D^2H)^{2.1165}$	0.91
	Leaf	$W_l = 0.02263(D^2H)^{1.5552}$	0.89
	Root	$W_r = 0.02745(D^2H)^{1.7670}$	0.95
Pinus sylvestris var. *mongolica*	Stem	$W_s = 0.06823(D^2H)^{2.1067}$	0.95
	Branch	$W_b = 0.0093(D^2h)^{2.1443}$	0.94
	Leaf	$W_l = 0.0524(D^2H)^{1.2486}$	0.92
	Root	$W_r = 0.0183(D^2H)^{2.2119}$	0.84
Ulmus pumila	Stem	$W_s = 0.02997(D^2H)^{1.0531}$	0.81
	Branch	$W_b = 0.008534(D^2H)^{2.4337}$	0.91
	Leaf	$W_l = 0.003371(D^2H)^{2.1687}$	0.85
	Root	$W_r = 0.0379(D^2H)^{2.4946}$	0.93

Table 2. Carbon storage in tree organs.

Species	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage
	Woody stems	Branches	Leaves	Roots
Populus × beijingensis	15.93 (0.75)a	5.47 (0.57)ab	1.16 (0.14)a	4.55 (0.55)a
Pinus sylvestris var. *mongolica*	10.94 (1.09)c	6.23 (0.80)a	1.49 (0.13)a	2.68 (0.48)b
Ulmus Pumila	12.09 (0.65)b	4.85 (0.57)b	1.18 (0.12)a	4.26 (0.44)a

Values are means (SD) of three replicates. Means in a row followed by the same lowercase letter are not significantly different ($P > 0.05$; SNK post hoc tests)

Table 3. Carbon storage in herbs and in standing litter.

Layer	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t
	FG[a]	UG[b]	MG[c]	CK[d]
Above-ground parts of herbs	1.09 (0.48)a	0.65 (0.10)bc	0.76 (0.01)b	0.45 (0.11)c
Underground parts of herbs	0.71 (0.19)a	0.61 (0.02)a	0.69 (0.24)a	0.21 (0.04)b
Standing litter	0.21 (0.05)a	0.19 (0.08)a	0.19 (0.02)a	0.19 (0.03)a

Values are means (SD) of three replicates. Means in a row followed by the same lowercase letter are not significantly different ($P > 0.05$; SNK post hoc tests);

[a] Grassland enclosed by a belt of trees;[b] Grassland afforested with *Ulmus pumila* ;[c] Grassland afforested with *Pinus sylvestris*var. *mongolica* ; [d] Desertified sandy grassland.

Table 4. Soil carbon content at varying depths up to 100 cm.

Soil depth (cm)	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)
	FG[a]	UG[b]	MG[c]	CK[d]
0–10	13.80 (0.22)a	11.98 (0.25)c	12.42 (0.74)b	2.12 (0.27)d
10–20	18.82 (0.23)a	12.36 (0.15)c	14.91 (0.92)b	5.93 (0.20)d
20–30	12.58 (1.37)a	9.40 (0.11)b	8.33 (0.05)c	1.83 (0.01)d
30–40	9.40 (0.24)a	8.92 (0.09)a	9.33 (0.11)a	2.09 (0.07)b

Soil depth (cm)	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)
40–50	6.14 (0.37)c	7.67 (0.05)b	8.33 (0.35)a	2.07 (0.12)d
50–60	6.51 (0.24)b	6.58 (0.10)b	8.13 (0.67)a	2.26 (0.20)c
60–70	3.74 (0.10)b	3.86 (0.09)b	6.55 (0.38)a	1.76 (0.05)c
70–80	2.04 (0.01)c	3.82 (0.05)b	4.88 (0.11)a	2.05 (0.27)d
80–90	1.16 (0.06)c	2.37 (0.09)b	3.92 (0.28)a	1.88 (0.17)c
90–100	0.98 (0.07)c	1.45 (0.15)b	3.18 (0.22)a	1.64 (0.21)b

Values are means (SD) of three replicates. Means in a row followed by the same lowercase letter are not significantly different (P > 0.05; SNK post hoc tests);

[a] Grassland enclosed by a belt of trees;[b] Grassland afforested with *Ulmus pumila* ;[c] Grassland afforested with *Pinus sylvestris*var. *mongolica* ; [d] Desertified sandy grassland.

Table 5. Carbon storage in different components (pools) of the ecosystem.

Component	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)	Carbon storage (t/ha)
	FG[a]	UG[b]	MG[c]	CK[d]
Trees	27.11 (0.47)a	22.38 (0.87)b	21.34 (0.50)b	—
Herbs	1.80 (0.12)a	1.26 (0.26)b	1.45 (0.86)ab	0.66 (0.25)c
Standing litter	0.21 (0.08)a	0.19 (0.04)a	0.19 (0.03)a	0.19 (0.03)a
Soil	75.17 (0.77)b	68.41 (1.21)c	79.98 (1.45)a	23.63 (0.83)d
Total	104.29 (1.71)a	92.24 (0.93)b	102.96 (1.35)a	24.48 (1.08)c

Values are means (SD) of three replicates. Means in a row followed by the same lowercase letter are not significantly different (P > 0.05; SNK post hoc tests);

[a] Grassland enclosed by a belt of trees;[b] Grassland afforested with *Ulmus pumila* ;[c] Grassland afforested with *Pinus sylvestris*var. *mongolica* ; [d] Desertified sandy grassland.

Figure captions

Fig. 1 Carbon concentrations in different organs of trees. Error bars are standard deviations (n = 3). The letters a and b indicate significant differences (P > 0.05; SNK post hoc tests).

Fig. 2 Biomass of different organs of trees. Error bars are standard deviations (n = 3). The letters a and b indicate significant differences (P > 0.05; SNK post hoc tests).

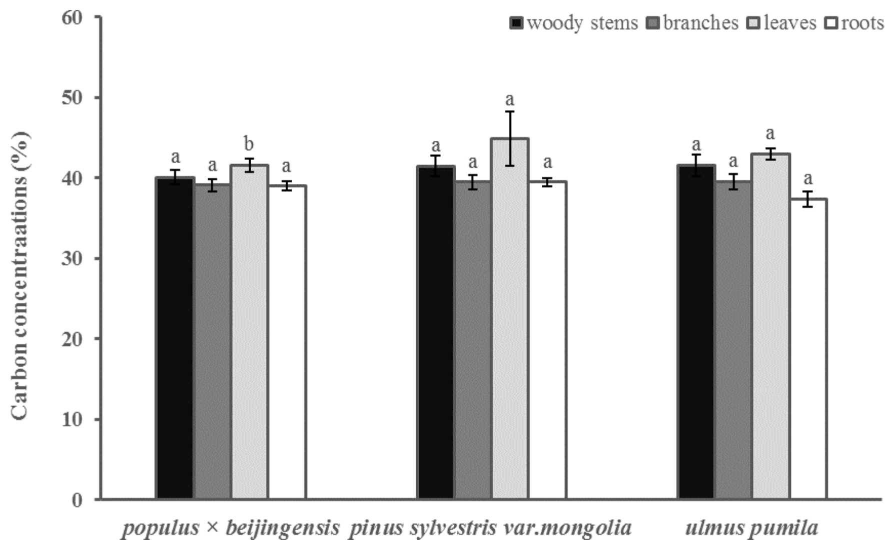
Fig. 3 Carbon concentrations in herb layer and in standing litter. Error bars are standard deviations (n = 3). The letters a and b indicate significant differences (P > 0.05; SNK post hoc

Fig. 4 Biomass of herbs and of standing litter. Error bars are standard deviations (n = 3). The letters a and b indicate significant differences (P > 0.05; SNK post hoc tests).

Fig. 5 Soil bulk density at varying depths up to 100 cm

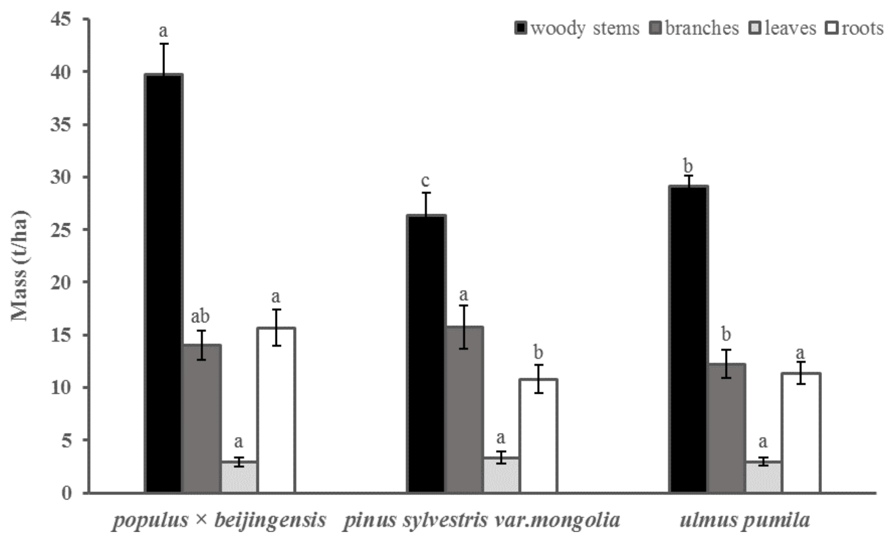
Fig. 6 Soil carbon concentrations at varying depths up to 100 cm.

Fig. 7 Ecosystem carbon storage. Error bars are standrad deviations (n = 3). The letters a and b indicate significant differences (P > 0.05; SNK post hoc tests).



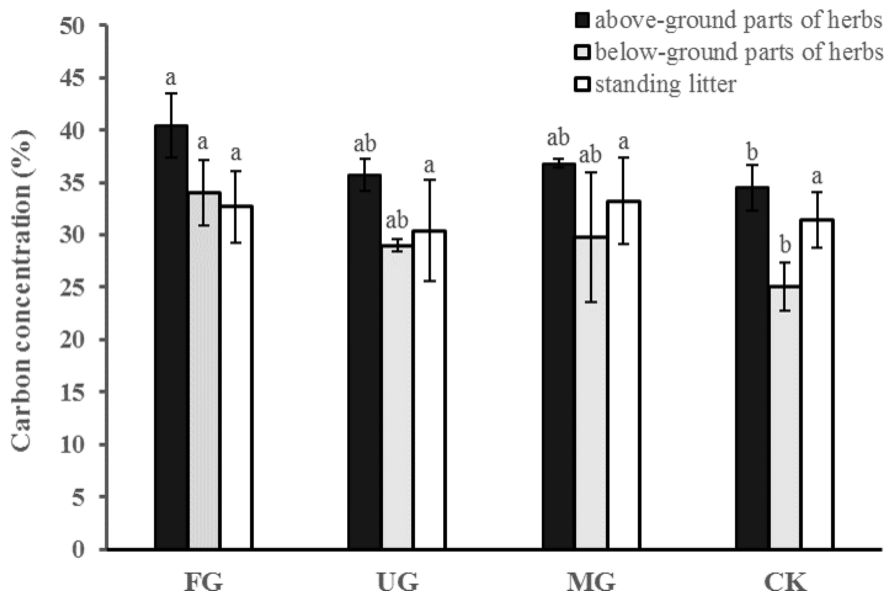
woody stems
branches
leaves
roots
Carbon concentraations (%)
0
10
20
30
40
50
60
a
a
b
a
a
a
a
a
a
a
a
a
populus × beijingensis
pinus sylvestris var.mongolia
ulmus pumila

Fig. 1



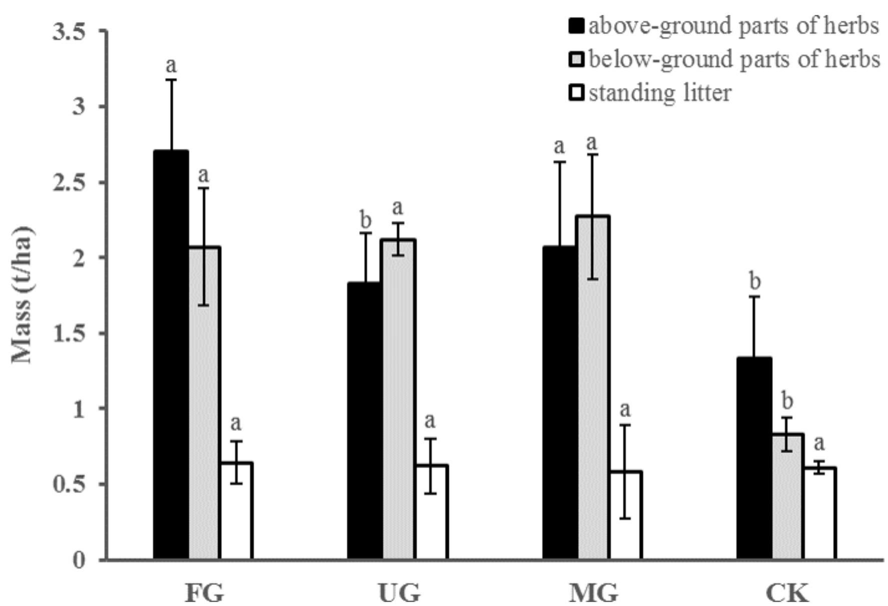
woody stems
branches
leaves
roots
Mass (t/ha)
0
5
10
15
20
25
30
35
40
45
a
ab
a
a
c
a
a
b
b
b
a
a
populus × beijingensis
pinus sylvestris var.mongolia
ulmus pumila

Fig. 2



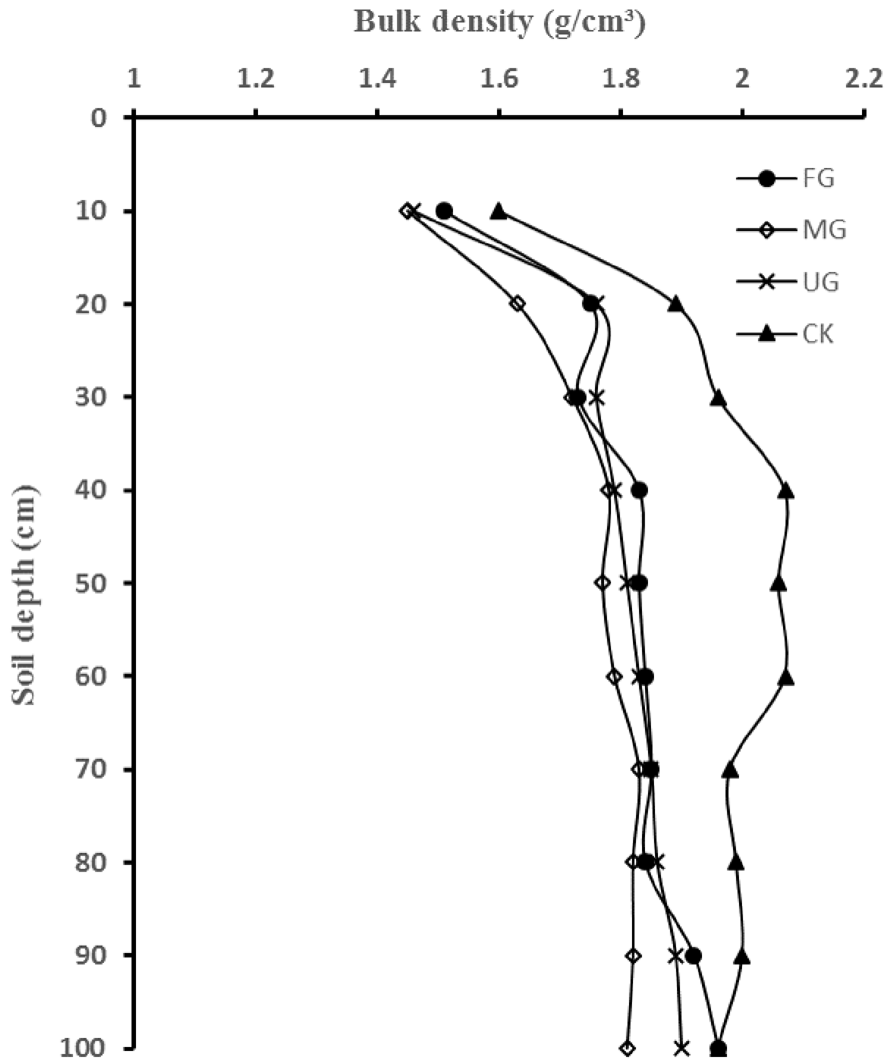
above-ground parts of herbs
below-ground parts of herbs
standing litter
Carbon concentration (%)
50
45
40
35
30
25
20
15
10
5
0
a
a
a
ab
ab
a
ab
ab
a
b
b
a
FG
UG
MG
CK

Fig. 3



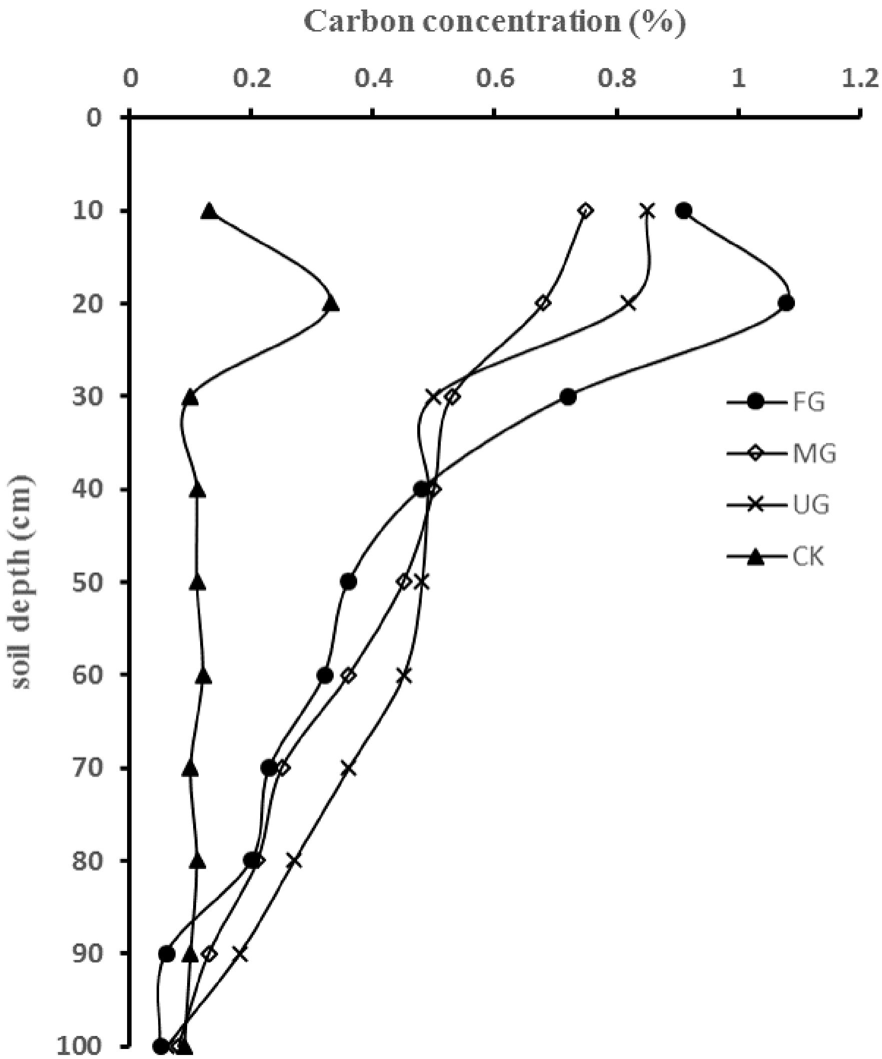
above-ground parts of herbs
below-ground parts of herbs
standing litter
Mass (t/ha)
3.5
3
2.5
2
1.5
1
0.5
0
a
a
a
b
a
a
a
a
a
b
b
a
FG
UG
MG
CK

Fig. 4



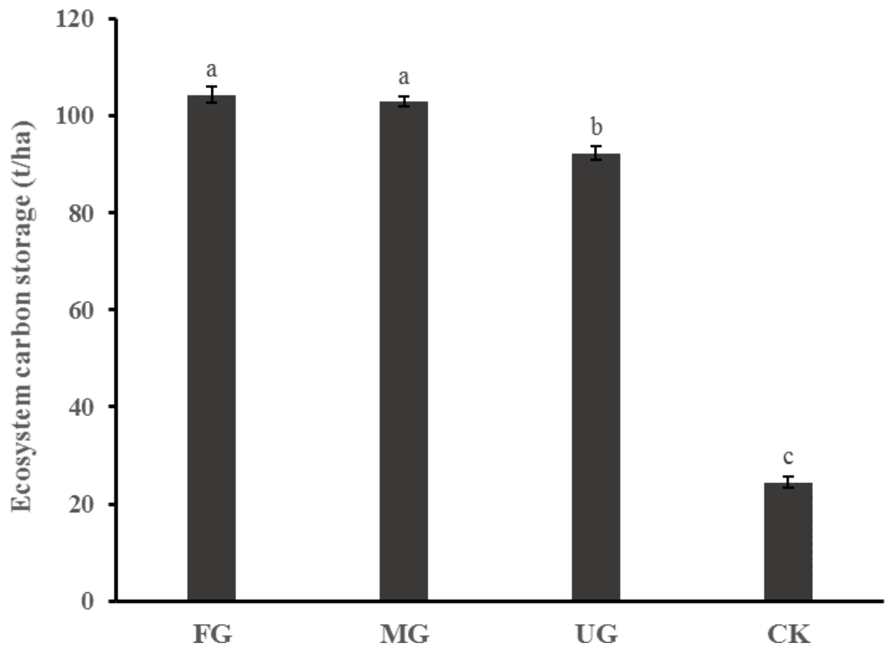
Bulk density (g/cm³)
1
1.2
1.4
1.6
1.8
2
2.2
0
10
20
30
40
50
60
70
80
90
100
Soil depth (cm)
FG
MG
UG
CK

Fig. 5



Carbon concentration (%)
0
0.2
0.4
0.6
0.8
1
1.2
0
10
20
30
40
50
60
70
80
90
100
soil depth (cm)
FG
MG
UG
CK

Fig. 6



Ecosystem carbon storage (t/ha)
120
100
80
60
40
20
0
a
a
b
c
FG
MG
UG
CK

Fig. 7