

Long-Term Elimination of Grazing Reverses the Effects of Shrub Encroachment on Soil and Vegetation on the Ordos Plateau

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Key Points:

- Soil texture and soil water holding capacity in the surface layer was improved in shrub-encroached grassland when grazing was eliminated
- Restoration of shrub-encroached grasslands was related to soil improvement in surface layer
- Shrub-encroached grasslands could be restored in about 20 years after grazing was eliminated

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Abstract Conversion from grasslands to shrublands resulting from overgrazing occurs worldwide and is reinforced by the global climatic change. The functioning, biogeochemical cycles (e.g., carbon and water), and the stability of grassland ecosystems are significantly affected by the conversion. To best manage and use those grassland ecosystems, it is imperative to examine how the shrub-encroached grasslands can be restored to native grassland ecosystems. In this study, aboveground biomass of all species, root biomass, soil water content, and soil texture in a set of fenced and unfenced plots were observed every three years from 1992 to 2011 in a semishrub encroached grassland on the Ordos Plateau of China. In the fenced plots, aboveground biomass increased for grass species and decreased for semishrub species. Root biomass and soil water content in the 0–10-cm soil layer increased, but decreased in the 10–30-cm soil layers. The fraction of soil particle sizes less than 0.01 mm increased in the 0–10-cm soil layer. In contrast, these variables remained constant in the unfenced plots over the experimental period. The semishrub encroached grassland dominated by *Artemisia ordosica* was replaced with a native perennial grass, *Stipa bungeana*, when grazing was eliminated in the fenced plots. Changes in soil texture and moisture after fencing were correlated with the native grassland restoration from semishrub encroached grasslands. This study suggests that, to restore the native grassland, grazing should be eliminated in the study region.

1. Introduction

Shrubs have encroached extensive areas of grasslands globally including North America (Bhark & Small, 2003; Jackson et al., 2002), Asia (Alward et al., 1999; Bhark & Small, 2003), and even the Arctic (Sturm et al., 2001). Shrub encroachment threatens grassy ecosystems worldwide (Case & Staver, 2017; D'Odorico et al., 2012; Mitchard & Flintrop, 2013; Stevens et al., 2017). These encroachments have generally been attributed to human disturbances (mainly overgrazing; Bhark & Small, 2003; Polley et al., 1994; Williams & Hobbs, 1989) and are likely to be accelerated due to global climatic changes (Alward et al., 1999; Sturm et al., 2001; Walther et al., 2002). Conversion from grasslands to shrublands has significant consequences for biodiversity conservation, livestock production, and other ecosystem services (Lautenbach et al., 2017; Ratajczak et al., 2016). The conversion process affects the structure and functioning of grassland ecosystems including their stability, life span, phenology, structural complexity, rooting depth and distribution, and biomass allocation (Brown et al., 2001; Chen, 1964; Christian, 2001; Diaz & Cabido, 1997; Jackson et al., 2002; Zavaleta, 2006; Zhou et al., 2017).

The encroachment process is determined by the interaction between grass and shrub species, and is mediated by the growth characteristics of the component species and their water and nutrient use (Williams et al., 1987) as well as human disturbance, such as overgrazing. Changes in soil water availability can influence shrub encroachment and thus grassland stability (Bhark & Small, 2003; Ridolfi et al., 2008; Schade & Hobbie, 2005). Bottom-up hypotheses related to the effects of resources such as soil water and nitrogen and top-down hypotheses about the effects of fire and herbivory (Case & Staver, 2017; Cipriotti et al., 2014; Ratajczak et al., 2017) were used to explain shrub encroachment. One of the bottom-up hypotheses is niche separation that assumes that water is the main limiting resource, stating that shrubs have deeper roots and use deeper soil water compared with grasses, so deeper infiltration of soil moisture favors shrub growth relative to grasses (Cipriotti et al., 2014; O'Connor et al., 2014). Specifically, Cipriotti et al. (2014) and

Ratajczak et al. (2017) stated that the two-layer hypothesis indicated that the selective use of deep-soil water increases the growth or persistence of woody species and shallow-soil water benefits the growth of grass species in dry ecosystems, while the resource pool hypothesis suggests that shallow-soil water benefits the growth of all plants and deep-soil water benefits woody species. However, there is no universal hypothesis to explain shrub encroachment worldwide.

Conversion from shrublands to grasslands is thought to be extremely difficult (Barrow & Havstad, 1995; Bhark & Small, 2003). There have been a number of studies to examine whether the shrub-encroached grasslands can convert to the native grasslands (Archer et al., 1995; Brown et al., 2001; Hobbs & Mooney, 1986; Knoop & Walker, 1985; Paruelo et al., 1998; Williams et al., 1987). These studies mostly focused on examining grassland recovery after relatively short periods of shrub encroachments, which have started 100–200 years ago, while shrub encroachments have started 600–700 years ago (e.g., in some areas of China). Thus, those studies had difficulties to draw sound conclusions about long-term processes of grassland conversion after shrub encroachment. Research is needed to examine the long-term restoration mechanisms in grassland ecosystems where shrubs have encroached and dominated native grass species for a long time.

In semiarid and arid areas, overgrazing can induce desertification (Xiao et al., 2003) through wind erosion. Because of sheep trample, fine particle size soils are blown away, and soils become sandy. Consequently, soil water after rain can easily infiltrate into deep soils (Xu & Li, 2003). This change in soil moisture benefits deep root shrub species, but not shallow root grass, and subsequently grasslands can be encroached by shrub (Cipriotti et al., 2014; O'Connor et al., 2014). We hypothesize that, when grazing is eliminated, the process of soil change reverses, and shrub encroached-grasslands can be converted to native grasslands gradually.

The semiarid temperate grasslands in China occupy 78% of the national grassland area and 27% of the area of temperate grasslands in the world (Chen & Wang, 2000). The Ordos Plateau, a region of semiarid temperate grasslands, lies in southern Inner Mongolia and is bounded by the Great Wall to the south and the Yellow River on the other three sides (Zheng et al., 2006), with a total area of about 120,000 km² (Figure A1). The grasslands of the Ordos Plateau were historically among the region's highest-quality grasslands (Zheng et al., 2005). Due to rapid desertification, however, much of the area has been turned into sandy semishrub steppe. The original dominant steppe grass species, *Stipa bungeana* Trin., has been replaced by the semishrub *Artemisia ordosica* Krasch (Zheng et al., 2005), distributed mainly in the Inner Mongolia and Ningxia Autonomous region (Cui, 1991). The degradation of the grasslands of the Ordos Plateau started 2,400 years ago (Xu et al., 2002), and resulted from a combination of adverse climatic change and human activities including grazing. The most severe grassland degradation and associated *Artemisia* encroachment, however, has occurred mainly over the last 600–700 years (during the Yuan Dynasty), especially in the most recent 300–400 years (Qing Dynasty), during which the *A. ordosica* community has become well established, and its dominance continues to the present (Wang et al., 1997; Zeng, 1964). Compared with *S. bungeana*, *A. ordosica* has low nutrition value and palatability. Furthermore, the ability to reduce wind and water erosions for grasslands encroached by *A. ordosica* was lower than that of *S. bungeana* steppe (Zhang, 1994). Thus, converting the semishrub encroached grasslands to native grasslands is highly necessary (Zheng et al., 2005). With this long history of semishrub encroachment and dominance by the *A. ordosica* community, the grasslands in the Ordos Plateau are ideal for examining the dynamic process of semishrub encroachment, understanding whether and how the encroached semishrub community maintains its stability, and identifying possible mechanisms for grassland restoration. Although *A. ordosica* is distributed mainly on the Ordos Plateau, the physical environment in most of the northwest China is well suited to its encroachment and establishment (Chen, 1964). Thus, such a study is of paramount significance as China has gone through an unprecedented economic development and imposed an increasing pressure on the environment.

This study used a 19-year experiment on the Ordos Plateau of Inner Mongolia to address two questions: (i) what are the major factors contributing to the long-term dominance of the *A. ordosica* community in steppe regions? and (ii) what will happen to soil and vegetation after grazing is eliminated from semishrub encroached grasslands? We hypothesized that a long-term elimination of grazing would increase the fraction of fine particles in upper soil layers, reduce the rate of moisture infiltration into deeper soils, and increase grass biomass in shrub-encroached grasslands of the Ordos Plateau.

2. Materials and Methods

2.1. Study Sites and Observations

The study site with typical *A. ordosica* vegetation was established in the Ejin Horo Banner of the Ordos Plateau, in the Inner Mongolia Autonomous Region, China (39°21'N, 109°51'E), by the Institute of Botany, Chinese Academy of Sciences (Figure A1). This region has average annual precipitation of 345.2 mm and annual mean temperature of 6.7 °C. The growing season extends from early April to early October. Sheep and goats are the main livestock in this area. The grazing density is 2–3 sheep units/ha per year (Qi, 1998).

In 1992, six 50 × 50-m plots were established and then three plots were randomly selected to have grazing eliminated by fencing (the “fenced” plots). In the fenced plots, native small herbivores, such as rabbits, were not excluded. On each of seven occasions (late September of 1992, 1996, 1999, 2002, 2005, 2008, 2011), a single 2 × 2-m quadrat was randomly located within each 50 × 50-m plot, giving three samples for each treatment per occasion. Within these quadrats, species composition and the number of species were recorded. In addition, a number of measurements were made (Schade & Hobbie, 2005; Wang & Li, 1994). First, gravimetric soil water content (g water per 100 g soil) was determined at depths of 5, 10, 20, 30, 40, 60, 80, and 100 cm using a 5-cm-diameter cutting ring (height is 5 cm). Soil cores were collected from eight soil depths (2.5–7.5, 7.5–12.5, 17.5–22.5, 27.5–32.5, 37.5–42.5, 57.5–62.5, 77.5–82.5, and 97.5–102.5 cm) with three replicates after removing any rocks and litters. Collected soil samples were immediately measured the fresh weight (FW), and then the soil samples were taken to the laboratory and oven dried at 105 °C to a constant weight to measure the dry weight (DW). Gravimetric soil water content (SWC) was calculated (g water per 100 g soil; Guo, 2000):

$$\text{SWC} = 100 \times (\text{FW} - \text{DW}) / \text{DW} \quad (1)$$

Second, soil particle size was determined within the top 10-cm depth of soil with three replicates using a 10-cm-diameter cutting ring (10 cm high). Soil samples were brought to laboratory, air dried, and the fraction of soil particle size was determined using different graded sieves. Sieves with mesh size of 0.5, 0.25, 0.1, 0.05, and 0.01 mm were used to determine the fraction of soil particle sizes of >0.5, 0.25–0.5, 0.1–0.25, 0.05–0.1, and 0.01–0.05 mm. The fraction of soil particle sizes of 0.005–0.01, 0.001–0.005, and <0.001 were determined with the hydrometer method (Bouyoucos, 1936). The organic matter was lower than 1% of the sandy soil (Zhang et al., 2018) and was removed from samples beforehand with pretreatment for further analysis.

Third, root biomass was measured by manually excavating roots. Specifically, soils were dug with a small shovel, soils and roots were collected for a 1 × 1-m area at 10-cm depth intervals to a depth of 1 m, and then these soil samples with roots were brought to laboratory, carefully washed with water in a wooden box. To avoid losing very fine roots, the bottom of the box was made of nylon mesh with mesh size of 0.12 × 0.12 mm. All roots were separated from soils and then bagged, oven dried (at 80 °C for three days), and weighed.

Fourth, aboveground plant biomass was determined by harvesting all standing biomass within a 2 × 2-m quadrat. Plants were separated according to species, bagged, oven dried (at 80 °C for three days), and then weighed. Late September is the time when biomass is at its highest (Wang & Li, 1994); gravimetric soil water content was measured at that time, if there has been no rain for a week; gravimetric soil water content was measured a week later when rain occurred. All soil and plant samples were collected almost at the same time. Each species within the quadrat was classified according to its life form into annual forb, perennial forb, rhizome grass, tussock grass, and semishrub according to Bei et al. (2004), and aboveground plant biomass was classified into four species groups.

At each of the three 50 × 50-m study plots in both fenced and unfenced plots, soil water content sensors (ECH₂O-10, Decagon Devices, Inc., Pullman, WA, USA) were used to measure the soil water content at 5-, 10-, 20-, 30-, and 40-cm soil layers. If soil water content increased obviously at the specific soil depth after rain, it meant that the rain infiltrated into that soil layer, the time for the infiltration after rain could be determined. These sensors were installed in May 2009 and soil water content was monitored for two years.

2.2. Diversity Indexes

Simpson's Index and Shannon's Entropy Index were used to calculate the species diversity of vascular plants in each plot. Simpson's Index is often used to quantify the biodiversity of a habitat in terms of species numbers and the abundance of each species (equation (2)). Shannon's Entropy Index (equation (3)) accounts for both abundance and evenness of the species in a habitat (Magurran, 1988):

$$C = \sum_{i=1}^S \frac{n_i(n_i - 1)}{n(n - 1)} \quad (2)$$

$$I = - \sum_{i=1}^S P_i \ln P_i \quad (3)$$

where n is the total number of individuals in all S species, n_i is the number of individuals in species i , and P_i is the proportion of individuals of species i in the total number of species.

2.3. Statistical Analysis

To examine the difference between the fenced and unfenced plots, observed data at different times and soil depths were analyzed using the repeated measures ANOVA of General Linear Model in SPSS (the Statistical Package for the Social Sciences version 17.0). Species group aboveground biomass were analyzed as a dependent variable, treatments (fenced and unfenced) and species group as the between-subject factors, and time as the within-subject factor. Samples for soil water content and root biomass were separated into different soil layers of 0–10, 10–30, 30–60, and 60–100 cm; these soil water content data represented the average of data at 5- and 10-, 20- and 30-, 40- and 60-, and 80- and 100-cm soil depths, respectively. Values were averaged within depth classes within each plot first, producing three averages per year per depth class as three replicates. Using three replicates, the mean and standard errors were then calculated. In the analyses, the dependent variables were soil water content and root biomass, the between-subject factors were treatments and soil depth class, and the within-subject factor was time. Samples for soil texture were grouped according to soil particle sizes including less than 0.001 mm in diameter, 0.001–0.005, 0.005–0.01, and larger than 0.01 mm. Fraction of fine soil particles (<0.001, 0.001–0.005, 0.005–0.01, and 0.01–0.05 mm in diameter) was analyzed as a dependent variable, treatment, particle sizes as the between-subject factors, and time as the within-subject factor. For each dependent variable, the effects estimated were treatment (unfenced and fenced), particle sizes, species group or soil depth class, time, and their two-way and three-way interactions.

Canonical correspondence analysis (CCA) was used to determine the relationship between plant communities and soils. Averaged aboveground biomass data for every species of three replicates in the fenced and unfenced plots in every observed year as main matrix, and gravimetric soil water content at depths of 5, 10, 20, 30, 40, and 60 cm, fraction of soil particles with size <0.001, 0.001–0.005, 0.005–0.01, 0.01–0.05, 0.05–0.1, 0.1–0.25, and 0.25–0.5 mm within the top 10-cm depth of soil were used as second matrix to conduct the CCA. CCA is a direct gradient analysis technique that correlates community species composition with environmental variation. CCA axes were evaluated statistically with a Monte Carlo test (Ozinga et al., 2005). All data variables were assessed for normality (ter Braak, 1986) prior to the CCA analysis. PC-ORD 5.0 (McCune & Mefford, 1999) was used for ordination analysis. All the default settings were used for CCA; the variables in the CCA biplots were indicated with arrows pointing to the direction of maximum variation, with their length proportional to the rate of change (Hyvönen et al., 2005).

3. Results

3.1. Differences of Aboveground Biomass and Species Composition Between the Fenced and Unfenced Plots

Fourteen species were recorded in six 50 × 50-m plots, with significant recruitment of species in the fenced plots (Table A1). The apparent increase in species richness in the fenced plots was not consistent over time. After an initial recruitment to 10 species (1996 and 1998), the numbers reduced to five (2002), then to four (2005), and then increased to six (2008, 2011; Table A1 and Figure 1). By 2002, the annual forbs (*Ixeris* and *Corispermum*) and the perennial forbs (*Allium*, *Cynanchum*, *Oxytropis*, and *Polygala*) had disappeared from

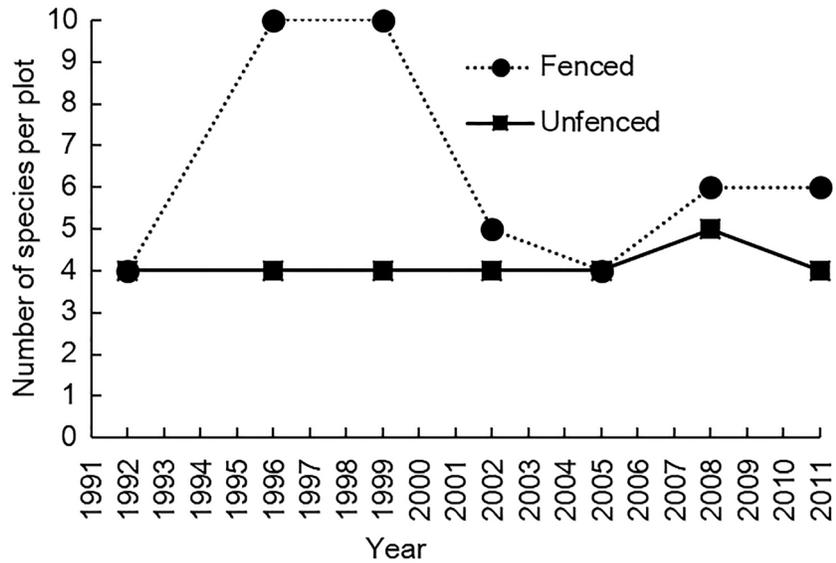


Figure 1. Species richness per plot over the observation period. The standard error bars are too small to appear on the graph at this scale.

the fenced plots. The rhizome grass (*Aneuolepidium*) only appeared once, in 1996. By 2011, the remaining species were tussock grasses (*Cleistogenes* and *Stipa*), annual forbs (*Euphorbia*), and semishrubs (*A. ordosica* and *Lespedeza*). For the unfenced plots, species stayed almost the same over the observation period (Table A1).

Obvious differences in the two diversity indices between the treatments (fenced and unfenced) were found. Both diversity indices for the unfenced plots remained fairly constant over time at 0.605 for Shannon's and 0.789 for Simpson's (Table A2). The Shannon's diversity in the fenced plots, however, increased from 1992 to 1999, decreased from 2002, while the Simpson's diversity was varying throughout.

Treatment (fenced and unfenced), species group, time, and their interaction had significant effects on the species group aboveground biomass (Table A3). The biomass of each group of semishrubs, perennial forbs, and annuals forbs per plot all declined with time in the fenced plots (Figure 2). The tussock grasses (*Stipa*, *Eragrostis*, and *Cleistogenes*) increased in the fenced plots (Figure 2). After an initial lag of four years, total biomass decreased in the fenced plots while it remained stable in the unfenced (Figure 2).

3.2. Impacts of Fencing on Soil Water Content

Treatment (fenced and unfenced), time, soil depth, and their interactions had significant effects on gravimetric soil water content (Table A4). The mean gravimetric soil water content was higher for the fenced area at the 0–10-cm depth from 1999 to 2011, lower at the 10–30- and 30–60-cm depths from 1999 to 2011, and

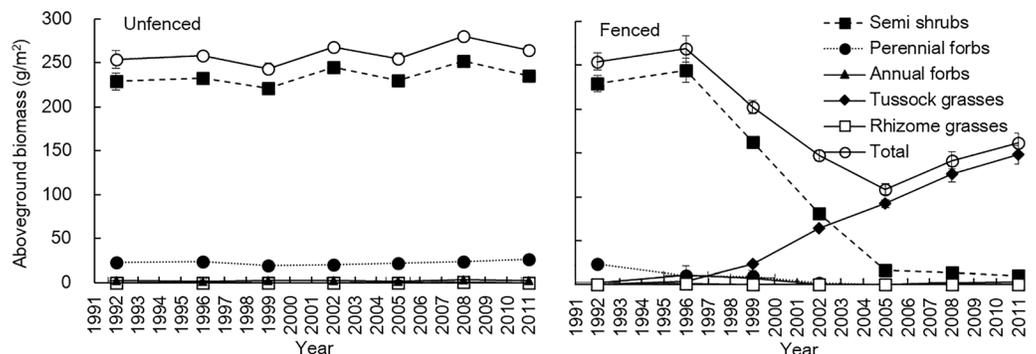


Figure 2. Biomass for life forms occurring in each treatment. Means and standard errors are shown.

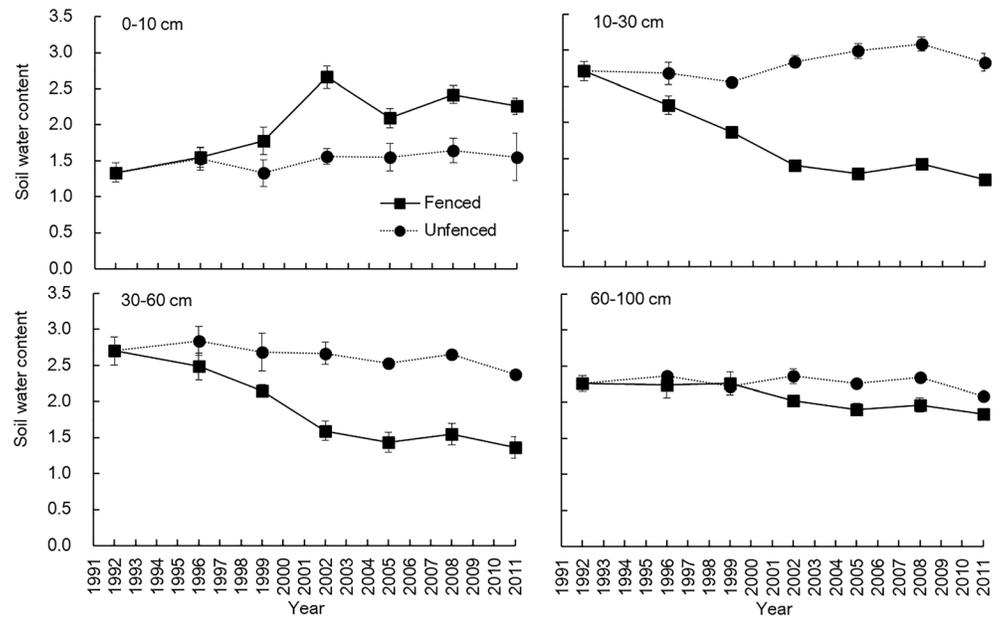


Figure 3. Changes in soil water content (g water per 100 g soil) over the observation period. Means and standard errors are shown.

similar throughout at the 60–100 cm in comparison with the unfenced area (Figure 3). In the fenced area, gravimetric soil water content increased in the surface layers (0–10 cm) from 1.34% (1992) to 2.66% (2002), although there were minor decreases in the layers 0–10 cm in 2005, 2008, and 2011. Gravimetric soil water content decreased in the layers at 10–30 and 30–60 cm, but remained stable at 60–100 cm (Figure 3). In the unfenced area, gravimetric soil water content remained stable.

As 10 mm was moderate intensity and regular individual rainfall in the growing season in the study area, it was taken as an example to show how rain infiltration time into soil can be affected (Qi, 1998). After a 10-mm rainfall, the time that soil water content started to increase and reached its maximum in the fenced plots was later than that in the unfenced plots in different soil layers. The deeper the soil layer is, the longer the time needed for soil water content to start to increase and reach its maximum is (Figure A2).

3.3. Root Biomass Differences

Treatment (fenced and unfenced), time, soil depth, and their interactions had significant effects on root biomass (Table A5). Root biomass increased for the fenced area in the top 10-cm depth in 2002, 2005, 2008, and 2011. In contrast, at deeper depths, root biomass decreased with time in the fenced area compared with the unfenced area. This was most pronounced in the 10–30-cm depth range, with difference increasing with time ($p < 0.05$). For the 30–60-cm depth range, the fenced area had lower root biomass in the later years (from 2002 to 2011, $p < 0.05$); the difference was least evident between 60 and 100 cm in 2002, 2005, 2008, and 2011 ($p < 0.05$); and root biomass in the fenced area was close to zero in 2011 (Figure 4). In the unfenced area, root biomass remained stable over time (Figure 4).

3.4. Impacts of Fencing on Soil Texture

Treatment (fenced and unfenced), time, the soil particle size, and their interactions had significant effects on soil texture (Table A6). The small particle sizes (<0.001, 0.001–0.005, and 0.005–0.01 mm) increased with time. The fraction of small particle sizes was higher in the fenced area in 2002, 2005, 2008, and 2011 in comparison with the unfenced area, while the fraction of coarse particle sizes (>0.01 mm) decreased and was lower in the fenced area in 2005, 2008, and 2011 in comparison with the unfenced area (Figure 5). The fraction of all particle sizes in the unfenced area remained stable (Figure 5).

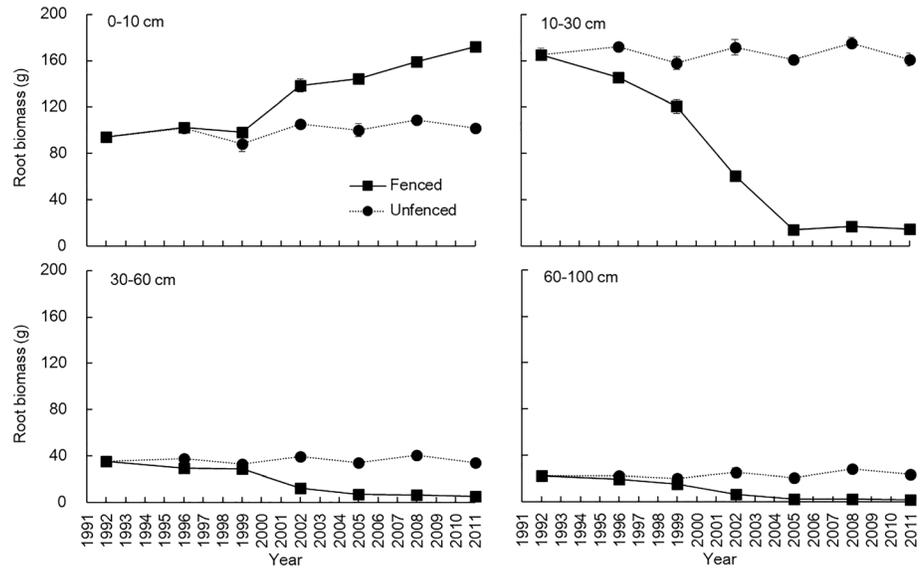


Figure 4. Changes in root biomass at four intervals: 0–10, 10–30, 30–60, and 60–100 cm over the observation period. Means and standard errors are shown.

3.5. The Relationship Between Plant Community and Soil Water Content and Texture

The first two axes of the CCA ordination could explain 95.2% variation of plant community. The fraction of soil particle with size 0.001–0.005 mm ($r = 0.99$), <0.001 mm ($r = 0.98$), and 0.005–0.01 mm ($r = 0.93$) and gravimetric soil water content in 5-cm depth ($r = 0.90$) were significantly and positively correlated with the axis 1; the gravimetric soil water content in 60-cm depth ($r = 0.93$), 40-cm depth ($r = 0.92$), 30-cm depth ($r = 0.90$), 20-cm depth ($r = 0.84$), and the fraction of soil particle with size 0.10–0.25 mm ($r = 0.87$) were significantly and negatively correlated with the axis 1. Plant communities dominated by *S. bungeana*, that is, fenced plots in 2005, 2008, and 2011, were located at the right side of axis 1; plant communities dominated

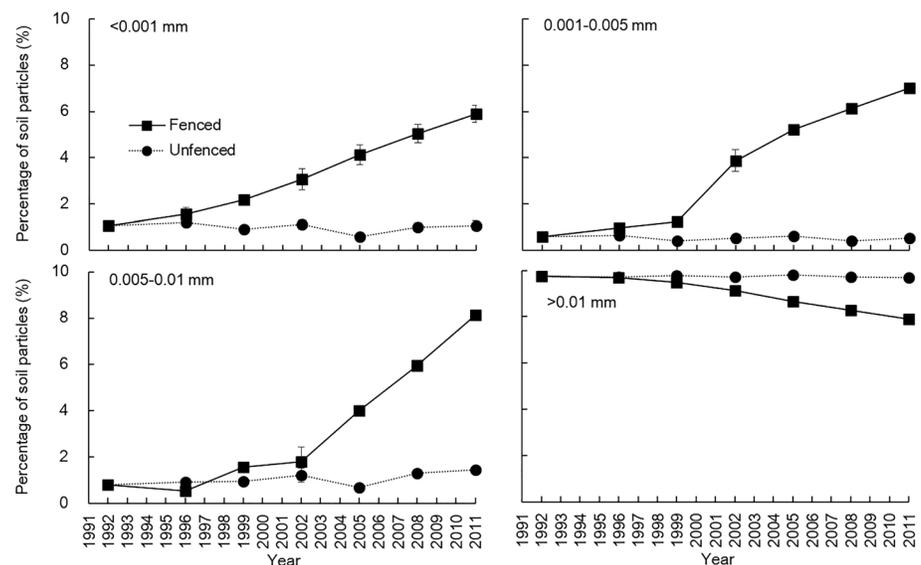


Figure 5. Changes in the proportion of soils in the low particle-size fraction (<0.01 mm) over the observation period. Means and standard errors are shown.

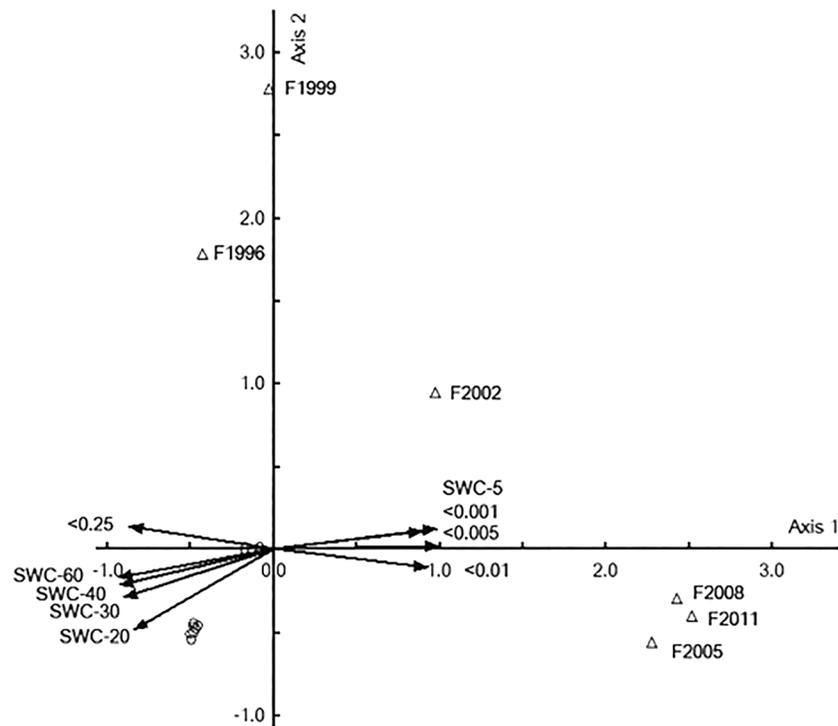


Figure 6. CCA ordination graph of the fenced and unfenced plots with the soil layers overlaid on it on the Ordos Plateau. The vectors represent soil variables. The length of the vector is proportional to its importance and the angle between a vector and each axis is related to its correlation with the axis. Variables with a correlation coefficient higher than 0.50 are presented. SWC-5, SWC-20, SWC-30, SWC-40, and SWC-60 denote gravimetric soil water content in 5-, 20-, 30-, 40-, and 60-cm depth, respectively. <0.001, <0.005, <0.01, and <0.25 denote fraction of soil particles with size <0.001, 0.001–0.005, 0.005–0.01, and 0.1–0.25 mm, respectively. The triangle symbols with labels (e.g., F2011) denote the averaged data (in 2011) observed in the fenced plots, and the circle symbols without labels denote the averaged data observed in the unfenced plots.

by *A. ordosica*, that is, fenced plots in 1992, and unfenced plots in 1992, 1996, 1999, 2002, 2005, 2008, and 2011, were located closely at the left side of axis 1, while the fenced plots in 1996, 1999, and 2002 were located between the unfenced plots and fenced plots. These results indicated that plant communities dominated by *S. bungeana* were related to increase of surface soil water content and fine soil particles after 19-year grazing exclusion, while plant communities dominated by *A. ordosica* were related to high soil water content in deep soil layers and high fraction of coarse soil particles in the initial stage of grazing exclusion and in the plots keeping grazing (Figure 6).

4. Discussion

Although the processes of grassland restoration have been widely studied, few studies have focused on restoration from shrub encroached grasslands (Germino & Reinhardt, 2014; Ridolfi et al., 2008; Walther et al., 2002). In our study, the enclosure was constructed in 1992, when neighboring fenced and unfenced plots had similar vegetation dominated by *A. ordosica*. Four years later, the biomass of *A. ordosica* in the fenced area showed a very minor increase; followed by a decrease in 1999, 2002, and 2005; and was close to zero in 2011. Besides low biomass (9.3 g/m²), *A. ordosica* in the fenced area appeared very sparse in 2011. In contrast, *S. bungeana* was absent in the fenced area from 1992 to 1996; its biomass increased after 1999 and reached 137.6 g/m² in 2011. If such a trend continues, it will not take long before *A. ordosica* is completely replaced by *S. bungeana*. In contrast, the unfenced area maintained similar species and biomass throughout the 19-year study period.

This finding is corroborated by field observations in Shapotou at the south eastern fringe of the Tengger desert in northern China, with mean annual precipitation of 187 mm (Li et al., 2002; Li et al., 2005; Xiao

et al., 2003; Xu & Li, 2003). Here, *A. ordosica* and other shrub species were originally planted in a check-board pattern to protect an important railway corridor from mobile sand dunes and grazing was strictly prohibited. After 32–35 years the planted shrubs have been replaced by annual and perennial grass species, while *Artemisia* species have almost completely disappeared (Li et al., 2002; Li et al., 2005; Xiao et al., 2003; Xu & Li, 2003). This case also indicates that a shrub-encroached system could revert to the local native grassland community if it is not subject to overgrazing for a long period of time, although the restoration may take longer in an arid area like Shapotou than in a semiarid area such as the Ordos Plateau (Li et al., 2005).

Climate has been shown to affect shrub encroachment (Alward et al., 1999; Sturm et al., 2001; Walther et al., 2002). Annual precipitation, a main factor affecting plant communities in semiarid areas, however, showed no significant changes, although the mean annual temperature significantly increased from 1971 to 1998 at our study sites (Li & Zheng, 2002). Our finding that the plant community and soil properties remained similar in the unfenced area from 1992 to 2011 indicates that climate was not the main reason for replacement of the *A. ordosica* by the *S. bungeana* plant community in the fenced area.

There are some reports on the effects of shrub encroachment on soil texture and soil water content in different soil layers. Sand in surface soil layer usually increased and clay decreased, and rainfall easily infiltrates into deep soil layers (D'Odorico et al., 2012; Germino & Reinhardt, 2014; Schade & Hobbie, 2005). The reversion from semishrub-encroached grasslands to native-grass-dominated grasslands changed soil texture differently comparing with the effects of shrub encroachment. Following the enclosure, the fine-particle components (<0.01 mm) of the soil gradually increased in the layer of 0–10 cm, while the coarse-particle components (>0.01 mm) decreased through time during the study period (Figure 5). This change implied that the soil water-holding capacity in the surface layer increased with enclosure time, and water infiltration from the surface soil to deep layers decreased. Indeed, the soil water content decreased in 10–30 and 30–60 cm after 1999, while the soil water content in 0–10 cm increased in 2002, although there was a little decrease in 2005. There was no difference for soil water content in 60–100 cm during the 19-year study period (Figure 3).

The change in soil water content was accompanied by a change in plant root distribution patterns. The root biomass in the layers at 10–30, 30–60, and 60–100 cm decreased with fencing time. Most notably, the root biomass at 10–30 cm decreased from 145.4 g/m² in 1996 to less than 14.4 g/m² in 2011. In contrast, root biomass at 0–10 cm first increased gradually and then increased in 2002 (Figure 4).

Based on these results, the plant community succession from deeper-rooted *A. ordosica* to shallow-rooted *S. bungeana* after the long-term grazing enclosure is clear and is consistent with our hypothesis. The process of replacement of *A. ordosica* by *S. bungeana* after grazing elimination could be explained as follows: after fencing, fine soil material accumulated beneath semishrubs during windy seasons because plants blocked material carried by wind. Such dust accumulation can reach 2–3 mm per year (Wang & Li, 1994). The fine soil material could stay there when grazing was eliminated and the surface soil was not damaged by animal trampling. As the surface soil texture changed, more water could be held in the surface layer and remain available to shallow-rooted grasses. Further, the absence of disturbance of the soil surface facilitated the formation of biological soil crust, a common and widespread feature of arid and semiarid landscapes throughout the world (Belnap, 2002). The thickness of the soil crust reached 0.3–0.4 mm in the *A. ordosica* community in our study area and in some typical fixed dunes on the Ordos Plateau (Guo, 2000), and it reached up to 0.8 mm in Shapotou after 40 years of enclosure (Li et al., 2005). The soil crust could lead to great changes in soil water movement and infiltration in different soil layers, especially in this area where light rainfall events are common (Guo, 2000; Yang et al., 1994). Rainfall was first intercepted by the surface layer, then runoff was induced by soil crusts, and consequently infiltration to deep soil layers was less in the fenced area than in the unfenced area, even after heavy rain. Because *A. ordosica* roots are distributed mainly in the 10–30-cm soil layer and may reach over 80 cm, while the roots of *S. bungeana* are found mainly at 0–10 cm (Yang et al., 1994), the decline of soil moisture in the root zone of *A. ordosica* and the increase of moisture in the surface layer were the main reasons for replacement of *A. ordosica*. This result could be

further proved by CCA results (Figure 6). In comparison, where overgrazing continued (the unfenced area), soil crusts could not form or were destroyed; therefore, soil water could reach deep layers and *A. ordosica* could sustain its dominance (Guo, 2000). Therefore, it was clear that the *A. ordosica* community was in an unstable successional stage; the surface layer soil texture could ensure that soil water infiltrate to relatively deep layers to maintain the dominance of *A. ordosica*. Grazing effects induced sandy soil and local semiarid climate are the major factors contributing to the long-term dominance of the *A. ordosica* community in this steppe region. Although fire does not occur frequently on the Ordos Plateau (Qi, 1998), together with the changes in soil nutrients, fire might affect the process of shrub encroachment. These factors deserve further studies.

Uptake of deep-soil water by woody plants in semiarid areas has also been confirmed in some studies such as *Larrea tridentate* and *Artemisia tridentata* in western North American deserts (Sala et al., 1989). Compared with shrub encroachment studies, our study on reversion to native grassland from semishrub-encroached grassland may be explained with a two-layer hypothesis. Encroachment may occur more quickly on sandy soils as deeper water infiltration favors deeper-rooted woody plant over shallower-rooted grasses and selective use of deep-soil water increases the growth or persistence of woody species (Leffler et al., 2004). However, few studies have directly tested these by manipulating deep-soil water availability and observing the long-term outcomes (Germino & Reinhardt, 2014). More such field experiments are needed for understanding semishrub encroachment and for management of these semiarid grassland ecosystems.

5. Conclusions

Our 19-year enclosure experiment in a semishrub-encroached grassland on the Ordos Plateau, Inner Mongolia, indicated that native species in the semishrub-encroached grassland could be re-established naturally. Semishrub-encroached grasslands dominated by *A. ordosica* were in an unstable successional stage, and grazing and the successional effects driven by climate on surface soil texture could maintain high soil water content in deep soil layers and maintain the dominance of semishrub species. In contrast, the semishrub-encroached grassland will be replaced by typical steppe vegetation dominated by the native perennial grass, *S. bungeana*, if overgrazing is constrained or eliminated through long-term grazing enclosure. However, in developing regions such as Inner Mongolia, livestock husbandry is necessary for local people to meet their needs. Policy guidelines for sustainable livestock husbandry are needed by decreasing grazing pressure while grassland restoration is part of environmental conservation effort on the Ordos Plateau.

Appendix A

Table A1
Species Occurrence at the Plots, Showing Life Form and Reproductive Mode

Species	Category	Reproductive mode	Unfenced	Fenced
<i>Allium mongolicum</i>	perennial forbs (D)	vegetative		1996, 1999
<i>Aneurolepidium dasystachys</i>	rhizome grass (C)	vegetative and seeds	2008	1996
<i>Artemisia frigida</i>	semishrubs (A)	seeds		1996
<i>Artemisia ordosica</i>	semishrubs (A)	seeds	ALL	ALL
<i>Cleistogenes squarrosa</i>	tussock grasses (B)	vegetative and seeds		1999–2011
<i>Corispermum candeladrum</i>	annual forbs (E)	seeds	ALL	1992, 1996
<i>Cynanchum komarovii</i>	perennial forbs (D)	seeds and vegetative	ALL	1992, 1996
<i>Eragrostis cilianensis</i>	tussock grasses (B)	seeds		1996, 1999, 2008, 2011
<i>Euphorbia humifusa</i>	annual forbs (E)	seeds		1999, 2008, 2011
<i>Ixeris chinensis</i> var. <i>graminifolia</i>	annual forbs (D)	vegetative and seeds		1996, 1999
<i>Lespedeza davurica</i>	semishrubs (A)	seeds and vegetative	ALL	ALL
<i>Oxytropis psammocharis</i>	perennial forbs (D)	seeds		1996, 1999
<i>Polygala tenuifolia</i>	perennial forbs (D)	seeds		1999
<i>Stipa bungeana</i>	tussock grasses (B)	vegetative and seeds		1999–2011

Table A2
Mean (Standard Error) of Shannon's and Simpson's Diversity Indices for Each Year of Observation

Year	Unfenced		Fenced	
	Shannon's	Simpson's	Shannon's	Simpson's
1992	0.560 (0.021)	0.808 (0.007)	0.560 (0.021)	0.808 (0.007)
1996	0.565 (0.021)	0.805 (0.007)	0.953 (0.021)	0.746 (0.007)
1999	0.527 (0.021)	0.824 (0.007)	1.592 (0.021)	0.541 (0.007)
2002	0.515 (0.021)	0.831 (0.007)	1.590 (0.022)	0.389 (0.007)
2005	0.552 (0.021)	0.812 (0.007)	0.896 (0.021)	0.673 (0.007)
2008	0.594 (0.029)	0.805 (0.017)	1.015 (0.027)	0.684 (0.013)
2011	0.605 (0.019)	0.789 (0.006)	0.900 (0.006)	0.732 (0.004)

Table A3
Results of the Repeated Measures ANOVA

Multivariate tests				
Effect	<i>F</i>	Hypothesis d.f.	Error d.f.	<i>p</i>
Time (<i>T</i>)	151.87	6	15	<0.001
Time × treatment (Tr)	296.24	6	15	<0.001
Time × species group (S)	11.28	24	72	<0.001
<i>T</i> × Tr × S	7.64	24	72	<0.001
Test of between-subject effects				
Source	d.f.	Mean square	<i>F</i>	<i>p</i>
Treatment (Tr)	1	12,473.55	94.77	<0.001
Species group (S)	4	218,673.11	1,661.41	<0.001
Tr × S	4	51,085.38	388.13	<0.001
Error	20	131.62		

Note. Mauchly's test of sphericity indicated that the assumption of sphericity had been violated. Pillai's trace in multivariate test was used. Species group aboveground biomass was analyzed as dependent variables, treatments (fenced and unfenced), species group as between-subject factors, and time as within-subject factor.

Table A4
Results of the Repeated Measures ANOVA

Multivariate tests				
Effect	<i>F</i>	Hypothesis d.f.	Error d.f.	<i>p</i>
Time (<i>T</i>)	40.40	6	11	<0.001
Time × treatment (Tr)	11.60	6	11	<0.001
Time × soil depth class (S)	4.33	18	39	<0.001
<i>T</i> × Tr × S	2.52	18	39	<0.01
Test of between-subject effects				
Source	d.f.	Mean square	<i>F</i>	<i>p</i>
Treatment (Tr)	1	5.94	22.69	<0.001
Soil depth class (S)	3	2.51	9.58	<0.001
Tr × S	3	5.06	19.31	<0.001
Error	16	0.26		

Note. Mauchly's test of sphericity indicated that the assumption of sphericity had been violated. Pillai's trace in multivariate test was used. Gravimetric soil water content was analyzed as dependent variables, treatments (fenced and unfenced), soil depth class as between-subject factors, and time as within-subject factor.

Table A5
Results of the Repeated Measures ANOVA

Multivariate tests				
Effect	<i>F</i>	Hypothesis d.f.	Error d.f.	<i>p</i>
Time (<i>T</i>)	70.18	6	11	<0.001
Time × treatment (Tr)	63.87	6	11	<0.001
Time × soil depth class (S)	3.73	18	39	<0.001
<i>T</i> × Tr × S	3.87	18	39	<0.001
Test of between-subject effects				
Source	d.f.	Mean square	<i>F</i>	<i>p</i>
Treatment (Tr)	1	21,947.43	179.78	<0.001
Soil depth class (S)	3	131,340.86	1,075.89	<0.001
Tr × S	3	25,666.15	210.25	<0.001
Error	16	122.08		

Note. Mauchly's test of sphericity indicated that the assumption of sphericity had been violated. Pillai's trace in multivariate test was used. Root biomass was analyzed as dependent variables, treatments (fenced and unfenced), soil depth class as between-subject factors, and time as within-subject factor.

Table A6
Results of the Repeated Measures ANOVA

Multivariate tests				
Effect	<i>F</i>	Hypothesis d.f.	Error d.f.	<i>p</i>
Time (<i>T</i>)	267.59	6	11	<0.001
Time × treatment (Tr)	160.22	6	11	<0.001
Time × soil particle sizes (S)	11.67	18	39	<0.001
<i>T</i> × Tr × S	4.37	18	39	<0.001
Test of between-subject effects				
Source	d.f.	Mean square	<i>F</i>	<i>p</i>
Treatment (Tr)	1	124.1	39.6	<0.001
Soil particle sizes (S)	3	208.0	66.3	<0.001
Tr × S	3	27.8	8.9	<0.001
Error	16	3.1		

Note. Mauchly's test of sphericity indicated that the assumption of sphericity had been violated. Pillai's trace in multivariate test was used. Fraction of fine soil particle size (<0.001, 0.001–0.005, 0.005–0.01, and 0.01–0.05 mm in diameter) was analyzed as dependent variables, treatments (fenced and unfenced), soil particle sizes as between-subject factors, and time as within-subject factor.

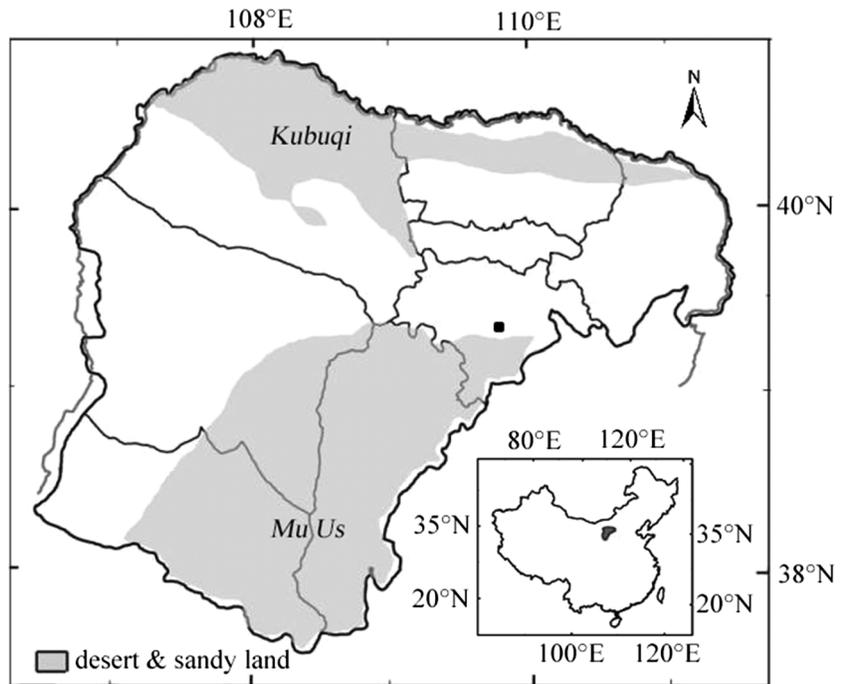


Figure A1. Location of the Ordos Plateau in the P. R. China and location of field plots within the Ejin Horo Banner of the Ordos Plateau, Inner Mongolia (black square).

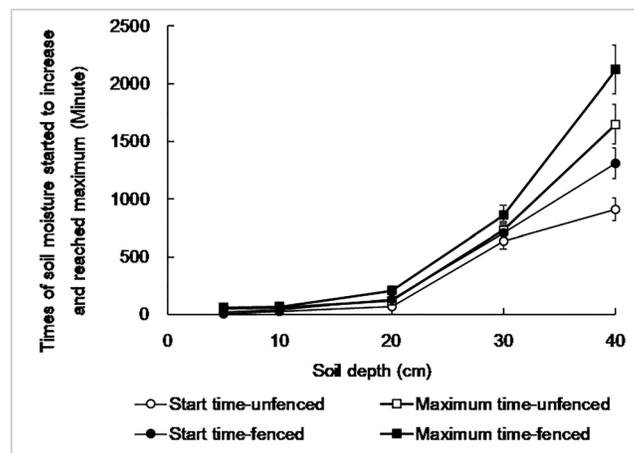


Figure A2. Time of soil water content started to increase and reached maximum in the fenced and unfenced plots followed a 10-mm rainfall started from 7 July, 18:20 in different soil layers in 2010.

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