Effects of Large-Scale Agriculture and Livestock Industry on the Vegetation and Soil Properties in the Steppe Region of Kazakhstan and Inner Mongolia, China

January 2014

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A Dissertation Submitted to
the Graduate School of Life and Environmental Sciences,
the University of Tsukuba
in Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy in Agricultural Science
(Doctor Program in Biosphere Resource Science and Technology)

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List of abbreviations

AC Abandoned Cropland

CCA Canonical Correspondence Analysis

CG Continuously Grazed

Cs Cesium

DCA Detrended Correspondence Analysis

DGGE Denaturing gradient gel electrophoresis

DSG Deferred Spring Grazing

EC Electrolytic Conductivity

INSPAN Indicator Species Analysis

K Köppen's rainfall index

OC Organic Carbon

PCR Polymerase Chain Reaction

Qs Sorensen's Similarity Index

RNA Ribonucleic Acid

SDR Summed Dominance Ratio

T-C Total Carbon

T-N Total Nitrogen

TWINSPAN Two-Way Indicator Species Analysis

WI Kira's Warmth Index

Chapter 1

General introduction

1-1 Scientific background

The major Eurasian grassland is called the “steppe” (Geoffrey, 1995). That is distributed over vast areas in the arid and semiarid regions of the Eurasian continent (Archibold, 1995). As shown in **Fig.1-1**, the steppe region extends over 8000 km from north-eastern China, via Mongolia, Russia and Ukraine to Hungary. According to Grousset (1988), it is divided into two major segments, the Eastern and Western Eurasian steppe. Altai mountains plays an important role for it. The Eurasian steppe is an important terrestrial ecosystem. There, the climate is unable to support forest vegetation, but there is enough moisture for a closed perennial herb vegetation (Walter, 1973), and these treeless plains are dominated by species belonging to the genus *Stipa* (Walter, 1973; Archibold, 1995; Hayashi, 2003; Gibson, 2009).

Desertification, defined in the UNCCD (1994) as “Land degradation in arid, semiarid and dry sub-humid areas resulting from various factors including climatic variability and human activities”. Desertificated areas account for 66%, 72% and 59% of the land area of the Kazakhstan, Mongolia and Inner Mongolia, respectively (MDGs, 2010; Dashzeveg B., 2011; IMD, 2008). Vegetation degeneration and land devastation, with desertification of steppe, are becoming serious, mainly due to droughts, over-grazing and cultivation (Bao *et al*., 2006; Batjargal, 1992; Liu and Hao, 2002; Wang and Zhang, 2012). Recently, many ecologists have conducted research on the process, mechanism, mitigation, monitoring and assessment of desertification, expecting the restoration of the steppe ecosystem.

Analysis of the floristic composition of plant communities provides an effective way to assess grassland conditions (Cheng and Nakamura, 2007). A significance of this study was to establish a systematical classification of vegetation that may serve to identify and predict future vegetation changes and to assess effects of conservation and management practices. It is very important to determine the vegetation change trends for combating the desertification in this area (Li *et al*., 2011). In order to control desertification and to manage ecosystems of the Eurasian steppes, it is important to study the relationship between environmental factors and plant communities in the region. The key correlation between the distribution of herbaceous species and environmental factors, such as water availability, has been well documented for the region (Ellis and Chuluun, 1993; Adler and Levine, 2007; Bai *et al.*, 2008; Yang *et al.*, 2011). Most researchers have focused on the relationship between the distribution of species and soil type (Huang *et al.*, 2007), soil properties (Casas and Ninot, 2003), temperature (Niu and Wan, 2008), scales (Ni *et al.*, 2007; Chandy and Gibson, 2009) and other environmental gradients (Fernandes-Gimenez and Allen-Diaz, 2001; Zemmrich *et al.*, 2010) at the species or community level. Although the distribution of species is commonly governed by several environmental gradients, most of these studies of the species-environment relationship have focused on a single factor. In addition, these fragile ecosystems are becoming degraded and suffering desertification because of unsustainable human activities, such as overgrazing and cultivation. Although changes in the characteristics of steppe vegetation communities in Mongolia and northern China, dominated by *Stipa grandis*, *Stipa krylovii* and *Stipa baicalensis*, have been well documented (Hayashi *et al.*, 1988; Nakamura *et al.*, 1998; Nakamura *et al.*, 2000; Wuyunna *et al.*, 1999; Kawada *et al.*, 2011), few studies have been carried out in Western Eurasian steppe (Cheng and Nakamura, 2006; 2007). Furthermore, little information is available to explain how human activities are influencing species loss and soil disturbance in this area.

Grassland regression in dry and semiarid lands is caused by the reclamation of grassland. In the 1950s Kazakhstan was designated as the Soviet Union's food base and its grassland was widely reclaimed in order to increase agricultural production (Wolman and Fournier, 1987). Under such socialist system, the Great Reclamation and the nature modification plan changed the vast, fertile lands in Eurasia, including Kazakhstan, to deserts (Kassas, 1977). In Inner Mongolia, vast areas were converted to cropland, and farmers have increased the size of these croplands during the past 40 years because of a rapid increase in the human population. Subsequently, many of the croplands were abandoned because of soil degradation and desertification caused by inappropriate agricultural management (He *et al*., 2005; Tong *et al*., 2004). Comparison to Kazakhstan or Inner Mongolia, where cultivation has been responsible for large-scale degradation Mongolian steppes are usually considered relatively intact (Babaev, 1999; Sneath, 1998). In the semiarid areas, the soil surface is easily influenced by external conditions, and once lost, it is extremely hard to recover the natural environment. The desertification caused by such a factor is generally thought to worsen the soil erosion through vegetation changes. Soil erosion is one of most widespread forms of soil degradation by which the topsoil is removed naturally by the wearing actions of rain and wind. In steppe ecosystem, especially, the cultivation and subsequent abandonment are accelerate the soil erosion (Schmitt *et al*., 2003). Because, as cultivation is increasing, the level of soil fertility of farmland is decreasing significantly. Consequently, it is abandoned, moving to new field. In this repeat, large scale plant mortality and losses of seeds from the soil (Lauenroth and Burke, 2008). To avoid this outcome, soil loss must be carefully monitored, and early detection of soil degradation in abandoned croplands is critical (Kawada *et al*. 2011). On the other hand, it is necessary to grasp the eroded soil and the current vegetation on it in order to maintain grassland vegetation and predict recovery.

In terrestrial ecosystems, aboveground and belowground components of ecosystems are strongly linked through a variety of direct and indirect interactions (Kardol and Wardle, 2010). Plants provide carbon sources and other nutrients for the soil fauna and microflora (Habekost *et al*., 2008), and, in turn, the soil biota, particularly its microbiota, decomposes soil organic matter, stabilizes soil structure and, through its essential role in the cycling of elements, releases nutrients for plant growth (Porazinska *et al*., 2003). In addition, soil abiotic factors, such as organic matter content (Kara *et al*., 2008), salinity (Lozupone and Knight, 2007) and pH (Fierer and Jackson, 2006) have been interpreted as the drivers of the vegetation patterns observed in terrestrial ecosystems. Therefore, in order to understand the restoration and sustainability of ecosystem, interactions between aboveground and belowground combined approach is required. Some studies have addressed that differences in aboveground plant species diversity, plant richness and plant biomass had significant effect on soil microbial processes, microbial biomass or colony-forming units of the major microorganisms (Hooper and Vitousek, 1998; Malý *et al*., 2000; Wardle and Nicholson, 1996). Most of above studies carried out in forest (Christine *et al*., 2001), grassland (Steenwerth *et al*., 2003) and cropland (Chen *et al*.,

2012) ecosystems, and relatively little is known in abandoned cropland ecosystem (Zhang *et al*., 2012), especially, in Eurasian steppe distributed in semi-arid regions.

Another cause of desertification is overgrazing. In countries in the steppe area such as Russia, Kazakhstan, Mongolia, and China, the overgrazing takes place in order to seek high production abilities. Overgrazing makes grassland vegetation degenerate, causing desertification (Akiyama and Kawamura, 2007; Han *et al*., 2008). In order to solve the problem, since 1990 Inner Mongolia in China has been implementing policies such as "land contract system", "ecological migrants", "enclosing transferring", "banned grazing system" and "deferred grazing system". These protection policies have helped pastoral management to transform into barn breeding and half barn breeding (Alatansha and Chitose, 2012). Grazing systems are restricted and modified while grassland resources are protected by limiting the grazing number and the grazing period (Jiyatu and Ono, 2009). Among them, "deferred grazing system" is one of a widely used system. The deferred grazing has long been recognized as a key grazing management tool in America (Brand and Goetz, 1986), and it was practiced and studied in Australia (Cocks, 1965) and New Zealand (McCallum *et al*., 1991), where precipitation is 300 mm or above. In Inner Mongolia, it has been used on a large area in the region of less than 200 mm of precipitation, however, observational studies have not been conducted to evaluate the effectiveness of it. Many ecologists have assessed the effectiveness of the grazing system through vegetation and soil responses (Martin and Severson, 1988; Wood and Blackburn, 1984). In terms of the most appropriate means to evaluate vegetation dynamics effectively in grasslands, many ecologists have accepted the dualistic approach of considering both equilibrium and nonequilibrium models (Briske *et al*., 2003; Illius and O'Connor, 1999; Walker and Wilson, 2002), and it can be indicated that grassland degradation is due to livestock overgrazing and climatic variability or one of which them by using the two models (Cheng *et al*., 2011).

1-2 General objectives

My studies were carried out in Kazakhstan and Inner Mongolian steppe region, including continuously grazing area, crop abandoned area and protected area. The general objectives of this study is to analyze and evaluate the effects of cultivation and grazing on the vegetation and soil properties in the steppe region, to evaluate the effects of different grazing system on the vegetation and the recovery process on the vegetation and soil properties in the steppe region. Additionally, I also have discussed what kinds of grassland protection should be adopted for maintaining the higher natural level of grassland.

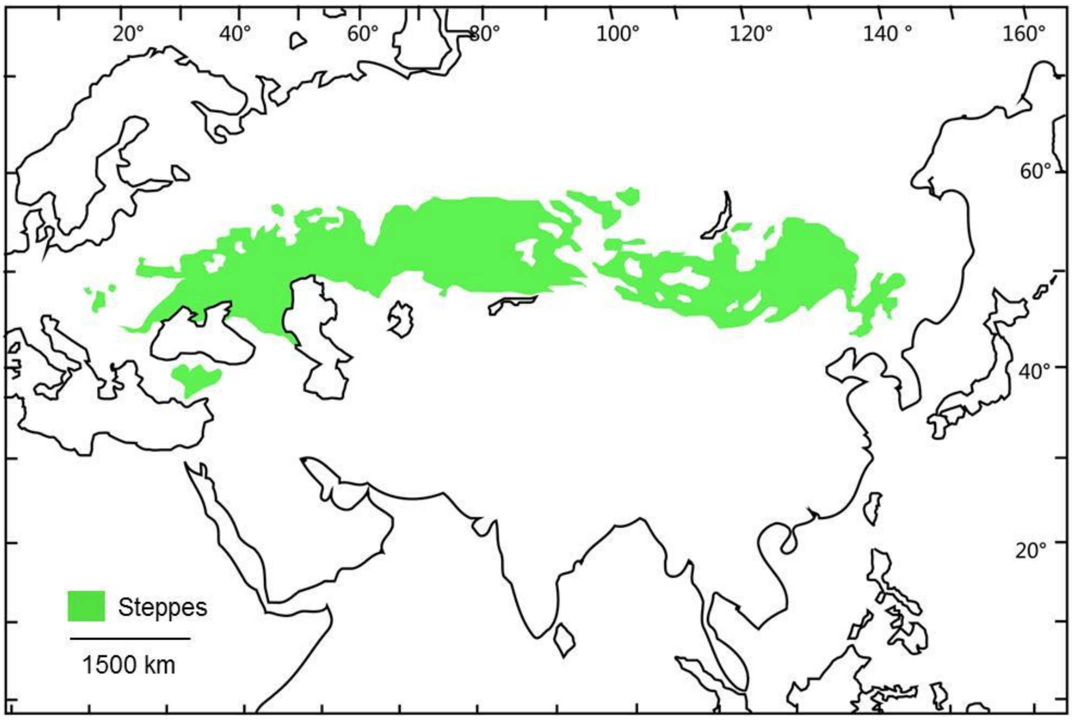

Fig. 1-1 Eurasian steppe zone (after Walter, 1973)

Chapter 2

Overview of the Study Area

2-1 Kazakhstan

2-1-1 Geographical position

Kazakhstan is located on the junction of two continents, Europe and Asia, within 45-87 °E and 40-55 °N, having a territory of 2,725,000 km^2 area, stretching 2,925 km east to west and 1,700 km north to south (Busscher *et al*., 2007). Kazakhstan borders Russia on the north, China on the east, Kyrgyzstan and Uzbekistan on the south, and Turkmenistan and Caspian Sea on the west (**Fig. 2-1**). Because the country is so large, it has a wide variety of climate, terrain, vegetation and soil type. For terrain, except Tian Shan and Altai Mountains, most of the country is flat lowlands (Brown, 2006).

2-1-2 Climate

The climate of Kazakhstan is extremely continental. There are cold and long winters and dry, short summers in the northern part, and vice-versa, short and low-snow winters and long, dry and hot summers in the south. Summer droughts accompanied by dusty storms and dry winds are very common (Gvozdetski and Nikolaev, 1971). Annual precipitation for northernmost Kazakhstan is 315 mm. In central Kazakhstan, annual precipitation is about 150 mm. In the foothills of the mountains, precipitation increases to 880 mm on the forested mountainsides (Brown, 2006). About the study sites, the mean annual air temperature is 2.0 °C in Akmola and 4.4 °C in Aktobe, while the mean annual precipitation amounts to 272.1-351.9 mm in Akmola and 239.8-266.8 mm in Aktobe.

2-1-3 Vegetation

The large size of the territory of Kazakhstan stipulates a high variety in natural landscapes. Five major natural zones are distinguished: high-mountains, forest-steppe, temperate steppe, semi-desert and desert landscape zone (Arkhipov, 1980). The plants of Poaceae have dominated in steppe zones. The dominant species were belonged to genus *Stipa* such as *S. capillata, S. pennata, S. lessingiana* and *S. zalesskii*.

2-1-4 Soil

In Kazakhstan, 86% of the whole territory is plains. There are mainly three types of soils on the plains: Chernozem, Kastanozems and Xerosols (FAO-Unesco, 1978). In the northern forest steppe and steppe, soils are deep, dark Chernozems (WWOOF, 2009). The Kastanozems of the steppes are fit for agriculture and animal farming. The soil fertility is decreasing to the south of the country. The Xerosols distributed at the southern deserts area.

2-1-5 Social environment

Kazakhstan consists mostly of deserts, semi-desert, and steppes, not suitable for cultivation and usable only for stockbreeding using grasslands. Therefore, until the 20th century, the Kazakh people were largely engaged in nomadic pastoralism with little settled agriculture (Robinson *et al*., 2003). As for forms of agriculture, Kazakhstan is divided into three areas: the grain production zone in the dry steppes in the north, the livestock zone from the central area to the south, and the irrigated farming area around the Chu and the Syr Darya River basins in the south (Frenken, 2013). It was developed in full swing for crop cultivation during the Soviet era in the 1950s, dubbed as the Virgin Lands Campaign, when vast lands of 19.9 million hectares were newly developed (Yamamura, 2007). Kazakhstan used to be one of the powerful agricultural production countries within the Soviet Union, along with Russia and Ukraine, among others. However, with the collapse of the Soviet Union in 1991, its agriculture sector faced a serious crisis, and its agricultural production decreased 46% during the 1990s, out of 220 million hectares of the total farmland area, only 16.2 million hectares have been used for crop cultivation as of 1999 (**Fig. 2-2**). Leaving many agricultural enterprises virtually bankrupt and the number of livestock heavily reduced (**Fig. 2-3**). By the late 2000s, the situation had improved, and from 2000 to 2005, the agricultural production increased 30%. In 2008, agriculture, forestry and fisheries had only a 5.3% share of the entire industry. As for rural residents, the Ministry of Agriculture stated that 42.6% of the whole population lived in agricultural communities. As for Kazakhstan ́s employed population, 29.5% worked in the agriculture, forestry and fisheries industry in 2008. Since 2003, its share had dropped 5.8% (only in five years), according to the National Bureau of Statistics (www.stat.kz/pages/default.aspx). The agricultural population is expected to decrease further in the future.

Fig. 2-1 Steppe zone in Kazakhstan (after IUCN, 2009)

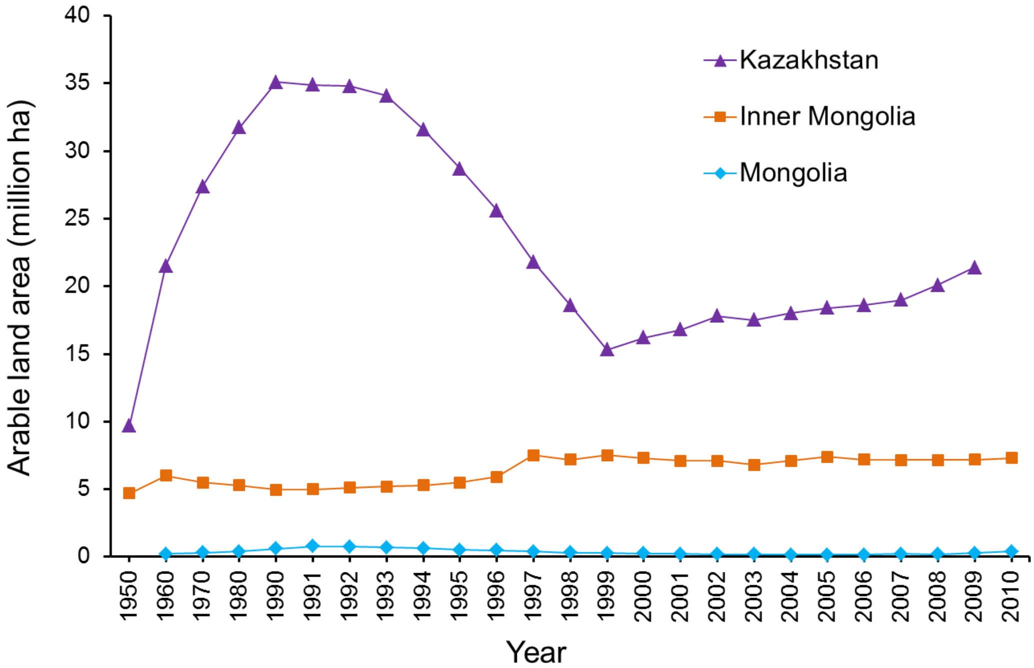

Fig. 2-2 Yearly change of arable land area in Kazakhstan, Mongolia and Inner Mongolia. (Agency of Statistics of Republic of Kazakhstan, 2010; Statistical Bureau of Inner Mongolia, 2011; MNSO, 2011)

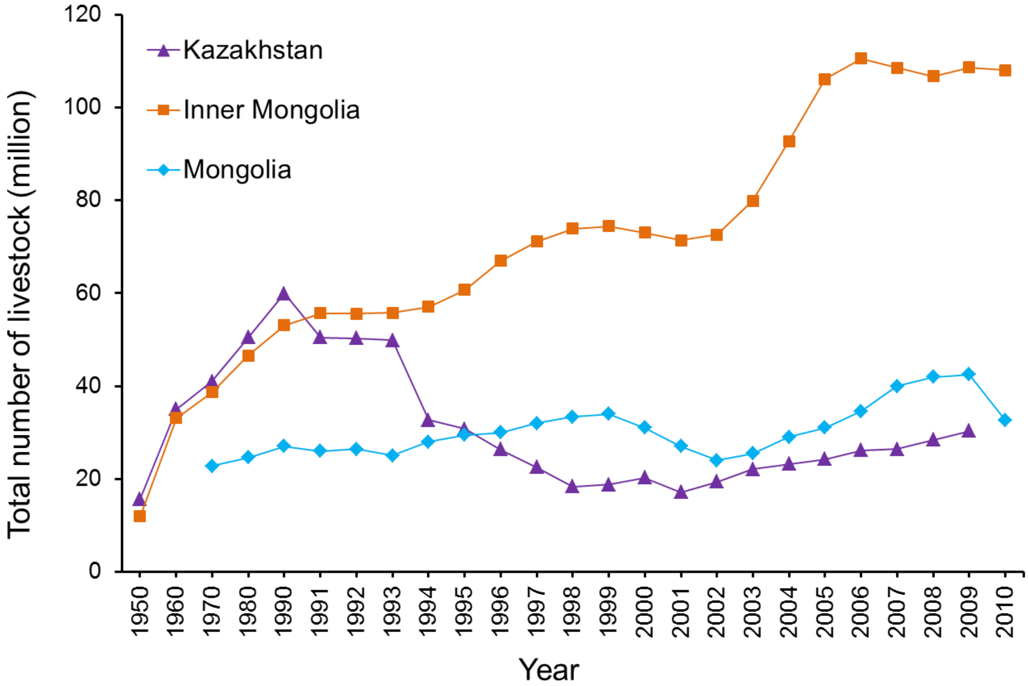

Fig. 2-3 Livestock dynamics in Kazakhstan, Mongolia and Inner Mongolia. (Agency of Statistics of Republic of Kazakhstan, 2010; Statistical Bureau of Inner Mongolia, 2011; MNSO, 2011)

2-2 Inner Mongolia

2-2-1 Geographical position

Inner Mongolia, located in eastern steppe regions of Eurasia. It is classified as one of the provincial-level divisions of China. The geographical coordinates are 97°12′-126°04′ E, 37°24′-53°23′ N. The landscape is characterized by Mongolian Plateau, features a long, narrow strip of land sloping from northeast to southwest. The region covers an area of 1.18 million km^2, or 12.3% of the country's territory. It stretches 2,400 km from west to east and 1,700 km from north to south. It neighbors eight provinces and regions in its south, east and west and Mongolia and Russia in the north. Investigation sites were set in HulunBuir, located in HulunBuir city (115°31′-126°04′ E, 47°05′-53°20′ N) that is northeast part, Xilingol, located in Xilingol league (111°09′-120°00′ E, 41°34′-46°52′ N) that is middle part, and Ordos, located in Ordos city (106°42′-111°27′ E, 37°35′-40°51′ N) that is south part in Inner Mongolia (**Fig. 2-4**).

2-2-2 Climate

Inner Mongolia has a temperate monsoon climate, which is featured by few and irregular rainfall, and drastic shift of temperature. Winter is freezing cold, lasting for a very long time, sometimes even more than half a year. The annual precipitation is about 50 mm in its western part and 450 mm in eastern part. January is the coldest month with temperature from -10 ℃ to -32 ℃. With a monthly average temperature of 16 ℃ to 27 ℃, summer is warm and short, lasting for only one or two month, and there is no summer in some area of Inner Mongolia. The hottest month always comes in July with a highest temperature of 36 ℃ to 43 ℃.

Hulunbuir, the annual average temperature is -5 ℃ to 2 ℃ and the annual precipitation is about 400 mm, Daxinganling Range of eastern exceeding 500 mm, and plains of western 250－300 mm.

Xilingol, the mean annual temperature is 0-3 ℃. In January, the coldest month, the average temperature is -20 ℃ while the mean temperature of the warmest month, July, is 21 ℃. The average annual rainfall is 295 mm, decreasing from east to west. Rainfall is high between June and September during the summer monsoon season. The Annual evaporation is 1500－2700 mm, increasing from east to west.

Ordos, The annual precipitation is 300 to 400 mm in the eastern part of the prefecture, and 190 to 350 mm in the western part. Most of the rain falls between July and September. The annual mean temperature is 6.7 ℃, with monthly mean temperatures below 5 ℃ from November to March and between 7.4 ℃ and 21.9 ℃ from April to October (Zheng *et al.*, 2005).

2-2-3 Vegetation

Inner Mongolian grassland divide into 5 vegetation zones: the cold-temperate bright coniferous forest zone, warm-temperate deciduous forest zone, moderate-temperate steppe zone, warm-temperate steppe zone and warm-temperate desert zone (Wang *et al*. 1979). Its primary natural resources are grasslands which cover more than 70% of the area (870,000 km^2) (Kobayashi *et al.*, 1994). About the main land use type, the total area of Inner Mongolia has 72.2 million ha of cultivated land, 6.11% of the region's total, and 186.7 million ha of forests, 15.8% of the region's total. The 788.8 million ha area of grassland is the largest land form in Inner Mongolia (Zhao *et al*. 2006).

HulunBuir, the grassland area is about 2.6×10^5 km^2, with a west to east distribution of arid steppe, semi-arid steppe, and meadow steppe.

Xilingol, the grassland spreads from east to west, meadow steppe, typical steppe, and desert steppe, with their areas available, 12.3%, 52.2%, and 15.6%, respectively, of the entire area. The Xilingol steppe area is about 202,580 km^2, approximately 88% of which can be utilized for grazing and farming.

Ordos region covers the bigger part of the desert. Our study area located in south of Ordos, where sandy and meadow steppe are mixed.

2-2-4 Soil

There are following soil types distributed in the investigation sites (FAO - Unesco, 1978). The main soil type in HulunBuir grassland is Kastanozems. In Xilingol grassland, Kastanozems is mainly distributed in typical steppe, and Xerosols is mainly distributed in desert steppe. The main soil type in Ordos grassland are Xerosols and Yermosols.

2-2-5 Social environment

Since ancient times, the Mongolians have led a nomadic life, protecting their living environments. However, Inner Mongolia, which has been damaged due to the long development, is hardly beautiful. Currently, Inner Mongolia is plagued with the degeneration and desertification of grassland (Liu and Hao, 2002; Bao *et al*., 2006). As cultivation developed in areas only suited to stockbreeding, traditional livelihood changed (Zhou *et al*., 1995; Zhang and Itohara, 2003; Nomura *et al*., 2008). These changes fall into two categories: from stockbreeding to farming and from nomadism to fencing. They cannot be clearly separated, but they have been occurring simultaneously. According to statistical data, the number of livestock in 1947 was 9.32 million. However, it had increased to 106.15 million by 2005 (**Fig. 2-3**). According to Inner Mongolia Environmental Protection Agency, the amount of hay needed to feed grazing livestock in Inner Mongolia for 1 year is 35.264 billion kg, but it needs 6.605 billion kg more, 18% short. In 28 of the 33 livestock farming enterprises in Inner Mongolia, overgrazing takes place, making hay 20% short (Wang, 2011). These data tell us that the overgrazing issue certainly exists.

On the other hand, the grassland was damaged by cultivation. “immigration policy” aimed to develop remote regions in China proposed by the government of the Ching dynasty at the beginning of the 20th century, they let farmers and manufacturers of handcrafts who exiled themselves from south because of bankrupts move into Inner Mongolia and settle them there. The full-scale (except Alshia league and a part of HulunBuir city) cultivation of Inner Mongolia began since then (Jin, 2007). Over the past half century, large areas of the rangeland have been converted to cropland (Han *et al*., 2004). Although area of farmland was 4.7 million hectare in 1949, 5.9 million hectare in 1996, and it was became 7.4 million hectare in 2005 (**Fig. 2-2**).

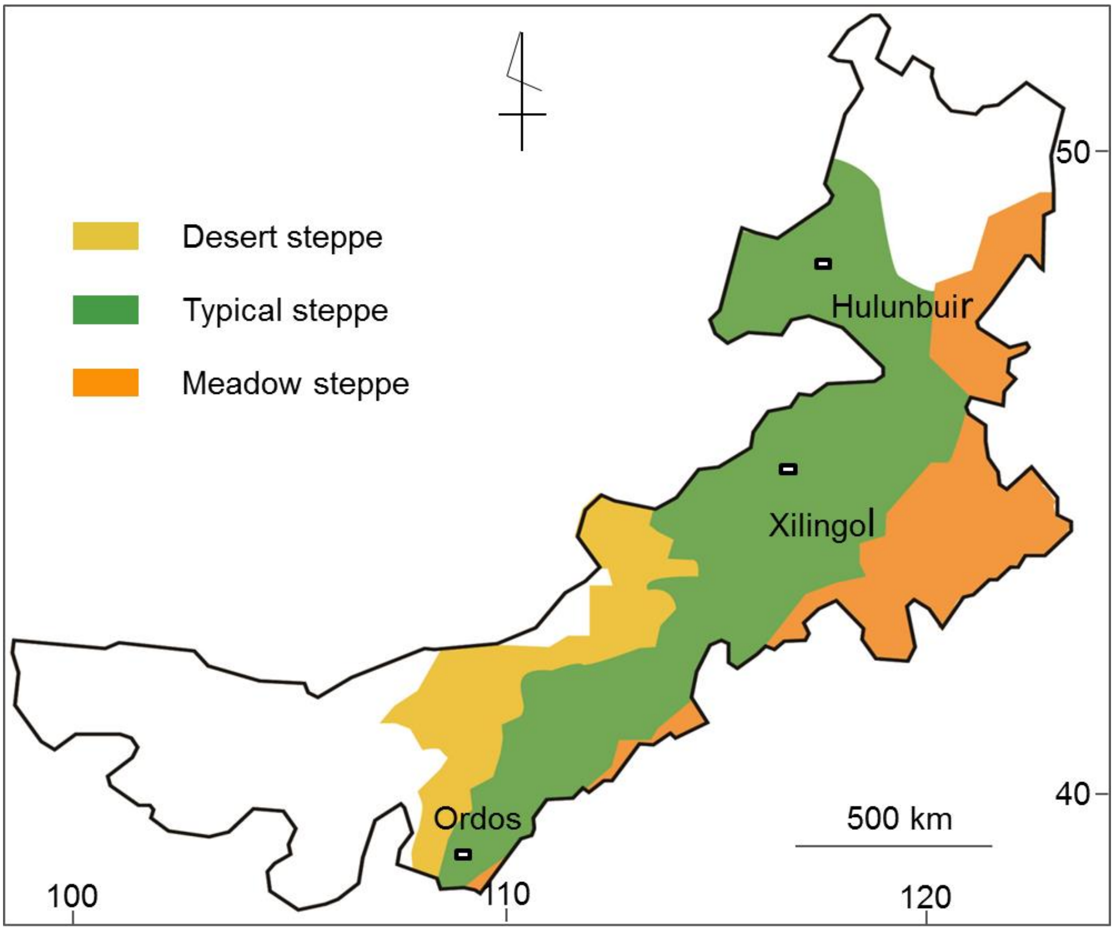

Fig. 2-4 Steppe zone in Inner Mongolia (after Kang *et al*., 2007)

Chapter 3

Relationships between the Species Composition of Vegetation and Environmental Factors in abandoned cropland and abandoned grazing area in Steppe Region of Kazakhstan

3-1 Introduction

Desertification has become a longstanding and increasingly severe problem in many lands of the steppe. The desertification process is often due to droughts, over-grazing and cultivation in terms of changes in surface vegetation and soil erosion on steppe (Bao *et al*., 2006; Liu and Hao, 2002; Squires, 2009; Zhao *et al*., 2011). Essentially, there is the very little land of steppe suitable for cultivation (Kovda, 1974). However, along with the rapid increase of human population, vast areas of steppe have been converted to cropland, and it is the farmers who have increased the size of these croplands (Liu *et al*., 2003).

The objectives of this study, therefore, were to: (1) classify and describe plant communities of the Midwestern Kazakhstan steppe, (2) demonstrate the relationship between environmental factors and plant species distribution and (3) identify the most effective factors for separating vegetation communities and for identifying and predicting changes in species composition. Detailed research background is explained in the General introduction (Chapter 1).

3-2 Materials and methods

3-2-1 Site description

This study was carried out in Akmola and Aktobe Provinces in northern Kazakhstan. Akmola Province is located centrally (71°16'-73°15'E, 51°31'-52°03'N), and Aktobe Province is in the west (57°15'-57°57'E, 49°56'-50°17'N) of the Kazakhstan steppes ecological region. Six study sites were included, these were in the villages of Ereymentau, Ivanovskoe and Klyuchi in Akmola, and Alga, Novorossiyskoye and Olke in Aktobe Province (**Fig. 3-1**).

To assess the effect of human activities on vegetation, I chose to examine sites with different land use patterns. The Ereymentau study site is a nature preservation area. Grazing has been prevented at the Klyuchi study site since 1956. The Alga and Novorossiyskoye sites are both on open steppe. The Olke site was cultivated until five years ago, and the Ivanovskoe site was cultivated or grazed until seven years ago (**Table 3-1**).

3-2-2 Sampling design

The vegetation data were collected from a total of 35 stands (4-225 m^2 in area) across the six study sites, according to the phytosociological method of Braun-Blanquet (1964), in July and August 2007. Plant communities were classified on the basis of their differential species (Braun-Blanquet, 1964; Mueller-Dombois and Ellenberg, 1974). Site location, latitude, longitude, and altitude were recorded using GPS, while gradient was recorded using a Suunto Clinometer; human disturbance and other activities were also recorded by interviews.

3-2-3 Data collection

3-2-3-1 Vegetation analysis

The species composition tables for hierarchical classification were constructed using the Database Management System, which allows input and collation of survey data from field records. Published data used were originated from the Forestry and Forest Products Research Institute (PRDB, 2006).

3-2-3-2 Soil analysis

Soil samples were collected from the upper 5 cm of the soil profile. In each stand, one soil sample was taken to determine pH (H_2O), electrolytic conductivity (EC), total nitrogen (T-N), total carbon (T-C) and C/N ratio.

The soil hardness was measured by using Nakayama-type penetrometer (company). Soil moisture was measured using a TDR (Time Domain Reflectometer, company) soil moisture device. Both characteristics were measured at the field. The soil samples were air dried, thoroughly mixed, and sieved through 2 mm and 0.5 mm sieves to remove stones and large fragments of organic material prior to chemical analysis. pH (H_2O) and

EC were determined using a subsample of the <2 mm fraction. The pH of a 1:2.5 soil: water mixture was measured with a digital pH meter (HM-30R, company). EC was measured using a portable electrical conductivity meter. Total carbon and total nitrogen were determined with a NC analyzer (machine name Shimazu Corporation, Japan); using the soil samples of less than 0.5 mm. The temperature of combustion tube was set at 830 ℃.

3-2-3-3 Climate data

Monthly and annual precipitation and temperature data (Willmott and Matsuura, 2007) were available for each research site for the period 1990-2006. Monthly averages for air temperature (*T*) and monthly total precipitation (*P*) were interpolated to a 0.5° by 0.5° degree latitude/longitude grid, where the grid nodes are centered on 0.25°.

Kira's Warmth Index (WI; Kira, 1977) and the rainfall index (K; Köppen, 1931), which have been shown to be useful when describing the vegetation-climate relationship in Eurasia (Araki, 2005), were calculated for each research site as follows:

$$WI = \Sigma(tn - 5)$$

$$K = P/2*(T + \alpha)$$

Where *tn* is the mean monthly temperature when it exceeds 5 ℃; *P* is the mean annual precipitation and *T* is the mean annual temperature; α is a constant, which equals 14 when the mean monthly precipitation is less than 60 mm, summer is the rainy season and winter is the dry season.

3-2-4 Statistical analysis

Vegetation data and the environmental factors were examined for each research site and the differences in species composition. The multivariate ordination technique detrended correspondence analysis (DCA; Hill, 1979) and canonical correspondence analysis (CCA; Ter Braak, 1986) were used to identify vegetation clusters, investigate patterns in species distribution and correlate the patterns with environmental variables. Canonical ordination is easier to apply and requires less data than regression. A Monte Carlo test (100 permutations) was used to analyze the significance of the first two canonical axes. These analyses were conducted using the R (Vegan) statistical package (R version 2.7.1, 2008-06-23; Copyright 2008 The R Foundation for Statistical Computing; ISBN 3-900051-07-0).

3-3 Results

3-3-1 Species composition of each community

Two phytosociological plant communities were recognized (**Table 3-2 and Appendix**): *Festuca sulcata-Stipa capillata* and *Artemisia vulgaris-Agropyron repens* communities. Furthermore, the *F. sulcata-S. capillata* community was sub-divided into the *Helictotrichon desertorum* lower unit and the *Helichrysum arenarium* lower unit. Almost all the study areas were characterized by gentle, flat lowlands. The altitude ranged from 252 m to 428 m.

Festuca sulcata-Stipa capilata **community**

The average number of species in a plot was 22, ranging from 16 to 34 for this community.

The community is characterized by *F. sulcata*, *S. capillata*, *Artemisia frigida*, *Carex physodes*, *Galium verum*, *Dianthus campestris*, *Koeleria cristata* and *Poa pratensis*. Out of a total of 35 study stands, 30 stands were identified as belonging to this community. The most widespread vegetation type of the Kazakhstan steppes from the central to the western study areas is the *F. sulcata-S. capillata* community.

Within this community, two first lower units were recognized. The *Helictotrichon desertorom* first lower unit is characterized by *H. desertorom*, *Phlomis tuberosa*, *Aster alpines*, *Plantago lanceolata*, *Artemisia schrenkiana* and *Thymus stepposus*. This lower - unit was very limited in its extent in the central part of the Kazakhstan steppe. The *Helichrysum arenarium* first lower unit is differentiated by the presence of *H. arenarium*, *Potentilla acaulis*, *Poa stepposa*, *Astragalus physodes* and *Artemisia scoparia*, and its presence was confirmed in the western study area.

Furthermore, even lower units were recognized below the *H. desertorom* sub-community. The first of these was the *Pulsatilla patens* second lower unit with the differential species, such as *P. patens*, *Veronica incana*, *Acroptilon repens*, *Artemisia* *nitrosa* and *Hieracium umbelliferum*. This second lower unit was found in the protected natural area of Ereymentau. The second was the *Stipa lessingiana* second lower unit, characterized by *S. lessingiana*, *Ferula soongarica* and *Inula hirta*. This unit was found in Klyuchi, in a field that had been abandoned for more than 50 years (since 1956). The third second lower unit represented the typical species composition and was found mainly in Ivanovskoe, but there was also one stand in the Ereymentau research area.

The *H. arenarium* first lower unit was also sub-divided into two lower-units: the *Euphorbia virgata* and the *Silene wolgensis* second lower units. Both were found in abandoned grazing areas of Alga and Novorossiyskoy. The soils were Kastanozems in Alga and Umbrisols in Novorossiyskoy.

***Artemisia vulgaris-Agropyron repens* community**

The average number of species in a plot was 24, ranging from 19 to 28 for this community.

The differential species for this community were *A. vulgaris*, *A. repens*, *Cichorium intybus* and *Ceratocarpus arenarius*. Large number of species were annual, such as *Helianthus annuus*, *Atriplex aucheri*, *Ceratocarpus arenarius* and *Cuscuta campestris*. The community was found in five stands in Olke, all in abandoned fields that have not been cultivated for five years. The soils were fertile chernozems.

3-3-2 Relationships between classified communities and land use history

I examined the relationship between communities land-use history (**Table 3-3**). Community I-A-a comprised three natural stands, two abandoned 50 years stands and one abandoned 27 years stands. Community I-A-b comprised two natural stands and four abandoned 50 years stands. Community I-B-e comprised four abandoned 27 years stands and two abandoned 13 years stands. All above means abandoned 50 years community is completely recovered, and recovery of abandoned 27 and 13 years communities are towards to natural. In comparison, communities that were abandoned grazing for 7 years and abandoned cultivation for 5 year are not found recovered. In other words, I-A-c and Ⅱ to preserve their original stands. It demonstrates that the recovery of the vegetation is related to the years that are more than 13, but not related to cultivation and grazing history.

3-3-3 Vegetation gradients and effect factors

The first axis of the survey stands (**Fig. 3-2**) has an eigenvalue of 0.573 and shows a trend running from the AC-5 to other all communities. On the first axis, the short abandoned site has a high score and is associated with land used for cultivation and as suggested by the results presented in Table 3-2 based on species composition. The second axis has an eigenvalue of 0.442 and represents a clear recovery process by abandoned years, running from abandoned 7 year site to 13 year, 27 year, 50 year sites and ultimately recovered to natural protected site.

There were not only the ages after the abandons of cultivation that affect species distribution, but also climatic, topographic and soil variables. The main environmental factors of each study stands as shown in **Table 3-4**. Soil moisture, T-C and T-N were represented high index in the higher precipitation regions, WI was influenced by lower precipitation, and pH, EC and C/N were represented high index in the cultivated areas.

The relationship between environmental factors and vegetation was examined by super-imposing the environmental data onto the ordination plot and looking for patterns and correlations (Janišova, 2005). All eleven environmental factors were collected in my study. However, high correlations were found between the environmental factors, such as T-C and T-N or K and WI (**Table 3-5**). This high correlation might influence the data set variation of CCA analysis. Based on the above information, I excluded the T-C and K from environmental factors. Also, elevation, slope and C/N were not obtained the *p* value as expected, thus, they were excluded. Finally, total six environmental factors were selected for CCA analysis. The results of the CCA are shown in **Fig. 3-3**. The 35 site scores are plotted along axes 1 and 2, and divided into six communities based on the community composition table. The species-environment correlations were high for the first three canonical axes, explaining (57.9%) of the cumulative variance. These results suggest a strong association between communities and the measured environmental parameters. In the biplot, several environmental variables explain a high percentage of the data set variation (**Table 3-6**). There was a significant correlation between species and environmental variables on the first and second CCA axes. Soil moisture (Monte Carlo test, $p < 0.01$) and WI ($p < 0.01$) were major variables associated with axis 1, whilst pH ($p < 0.01$) and EC ($p < 0.01$) were strongly associated with axis 2, that based on species distribution being determined by particular environmental variables. The communities found in the western study region, namely the AG-13 and the AG-27, were strongly associated with high WI in the lower right of the ordination diagram. The characteristic species included *E. virgata*, *Silene odoratissima* and *S. wolgensis*. All these community types were found in the abandoned grazing areas. However, examination of the feces of the domestic animals in the study region suggests that only a few of these plants are consumed by these animals. In the left part of the diagram, the NP and the AG-7 of the *Helictotrichon desertorum* first lower unit were associated with high soil moisture. These communities were found in the nature reserve and the abandoned grazing area of the central study region. In these areas, no domestic animals or their feces were observed. The AC-5 and the AC-50 were present in the upper right part of the diagram and were associated with high pH and EC. These communities were found in areas where cultivation had been abandoned in the central and western study regions.

3-4 Discussion

My study demonstrates that *S. capillata-F. sulcata* community is the dominant community of the Midwestern Kazakhstan steppes. This community is very similar to the *S. capillata-F. sulcata* community found in Ukraine and the *S. capillata* community in East Kazakhstan, identified by Cheng and Nakamura (2006, 2007). These communities were dominated by *S. capillata*, *F. sulcata* and *K. cristata*. It has been suggested that *S. capillata* is often dominant in western Eurasian regions, including the Ukraine and Kazakhstan steppes (Keller, 1927; Kawada *et al*., 2005), and *F. sulcata* and *K. cristata* are common members of the *S. capillata* community in most natural grasslands in the Western Eurasian steppe. However, except for the dominant species, there are few similarities at the species level between the Midwestern Kazakhstan community and the other Western Eurasian steppe communities described by Cheng and Nakamura (2006, 2007). This indicates that the variation in environmental conditions supports the high species diversity of vegetation in the Kazakhstan steppe.

Previous research suggests that the composition of the Kazakhstan steppe vegetation is similar at the genus level to that of the Mongolian and Inner Mongolian steppes (Nakamura *et al.*, 1988; Hilbig, 1995), although there are few similarities at the species level (Cheng and Nakamura, 2007). However, in my research, approximately one third of the all species which were confirmed are the same as the species found in Inner Mongolia, including *A. frigida*, *Aster alpinus*, *Phlomis tuberosa* and *Galium verum*. This means that different regions with similar environmental conditions support the same species. Of these, *A. frigida* and *A. scoparia* are commonly considered to be indicator species - their presence suggestion that grassland is degraded (Zhao and Zhou, 1999; Li *et al.*, 2002). *Potentilla bifurca* is generally considered to be an indicator of overgrazing (Zemmrich *et al.*, 2010). In addition, *P. tuberosa* and *G. verum* are have been identified as C_3 species on the Inner Mongolian meadow steppe (Liu *et al.*, 2004). The species present in many of the communities in my study included all the above indicator species. Only second lower units, *Stipa lessingiana* and *Euphorbia virgata*, were considered to represent a native vegetation type different from any found on the Mongolian and Inner Mongolian steppes.

Combining the results of my hierarchical classification and CCA, I found that the first ordination axis was associated with climatic factors. The WI was significantly correlated with vegetation composition: in drier locations with high WI, communities were characterized by the cushion-shaped shrubs *Caragana*, whilst more grasses were found on the west Kazakhstan steppe. Soil moisture was also important, with communities in relatively cooler climates with high soil moisture characterized by *Sanguisorba officinalis*, *Vicia cracca* and *Filipendula ulmaria*. Thus, climatic factors play a key role in controlling the species composition of vegetation communities at a large scale.

Species composition changes over time independently of community composition, because the composition and spatial pattern of patches can change. The results of the current study and that by Cheng and Nakamura (2007) on the Kazakhstan steppe indicate that most of the species of cultivated fields abandoned 1-5 years ago are annual or biennial herbs, fields abandoned 7 years ago are dominated by perennial forbs, and those abandoned 11 years ago have reverted to feather grass. This pattern of succession suggests that the vegetation is developing towards a climatic climax. In my results, although vegetation which was destroyed by cultivation had begun to recover after 13 years, the soil, which is fundamental for vegetation communities, has not yet recovered. My results indicate that the pH and EC values of soil that has been subjected to 50 years of cultivation in this region are higher than for undisturbed soils, and are generally alkaline. EC values between 0 and 1.5 dS $m^{-1}$and pH values between 6 and 7.5 are suitable, in general, for plant growth and microbial activity (Barbercheck *et al.*, 2009). At most of my sites, the soil pH was still within the range for optimum plant growth, although at the abandoned field sites the original neutral soil pH had increased resulting in slightly alkaline conditions caused by cultivation, which may be an important factor contributing to land disturbance on the steppe. As the results CCA analysis, no correlation were found between C/N, T-N in soil and the vegetation of abandoned cultivation area, but according to the result of measurement, it showed that the C/N value of abandoned cultivation area was higher than the value of abandoned grazing and natural protect area, and T-N value was lower in the abandoned cultivation area. The two factors, C/N and T-N, were also considered to be the reasons which related to the disturbance in steppe.

Fig. 3-1 Map of the study area and study sites with their locations in Kazakhstan (site.1: Erementau, site.2: Ivanovskoe, site.3: Klyuchi, site.4: Alga, site.5: Novorossiyskoy, site.6: Olke)

Table 3-1 Summerized information for studied area in Kazakhstan steppe

Study area	Akmola province			Aktobe province		
	Erementau	Ivanovskoe	Klyuchi	Alga	Novorossiyskoy	Olke
Location	51°31'N 73°15'E	52°03'N 71°34'E	51°33'N 71°16'E	49°56'N 57°15'E	50°17'N 57°32'E	50°14'N 57°57'E
Altitude(m)	391-398	341-347	425-428	320-326	252-257	419-428
Mean temperature (℃)*	1.1	2.3	2.6	4.8	4.3	4.2
Precipitation (mm)*	351.9	272.1	282.4	239.8	245.2	266.8
Land use	Natural protect area	Abandoned grazing area for 7 years	Protect area since 1956y (abandoned field)	Abandoned grazing area for 13 years	Abandoned grazing area for 27 years	Abandoned field for 5 years
Vegetation type	I-A-a	I-A-c	I-A-b	I-B-d	I-B-e	Ⅱ
Soil type	Chernozems	Chernozems	Chernozems	Kastanozems	Umbrisols	Chernozems

*data source from monthly and annual precipitation, temperature data referenced by Willmott and Matsuura (2007).

I-A-a: *Pulsatilla patens* second lower unit
I-A-b: *Stipa lessingiana* second lower unit
I-A-c: Typical second lower unit
I-B-d: *Euphorbia virgata* second lower unit
I-B-e: *Silene wolgensis* second lower unit
Ⅱ: *Artemisia vulgaris-Agropyron repens* community

Table 3-2 Synthesis table of steppe vegetation in Kazakhstan steppe

Ⅰ. *Festuca sulcata-Stipa capillata* community
A. *Helictotrichon desertorum* first lower unit
a. *Pulsatilla patens* second lower unit
b. *Stipa lessingiana* second lower unit
c. Typical second lower unit
B. *Helichrysum arenarium* first lower unit
d. *Euphorbia virgata* second lower unit
e. *Silene wolgensis* second lower unit
Ⅱ. *Artemisia vulgaris -Agropyron repens* community

Community type	Ⅰ					Ⅱ
	A			B		
	a	b	c	d	e	
Number of stands	5	6	7	7	5	5
Mean of total number of species	29	26	20	18	16	24
Differential species of *Festuca sulcata-Stipa capillata* community						
Festuca sulcata	Ⅴ	Ⅴ	Ⅴ	Ⅴ	Ⅴ	Ⅰ
Stipa capillata	Ⅳ	Ⅴ	Ⅴ	Ⅴ	Ⅴ	.
Artemisia frigida	Ⅴ	Ⅳ	Ⅴ	Ⅴ	Ⅴ	Ⅲ
Carex physodes	Ⅴ	Ⅴ	Ⅳ	Ⅱ	Ⅱ	.
Galium verum	Ⅴ	Ⅴ	Ⅴ	Ⅰ	Ⅰ	.
Dianthus campestris	Ⅴ	Ⅱ	Ⅱ	Ⅲ	Ⅴ	.
Koeleria cristata	Ⅴ	Ⅲ	Ⅲ	Ⅴ	.	.
Poa pratensis	.	Ⅳ	.	Ⅰ	Ⅲ	.
Differential species of *Helictotrichon desertorum* first lower unit						
Helictotrichon desertorum	Ⅳ	Ⅴ	Ⅰ	.	.	.
Phlomis tuberosa	Ⅲ	Ⅴ	Ⅴ	.	.	.
Aster alpinus	Ⅳ	Ⅴ	Ⅲ	.	.	.
Plantago lanceolata	Ⅱ	Ⅴ	Ⅲ	.	.	.
Veronica longifolia	Ⅲ	Ⅴ	Ⅱ	.	.	.
Potentilla bifurca	Ⅲ	Ⅳ	Ⅲ	.	.	.
Artemisia schrenkiana	Ⅱ	Ⅲ	Ⅳ	Ⅱ	.	.
Thymus stepposus	Ⅱ	Ⅳ	Ⅰ	.	Ⅱ	.
Filipendula hexapetala	Ⅲ	Ⅰ	Ⅲ	.	.	.
Tanacetum vulgare	Ⅲ	Ⅰ	Ⅱ	.	.	.
Scabiosa ochroleuca	Ⅱ	Ⅱ	Ⅰ	.	.	.
Differential species of *Pulsatilla patens* second lower unit						
Pulsatilla patens	Ⅴ	Ⅰ	.	.	.	.
Veronica incana	Ⅴ	.	Ⅰ	.	Ⅰ	.
Acroptilon repens	Ⅲ	.	.	.	.	Ⅰ
Artemisia nitrosa	Ⅳ	.	.	.	.	.
Hieracium umbelliferum	Ⅳ	.	Ⅰ	.	.	.
Differential species of *Stipa lessingiana* second lower unit						
Stipa lessingiana	.	Ⅴ	.	Ⅰ	Ⅰ	.
Ferula soongarica	.	Ⅴ	.	.	.	.
Inula hirta	.	Ⅴ	.	.	.	.
Differential species of *Helichrysum arenarium* first lower unit						
Helichrysum arenarium	.	.	.	Ⅴ	Ⅴ	.
Potentilla acaulis	.	.	.	Ⅲ	Ⅴ	.
Poa stepposa	.	.	.	Ⅳ	Ⅱ	.
Astragalus physodes	.	.	.	Ⅳ	Ⅴ	.
Artemisia scoparia	Ⅰ	Ⅰ	Ⅰ	Ⅴ	Ⅰ	.
Differential species of *Euphorbia virgata* second lower unit						
Euphorbia virgata	.	.	.	Ⅴ	.	.
Silene odoratissima	.	.	.	Ⅴ	.	.
Differential species of *Silene wolgensis* second lower unit						
Silene wolgensis	.	.	.	.	Ⅴ	Ⅰ
Koeleria glauca	.	.	.	.	Ⅳ	.
Artemisia arenaria	.	.	.	.	Ⅳ	.
Centaurea ruthenica	.	.	.	.	Ⅲ	.
Differential species of *Artemisia vulgaris -Agropyron repens* community						
Artemisia vulgaris	.	.	.	.	.	Ⅴ
Agropyron repens	.	Ⅱ	Ⅰ	.	.	Ⅴ
Cichorium Intybus	.	.	.	.	.	Ⅴ
Helianthus annuus	.	.	.	.	.	Ⅴ
Kochia prostrata	.	.	.	Ⅰ	Ⅰ	Ⅴ
Carduus stenocephalus	.	.	.	.		Ⅴ
Atriplex aucheri	.	.	.	Ⅰ	.	Ⅴ
Ceratocarpus arenarius	.	.	.	.	.	Ⅳ
Orobanche arenaria	Ⅱ	.	.	.	.	Ⅳ
Fagopyrum sagittatum	.	.	.	.	.	Ⅳ
Convolvulus arvensis	.	.	.	.	.	Ⅳ
Cuscuta campestris	.	.	.	.	.	Ⅳ
Raphanus sativus	.	.	.	.	.	Ⅲ
Matricaria recutita	.	.	.	.	.	Ⅲ
Companions						
Achillea millefolium	Ⅰ	Ⅲ	Ⅳ	Ⅲ	Ⅲ	Ⅴ
Medicago falcata	Ⅳ	Ⅴ	Ⅲ	.	.	Ⅳ
Polygonum aviculare	Ⅱ	.	Ⅰ	Ⅲ	.	Ⅴ
Artemisia lercheana	.	.	.	.	Ⅱ	Ⅴ
Erysimum marschallianum	.	.	Ⅱ	Ⅱ	.	Ⅴ
Potentilla erecta	Ⅴ	.	.	Ⅲ	Ⅳ	.
Scaligeria setacea	Ⅰ	Ⅱ	Ⅰ	Ⅳ	.	.
Potentilla impolita	.	Ⅴ	Ⅳ	Ⅰ	.	.
Linaria vulgaris	.	Ⅲ	.	Ⅲ	Ⅰ	Ⅳ
Eryngium planum	Ⅰ	.	Ⅴ	Ⅱ	.	.
Caragana frutex	Ⅳ	.	.	Ⅱ	.	.
Onosma tinctorum	Ⅱ	.	.	.	.	.
Limonium gmelinii	.	Ⅱ	Ⅲ	.	.	.
Spiraea hypericifolia	.	.	.	.	Ⅰ	.
Silene gebleriana	.	Ⅰ	Ⅱ	Ⅰ	.	.
Linosyris tatarica	.	Ⅲ	Ⅱ	.	Ⅰ	.
Sanguisorba officinalis	.	Ⅰ	Ⅰ	.	.	.
Vicia cracca	.	.	Ⅰ	.	.	.
Senecio erucifolius	.	.	Ⅰ	.	.	.
Bromus inermis	Ⅰ	Ⅰ	Ⅰ	.	.	.
Plantago major	Ⅰ	.	Ⅱ	.	.	.
Tragopogon tatarica	Ⅱ	Ⅰ		.	.	.

Table 3-2 Continued

Ⅰ. *Festuca sulcata-Stipa capillata* community
  A. *Helictotrichon desertorum* first lower unit
    a. *Pulsatilla patens* second lower unit
    b. *Stipa lessingiana* second lower unit
    c. Typical second lower unit
  B. *Helichrysum arenarium* first lower unit
    d. *Euphorbia virgata* second lower unit
    e. *Silene wolgensis* second lower unit
Ⅱ. *Artemisia vulgaris -Agropyron repens* community

Community type	Ⅰ					Ⅱ
	A			B		
	a	b	c	d	e	
Number of stands	5	6	7	7	5	5
Mean of total number of species	29	26	20	18	16	24
Inula britannica	.	.	Ⅱ	.	.	.
Lappula microcarpa	.	.	Ⅰ	.	.	Ⅲ
Salvia stepposa	.	.		.	Ⅰ	
Leymus angustus	.	Ⅰ	Ⅱ	.	.	
Equisetum fluviatile	.	.	.	.	.	Ⅱ
Avena sativa	.	.	.	.	.	Ⅱ
Chenopodium album	.	.	.	.	.	Ⅱ
Saussurea elata	.	Ⅰ	Ⅰ	Ⅰ	.	.
Alyssum desertorum	.	.	.	Ⅱ	.	Ⅰ
Jurinea cyanoides	.	.	.	Ⅱ	.	.
Astragalus danicus	Ⅱ	.	.	.	.	.
Senecio jacobaea	Ⅱ	.	Ⅰ	.	.	Ⅰ
Saussurea laciniata		Ⅱ	.	Ⅰ	.	
Onosma simplicissimum	Ⅰ	Ⅰ	.	.	.	.
Asperula humifusa		Ⅰ	Ⅰ	.	.	.
Acroptilon sp	Ⅰ	Ⅱ	.	.	.	.
Scorzonera austriaca	Ⅱ	.	.	.	.	.
Phragmites australis	Ⅰ	.	.	.	.	.
Glyceria sp	Ⅰ	.	Ⅰ	.	.	.
Onobrychis viciifolia	Ⅰ	Ⅰ	.	.	.	.
Campanula sibirica	Ⅰ	.	.	.	.	.
Teucrium scordium	.	.	Ⅱ	.	.	.
Veronica biloba	.	Ⅱ	.	.	.	.
Senecio jacobea	.	Ⅱ	.	.	.	.
Lactuca tatarica	.		.	Ⅰ	.	.
Fragaria vesca	.	Ⅰ	Ⅰ	.	.	.
Allium inaequale	.	Ⅱ	.	.	.	.
Bromus tectorum	.	.	.	.	.	.
Verbascum blattaria	Ⅰ	.	.	.	.	Ⅱ
Verbascum sp	Ⅰ	.	.	.	.	.
Androsace turczaninovii	Ⅰ	.	.	.	.	.
Potentilla humifusa	Ⅰ	.	.	.	.	.
Ceratoides sp	Ⅰ	.	.	.	.	.
Polygonum hybridm	Ⅰ	.	.	.	.	.
Polygonum amphibium	.	.	Ⅰ	.	.	.
Elysimum marschlianum	.	.	Ⅰ	.	.	.
Apiaceae sp	.	.	Ⅰ	.	.	.
Pedicularis achilleifolia	.	.	Ⅰ	.	.	.
Urtica dioica	.	.	Ⅰ	.	.	.
Stellaria graminea	.	.	Ⅰ	.	.	.
Glycyrrhiza glabra	.	.	Ⅰ	.	.	.
Tripleurospermum inodorum	.	.	Ⅱ	.	.	.
Seseli ledebourii	.	.	Ⅰ	.	.	.
Allium globosum	.	Ⅰ	.	.	.	.
Glycyrrhiza uralensis	.	Ⅰ	.	.	.	.
Erigeron podolicus	.	Ⅰ	.	.	.	.
Thalictrum simplex	.	Ⅰ	.	.	.	.
Iris ruthenica	.	Ⅰ	.	.	.	.
Phlomis sberosa	.	Ⅰ	.	.	.	.
Astragalus puberulus	.	Ⅰ	.	.	.	.
Taraxacum officinale	.	Ⅰ	.	.	.	.
Seratula coronata	.	Ⅰ	.	.	.	.
Ferula tatarica	.	Ⅰ	.	.	.	.
Ephedra distachya	.	.	.	Ⅱ	.	.
Centaurea sibirica	.	.	.	Ⅰ	.	.
Astragalus tauricus	.	.	.	Ⅰ	.	.
Agropyron desertorum	.	.	.	Ⅰ	.	.
Gypsophila altissima	.	.	.	.	.	Ⅰ
Gypsophila paniculata	.	.	.	Ⅰ	.	.
Anisantha tectorum	.	.	.	.	.	Ⅰ
Erigeron canadensis	.	.	.	.	.	Ⅰ
Elysimum marschalianum	.	.	.	.	.	Ⅱ
Onosma tinctorium	.	.	.	.	Ⅰ	.
setaria viridis	.	.	.	.	.	Ⅰ
Leonurus glaucescens	.	.	.	.	.	Ⅰ
Allium lineare	.	.	.	.	.	Ⅰ
Melilotus officinalis	.	.	.	.	.	Ⅰ
Polygonum convolvulus	.	.	.	.	.	Ⅰ

Constancy, calculated by the 'cover degree value' for each species from community table; Ⅴ, 80.1-100% high constancy; Ⅳ, 60.1-80% moderately high constancy ; Ⅲ, 40.1-60% intermediate constancy; Ⅱ, 20.1-40% low constancy; Ⅰ, <20% very low constancy.

Table 3-3 Communities from hierarchical classification responded to land use type

Land use type	Communities					
	Ⅰ-A-a	Ⅰ-A-b	Ⅰ-B-e	Ⅰ-B-d	Ⅰ-A-c	Ⅱ
NP	3	2				
AC-50	2	4				
AG-27	1		4			
AG-13			2	5		
AG-7					7	
AC-5						5
Total stands	6	6	6	5	7	5

NP: Natural Protect area; AC: Abandoned Cropland; AG: Abandned Grazed area

I-A-a: *Pulsatilla patens* second lower unit
I-A-b: *Stipa lessingiana* second lower unit
I-A-c: Typical second lower unit
I-B-d: *Euphorbia virgata* second lower unit
I-B-e: *Silene wolgensis* second lower unit
Ⅱ: *Artemisia vulgaris-Agropyron repens* community

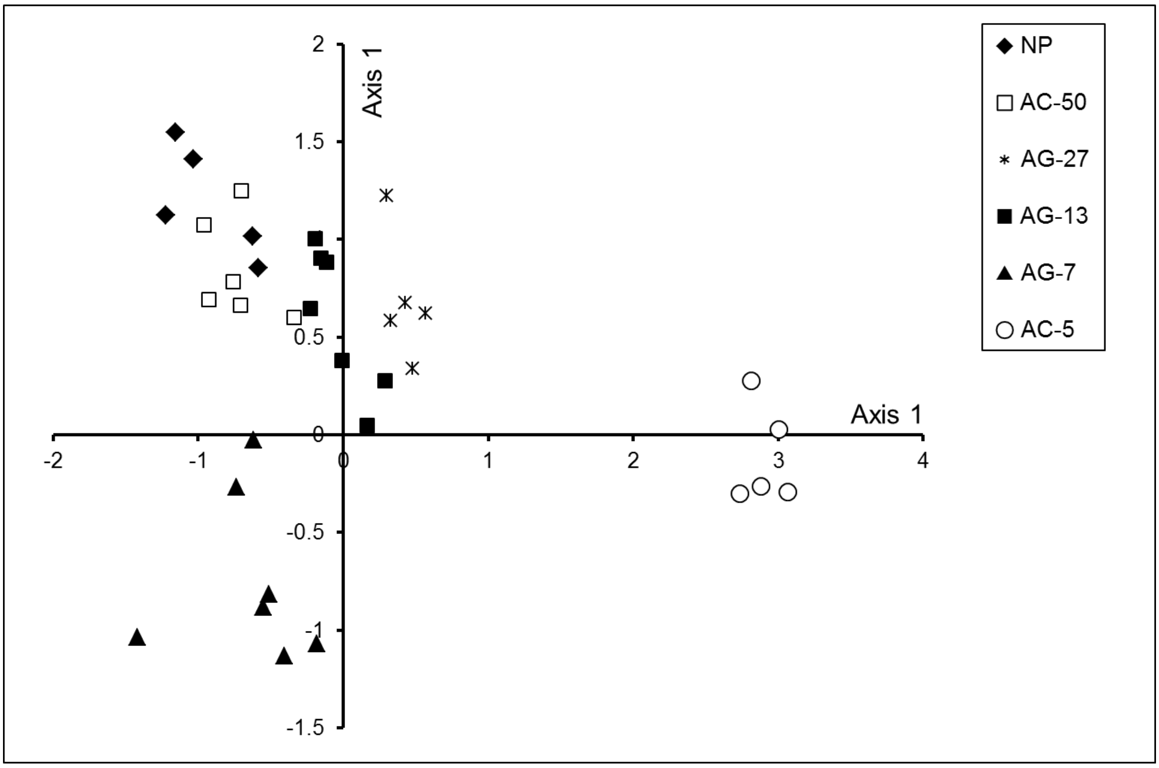

Fig. 3-2 DCA analysis, ordination of sites by different land use type in Kazakhstan steppe. Eigenvalue for axis 1 and 2 was 0.573 and 0.442, respectively.

Table 3-4 Values of environmental factors at the all study sites

Properties	NP	AC-50	AG-27	AG-13	AG-7	AC-5
Elevation (m)	395.2 ± 2.5 a	426.8 ± 2.7 a	254.7 ± 3.5 c	324.3 ± 3.1 b	343.5 ± 2.8 b	424.4 ± 4.2 a
pH (H_2O)	6.58 ± 0.45 b	8.39 ± 0.18 a	6.68 ± 0.14 b	6.46 ± 0.15 b	7.03 ± 0.73 b	8.23 ± 0.18 a
EC (dS/m^{-1})	0.08 ± 0.04 c	0.18 ± 0.07 a	0.06 ± 0.01 c	0.06 ± 0.01 c	0.12 ± 0.05 b	0.21 ± 0.06 a
Soil moisture (%)	17.75 ± 2.27 a	9.62 ± 0.81 b	7.54 ± 1.15 b	7.82 ± 0.74 b	16.18 ± 1.44 a	9.57 ± 1.53 b
Soil hardness (mm)	15.90 ± 2.30 b	14.33 ± 2.17 b	14.52 ± 4.99 b	18.27 ± 3.52 a	12.84 ± 1.85 c	15.08 ± 6.21 b
T-C (g/kg)	39.89 ± 8.86 a	38.24 ± 3.06 a	21.70 ± 2.73 c	18.59 ± 3.31 c	29.84 ± 7.02 b	16.95 ± 4.81 c
T-N (g/kg)	4.11 ± 0.73 a	2.77 ± 0.47 b	1.96 ± 0.23 c	1.71 ± 0.24 c	2.90 ± 1.09 b	1.15 ± 0.29 c
C/N	9.71 ± 0.44 c	13.82 ± 1.15 a	11.07 ± 1.22 b	10.87 ± 0.51 b	10.29 ± 1.17 c	14.75 ± 0.88 a
K index	11.50	8.49	6.69	6.30	8.34	7.33
WI index	47.12	57.71	63.48	66.57	53.51	64.52

* Mean (± SE) values compared with Tukey-test, the significant differences are indicated by different letters.

Table 3-5 Correlation among environmental factors in study stands

Environmental factor	elevation	pH (H_2O)	EC	soil moisture	soil hardness	slope	T-N	T-C	C/N	K	WI
Elevation	—										
pH (H_2O)	-0.169	—									
EC	-0.127	0.545*	—								
Soil moisture	0.218	-0.014	0.647*	—							
Soil hardness	-0.314	-0.563	0.602*	-0.338	—						
Slope	0.157	0.147	0.207	0.103	-0.011	—					
T-N	0.152	0.199	0.755**	0.866**	-0.435	0.030	—				
T-C	0.135	0.247	0.779**	0.847**	-0.465	0.116	0.985**	—			
C/N	-0.238	0.148	-0.127	-0.419	0.011	0.387	-0.383	-0.228	—		
K	0.363	-0.020	0.414	0.818**	-0.184	0.295	0.652*	0.681*	-0.181	—	
WI	-0.407	0.093	-0.455	-0.916	0.210	-0.197	0.733**	0.730**	0.351	0.940**	—

**-coefficients significant at 0.01 level, *-coefficients significant at 0.05 level (definition of pearson's correlation coefficient);

EC: electrolytic conductivity; T-N: total nitrogen; T-C: total carbon; K: Köppen's rainfall index; WI: warmth index

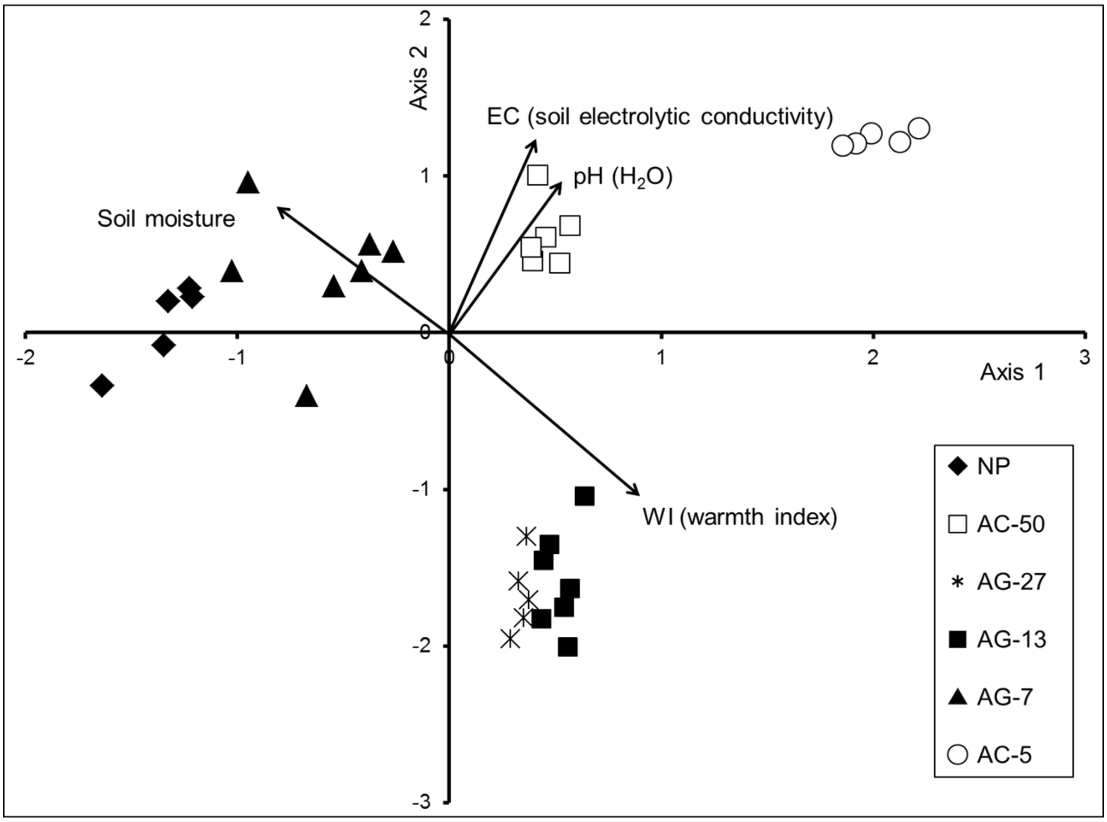

Fig. 3-3 CCA analysis, different land use stands with their influence factors on the standard two axes (p < 0.01). Eigenvalue for axis 1 and 2 was 0.533 and 0.338, respectively. (I-A-a: *Pulsatilla patens* second lower unit, I-A-b: *Stipa lessingiana* second lower unit, I-A-c: Typical second lower unit, I-B-d: *Euphorbia virgata* second lower unit, I-B-e: *Silene wolgensis* second lower unit, Ⅱ :*Artemisia vulgaris-Agropyron repens* community).

Table 3-6 Canonical correspondence analysis, correlation of species ordination axes with environmental factors, eigenvalues and percentage variances explained for 6 factors.

Properties	Axis 1	Axis 2	Axis 3
1. pH (H_2O)	0.671 *	0.740 **	-0.405
2. EC	0.377	0.937 **	-0.031
3. Soil moisture	-0.874 **	0.511	0.216
4. Soil hardness	0.121	-0.317	0.079
5. T-N	-0.493	0.548 *	0.397
6. WI	0.915 **	-0.783 *	0.046
Eigenvalue	0.53	0.34	0.22
% variance	8.9	6.4	3.7

Significant correlation (Monte Carlo test, **P<0.01, *P<0.05)

EC: electrolytic conductivity; T-N: total nitrogen; WI: warmth index

Chapter 4

Effects of soil erosion rates on species composition and soil physicochemical properties in abandoned steppe croplands of Inner Mongolia

4-1 Introduction

Chapter 3 described the relationship between vegetation and environmental factors. The distribution of vegetation communities in abandoned field sites differed from that elsewhere due to differences in soil pH and EC. This may increase the rate of soil erosion in abandoned field sites relative to that seen on natural grasslands on the steppe.

Inner Mongolia is one of the most desertified regions in China: 35.6% of its total land area is undergoing or has experienced desertification (Gao *et al*., 2006). The Chinese government introduced the Cropland Conversion to Forest and Grassland (CCFG) program in 2002 to provide environmental countermeasures to this process. Efforts made within this program have restored some vegetation and stopped the spread of desertification in some places (Wang *et al*., 2009). However, no environmental improvements have been observed in most areas where the program has been implemented (Batu, 2007). In addition, it seems that some areas of environmental destruction will continue to grow due to the project's implementation (Osawa, 2005). It is therefore likely that the extent of soil erosion in Inner Mongolia will increase in future.

The aims of this study were to: (1) describe vegetation patterns in natural grasslands and abandoned croplands in different regions of the steppe, (2) measure soil erosion rates at each site using the ^{137}Cs technique and explore the soil's physicochemical properties, and (3) characterize the relationship between soil erosion rates, the local plant species composition, and soil physicochemical properties.

4-2 Materials and methods

4-2-1 Site description

Sites at three locations in the ecological region of the Inner Mongolian steppe: HulunBuir city in northeastern Inner Mongolia, Xilingol league in the center of the region, and Ordos city in the west of the region. Thirteen study sites were investigated. Four were in the ShinBarag-you-qi (SB) area of HulunBuir city: the non-cultivated site SB-1 and the cultivated sites SB-2, SB-3, and SB-4. Three were in Xilinhot city (X) and another three were in Sonid-you-qi (S), both of which lie within Xilingol league: the non-cultivated sites X-1 and S-1, and the cultivated sites X-2, X-3, S-2, and S-3. Finally, there were three sites in the vicinity of Wushin-qi in Ordos city: the non-cultivated site W-1 and the cultivated sites W-2 and W-3. The sites' locations are shown in **Fig. 4-1**.

To assess the effects of crop cultivation and abandonment on vegetation, two site types were studied in each area: non-cultivated sites that had experienced little or no grazing, and croplands that had been abandoned at various points in time. Details on the study sites and their history are presented in **Table 4-1**.

4-2-2 Sampling design and data collection

4-2-2-1 Vegetation analysis

The quadrat method was used to characterize the vegetation cover at each site. Five plots of $1 m^2$ each were established at each sampling site in August 2010, along 50 m. A total of 65 plots were thus established across the 13 sites and four study areas.

Vegetation data were collected for each plot. I measured the height of each species and estimated their cover values, recording each individual plant that appeared within the plots. In addition, for each site, I recorded the total vegetation cover according to Penfound–Howard (1940). I also calculated the summed dominance ratio (SDR_3; Numata, 1969) for each identified species based on the height, cover and frequency of occurrence at each site, using the following equation:

$$SDR_3 = (T' + C' + F') / 3$$

Here, T' is the sum of the species' height in all quadrats within a site, expressed as a percentage of the maximum observed height for that species; C' is the total coverage of the species in all quadrats, expressed as a percentage of the maximum; and F' is the total frequency of each species for all quadrats at the site in question, expressed as a percentage of the maximum.

4-2-2-2 Soil analysis

Soil sampling was performed in two ways. The first involved collecting samples from the uppermost 5 cm of the soil profile in each plot at the cultivated and non-cultivated sites. The soil from the non-cultivated site was used as a reference when establishing the ^{137}Cs inventory for the studied region. Changes in the inventory at both erosion and deposition sites were then evaluated by comparison to the reference. The second sampling method involved collecting 2 cm thick soil columns to a depth of 20 cm at abandoned cropland sites only. The collected soil samples were air-dried and passed through a 2 mm mesh sieve to remove pebbles and plant roots, then placed in a plastic container for use in gamma spectrometric assays.

To investigate the relationship between the ^{137}Cs inventory and soil characteristics, the samples' texture and nutrient content were quantified. The soil texture was determined by mechanical analysis using the pipette method (Day, 1965), and the soil phosphorus content was determined by the method of Truog (Truog and Meyer, 1929). Other soil properties such as pH (H_2O), electrolytic conductivity (EC), total nitrogen (T-N), organic carbon (OC) and C/N ratio were measured as described in chapter 3.

4-2-2-3 Cs^{137} as tracers of soil erosion

The Cs^{137} technique was used to monitor soil erosion. Radio-caesium (Cs^{137}) does not occur naturally on the earth and is exclusively anthropogenic. As a result of atomic bomb tests during the 1950s and 1960s, radioactive fission products were launched into the atmosphere, spread around the world, and gradually deposited on the land. One of the most abundant of these fission products today is Cs^{137}, due to its relatively long half life (30 years). When Caesium-137 reaches the soil surface via wet or dry deposition processes, it is quickly and strongly adsorbed by the soil and is almost nonexchangeable in most environments. Therefore wind and water are the dominant agents that move soil particles containing Cs^{137}. In essence, this radio-isotope becomes adsorbed on fine soil particles and cannot be taken up by plants or react with other chemicals in the soil. It therefore travels with wind and water in the same way as the soil itself, and can effectively be used as a tracer for soil particles (Ritchie and McHenry, 1990; Zapata, 2002). Accurately measuring Cs^{137} activities in environmental samples is relatively easy. Thus, by measuring the distribution of Cs^{137} across diverse sites, one can obtain information on average soil redistribution rates and patterns.

The distribution of Cs^{137} in soil can be measured using gamma-spectrometry. Cs^{137} is quantified in terms of activity (Bq kg) or inventory, which is the activity per unit area (Bq m^{-2}). At any erosion study site, this activity will decrease every year due to radioactive decay and erosion. To measure the local fallout inventory, it is necessary to find a place in the study area that has not undergone erosion or deposition such that the only factor causing radio-caesium degradation is radioactive decay. This site is referred to as the reference. The concentration of Cs^{137} at an erosion site that has lost some fine Cs^{137}-tagged particles will be lower than that at the reference site whereas a deposition site will accumulate Cs^{137}-tagged soil particles and so the opposite will be true. If a relationship between caesium-137 loss or gain and soil loss or gain can be established, one can estimate the rates of soil erosion and deposition by measuring the Cs^{137} content of soil samples.

The radioactivity of ^{137}Cs in the soil samples was measured by counting 662 keV gamma-ray emissions for 12 h using germanium detectors (Canberra, BE5025) at the National Institute for Agro-Environmental Sciences of Tsukuba, Ibaraki, Japan. The standard date used for decay correction was August 2010. ^{137}Cs activity was measured in mass-based units (Bq / kg^{-1}). Total activity, i.e. the ^{137}Cs inventory (expressed in area-based units, Bq m^{-2}), was calculated from the vertical profile of ^{137}Cs activity, soil bulk density, layer depth, and the number of layers.

4-2-3 Statistical analysis

Detrended correspondence analysis (DCA; Hill, 1979) was used to visualize the vegetation community and structure. DCA is an indirect gradient analysis technique in which the distribution of species is not controlled by environmental variables. Instead, it focuses on analyzing the pattern of species distribution (Ahmad and Yasmin, 2011). Presence (1) or absence (0) data for each species in each plot were used in these analyses. The Sorensen similarity index for two sites (X and Y) was calculated as $Qs = 2a / (2a + b + c)$, where a is the number of species common to at the two sites, b is the number of species at site X, and c is the number of species at site Y (Magurran, 2004). Vegetation data and the environmental factors were examined for each research site and their differences in vegetation composition were investigated. The multivariate ordination technique canonical correspondence analysis (CCA; Ter Braak, 1986) was used to identify vegetation clusters, investigate patterns of species distribution, and correlate the patterns with environmental variables. Canonical ordination is easier to apply and requires less data than regression. A Monte Carlo test (100 permutations) was used to analyze the significance of the first two canonical axes. Multivariate analysis was conducted using the R (Vegan) statistical package (R version 2.15.2, 2012-10-26; Copyright 2012 The R Foundation for Statistical Computing; ISBN 3-900051-07-0).

Other statistical analysis was performed using the SPSS statistical software package (version 19.0; SPSS, Inc., Chicago, IL, United States). In all tests, significant effects/interactions were those with a P value that was < 0.05. Multiple comparisons were performed using Tukey's test to compare differences in vegetation and soil parameters in each area. Spearman's rank correlations were performed to compare differences in soil inventory and other environmental factors between sites.

4-3 Results

4-3-1 Vegetation patterns in each study area

In each studied area, both the mean number of species and mean vegetation coverage (%) at cropland sites that had recently been abandoned were significantly lower (P < 0.05) than those at natural grasslands (**Fig. 4-2**, **Fig. 4-3**). However, the coverage of long-term abandoned sites (other than those in the Sonid-you-qi area) did not differ significantly from that for natural sites. Interestingly, the coverage at the SB(1) site differed significantly from that at the SB(5) site and was not significantly different from that for the SB(Natural) site. In each studied area, the coverage of perennial and shrub species at short-term abandoned sites was significantly lower than at natural sites. Conversely, annual species were significantly more abundant at short-term abandoned sites than at natural sites (**Fig. 4-4**).

In the ShinBarag-you-qi area, the SDR_3 values of *Stipa krylovii* and *Astragalus galactites* were significantly higher at the natural site than at the recently abandoned sites while those for *Cleistogenes squarrosa*, *Carex korshinskyi*, *Potentilla bifurca* and

Chenopodium acuminatum were significantly lower at the natural site than at the recently abandoned site (**Table 4-2**). In total, eight common species were found in the Xilinhot region (**Table 4-3**). The SDR values for two of these common species, *Stipa grandis* and *Leymus chinensis*, were significantly higher at the natural site than the recently abandoned sites. Conversely, those for *Cleistogenes squarrosa*, *Carex korshinskyi*, *Artemisia frigida* and *Chenopodium glaucum* were significantly lower at the natural site than the recently abandoned sites. No common species were identified at all three study sites in the Sonid-you-qi area (**Table 4-4**). The natural site in this area was dominated by *Stipa breviflora*. In contrast, the S(21) site was dominated by *Agropyron cristatum* but also contained numerous annual species such as *Eragrostis pilosa*, *Lappula myositis* and *Corispermum declinatum*. Species that were common to the S(8) and S(Natural) sites included *Salsola collina*, *Cleistogenes squarrosa* and *Caragana microphylla*; the SDR values for all of these species were higher at the S(8) site. In addition, *Stipa krylovii* was found at the S(8) site exclusively. Three common species were found in the Wushin-qi area (**Table 4-5**). Of these, *Leymus secalinus* had a significantly higher SDR value at the natural site than the recently abandoned site whereas the opposite was true for *Stipa glareosa*.

4-3-2 ^{137}Cs distribution profile, ^{137}Cs loss (soil erosion) rates and soil physicochemical parameters

The ^{137}Cs activity was distributed irregularly and evenly in some soil samples collected using the second sampling method, and some cases distributed shallower or not (**Fig. 4-5**). Two distinct ^{137}Cs distribution patterns were thus identified: an anthropogenic disturbance profile and an eroded profile. In addition, the variation in ^{137}Cs concentrations at recently abandoned sites was lower than that at long-term abandoned sites in each studied area (**Fig. 4-5**).

The rates of ^{137}Cs loss from the soil surface at each site calculated based on the ^{137}Cs inventories are shown in **Table 4-6**. The natural grassland (reference sites) ^{137}Cs inventories were relatively high in all of the studied areas. However, I was unable to determine a ^{137}Cs inventory for the S(8) site. In the SB area, the calculated ^{137}Cs loss rates for sites that had been abandoned for 25 years were approximately one third as high as those for sites that had been abandoned for only 1 or 5 years. The soil loss rates for the X(34) and X(11) sites in the Xilingol area were very similar to those for the W(30) and W(13) sites in the Wushin-qi area.

Table 4-7 show the soil physicochemical properties determined during the field trial. In each area, the clay, silt and organic matter contents of the soil were significantly higher at natural sites than recently abandoned ones. Conversely, the sand and pH (H_2O) levels at natural sites were significantly lower than those at recently abandoned ones. The electrolytic conductivity (EC) was significantly lower at natural sites than recently abandoned ones in the Sonid-you-qi and Wushin-qi areas, but no clear trend was detected for the ShinBarag-you-qi and Xilinhot areas.

4-3-3 Gradients of vegetation and soil erosion

Multivariate analysis using DCA was used to compare the compositions of the communities in the studied plots (see **Fig. 4-6**). The eigenvalues and gradient lengths of the first and second DCA axes with respect to species were relatively high, indicating that the different species were well separated along the first and second axes. The eigenvalues for each group in the SB area were 0.64 (axis 1) and 0.26 (axis 2); those for the X area were 0.47 (axis 1) and 0.18 (axis 2); those for the S area were 0.64 (axis 1) and 0.27 (axis 2); and those for the W area were 0.41 (axis 1) and 0.21 (axis 2). The species composition gradient along the first DCA axis was generally consistent with the amount of time that had elapsed since site abandonment, dividing the species into three or four separate groups. That is to say, when the communities were arranged along the first DCA axis in order of the time since the abandonment of the corresponding sites, they formed a strong gradient with the natural sites at one extreme and the most recently disturbed sites at the other. The species compositions at sites that had been abandoned for 34 years in the Xilinhot area and another that had been abandoned for 30 years site in the Wushin-qi area had recovered fully from their use as croplands and were identical to those for natural grassland, while a site that had been abandoned for 25 years site in the ShinBarag-you-qi area had almost fully reverted to the “natural” state. However, no trend towards recovery was observed for sites that had been abandoned for 21 years in Sonid-you-qi area (**Fig. 4-6**). There was no clear trend in species composition relative to the second DCA axis.

As shown in **Figure 4-7**, there was a significant negative correlation between the value of the pairwise similarity index when comparing natural and abandoned sites and the rate of soil erosion at the two sites ($p < 0.05$, $r = 0.825$). The highest values of the Sorensen similarity index (0.64, 0.59) were calculated for pairs of sites with low levels of erosion, while the lowest similarity index value (0.19) was obtained in a comparison of a highly eroded site and a natural site.

4-3-4 Relationship between soil erosion, soil physicochemical parameters and vegetation

Canonical correlation analysis (CCA) was used to identify correlations between seven environmental factors (soil erosion rate, clay, silt, sand, and organic matter content, pH, and EC) and ^{137}Cs loss rates. The results of the CCA are shown in **Fig. 4-8**. The scores for 15 to 20 sites were plotted along axes 1 and 2 and cluster into three or four communities based on species composition. The species-environment correlations were high for the first two canonical axes, and explained the following proportions of the cumulative variance for each area: SB, 57.7%; X, 57.9%; S, 61.6%; and W, 57.3%. These results suggest a strong association between community composition and the studied environmental parameters. Biplots for each study area revealed significant correlations between species composition and environmental variables on the first and second CCA axes. High clay (Monte Carlo test, $p < 0.05$) and organic matter ($p < 0.05$) contents were strongly associated with natural communities, while high soil erosion rates ($p < 0.05$) and pH values ($p < 0.05$) were strongly associated with communities at recently abandoned sites.

4-4 Discussion

Detailed measurements of soil erosion rates were acquired, making it possible to investigate the relationship between soil erosion rates, vegetation cover, and soil physicochemical properties in natural sites and abandoned croplands.

The rates of ^{137}Cs loss (soil erosion) were generally highest for recently abandoned sites followed by long-term abandoned sites and then natural sites (**Table 4-6**). This implies that periods of cultivation followed by abandonment influence soil erosion in steppe regions. However, it was also found that the range of ^{137}Cs concentrations measured in recently abandoned cropland is greater than that for long-term abandoned plots (**Fig. 4-5**). This is probably because the recently abandoned croplands had been cultivated for longer periods of time (**Table 4-1**), and large area of cropland are left bare after harvesting, causing extensive wind erosion of the dry and unconsolidated soil (Aguilar *et al*., 1988; Liu *et al*., 2003). Consequently, the Cs content of sites that were cultivated for short periods of time was greater than that for those that had been cultivated for longer periods. Sites that had been cultivated for extended periods exhibited evidence of extensive soil erosion as a result of their cultivation. Notably, no Cs was detected at the S(8) site, which is located near the desert steppe and had been used as cropland for a long period of time, so it is likely that it would have experienced extensive wind erosion.

Vegetation coverage and number of species are negatively associated with the degree of soil erosion (Nunes *et al*., 2011; Zhou *et al*., 2008). In this work, the mean number of species, mean vegetation coverage and the number of perennial species decreased as ^{137}Cs loss rates increased (**Fig. 4-2**, **4-3** and **4-4**). Conversely, annual species numbers increased with the ^{137}Cs loss rate. This indicates that the natural vegetation community of the steppe was replaced by tolerant annual plant species, and that the pattern of intense plant species diversity reduction due to soil erosion has an ecological basis (García-Fayos *et al*., 2010). In addition, the degree of vegetation coverage was quite variable, particularly in the recently abandoned sites. This suggests that the vegetation at these sites is not in equilibrium, and that the number of species is likely to fluctuate with the vegetation coverage. Consequently, with the exception of one site that had been abandoned for only one year in the SB area, the vegetation coverage at most recently abandoned sites was not consistent with recovery to the natural state. However, the vegetation coverage at a site that had been abandoned for only one year was inconsistent with this trend, being significantly higher than that for a five year abandoned site. This suggests that fast growing annual plants become established over a large area following disturbance (James *et al*., 2011), and that livestock do not like to eat them (Xu *et al*., 2005).

The species composition measurements (**Table 4-2** to **4-5**) indicate that the dominant species in natural grassland such as *Stipa grandis*, *Leymus chinensis*, *Kochia prostrata* and *Leymus secalinus* (Nakamura *et al*., 1998, 2000; Zhu *et al*., 2009) were present at similar or higher levels at sites with low levels of erosion. In contrast, indicator species of grassland degradation such as *Salsola collina*, *Cleistogenes squarrosa*, *Carex korshinskyi*, *Artemisia frigida* and *Artemisia scoparia* (Kawada *et al*., 2011; Li *et al*., 2002; Zhao and Zhou, 1999) were present, often at high levels, in highly eroded sites. This implies that the invasion of drought, sand, and wind tolerant species reduced the competitive capacity and dominance of native plants, increasing the likelihood of their ultimate disappearance due to erosion. *Stipa krylovii* is often the dominant species in natural steppe sites. However, in the Sonid-you-qi area, it was not present at natural or long-term abandoned sites but was present at recently abandoned sites. This was tentatively explained by suggesting that the species composition at the recently abandoned site may have been different to that at the other sites in the area prior to the start of cultivation. However, with the exception of the slope, there were no significant differences between the properties of these sites (see **Table 4-1**). The slope at a site that had been abandoned for 8 years was 5° while that at a site that had been abandoned for 21 years was 1°. The difference between the slopes is small and would not be sufficient to explain the proposed difference in initial vegetation cover. It was therefore assumed that the difference was due to erosion.

In drier areas, the recovery of abandoned land is slow in ecological terms (Zambel *et al*. 2000). Based on my species composition data and DCA, I found that the first ordination axis was closely related to the time since abandonment in all of the studied areas. As the time since a site's abandonment increased, its species composition became more similar to that for natural grassland. This conclusion was corroborated by calculated similarity index values (**Fig. 4-7**). Thus, at all sites other than those in the Sonid-you-qi area, the vegetation in long-term abandoned plots had fully or almost fully reverted to that found on natural grassland. More recently abandoned sites were well separated from natural and long-term abandoned sites on the DCA biplot (**Fig. 4-6**). Although vegetation destroyed by human activities typically begins to recover within ten years of the activity's cessation (Cheng and Nakamura, 2007; Park *et al*., 2013), the vegetation community at a site that had been abandoned for 21 years site in the Sonid-you-qi area had not yet recovered. This may indicate that recovery after the heavy erosion experienced at this site will be slow or impossible.

In semiarid grassland, soil erosion caused by strong winds is the primary process of desertification and leads to losses of soil particle including clay and organic matter. High soil clay and organic matter content correlated significantly and positively with natural sites but negatively with erosion rates based on CCA biplots (**Fig. 4-8**). This finding is consistent with previous reports discussing the mechanism of ^{137}Cs adsorption and its movement during soil erosion (Ritchie and McHenry, 1990; Walling and Quine, 1991; Walling 1998). The pH and soil erosion rate correlated significantly with recent abandonment in each area (**Fig. 4-8**). In general, soils with pH values between 6 and 7.5 are suitable for plant growth and microbial activity (Barbercheck *et al*., 2009). At the natural sites examined in this work, the soil pH was within the optimal range for plant growth. However, at the abandoned field sites, the originally neutral soil pH had increased due to cultivation, yielding slightly alkaline conditions. This may be an important factor that contributes to land disturbance on the steppe (Shinchilelt *et al*., 2013).

In this investigation, the ^{137}Cs technique was used to characterize soil erosion at disturbed and undisturbed sites, revealing that vegetation degradation and restoration are clearly linked to soil losses. Similar results have been obtained in previous studies, which found that ^{137}Cs is concentrated in the uppermost 5 cm of undisturbed soil and that activity decreases exponentially with depth. One flaw of the study reported herein was that soil samples were only collected to a depth of 5 cm at the reference sites; ideally, I would have determined the distribution of ^{137}Cs at depths of up to 20 cm at the reference sites as well as the disturbed ones.

Following disturbances, vegetation communities adjust to the new soil conditions at the site, after which changes in species distribution may occur. Vegetation that was destroyed by cultivation may initially recover quickly. However, the physicochemical properties of the soil have a profound impact on vegetation communities and recover much more slowly. My results indicate that soils at formerly cultivated sites in the studied regions have higher pH and EC values than undisturbed soils and are generally alkaline. The conversion of grassland into cropland promotes desertification in arid and semiarid steppe regions. Therefore, quantitative analyses of the interactions between soil erosion rates, changes in vegetation cover, and soil properties can provide vital insights into the ecological condition of the steppe. Such data are also valuable for planning restoration projects and assessing their success. Further research will be required in order to obtain quantitative data of this sort for other regions of Inner Mongolia. In addition, it will be important to study the biotic and abiotic properties of the soil, which will enable a more comprehensive and accurate characterization of its properties and their impact on desertification.

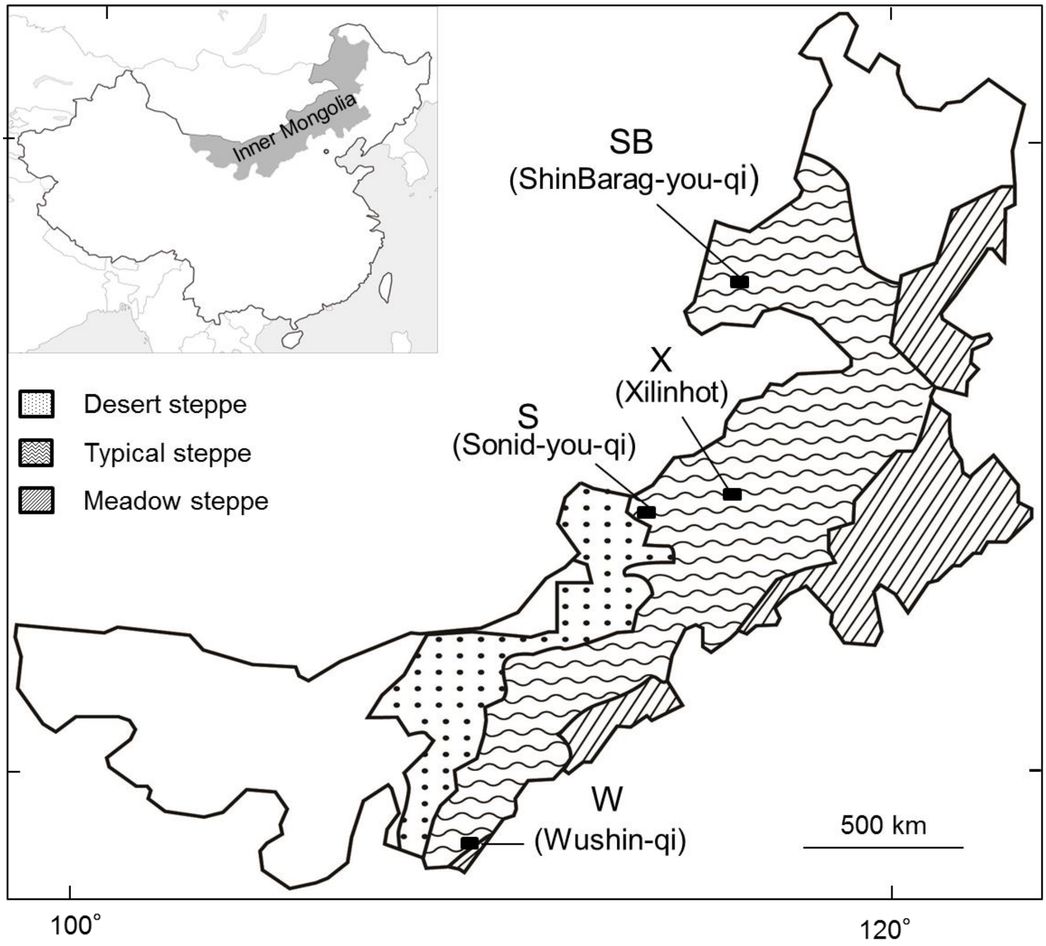

Fig. 4-1 Map of the study area and study sites with their locations in Inner Mongolia

Table 4-1 Summerized information for studied sites in the Inner Mongolian steppe

Study sites	Abandoned years	Latitude	Longitude	Altitude (m)	Slopes (°)	Annual mean temperature (℃)*	Annual precipitation (mm)*	Dominant species	cultivated years
W	Natural	38 09'N	108 36'E	1252	1	8.9	338.3	*Leymus secalinus*	—
	30	38 08'N	108 37'E	1246	1	8.9	338.3	*Leymus secalinus*	2
	13	38 08'N	108 39'E	1240	-	8.9	338.3	*Salsola collina*	4
X	Natural	43 54'N	116 25'E	1189	-	3.1	287.8	*Stipa grandis*	—
	34	43 55'N	116 25'E	1204	3	3.1	287.8	*Stipa grandis*	3
	11	43 53'N	116 21'E	1186	2	3.1	287.8	*Stipa grandis*	12
SB	Natural	49 14'N	116 55'E	727	2	1.4	249.0	*Stipa krylovii*	—
	25	48 37'N	117 01'E	548	2	1.4	249.0	*Stipa krylovii*	2
	5	48 38'N	117 00'E	550	-	1.4	249.0	*Artemisia eriopoda*	3
	1	48 38'N	116 57'E	545	-	1.4	249.0	*Eragrostis pilosa*	7
S	Natural	42 08'N	113 03'E	1307	-	4.7	241.0	*Stipa breviflora*	—
	21	42 02'N	113 08'E	1389	1	4.7	241.0	*Agropyron cristatum*	2
	8	42 00'N	113 10'E	1438	5	4.7	241.0	*Elymus duhuricus*	6

SB: ShinBarag-you-qi; X: Xilinhot; S: Sonid-you-qi; W: Wushin-qi. *Data(1970-2010) provided by http://climate.geog.udel.edu/~climate/html_pages/download.html

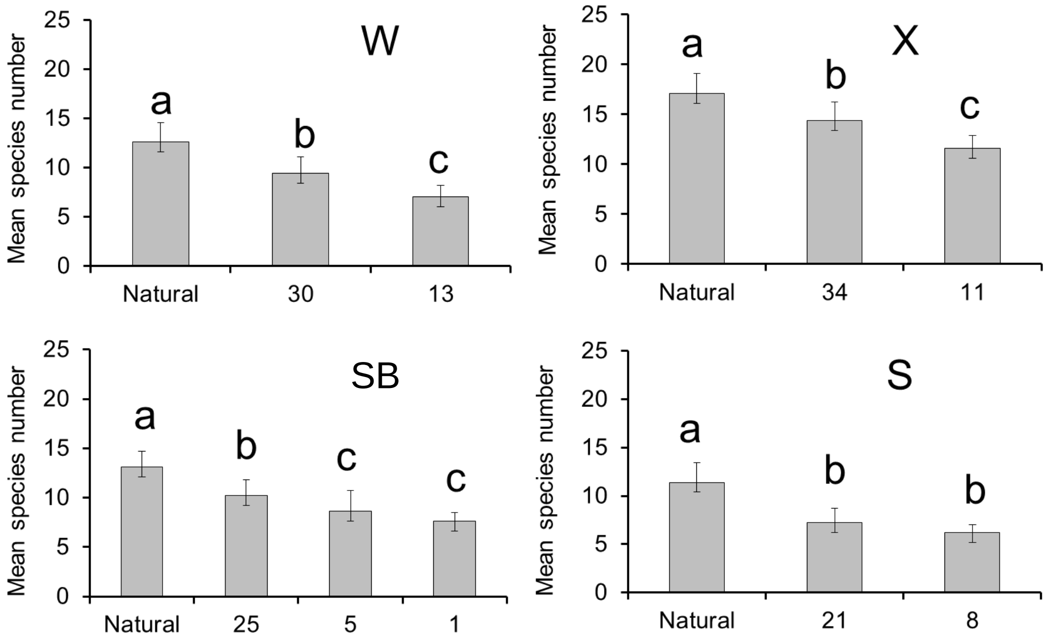

Fig. 4-2 Mean number of species relation to the cropland abandoned gradient. Different alphabetical in parentheses indicate a significant difference (Tukey test, P < 0.05).

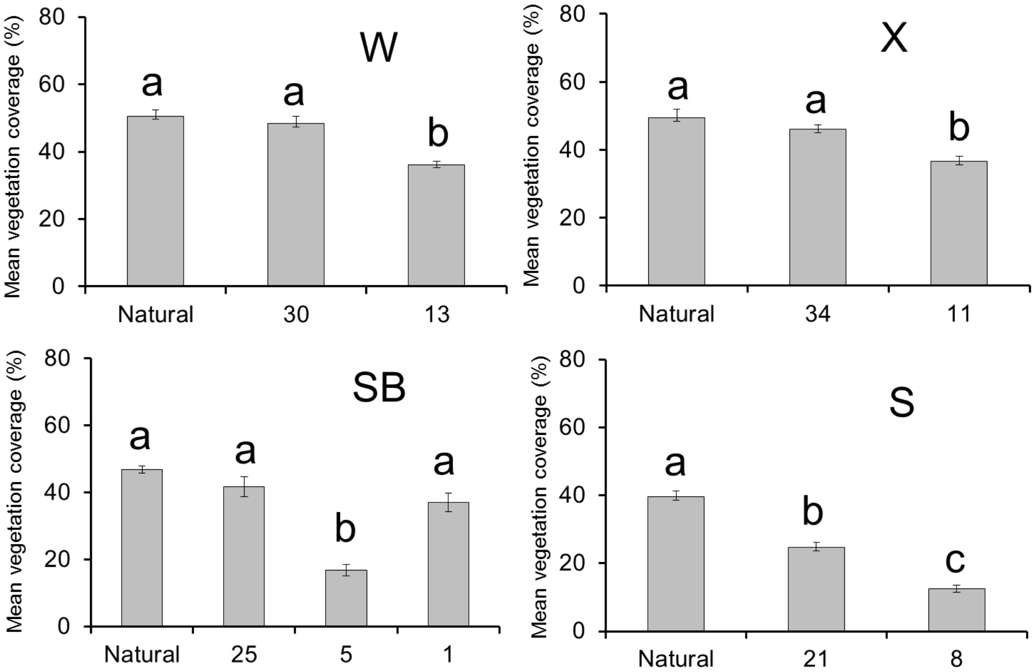

Fig. 4-3 Mean vegetation coverage (%) relation to the cropland abandoned gradient. Different alphabetical in parentheses indicate a significant difference (Tukey test, P < 0.05).

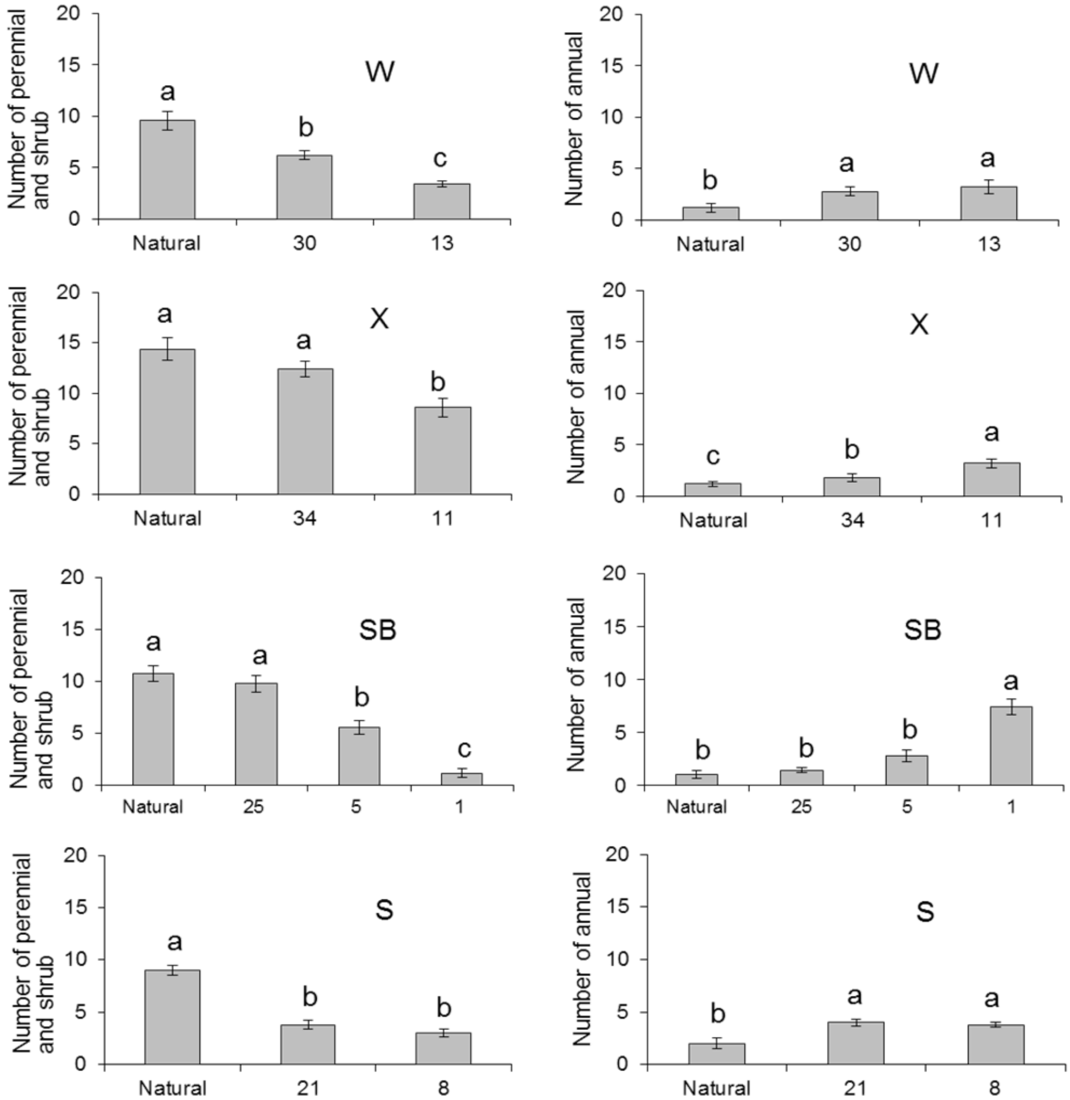

Fig. 4-4 Number of perennial, shrub and annual species relation to the cropland abandoned gradient. Different alphabetical in parentheses indicate a significant difference (Tukey test, $P < 0.05$).

Table 4-2 General species composition and each species SDR_3 of the grasslands in non-cultivated and abandoned cultivated fields in ShinBarag-you-qi

Species	Life-forms	SB(Natural)	SB(25)	SB(5)	SB(1)
Salsola collina	Annual	8.6 (c)	11.9 (c)	43.8 (b)	57.9 (a)
Stipa krylovii	Perennial	100.0 (a)	59.3 (b)	32.2 (c)	
Leymus chinensis	Perennial	46.6 (a)	44.2 (a)	47.3 (a)	
Cleistogenes squarrosa	Perennial	13.3 (c)	26.4 (b)	38.4 (a)	
Carex korshinskyi	Perennial	11.4 (c)	29.9 (b)	51.6 (a)	
Potentilla bifurca	Perennial	7.5 (b)	9.3 (b)	15.0 (a)	
Astragalus galactites	Perennial	15.8 (a)	9.0 (b)		8.5 (b)
Chenopodium acuminatum	Annual		7.0 (c)	20.1 (b)	59.0 (a)
Artemisia frigida	Shrub	21.5	51.2		
Allium polyrhizum	Perennial	8.2	11.0		
Ptilotricum canescens	Shrub	6.7	9.0		
Allium tenuissimum	Perennial	8.4		18.1	
Euphorbia esula	Perennial	4.9			10.6
Artemisia eriopoda	Perennial		46.3	67.3	
Setaria viridis	Annual			52.6	52.1
Eragrostis pilosa	Annual			13.3	71.5
Agropyron cristatum	Perennial	18.1			
Festuca dahurica	Perennial	9.8			
Bupleurum angustissimum	Perennial	8.6			
Melissitus ruthenica	Perennial	7.2			
Thermopsis lanceolata	Perennial	6.7			
Caragana stenophylla	Shrub	6.2			
Allium ramosum	Perennial		43.1		
Silene angustifolia	Perennial		25.5		
Allium anisopodium	Perennial		17.4		
Iris tenuifolia	Perennial		11.6		
Achnatherum splendens	Perennial			31.2	
Polygonum angustifolium	Perennial			29.9	
Potentilla acaulis	Perennial			19.0	
Hypecoum erectum	Annual			9.0	
Erodium stephanianum	Annual				54.8
Calystegia haderacea	Annual				49.7
Artemisia annua	Annual				33.1
Polygonum bungeanum	Annual				27.2
Artemisia pectinata	Annual				15.3
Chenopodium aristatum	Annual				12.1
Suaeda glauca	Annual				5.5

*Vacant places in the table indicate that the plant was not found in the corresponding plot.
**The letters indicate the significant differences of SDR, values between the areas of the species. (the species that appeared three or more in the plot, were tested by Tukey's test, $p < 0.05$).

Table 4-3 General species composition and each species SDR_3 of the grasslands in nature steppe and abandoned cultivated fields in Xilinhot

species	Life-forms	X(Natural)	X(34)	X(11)
Stipa grandis	Perennial	100.0 (a)	82.4 (b)	76.2 (b)
Leymus chinensis	Perennial	57.7 (a)	53.0 (a)	28.4 (b)
Cleistogenes squarrosa	Perennial	30.1 (b)	43.2 (a)	44.5 (a)
Heteropappus altaicus	Perennial	27.8 (a)	25.9 (a)	22.9 (a)
Phlomis umbrosa	Perennial	27.0 (c)	45.8 (a)	36.4 (b)
Carex korshinskyi	Perennial	26.4 (b)	35.3 (a)	39.9 (a)
Artemisia frigida	Shurb	16.6 (b)	29.7 (a)	27.8 (a)
Chenopodium glaucum	Annual	16.3 (b)	19.4 (b)	26.7 (a)
Agropyron cristatum	Perennial	32.0	29.0	
Melilotoides ruthenica	Perennial	25.0	24.3	
Potentilla bifurca	Perennial	22.2	32.6	
Allium bidentatum	Perennial	21.7	27.7	
Thalictrum petaloideum	Perennial	19.1	42.9	
Cymbaria dahurica	Perennial	12.8	19.3	
Allium senescens	Perennial	9.6	16.2	
Sibbaldia adpressa	Perennial	8.2	15.2	
Artemisia sieversiana	Annual	17.8		41.8
Inula britanica	Perennial	15.0		30.8
Geranium pratense	Perennial	11.4		13.8
Potentilla anserina	Perennial	3.5		9.8
Anemarrhena asphodeloides	Perennial		24.3	11.2
Aristida adscenionis	Annual		8.8	15.9
Kochia prostrata	Shurb	12.1		
Saposhnikovia divaricata	Perennial	9.7		
Vicia cracca	Perennial	9.4		
Potentilla tanacetifolia	Perennial	3.0		
Bupleurum scorzonerifolium	Perennial		25.1	
Allium condensatum	Perennial		22.2	
Serratula centauroides	Perennial		16.9	
Filifolium sibiricum	Perennial		16.9	
Caragana microphylla	Shurb		16.4	
Haplophylum dauricum	Perennial		10.9	
Artemisia pectinata	Annual		6.9	
Lappula myosotis	Annual			46.4
Caragana stenophylla	Shurb			37.6
Salsola collina	Annual			29.7
Artemisia eriopoda	Perennial			12.1

Table 4-4 General species composition and each species SDR_3 of the grasslands in nature steppe and abandoned cultivated fields in Sonid-you-qi

species	Life-forms	S(Natural)	S(21)	S(8)
Agropyron cristatum	Perennial	33.7	100.0	
Neopallasia pectinata	Annual	14.7	17.3	
Iris lactea	Perennial	11.6	31.8	
Salsola collina	Annual	20.3		22.2
Cleistogenes squarrosa	Perennial	17.7		31.1
Caragana microphylla	Shrub	10.4		53.3
Elymus dahuricus	Perennial		55.3	76.5
Eragrostis pilosa	Annual		54.1	18.3
Corispermum declinatum	Annual		31.0	52.9
Lespedeza bicolor	Shrub		16.3	9.8
Tribulus terrestris	Annual		11.1	71.1
Stipa breviflora	Perennial	86.3		
Artemisia frigida	Shrub	44.6		
Asparagus gobicus	Perennial	34.9		
Peganum nigellastrum	Perennial	25.5		
Kochia prostrata	Perennial	24.4		
Allium polyrhizum	Perennial	21.7		
Euphorbia humifusa	Annual	21.6		
Caragana stenophylla	Shrub	17.6		
Bassia dasyphylla	Annual	14.3		
Hippolytia trifida	Shrub	11.3		
Allium mongolicum	Perennial	9.5		
Lappula myosotis	Annual		45.9	
Artemisia eriopoda	Perennial		30.7	
Xanthium sibiricum	Annual		12.1	
Inula britanica	Perennial		11.1	
Echinochloa crusgalli	Annual		11.1	
Stipa krylovii	Perennial			40.8
Chenopodium album	Annual			11.0

Table 4-5 General species composition and each species SDR_3 of the grasslands in nature steppe and abandoned cultivated fields in Wushin-qi

	Life-forms	W(Natural)	W(30)	W(13)
Leymus secalinus	Perennial	100.0 (a)	98.3 (a)	50.3 (b)
Stipa glareosa	Perennial	36.6 (c)	49.1 (b)	53.7 (a)
Oxytropis tannis	Perennial	12.4 (b)	18.7 (a)	9.9 (b)
Artemisia annua	Annual	40.2	81.2	
Artemisia frigida	Shrub	37.8	32.4	
Astragalus adsurgens	Perennial	37.5	28.9	
Carex duriuscula	Perennial	27.4	23.0	
Poa annua	Annual	18.2	18.4	
Oxytropis glabra	Perennial	16.6	38.2	
Thymus serpyllum	Shrub	16.3	18.7	
Achnatherum splendens	Perennial	43.4		35.9
Salsola collina	Annual		22.4	85.0
Caragana pygmaea	Shrub		41.5	59.0
Heteropappus tataricus	Annual		37.1	31.1
Lepidium apetalum	Annual		15.2	18.9
Agropyron dasystachys	Perennial	22.9		
Cleistogenes squarrosa	Perennial	20.7		
Messerschmidia sibirica	Perennial	14.0		
Lespedeza potaninii	Shrub	11.3		
Glaux maritima	Perennial	8.4		
Taraxacum dealbatum	Perennial	7.6		
Swainsonia salsula	Perennial	5.3		
Artemisia capillaris	Shrub		45.8	
Potentilla multicaulis	Perennial		38.4	
Potentilla anserina	Perennial		37.9	
Sonchus oleraceus	Annual		10.4	
Artemisia scoparia	Annual			28.8
Allium tenuissimum	Perennial			20.4
Sonchus arvensis	Annual			17.0

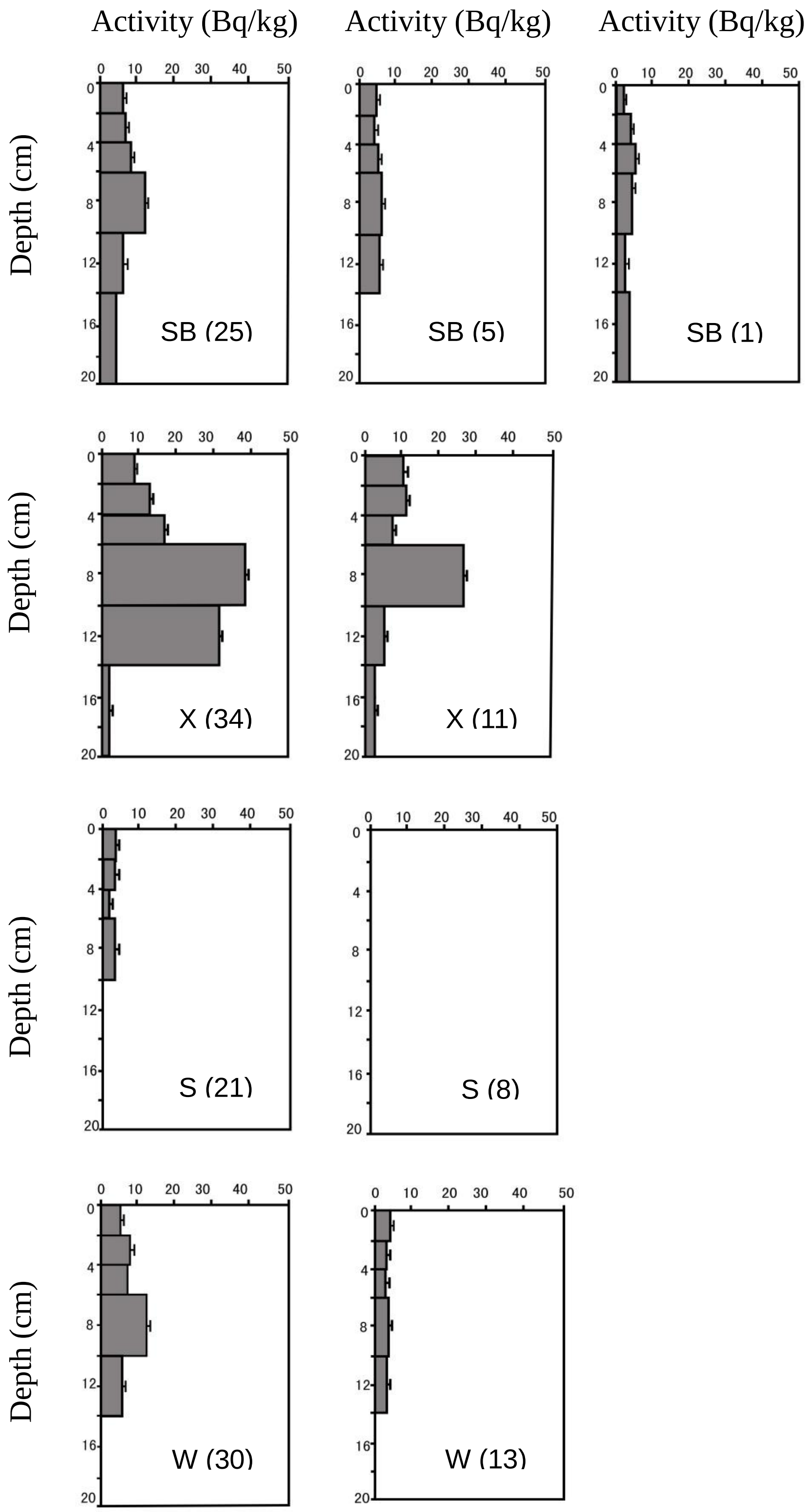

Fig. 4-5 Vertical distribution of ^{137}Cs activities in abandoned cultivated soils at sample sites. SB: ShinBarag-you-qi; X: Xilinhot; S: Sonid-you-qi; W: Wushin-qi. The number in () represents the abandoned period of each cultivated sites.

Table 4-6 Calculated ^{137}Cs inventory conversion and ^{137}Cs loss (%) in each site

sites	Abandoned year	^{137}Cs inventory (Bq/m^{-2})	^{137}Cs loss (%)
W	Natural	942.4	standard
	30	343.3	63.5
	13	320.3	66.0
X	Natural	1702.4	standard
	34	790.6	53.6
	11	786.2	53.8
SB	Natural	909.1	standard
	25	643.3	29.2
	5	257.8	72.3
	1	268.9	70.4
S	Natural	788.0	standard
	21	144.7	81.6
	8	0.0	100.0

Table 4-7 Change to soil physicochemical properties across the field trial in the each site

Sites	Abandoned years	Clay (%)	Silt (%)	Sand (%)	OC (g/kg^{-1})	TN (g/kg^{-1})	Av-P ($mg\ kg^{-1}$)	pH (H_2O)	EC (dS/m)
W	Natural	12.7 ± 0.6 a	11.3±1.5 a	76.0 ± 5.9 a	11.29 ± 2.01 a	1.16 ±0.21 a	12.63 ± 1.20 a	8.3 ± 0.3 a	1.37 ± 0.11 a
	30	9.5 ± 0.6 b	9.7 ± 0.6 a	80.8 ± 4.2 a	8.57 ± 2.24 a	0.82 ± 0.35 a	11.17 ± 1.65 a	8.9 ± 0.3 b	1.46 ± 0.34 a
	13	4.8 ± 0.8 c	14.7 ± 2.7 b	80.5 ± 4.2 a	5.67 ± 1.57 b	0.54 ± 0.19 b	6.31 ± 1.04 b	8.9 ± 0.2 b	2.35 ± 0.67 b
X	Natural	27.5 ± 1.8 a	12.4 ± 1.3 a	60.2 ± 4.1 a	13.83 ±2.61 a	1.42 ± 0.25 a	15.86 ± 1.90 a	7.2 ± 0.5 a	0.50 ± 0.08 a
	34	22.4 ± 1.3 b	14.4 ± 1.1 b	63.3 ± 4.6 a	11.55 ± 1.53 b	1.07 ± 0.14 b	13.50 ± 1.39 a	8.0 ± 0.4 b	1.16 ± 0.07 b
	11	20.9 ± 2.6 c	10.4 ± 1.4 bc	68.6 ± 4.2 b	11.11 ± 0.57 b	1.08 ± 0.03 b	13.32 ± 1.52 a	8.2 ± 0.2 b	0.96 ± 0.01 b
SB	Natural	16.9 ± 0.3 a	12.1 ± 0.7 a	71.0 ± 3.1 a	23.47 ± 1.83 a	2.40 ± 0.25 a	23.66 ± 4.24 a	7.1 ± 0.4 a	0.43 ± 0.07 a
	25	17.8 ± 1.1 a	9.4 ± 1.3 b	72.8 ± 4.3 a	9.16 ± 1.51 b	0.84 ± 0.12 b	12.18 ± 1.52 b	8.2 ± 0.4 b	0.73 ± 0.02 b
	5	11.5 ± 0.4 b	7.1 ± 0.5 c	81.3 ± 6.1 b	7.23 ± 0.64 c	0.65 ± 0.06 b	9.51 ± 1.51 c	8.6 ± 0.2 b	1.26 ± 0.22 c
	1	12.9 ± 0.5 c	5.7 ± 1.4 c	81.3 ± 4.1 c	6.55 ± 0.96 c	0.60 ±0.09 b	11.85 ± 1.15 b	7.7 ± 0.3 ac	0.43 ± 0.06 a
S	Natural	12.5 ± 1.1 a	10.7 ± 0.6 a	76.8 ± 4.9 a	7.35 ± 3.04 a	0.78 ± 0.24 a	8.43 ± 0.80 a	7.7 ± 0.4 a	0.65 ± 0.01 a
	21	10.8 ± 0.9 a	8.1 ± 2.4 a	81.1 ± 3.3 a	3.89 ± 0.69 b	0.36 ±0.05 b	5.74 ± 1.14 b	8.6 ± 0.2 b	0.86 ± 0.04 b
	8	7.6 ± 1.1 b	7.3 ± 2.4 a	85.2 ± 8.8 b	2.64 ± 0.30 c	0.24 ± 0.07 b	3.42 ± 0.83 c	8.8 ± 0.2 b	0.87 ± 0.05 b

* Mean (± SE) values compared with Tukey-test, the significant differences are indicated by different letters. SB: ShinBarag-you-qi; X: Xilinhot city;
S: Sonid-you-qi;W: Wushin-qi; SB-1, X-1, S-1 and W-1 were non-cultivated sites; SB-2 to 4, X-2 and 3, S-2 and 3, W-2 and 3 were cultivation abandoned sitis

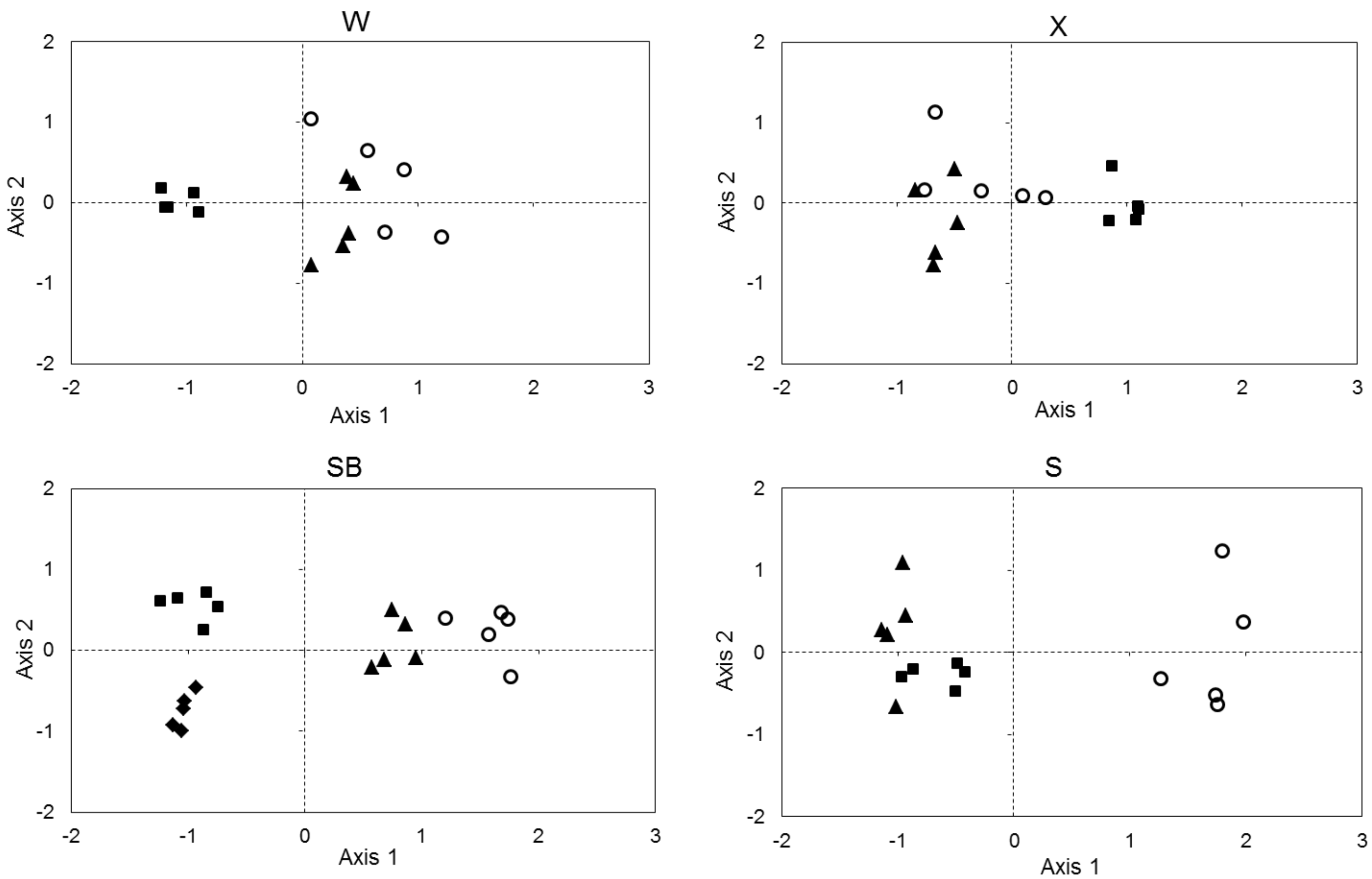

Fig. 4-6 Detrended correspondence analysis (DCA) based on frequency of plant species at the four research sites along the period of crop abandonment. ○: site of nature steppe, ▲: abandoned -for- 21 ~ 34 years sites, ■: abandoned –for- 5 ~ 13 years sites and ♦ : abandoned –for- 1 year site.

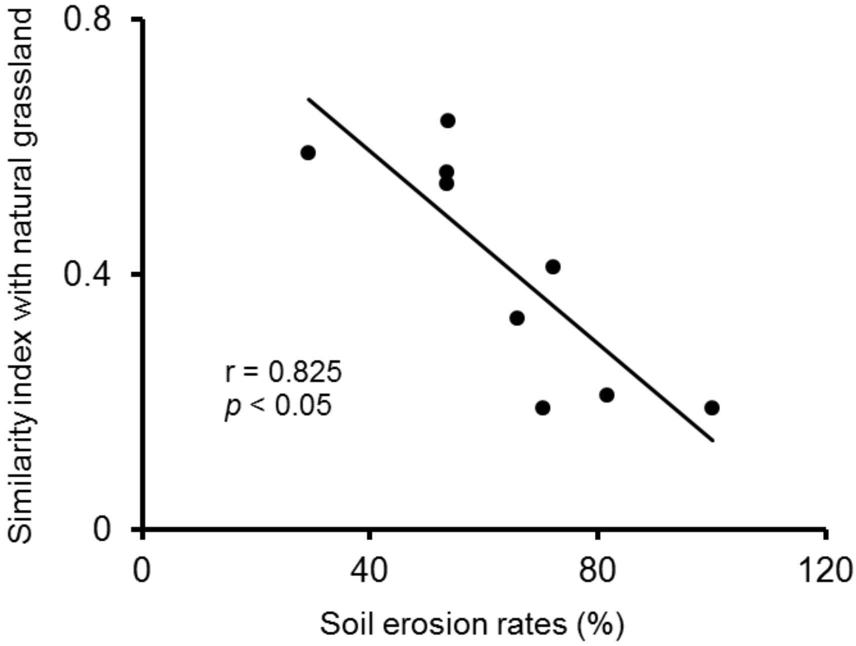

Fig. 4-7 Relationships between the Sorensen similarity indexes and soil erosion rates in each site.

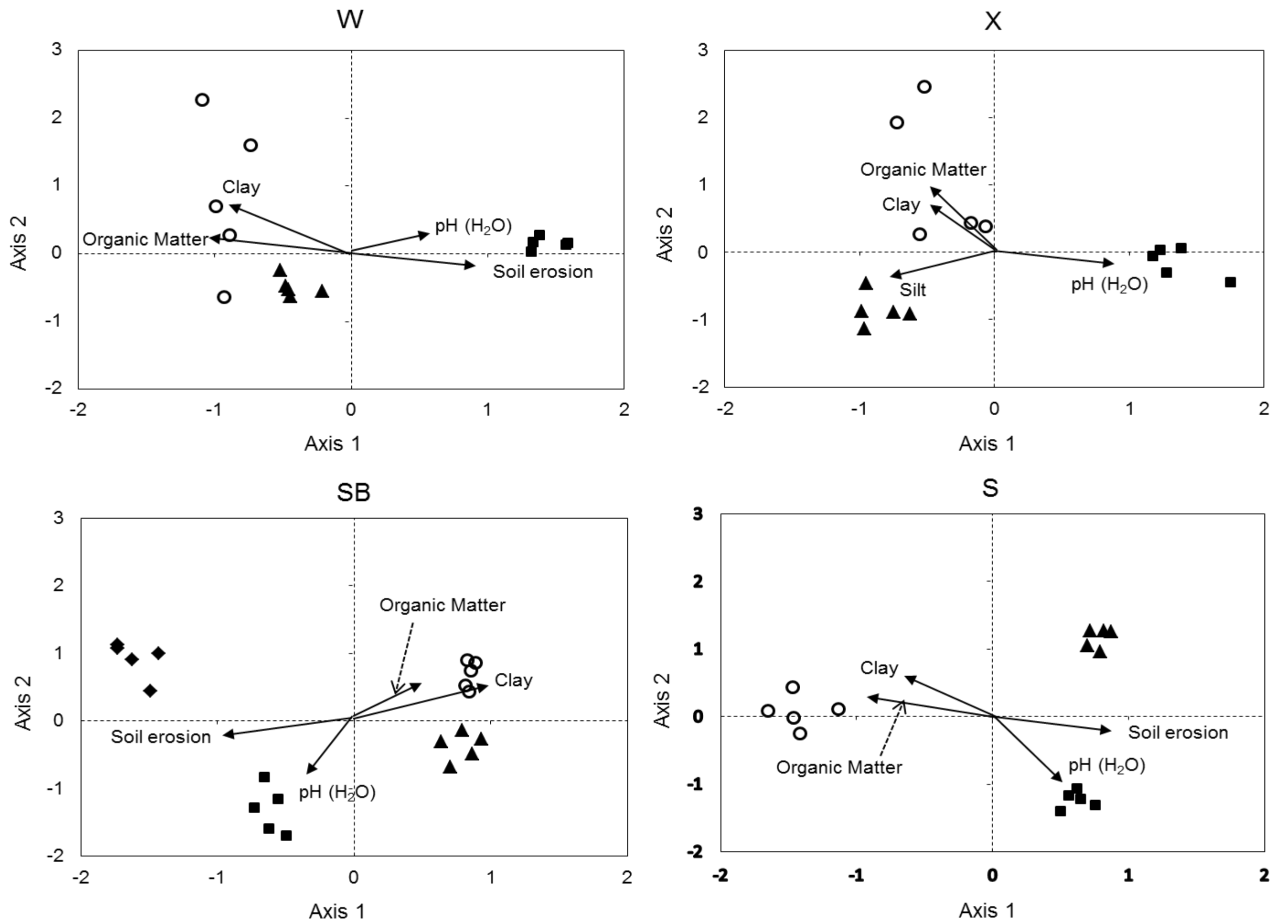

Fig. 4-8 CCA analysis, different land use sites with their influence factors on the standard two axes ($p < 0.05$). ○: site of nature steppe, ▲: abandoned -for- 21 ~ 34 years sites, ■: abandoned –for- 5 ~ 13 years sites and ♦ : abandoned –for- 1 year site.

Chapter 5

Soil microbial properties of abandoned cropland and their effects on vegetation in the steppe of HulunBuir, Inner Mongolia

5-1 Introduction

Chapter 4 described the effects of soil erosion rates effects on vegetation recovery in former steppe croplands. The above- and below-ground components of terrestrial ecosystems are strongly linked via diverse direct and indirect interactions (Kardol and Wardle, 2010). Plants provide carbon sources and other nutrients for the soil fauna and microflora (Habekost *et al*., 2008); in turn, the soil biota (particularly its microbiota) decompose soil organic matter, stabilize the soil structure and, via its essential role in the cycling of elements, release nutrients for plant growth (Porazinska *et al*., 2003). In addition, soil abiotic factors, such as organic matter content (Kara *et al*., 2008), salinity (Lozupone and Knight, 2007) and pH (Fierer and Jackson, 2006) have been interpreted as drivers of the vegetation patterns observed in terrestrial ecosystems. Therefore, in order to understand ecosystem restoration and sustainability, it is necessary to study the interactions between above- and belowground components in a holistic way. Some studies have demonstrated that differences in aboveground plant species diversity, plant richness and plant biomass can have significant effects on soil microbial processes, microbial biomass, and the colony-forming units of major soil microorganisms (Hooper and Vitousek, 1998; Malý *et al*., 2000; Wardle and Nicholson, 1996). Most of these studies have been conducted in forest (Christine *et al*., 2001), grassland (Steenwerth *et al*., 2003) and cropland (Chen *et al*., 2012) ecosystems, and relatively little is known about how these interactions affect abandoned cropland ecosystems (Zhang *et al*., 2012), especially, in Eurasian steppes located in arid and semi-arid regions. While the investigations discussed in previous chapters provided some insights into these ecosystems, they did not address the impact of soil microbes. Therefore, the studies presented in this chapter were conducted to investigate the soil microbial diversity and function in such sites, to enable a more accurate evaluation of their soil condition. The General introduction to this thesis (Chapter 1) provides a detailed overview of the roles of biotic and abiotic factors in soil processes (Chapter 1).

In this Chapter, I selected different crop abandoned area to assess the effects of the soil biotic and abiotic factors on vegetation recovery.

5-2 Materials and methods

5-2-1 Site description

The study area is located in the HulunBuir grassland (115°31′–126°04′ E, 47°05′–53°20′ N) in northeastern Inner Mongolia, China. I established 4 sites that consist of a series of 3 abandoned croplands and 1 light-grazing steppe grassland (**Fig. 5-1**). The 3 croplands were abandoned 1, 5, and 25 years ago (AC-1, site 1; AC-5, site 2; and AC-25, site 3), and site 4 is lightly grazed native grassland (AC-non). In the AC-1, AC-5, and AC-25 sites, corn, sunflower, and *Elymus cylindricus* had been rotated for about 40 years, after which the sites were abandoned due to degradation (declines in fertility).

The coordinates and elevations of the sampled sites are as follows: AC-non, 49° 10′ 58″ N, 116° 55′ 50″ E, 720 m; AC-1, 48° 38′ 43″ N, 116° 57′ 56″ E, 545 m; AC-5, 48° 38′ 50″ N, 117° 00′ 48″ E, 550 m; and AC-25, 48° 38′ 45″ N, 117° 01′ 56″ E, 545 m. The mean temperature and precipitation from 1970 to 2010 for each site (26) were as follows: AC-non, 0.7 ℃ and 249 mm; AC-1, AC-5, and AC-25, 1.4 ℃ and 222.5 mm. Therefore, mean temperature and precipitation were not significantly different between sites.

5-2-2 Sampling design and data collection

5-2-2-1 Vegetation analysis

Plant surveying and soil sampling were conducted in August 2010. All of the sites were selected for their similar topography (flat). Each site contained 3 plots and 15 replicate sub-plots: site 1, sub-plots 1-15; site 2, sub-plots 16-30; site 3, sub-plots 31-45; and site 4, sub-plots 46-60. The plots were established as follows. First, a 50 m transect was created, and homogenous sample sub-plots (quadrat, 1 m^2) were established at 10 m intervals along it; these plots were termed transect plots. In addition, at every fifth transect plot, two additional sub-plots (quadrat, 1 m^2) were established by moving a random distance to either side of the transect. These plots were termed random plots one and two and are considered to be true replicates because the distances between them exceeded the spatial dependence of most soil chemical and microbial variables (Mariotte *et al.*, 1997). In each plot, the species composition was recorded. Plant communities were classified on the basis of their differential species; all species were identified and measured for cover, height, and density, and Shannon's diversity index was calculated. Soil moisture was measured with a TRIME-FM (Ettlingen, Germany). Above-ground plant biomass was also determined by clipping the plants at ground level, sorting by species, drying at 60 ℃ for 48 h, and weighing the samples.

The expressions used to calculate the estimated biomass, percentage of vegetation cover and diversity index for each site are presented in Chapter 4.

5-2-2-2 Soil chemical analysis

Soil samples were collected from the upper 10cm of the soil profiles in each plot at the cultivated and non-cultivated sites, and the non-cultivated site was used as the reference site. After carefully removing the surface organic materials and fine roots, each mixed sample was divided into 2 parts. One part was air-dried for the analysis of soil physicochemical properties. The other was sifted through a 2 mm sieve, sealed in sample vials, kept on ice for transport to the laboratory, and stored at -30 ℃ for microbial assays. Three of these samples were selected at random and pooled to form a single sample. The number of samples after pooling was as follows: site 1, samples a1 – a5; site 2, samples b1 – b5; site 3, samples c1 – c5; and site 4, samples d1 – d5.

The concentrations of NO_3-N and NH_4-N in KCl extracts of the soil samples were determined by the zinc reduction-naphthylethylenediamine method for NO_3-N (Leonardo, 2009) and the indophenol blue colorimetric method for NH_4-N (Motsara and Roy 2008). The soil phosphorus content was determined by the Truog method (Truog and Meyer 1929). The methods for measuring OC and T-N, pH, EC, soil moisture and soil hardness are as discussed in Chapter 3.

5-2-2-3 Soil Microbiological analysis

I investigated the molecular ecology of the total bacterial communities in soils from steppe grasslands and croplands that had been abandoned for different periods (1, 5, and

25 years) and compared the degree of recovery to the "natural" state in each case. Quantitative PCR (qPCR) was used to analyze the 16S rRNA genes present in each sample in order to study the total bacterial community. Bacteria can be detected by extracting nucleic acids directly from soil samples and identifying the nucleotide sequences of PCR-amplified 16S rRNA genes (Ward *et al*., 1992). The microbial properties of the soil in terms of the diversity and quantity of bacteria it contained were determined by denaturing gradient gel electrophoresis (DGGE) and real-time PCR methods.

DNA extraction and PCR: DNA from each of the 20 samples (4 sites × 5 replicates) was extracted from 3 subsamples of 0.5 g soil each using the FastDNA Spin Kit for soil (MP Biomedicals, Illkirch, France) according to the manufacturer's protocol. The quality and quantity of the DNA extracts were checked with a SmartSpec Plus spectrophotometer (Bio-Rad Laboratories, United States). The samples were pooled and stored at -20 ℃ until use.

Fragments of approximately 200 bp, corresponding to the V3 region of the 16S rRNA gene (Muyzer *et al*., 1993), were amplified using a reaction mixture that contained the reagent mixture described above with the addition of 0.25 μL of 100 μM 357F-GC (CCTACGGGAGGCAGCAG-GC-clamp) and 0.25 μL of 100 μM 518R (GTATTACCGCGGCTGG). Amplification was achieved using a touchdown thermocycling program (Muyzer *et al*., 1993). All of the PCR amplicons were electrophoresed on an agarose gel to ascertain their sizes and purified using the UltraClean PCR Clean-Up Kit (MO BIO Laboratories, Carlsbad, CA, United States).

DGGE: DGGE was performed using the D-Code system (Bio-Rad Laboratories, Hercules, CA, United States) as described by Baxter and Cummings (Baxter and Cummings, 2006). The polyacrylamide concentration of the gel was 8%, and the linear denaturing gradient was 30% to 60% (100% denaturant corresponds to 7 M urea and 40% deionized formamide). The gel was run at 36 V for 18 h at 60 ℃ in 0.5 × TAE buffer. It was then stained for 30 min with 1:10,000 (v/v) SYBR Gold, rinsed with 0.5 × TAE, and scanned on a transilluminator. Bands were identified, and relative intensities were calculated based on the percentage of intensity for each band in a lane. This was done using an image-analyzing system (Image Master; Amersham Pharmacia Biotech, Uppsala, Sweden). Shannon's diversity index (H') was calculated using the expression $H' = -\Sigma\, p_i \log_2 p_i$, where p_i is the ratio of the relative intensity of band i compared to that of the lane.

Real-time PCR assay: Reactions were set up using SYBR green (Bio-Rad Laboratories, the Netherlands) according to Baxter and Cummings (Baxter and Cummings, 2008) with the LightCycler 1.5 system (Roche Applied Sciences, Indianapolis, IN, United States). Reaction mixtures were heated to 95 ℃ for 15 min to denature the DNA before completing 40 cycles of denaturation (95 ℃ for 15 s [16S rRNA]), annealing (65 ℃ for 15 s [16S rRNA]), and extension (72 ℃ for 15 s [16S rRNA]). Soil DNA extracts were diluted 1:100 to prevent inhibition of PCR by soil contaminants (e.g., by co-extracted humic substances), and each run included triplicate reactions for each DNA sample, the standard curve, and the no template control. The average copy number was converted into copies of the gene per gram of soil. The 357F and 518R primers were used for total bacteria qPCR. Dilution series of pGEM-T Easy vector (Promega, Madison, WI, United States) DNA with cloned bacterial 16S rRNA gene fragments were used to generate standard curves ranging from 10^1 to 10^7 gene copies$\cdot\mu L^{-1}$ for DNA quantification. The specificity of the amplified products was checked by the observation of a single melting peak and the presence of a unique band of the expected size in a 2% agarose gel stained with ethidium bromide. The standard curve produced was linear ($r^2 > 0.98$), and the PCR efficiency (Eff = $10^{(-1/\mathrm{slope})} - 1$) was > 0.90.

5-2-3 Statistical analysis

The plant community types in each subplot were classified using two-way indicator species analysis (twinspan; Hill 1979). I divided the vegetation data until the second cut level (yielding three divisions as a dendrogram) and selected endpoints of divisional standards using a v2 test ($P < 0.05$). Indicator species for each group were then identified using indicator species analysis (inspan; Hill *et al*. 1975). An indicator criterion of $P < 0.05$ was applied in this analysis. The TWINSPAN and INSPAN analyses were performed using PC–ORD for Windows version 4.25 (MjM Software Design, Gleneden Beach, OR, USA).

Other statistical analysis was performed using SPSS statistical software package (version 19.0; SPSS, Inc., Chicago, IL, United States). In all tests, significant effects/interactions were those with a *P* value that was < 0.05. Multiple comparisons were performed using Tukey's test to compare differences in vegetation and soil parameters between plots. Differences between main effects were tested by analysis of variance (ANOVA).

The choice between a linear or unimodal species response model depends on the underlying gradient length, which is measured in standard deviation units along the first ordination axis and can be estimated by detrended correspondence analysis (DCA). It is recommended to use linear methods when the gradient length is < 3, unimodal methods when it is > 4, and any method for intermediate gradient lengths (Ter Braak and Smilauer, 2011). The DCA gradient length for *nifH* gene patterns was 1.77, and that for 16S rRNA patterns was 1.11. The linear species response model known as canonical correspondence analysis (CCA; Ter Braak, 1986) was therefore used to perform multivariate statistical analysis, identify patterns of species distribution, and correlate these patterns with environmental (soil physicochemical) variables. Canonical ordination is easier to apply and requires less data than regression. A Monte Carlo test (100 permutations) was used to analyze the significance of the first two canonical axes. These analyses were conducted using the R (Vegan) statistical package.

5-3 Results

5-3-1 Species composition

I identified 37 species (including 1 unknown) in the 60 plots. The plant communities were dominated by *Stipa krylovii* in AC-non, *Cleistogenes squarrosa* in AC-25, *Artemisia eriopoda* in AC-5 and *Chenopodium acuminatum* in the AC-1 site, respectively. Other high frequency species were *Cleistogenes squarrosa* (50%), *Leymus chinensis* (50%), *Stipa krylovii* (55%), *Carex korshinskyi* (60%) and *Salsola collina* (55%).

5-3-2 Classified groups and indicator species

Based on the TWINSPAN results, the 60 plots were classified into two categories in the first division and four groups in the second division (**Fig. 5-2**). There were a total of 13 species (including 1 unknown) in group 1. The indicator species based on the INSPAN analysis for group 1 were *Salsola collina, Setaria viridis, Chenopodium acuminatum, Eragrostis pilosa, Calystegia haderacea* and *Erodium stephanianum*. Group 2 contained 15 species in total, with the indicator species being *Artemisia annua*. Group 3 contained 16 species in total; its indicator species were *Artemisia frigida, Artemisia eriopoda, Cleistogenes squarrosa, Carex korshinskyi* and *Allium ramosum*. Group 4 included 10 species and *Stipa krylovii, Leymus chinensis* and *Agropyron cristatum* were its indicator species.

The relationship between land use history and group is shown in **Table 5-1**. Groups 1 and 2 consisted of cropland plots at sites that had been abandoned for one (AC-1) or five (AC-5) years, respectively. Group 3 comprised 11 plots at the cropland site which had been abandoned for 25 years (AC-25), while group 4 comprised 15 plots that had never been cultivated (AC-non) and 4 AC-25 plots.

5-3-3 Variation in vegetation parameters with years of abandonment

Table 5-2 shows how the aboveground vegetation changes as the number of years of abandonment increased. The lowest levels of both vegetation cover and biomass were observed in the croplands that had been abandoned for 5 years, while the cropland that had been abandoned for 25 years was very similar to uncultivated land. The diversity index (*H'*) increased from 1.6 in the 1-year abandoned cropland to 1.8 in the 25-year abandoned cropland, and then decreased to 1.6 in the uncultivated land. The percentage of perennial plants increased from 13.1 in the 1-year abandoned cropland to 89.5 in the uncultivated land. The highest levels of litter were found in the uncultivated AC-non site.

5-3-4 Variation in soil properties with years of abandonment

Some biotic and abiotic properties of the studied soils are listed in **Table 5-3**. Most parameters tended to return to their native levels during the progression from AC-1 to AC-non. However, the values of most parameters for the AC-5 site were higher than those at the other sites, and the C/N value for AC-5 was significantly higher than at the other sites. The pH for AC-5 was comparatively high and that at AC-non was comparatively low. In addition, the EC tended to increase from AC-1 to AC-25 and then decreased in AC-non.

5-3-5 Total bacterial diversity and qPCR analysis of the 16S rRNA gene

DGGE gels showing the diversity of total bacteria in the soil samples are shown in **Fig. 5-3**. The ANOVA results for Shannon's diversity index of the 16S rRNA gene indicated that the abandoned period ($P < 0.001$) was a significant factor. Therefore, each sample was analyzed separately by multiple comparison. Soils from AC-25 and AC-non (both almost equal, 3.17 ± 0.07) had higher levels of 16S rRNA diversity than those from AC-1 (2.8 ± 0.04) and AC-5 (3.06 ± 0.01) (**Fig. 5-4**).

The aim of the qPCR analysis was to compare the differences between each site rather than absolute quantification. There were significant differences between the total bacterial (16S rRNA) populations at each site ($P < 0.03$). Therefore, separate analyses were conducted for each site individually (**Fig. 5-5**). The copy numbers for the 16S rRNA gene at all four sites were similar, ranging from 4.4×10^7 to 1.2×10^8 copies g^{-1} of soil. However, the copy number at the AC-non site was higher than that for the other soils. There were no significant differences between the copy numbers for the different abandoned cropland sites (AC-1, AC-5, and AC-25), which ranged from 4.4×10^7 to 8.3×10^7 copies g^{-1} of soil (**Fig. 5-5**).

5-3-6 Relationship between soil properties and TWINSPAN groups

The results of the CCA are shown in **Fig. 5-6** and **Table 5-4**. The scores for the 60 quadrats are plotted along Axes 1 and 2, and divided into four groups based on the TWINSPAN clusters, which are superimposed on the plot. The four groups were labeled AC-1, AC-5, AC-25 and AC-non (**Fig. 5-3**).The CCA analysis was conducted to optimize the site scores. The correlations for sites and soil physicochemical properties were high for the first two canonical axes, explaining 53.5% of the cumulative variance of all edaphic parameters. In the biplot, significant effects/interactions were those with a P value that was < 0.05. The soil moisture and two of the microorganism-related variables explain a large proportion of the variation in the data set (**Table 5-4**).

Significant positive correlations were identified between the soil moisture, 16S rRNA gene amount and NH_4-N for Axis 1; these were primarily due to the AC-25 and AC-non sites. The NO3-N and 16S rRNA gene-*H'* correlated positively for Axis 2 and were mainly due to the AC-5y and AC-1y sites.

5-4 Discussion

Based on the TWINSPAN and INSPAN results, the study plots were classified into four vegetation groups associated with different land use patterns and time since abandonment (**Fig. 5-2** and **Table 5-1**). On going from sites that had been abandoned for 1 year to 5 years and then 25 years of abandonment, and finally non-cultivated grassland, there was a tendency for weed species to give way to typical grassland species, increases in the abundance of the Graminaceae and perennial species, and increases in vegetation coverage and biomass (**Fig. 5-2** and **Table 5-2**).

Kawada *et al*. (2011) described indicator species that can be used to quantify grassland degradation due to cropland abandonment in Inner Mongolia. In this work, the annual species *Chenopodium acuminatum* and *Artemisia eriopoda* were found to be dominant in recently (1 to 5 years ago) abandoned croplands. *Cleistogenes squarrosa*, which Kawada *et al*. (2011) identified as an indicator species of abandoned croplands, was dominant in long-term (25 years) abandoned cropland, while the feather species *Stipa krylovii* was dominant in natural grassland. This progression is consistent with the successional pattern of vegetation in grassland reported by Cheng and Nakamura (2007).

The soils at the different sites considered in this study were of the same type, but the water holding capacity of each soil differed. The soil moisture content decreased in the following order: AC-non > AC-25 > AC-5 > AC-1 (Table 5-3). The same progression was observed for the level of aboveground litter at each site, implying that the presence of litter has a strong effect on the rate of soil evaporation (Villegas *et al*. 2010). The permeability and water-holding capacity of the soil is determined by its balance of macropores and micropores (Cogger, 2000). In addition, the biological decomposition of organic materials produces natural glues that bind and strengthen soil aggregates (Cogger, 2000), and helps soils to hold water and nutrients; over time, this may change the relative abundance of macro- and micropores. Organic matter also is a long-term, slow-release storehouse of nitrogen, phosphorus, and sulfur (Cogger, 2000). Accordingly, organic matter had significant effects on the vegetation parameters (**Table 5-3**): the AC-5 site had low levels of OC, TN and NH_4-N as well as a low percentage of vegetation cover and biomass site (**Table 5-2**). This may have been because of the significant difference in soil organic matter content between the abandoned croplands and the non-cultivated site ($P < 0.05$) (**Table 5-3**). Additionally, the soil hardness at the AC-5 site differed significantly from that at the other sites (**Table 5-3**). This is also consistent with a difference in the soils' water-holding capacities. Previous studies indicated that pH and EC have important effects on the structure of the bacterial community. EC values between 0 and 1.5 dS $m^{-1}$and pH values between 6 and 7.5 are generally suitable for plant growth and microbial activity (Barbercheck *et al.*, 2009).

The distribution of 16S rRNA genes in the soil from the AC-25 site is more similar to that for the AC-non site than those for AC-1 or AC-5. This suggests that the function and diversity of bacterial communities of abandoned croplands recovers over time, to some extent. Cheng and Nakamura (2007) found that plant communities recovered over time in abandoned croplands, so it is perhaps not surprising that the bacterial community would do the same as the soil regained its original characteristics.

Larkin and Honeycutt (2006) reported that plant effects are the most important drivers of soil microbial community characteristics within a given site and soil type. The CCA revealed that biotic factors strongly influence the vegetation structure, and that gene copy numbers of total bacteria and diversity of total bacteria were positively correlated with AC-non or AC-25 (**Fig. 5-6**). This is caused by interactions between plants and microorganisms. Above-ground net primary productivity was expected to increase soil carbon input by enhancing the turnover of plant biomass and root exudation, and may therefore influence carbon-limited microbial communities in the soil (Niklaus *et al.*, 2003; Zak *et al.*, 2003). Concurrently, microorganisms also affect plants by producing canonical plant growth-regulating substances such as auxins or cytokinins (Ortiz-Castro *et al.*, 2009). Carney and Matson (2005) found that plant diversity had a significant effect on the microbial community composition that were reflected in alterations in microbial abundance rather than community composition.

Other researchers have also reported that plants are likely to have profound effects on bacterial communities via processes such as root exudation, which may have feedback effects on plant growth (Kennedy *et al*., 2004; Nunan *et al*., 2005). Thus, the results of this study suggest that advances in desertification can be prevented by adjusting environmental features of abandoned croplands such as the soil’s content of moisture and organic matter, which will enhance the functioning of its microbial community. This would support increased plant biomass production and expedite the recovery of the original ecosystem.

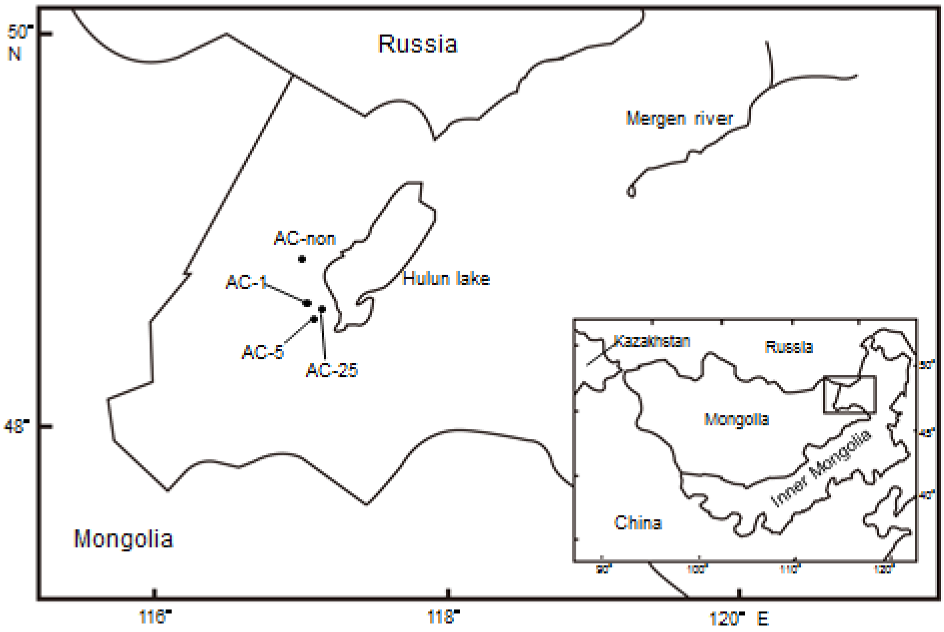

Fig. 5-1 Map of the study area and study sites with their locations in HulunBuir grassland

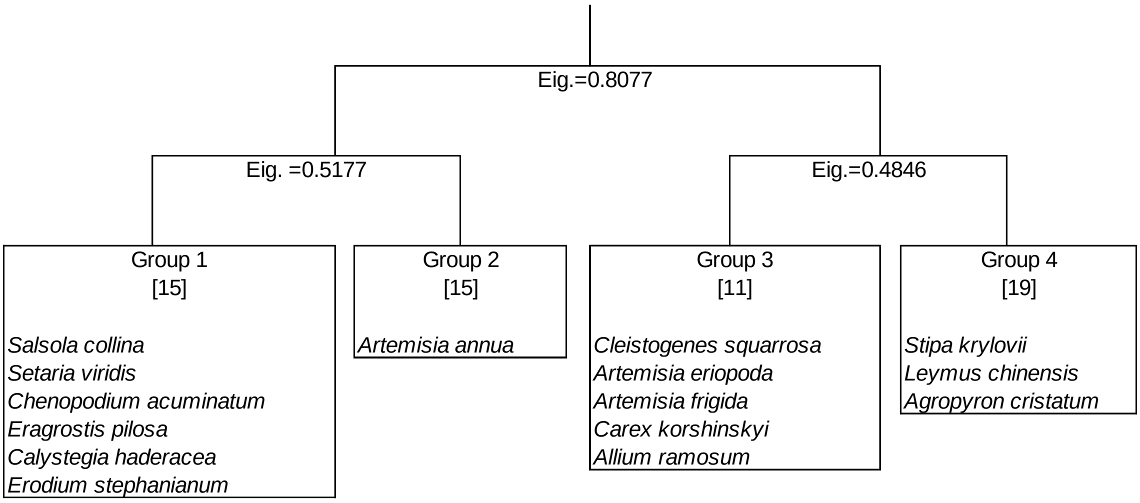

Fig. 5-2 Four vegetation types (Groups 1–4) based on classifications of 60 plots by TWINSPAN and the species name of each group is the indicator species based on INSPAN.

Table 5-1 Groups from TWINSPAN responded to land use type

groups	land use type			
	AC-1y	AC-5y	AC-25y	AC-non
group 1 (n=15)	15	-	-	-
group 2 (n=15)	-	15	-	-
group 3 (n=15)	-	-	11	4
group 4 (n=15)	-	-	-	15
Total of plots	15	15	11	19

AC: Abandoned Cropland; AC-non: light-grazing grassland that had never have been cultivated

Table 5-2 Above-ground vegetation parameters in the four land use type

parameters	AC-1y	AC-5y	AC-25y	AC-non
cover (%)	37.0 ± 5.7 a	18.8 ± 5.2 b	43.6 ± 4.7 ac	56.0 ± 6.5 d
biomass	48.5 ± 10.9 a	17.2 ± 4.9 b	79.7 ± 15.1 c	142.0 ± 6.2 d
diversity index (*H'*)	1.6 ± 0.4 a	1.7 ± 0.5 a	1.8 ± 0.2 a	1.6 ± 0.3 b
perennial plant (%)	13.1 ± 2.1 a	46.7 ± 7.8 b	87.5 ± 12.3 c	89.5 ± 6.6 c
litter	29.8 ± 4.4 a	36.6 ± 7.5 ab	40.1 ± 3.0 b	57.8 ± 4.1 c

Significant differences are indicated by different letters.

Table 5-3 Variables of soil properties at the four land use type

Soil properties	AC-1y	AC-5y	AC-25y	AC-non
pH (H_2O)	7.72 ± 0.13 a	8.57 ± 0.37 b	7.82 ± 0.17 a	6.21 ± 0.14 c
EC (dS/m)	0.37 ± 0.02 a	1.45 ± 0.05 b	1.04 ± 0.05 a	0.44 ± 0.03 a
NO_3-N (mg kg^{-1})	2.35 ± 0.42 a	2.54 ± 0.53 a	2.99 ± 0.67 b	4.21 ± 0.06 c
NH_4-N (mg kg^{-1})	1.59 ± 0.64 a	0.76 ± 0.04 a	1.09 ± 0.21 a	4.84 ± 0.37 b
C/N	9.74 ± 0.15 a	10.91 ± 1.35 b	9.64 ± 0.19 a	9.94 ± 0.30 a
Organic C (g kg^{-1})	9.16 ± 1.51 a	6.55 ± 0.96 a	7.23 ± 0.64 a	24.84 ± 3.09 b
Total N (g kg^{-1})	0.94 ± 0.12 a	0.60 ± 0.09 b	0.75 ± 0.06 ab	2.5 ± 1.71 c
P (mg kg^{-1})	12.38 ± 4.09 a	12.92 ± 2.68 a	12.5 ± 1.8 a	24.48 ± 5.67 b
Soil moisture (%)	8.00 ± 0.00 a	9.06 ± 0.13 ab	10.7 ± 1.09 b	12.12 ± 1.4 bc
Soil hardness	19.6 ± 1.70 a	23.2 ± 1.90 b	20.6 ± 1.10 ab	20.2 ± 1.60 a

Significant differences are indicated by different letters.

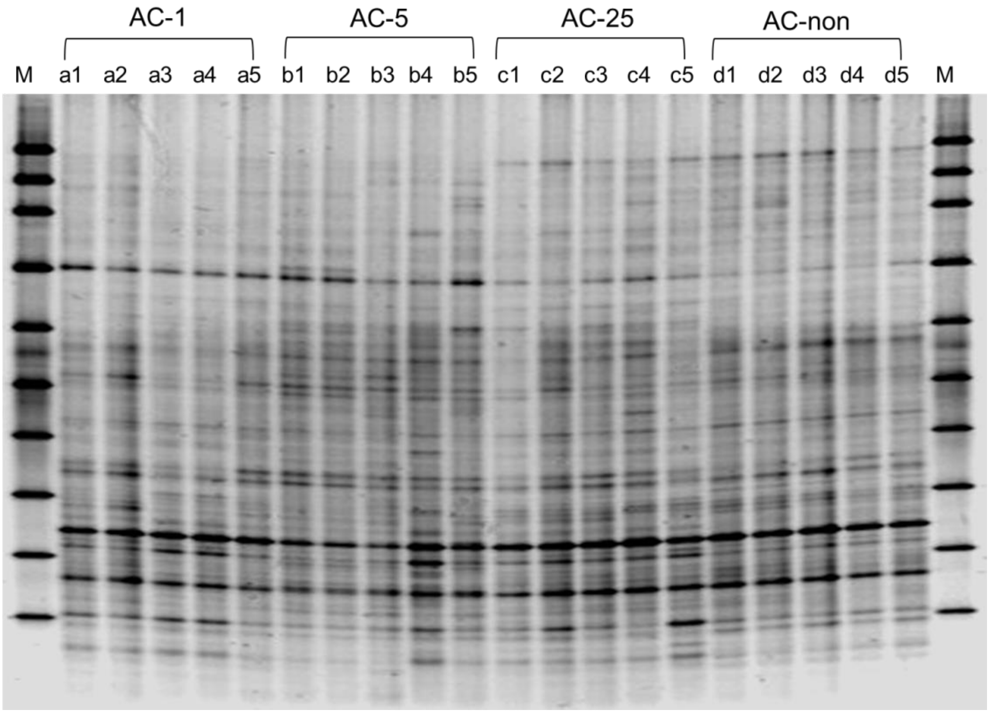

Fig. 5-3 Denaturing gradient gel electrophoresis (DGGE) profiles of the 16S rRNA genes. For all images, the numbers refer to the plot numbers in the sample areas. M: Marker.

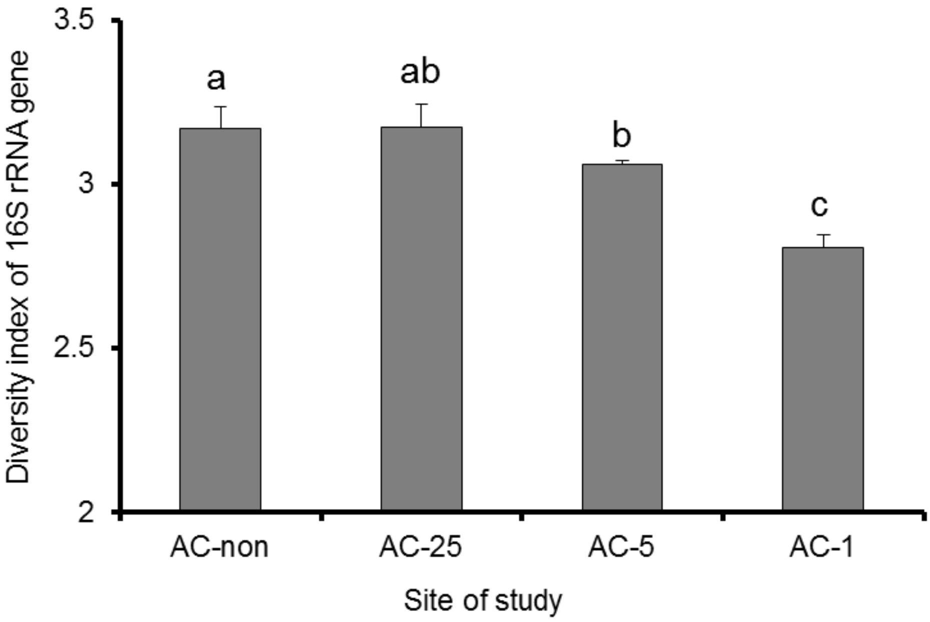

Fig. 5-4 Shannon's diversity index values for 16S rRNA genes. Data sets and results of analysis of variance (ANOVA) in the abandoned cropland (AC-1, AC-5, and AC-25) and light-grazing grassland that had never have been cultivated (AC-non) soils (n = 5; error bars represent standard deviations). Significant differences are indicated by different letters.

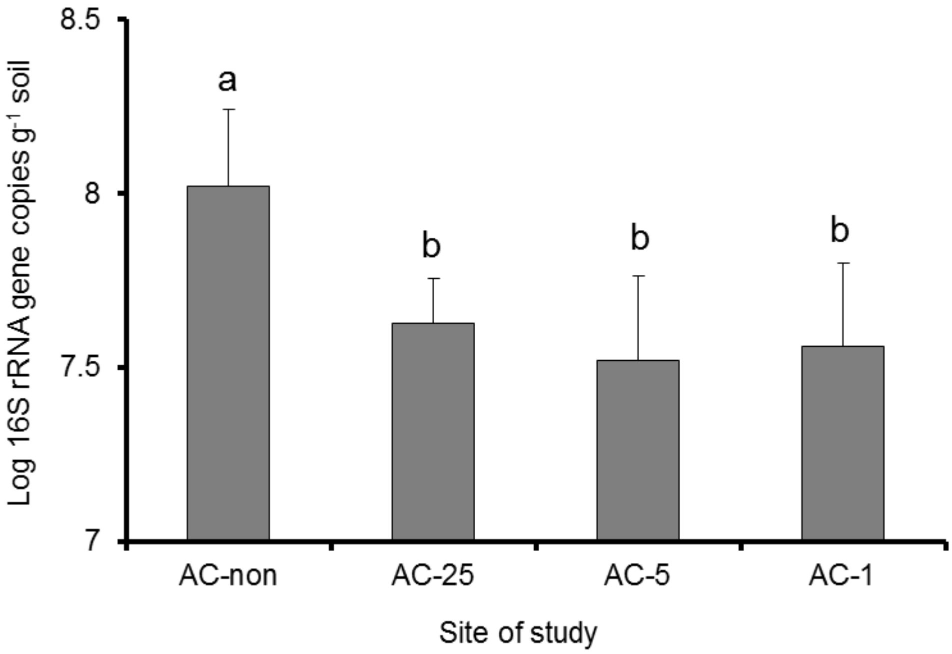

Fig. 5-5 Copy numbers of the 16S rRNA genes and results of analysis of variance (ANOVA) in the abandoned cropland (AC-1, AC-5, and AC-25) and light-grazing grassland that had never have been cultivated (AC-non) soils (n = 5; error bars represent standard deviations). Significant differences are indicated by different letters.

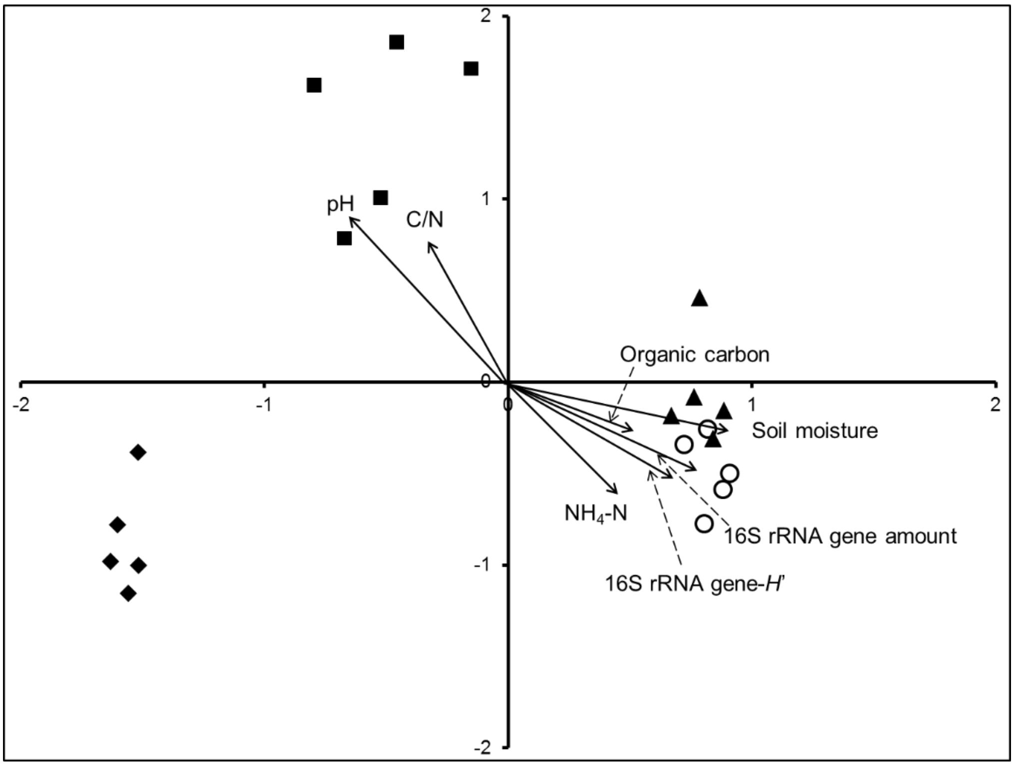

Fig. 5-6 Ordination plots of vegetation community associated with abandoned croplands for different times: AC-1 (♦), AC-5 (■), and AC-25 (▲), and non-cultivation with little grazing (AC-non, ○). The plots were generated by CCA of the denaturing gradient environmental factors. Eigenvalue for axis 1 and 2 was 0.667 and 0.294, respectively.

Table 5-4 Correlation of species ordination axes with soil physicochemical parameters, eigenvalues and percentage variances explained from CCA

Variable	Axis 1	Axis 2	
Soil moisture	0.92	-0.17	**
NO_3-N	-0.29	0.77	*
NH_4-N	0.53	-0.41	*
P	-0.19	0.92	
OC	0.54	-0.20	
TN	0.56	-0.24	
pH (H_2O)	-0.49	0.83	
16S rRNA gene-*H′*	-0.97	-0.13	**
16S rRNA gene amount (copies g^{-1})	0.76	0.36	**
C/N	-0.23	0.78	
Soil Hardness	-0.07	0.54	
EC(μs/cm)	0.38	0.47	
Eigenvalue	0.66	0.28	
% variance	37.69	15.82	

Significant correlation (Monte Carlo test, $^{**}P<0.01$, $^{*}P<0.05$)

Chapter 6

Effect of deferred spring grazing on vegetation of the Xilingol grassland, Inner Mongolia

6-1 Introduction

Chapter 3 to 5 has demonstrated the influences of cultivation on grassland. This chapter focuses on another human factor of desertification that is over-grazing on grassland and policies of preventing desertification.

Grassland ecosystems, like all natural systems, are affected by a number of factors. For example, plant species composition and community structure vary both spatially and temporally (Reitalu *et al.*, 2008; Thompson, 2005). In addition, grazing management is often associated with plant productivity, water quality and soil fertility (Carter *et al.*, 2011; Hubbard *et al.*, 2004). As a consequence, in order to evaluate the effectiveness of grassland protection policies in the context of vegetation science, it is essential to establish study sites within each vegetation type, reflecting differences in grazing management, and to include both time and grazing as variables under consideration. The Xilingol league government decided to implement policies such as deferred spring grazing, grazing prohibition, rotational grazing and cropland conversion to grassland in the Xilingol for a period of 9 years (2002–2010) in order to restore the vegetation. In 2002 deferred spring grazing was applied to approximately 9% of the grassland area of the Xilingol, in 2005 the proportion covered was 83.3%, and in 2009 it was 90% of the area (Guo *et al.*, 2006; Tumenulji, 2010). Many ecologists have examined the validity of deferred spring grazing in the Xilingol grassland (Li *et al.*, 2005; Liu *et al.*, 2006; Zhu *et al.*, 2008). Most of these studies were conducted in the typical steppe and meadow steppe, and the desert steppe has been largely neglected. In addition, in these regions there are not enough case studies examining the effects of grazing pressure and rainfall on vegetation to confirm the effectiveness of the grazing regimes. Detailed research background is explained in the General introduction (Chapter 1).

I located my study sites in areas where precipitation and grazing pressure differ, and I collected data in 2006 and 2010. My aims were to examine the vegetation response across a precipitation gradient, with differences in annual rainfall and grazing pressure, so as to evaluate the effect of deferred grazing.

6-2 Materials and methods

6-2-1 Study area and site description

The Xilingol steppe is an important ecosystem in Eurasia and has long been used by pastoral nomads. The study area is located in the Xilingol league (111°09′–120°00′ E, 41°34′–46°52′ N) in middle Inner Mongolia, China (**Fig. 6-1**). The Xilingol steppe covers about 202,580 km^2, approximately 88% of which can be used for grazing and other farming activities.

This area has a semi-arid continental climate; this is characterized by strong winds, dry conditions and low temperatures. The mean annual temperature is 0-3 ℃. In January, the coldest month, the average temperature is –20 ℃ while the mean temperature of the warmest month, July, is 21 ℃. The average annual rainfall is 295 mm, decreasing from east to west. Rainfall is high between June and September, which represents the summer monsoon season. Annual evaporation is 1500–2700 mm, increasing from east to west (Gou *et al*., 2010). Moving from east to west, the grassland starts as meadow gradually transforming into typical, and desert steppe, covering 12.28%, 52.2%, and 15.6%, respectively, of the entire area (Tong *et al*., 2004).

Using the Annual rainfall distribution map for Inner Mongolia (Fujiwara *et al*., 2007), I selected areas located between isohyets of below 200 mm, 200–300 mm, and 300–400 mm, and chose three natural grasslands where deferred spring grazing is implemented as a government policy: site 1 (desert steppe), site 2 (typical steppe), and site 3 (typical steppe). I collected data from an area that had been continuously grazed and an area subjected to deferred grazing at each of the three sites in early August 2006 and 2010 (**Fig. 6-1**). For small-scale farmers, continuously grazed areas are separated from deferred spring grazing areas by fences; the latter are not grazed for a specified period of the year (for this study 45 days in spring, from late April to early June). About a third of the land used by small-scale farmers has been fenced at each site. **Table 6-1** presents basic information about the study sites.

6-2-2 Sampling design and data collection

The primary objective of the study was to ascertain how reduced grazing pressure affects the vegetation within the exclosures. Thus, I used a paired design approach which compared data from the DSG (deferred spring grazing) and CG (continuously grazed) areas for each site. I assumed that the paired sites were the same at the time when the fences were erected, and that any differences in the vegetation were the result of grazing differences. The size of CG and DSG areas are 460 hectare (ha) and 236 (ha) for site 1; 247 (ha) and 133 (ha) for site 2; 154 (ha) and 105 (ha) for site 3, respectively.

At each site I used quadrats to sample the vegetation. In 2006, in each site, I selected two areas (DSG and CG), and set up a 30 m transects in each area, where vegetation was uniform; I collected data from a 1 m^2 quadrat every 10 m along each transect (i.e. at 10 m, 20 m and 30 m), amounting to 18 plots in three sites. In 2010, I placed 1 m^2 quadrats along 50 m transects in approximately the same areas as the 2006 transects and again sampled every 10m, resulting in total 30 plots in three sites. The transect distance from fences 1500-5000 m in each area in 2006 and 2010. In both years I measured the height of each species and estimated their cover values, recording each individual plant that appeared within the plots. In addition, for each site, I recorded total vegetation cover according to Penfound–Howard (1940). I also calculated the summed dominance ratio (SDR_3) for each species. That was referenced to the Chapter 4 method for calculation.

In order to ascertain livestock preferences, I referred to Commissione redactorum florae intramongolicae (1989, 1990, 1992, 1994, 1998) and the Editorial Committee of Grassland plants of Sunitezuo County (2009) and allocated the plants to one of four categories: a – species eaten by five different types of livestock; b – species eaten by three or four types of livestock; c – species eaten by one or two types of livestock; and d – species rarely eaten by any livestock.

After recording species names, heights and cover in all the quadrats, I selected one plot at random and harvested the aboveground biomass of every plant. The harvested material was separated into the different species. I recorded the fresh weight of this material, then dried it at 80 ℃ for 24 hours before reweighing.

In order to estimate biomass, I calculated each plant's *v*-value for each quadrat. These *v*-value were calculated as multiplied plant height (cm) by coverage of species (cm^2), expressed as cm^3 (Kawada *et al*., 2005). I used following formula to calculated the *v′*–value to estimate whole biomass in each quadrat (Kawada *et al*., 2007)

$$v' = \sum_{i=1}^{n} (Ci * Hi)$$

where: *Ci* is the coverage of species *i* in each quadrat; *Hi* is the height of species *i* in each quadrat.

The coverage values were subsequently allocated to classes as follows: 4: 8800 cm^2, 3: 6300 cm^2, 2: 3800 cm^2, 1: 1600 cm^2, 1’: 300 cm^2 and +: 50 cm^2. The correlations between measured biomass and *v*–values were evaluated for each site.

In order to compare species diversity for the different treatments, I used Shannon’s diversity index, *H′* (Shannon and Weaver, 1963).

$$H' = -\sum pi \log_2 pi$$

where: *pi* is the probability of finding species *i* among all species within a quadrat at each site.

To measure evenness, I used Pielou’s equitability index, *J′* (Pielou, 1969):

$$J' = H' / \log_2 S$$

where: *H′* is Shannon’s diversity index; *S* is the total number of species.

I calculated the grazing pressure from the number of livestock units, grazing area, and the number of grazing days at each research site. My figures were based on the official recommendations from the Chinese Ministry of Agriculture (2002). One livestock unit is equivalent to one adult sheep; the grazing impact of other livestock can be determined using the following factors: goat=0.8, cow=6.0, horse=6.0, camel=7.5.

Monthly and annual precipitation and temperature data (Willmott and Matsuura, 2010) were available for each research area for the period 2000–2010. Monthly averages for air temperature and monthly total precipitation were interpolated to a 0.5° by 0.5° degree latitude/longitude grid, with the grid nodes centered on 0.25°.

6-2-3 Statistical analysis

In all tests, probabilities lower than 0.05 were regarded as statistically significant. The normal distribution of the residuals was checked using histograms and the

Kolmogorov-Smirnov test. To counter heteroscedasticity, data were log transformed prior to analysis when necessary. The relationship between estimated and measured biomass was examined using an *F*–test. I tested the overall effects of grazing management and year on vegetation parameters using two-way ANOVA. Sampling numbers of CG and DSG are 3 and 3 for 2006; 5 and 5 for 2010, respectively. Detrended correspondence analysis, DCA (Hill, 1979) was used to determine any patterns within the full data set with respect to species distribution and structure. Plant cover data for each species in each plot were used in these analyses. All statistical analyses were performed with SPSS statistical software package (version 19.0; SPSS, Inc., Chicago, IL, United States) and R software (version 2.15.2, R Development Core Team., 2012).

6-3 Results

6-3-1 Species composition responses to precipitation and grazing gradients

Site 1 was desert steppe dominated by *Stipa gobica*, which has been fenced since 2003 (**Table 6-1**). Site 2 and 3 represents typical steppe dominated by *Stipa grandis*. Both sites have been fenced since 2001 and 2002, respectively (**Table 6-1**).

Figure 6-2 shows the DCA ordination plot for species composition (eigenvalues of the axis 1 and 2 are 0.64 and 0.28 for site 1; 0.55 and 0.29 for site 2; 0.58 and 0.31 for site 3, respectively). The variation along axis 1 is generally linked to the grazing intensity, with an inverse relationship between grazing pressure and DCA scores at the all sites. The gradient along axis 2 generally reflects the annual precipitation between 2006 and 2010, with a positive relationship between rainfall and DCA score (**Table 6-1, Fig. 6-2**).

6-3-2 Dominant species, common species and species palatability in different grazing regimes at each site

The dominant and common species were calculated by SDR_3 of all species in each site. The common species of *Salsola collina* and *Artemisia frigida* are significantly affected by both grazing pressure and years in site 1 (**Table 6-2**), while do not vary along with the years in site 2 and site 3 (**Table 6-5, Table 6-3 and Table 6-4**). The dominant species (species take up for more than 50% of proportion of SDR_3) tend to be differed by sites (**Table 6-5**). On site 1, most dominant species were significantly affected by the effect of year, only *Echinops gmelini* was affected by grazing pressure in reverse. On site 2, the dominant species were significantly affected by all the treatments. On site 3, *Stipa grandis* and *Leymus chinensis* were significantly affected by grazing pressure, while *Allium odorum* was differed by year and grazing pressure-years interaction.

The palatability levels of all species and their SDR_3 values are given in **Fig. 6-3**. Species (a) and (b) are preferred by livestock, whilst (c) and (d) are less palatable. On site 1, the unpalatable species were abundant in CG in 2006, whilst there were similar numbers of species of all levels of palatability in the DSG area. On site 2, the species that were unpalatable to livestock were most abundant in CG in 2006. In 2010, the palatability levels were very similar in the DSG and CG areas. On site 3, unpalatable species were uncommon in the DSG area in 2006, but became more common in 2010.

6-3-3 Plant species biomass, richness, diversity and evenness under different grazing regimes

I found correlations between the measured aboveground biomass of each species and the *v*-value at every site (**Table 6-6**, *F*–test, $P < 0.001$), so I were able to use *v*–values to estimate biomass.

The results of two-way ANOVA for the species biomass, richness, diversity and evenness indicated that parameters except for species richness in site 1 and 2, were significantly interacting between the treatments (grazing pressure and year; $P < 0.05$) shown in **Table 6-7**. On site 1, species biomass, diversity and evenness were differed substantially between the years, not significantly affected by grazing pressure and grazing pressure-year interaction. On site 2, species biomass, diversity and evenness were significantly affected by grazing pressure, year and grazing pressure-year treatments. On site 3, except species richness, other vegetation parameters were significantly correlated with grazing pressure, while not varying by effect of years.

6-4 Discussion

In arid and semiarid regions, the effects of herbivores may sometimes be unclear because they are masked by the effect of significant variation in precipitation; in addition, herbivores in natural ecosystems do not exhibit predictable fluctuations in productivity (Ellis *et al*., 1993). In my DCA (**Fig. 6-2**), axis 1 represents for grazing pressure, and axis 2 for years (my proxy for annual precipitation). It was suggested that variation in vegetation structure is affected by rainfall variability and grazing pressure across the whole of the study area.

Plant species biomass and diversity are negatively correlated with grazing pressure (Sankey *et al*., 2006; Zemmrich *et al*., 2007) and positively correlated with the amount of precipitation (Bates *et al*., 2006; Wuyunna *et al*., 2012) in desert and typical steppe. According to my results of two-way ANOVA, after implementing deferred spring grazing, the grazing pressure and precipitation affects differently to biomass, diversity and evenness in different regions (**Table 6-7**). On site 1, there was no significant correlation between grazing pressure and grazing pressure-year interaction. However, species biomass, diversity and evenness differed in each year, even though grazing pressure remained about the same in each of the areas in the different years (**Table 6-1**). This implies that grazing had little influence on species biomass, diversity and evenness. This result supports the suggestion that annual precipitation variation is a more important determinant of species biomass, diversity and evenness than grazing pressure in the desert steppe. On site 2, species biomass, diversity and evenness varied along with grazing pressure and years. In addition, the varying of species biomass along with grazing pressure-year interaction was also significant, suggesting vegetation of this site sensitive to both rainfall and grazing by geographical conditions. On site 3, the above vegetation parameters varied with grazing pressure but not between years. Although there was sufficient rainfall during the growing season (April–July) in both 2006 and 2010, and the abundant precipitation should encourage faster recovery of degraded vegetation (Johansen *et al*., 1993), grazing pressure differed greatly between DSG and CG area (**Table 6-1**). This may indicate that grazing pressure plays a more important role in controlling vegetation than rainfall variability.

It is generally described as a stable, mature grassland that steppe dominated by genus *Stipa* (Bai *et al*., 2004; Hayashi, 2003; Walter, 1973). Most of my study areas were dominated by *Stipa gobica* or *S. grandis*. However, unpalatable species such as *Echinops gmelini* and *Allium odorum* were dominated in the continuous grazing area (**Table 6-5**), and these dominants also strongly affect vegetation quantitatively because dominant species affect the species biomass more than other species in the community (Huston, 1997), and the dominant could also control the effect of diversity on community stability (Cingolani *et al*., 2005) which implies that the palatability of dominant was important for quality of vegetation community. *Artemisia frigida* is an indicator of excessive grazing (Burkhard *et al*., 2008), and *Salsola collina* is a grazing tolerant species (Wesche and Retzer, 2005), hence both the common species could be found independently of grazing pressure (site 2 and site 3). Similarly, the *E. gmelini* was only affected by grazing pressure in site 1, which makes it described as an indicator of grazing in desert steppe after implementing deferred spring grazing.

The palatability levels of species did not change as a result of reduced grazing pressure (**Fig. 6-3**). Although unpalatable species such as *E. gmelini* were abundant in the CG area, other unpalatable species, for example *S. collina*, were also found in the DSG area, suggesting that their presence may be affected by precipitation, trampling and other factors. On the steppe, water is the key limiting resource and a slight change in rainfall greatly affects species composition (Cheng *et al*., 2011); in addition, palatable plants are more susceptible to trampling under such conditions (Hussain and Durrani, 2009). So, palatability is an important feature of the species of rangeland steppes. In spite of the high vegetation diversity in terms of overall species composition, if there are a few palatable species, the grazing value of the steppe is low (Wuyunna *et al*., 1999).

In general, vegetation restoration was found to be proportional to grazing pressure. The inter-annual variations in precipitation and monthly rainfall are important factors that influence vegetation restoration, and determine quality changes in vegetation, particularly in the desert steppe. Although these findings improve the understanding of the effect of deferred spring grazing in steppe regions, their applicability is limited because only three locations and two years were studied.

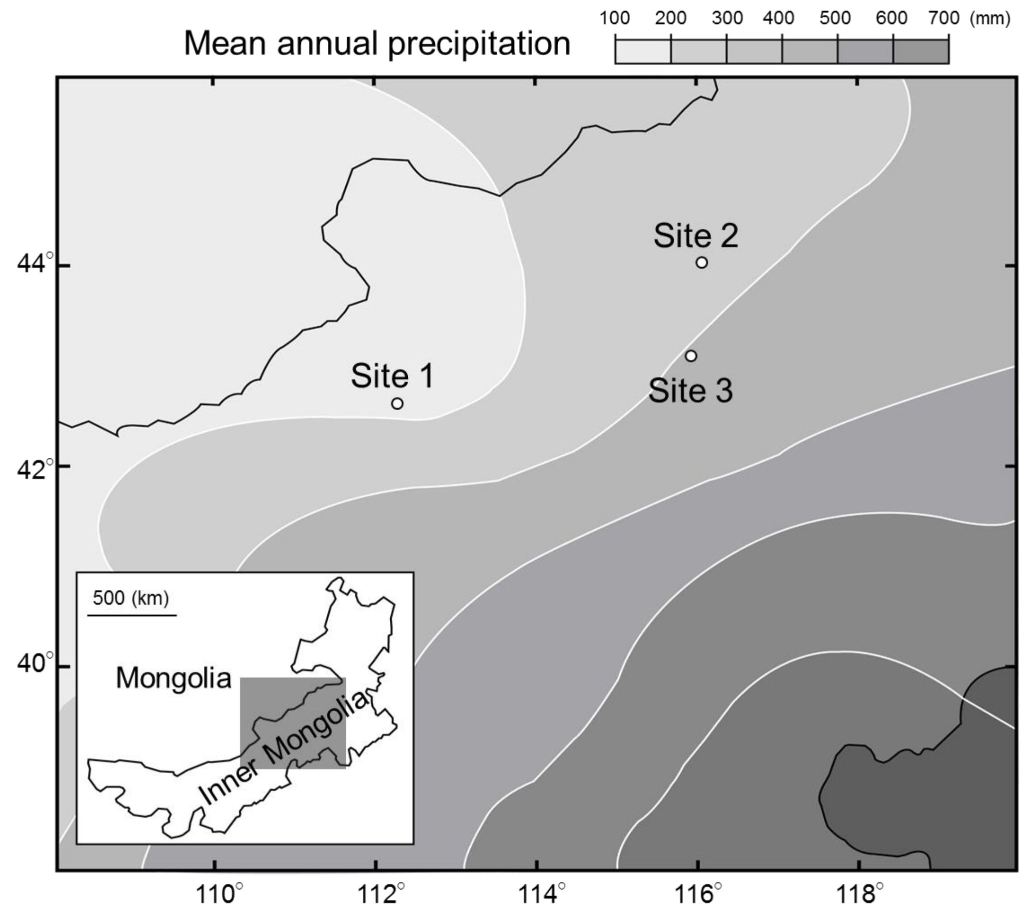

Fig. 6-1 Map of studied sites in Xilingol grassland. Contours show annual precipitation interpolated from sites of known precipitation obtained from Fujiwara *et al*. (2007)

Table 6-1 Summerized information for studied sites in the Xilingol grassland

Study site (area*)		Site1		Site2		Site3	
		DSG	CG	DSG	CG	DSG	CG
Latitude		42 42' N	42 42' N	44 05' N	44 04' N	43 23' N	43 22' N
Longitude		112 14' E	112 13' E	116 06' E	116 05' E	115 55' E	115 55' E
Altitude (m)		1202	1191	979	981	1375	1365
Annual mean temperature (°C)**		3.4	-	2.3	-	1.6	-
Annual precipitation (mm)**		171.9	-	276.1	-	341.9	-
Type of steppe***		DS	DS	TS	TS	TS	TS
Year of DSG establishment		2003		2001		2002	
Land use type****		DSG (45d)	grazing	DSG (45d)	grazing	DSG (45d)	grazing
Grazing pressure (2006)*****		1.6	3	2.6	3.8	2.5	4.9
Grazing pressure (2010)*****		1.4	2.7	2.9	3.3	2.9	5.1
Monthly	2000-2010 ave.	87.9	-	148.3	-	190.5	-
rainfall (Apr.-	2006	95.2	-	131.5	-	213.9	-
July; mm)	2010	79.6	-	158.9	-	201.2	-

*DSG, Deferred Spring Grazing; CG, Continuous Grazing. **Data (2000-2010) provided by http://climate.geog.udel.edu/~climate/html_pages/download.html. ***DS, Desert Steppe; TS, Typical Steppe. ****Land use type was the same in 2006 and 2010; DSG (45d), The 'no grazing' period in the DSG area is from late April to early June (delayed by 45 days relative to standard practice) every spring. *****Unit of grazing pressure is sheep/ha-1.

Table 6-2 Species composition, all species SDR_3 and number of species in DSG and CG area with different year in site1

	SDR_3 (2006)		SDR_3 (2010)	
Species name	DSG	CG	DSG	CG
Stipa gobica	77.2	67.8	100.0	80.4
Astragalus scaberrimus	94.4	33.8	98.1	47.8
Salsola collina	92.6	20.1	41.3	12.2
Agropyron desertrum	63.4	—	82.7	24.4
Artemisia frigida	57.5	28.7	68.2	41.4
Stipa glareosa	53.1	14.1	68.6	17.9
Allium mongolicum	51.6	44.8	53.9	40.3
Haplophyllum dahuricum	51.3	24.2	55.1	30.9
Asparagus gobicus	49.1	30.2	43.3	45.7
Caragana microphylla	46.7	46.8	49.4	41.4
Peganum nigellastrum	43.3	38.9	—	—
Cleistogenes soongarica	42.5	34.6	69.0	39.4
Eragrostis pilosa	31.5	38.9	23.4	—
Lagochilus ilicifolius	21.2	32.4	24.7	23.6
Artemisia pectinata	18.5	15.9	—	28.3
Echinops gmelini	—	79.7	—	26.8
number of species	15	15	13	14

DSG: Deferred Spring Grazing area, G: Grazing area

Table 6-3 Species composition, all species SDR_3 and number of species in DSG and CG area with different year in site2

Species name	SDR_3 (2006)		SDR_3 (2010)	
	DSG	CG	DSG	CG
Agropyron cristatum	100.0	16.6	73.8	53.1
Stipa grandis	63.4	82.0	87.2	83.2
Leymus chinensis	54.4	21.7	60.7	56.9
Pocockia ruthenica	39.9	—	51.3	45.2
Astragalus melilotoides	31.2	16.8	24.2	—
Artemisia scoparia	31.2	—	—	—
Allium bidentatum	30.5	22.3	41.9	35.4
Allium tenuissimum	27.2	16.8	33.6	—
Carex korshinskyi	17.8	18.3	32.3	37.1
Artemisia frigida	17.8	35.5	14.0	26.9
Salsola collina	17.3	19.5	9.6	12.6
Convolvulus ammannii	17.2	—	24.5	18.6
Cleistogenes squarrosa	15.6	34.8	19.5	30.6
Artemisia pectinata	—	28.4	—	—
Artemisia commutata	—	25.0	—	—
Heteropappus altaicus	—	13.4	16.8	20.6
Chenopodium glaucum	—	12.9	—	17.3
Allium odorum	—	11.3	18.0	16.2
Setaria viridis	—	10.2	—	—
Potentilla acaulis	—	7.7	10.7	19.0
Filifolium sibiricum	—	—	26.6	20.1
Koeleria cristata	—	—	12.8	—
number of species	13	17	17	15

Table 6-4 Species composition, all species SDR_3 and number of species in DSG and CG area with different year in site3

	SDR_3 (2006)		SDR_3 (2010)	
Species name	DSG	CG	DSG	CG
Stipa grandis	95.7	30.8	100.0	62.5
Leymus chinensis	59.5	30.9	33.8	24.8
Allium tenuissimum	59.3	21.9	29.6	29.4
Chenopodium glaucum	40.3	25.8	36.7	30.9
Poa sphondylodes	35.9	—	17.8	21.5
Adenophora stenanthina	34.4	—	30.3	15.3
Allium condensatum	33.5	—	20.6	—
Rhaponticum uniflorum	31.3	—	29.7	—
Kochia prostrata	29.4	—	—	—
Allium senescens	27.8	—	—	—
Achnatherum sibiricum	26.0	—	—	—
Carex duriuscula	17.2	9.3	13.5	13.3
Bupleurum scorzonerifolium	24.1	7.5	18.9	13.6
Caragana microphylla	24.1	—	28.9	24.7
Thalictrum petaloideum	23.8	—	25.0	20.7
Allium bidentatum	23.4	20.1	18.9	22.2
Iris tenuifolia	23.4	—	24.8	25.8
Puccinellia tenuiflora	20.9	10.3	—	—
Saussurea salicifolia	19.7	—	—	—
Cleistogenes squarrosa	17.8	14.0	23.8	24.1
Koeleria cristata	15.7	8.5	11.8	8.1
Saposhnikovia divaricata	14.7	—	17.1	—
Serratula centauroides	13.4	—	15.9	16.8
Artemisia frigida	12.9	20.2	17.0	29.9
Allium odorum	9.1	82.0	51.5	57.7
Potentilla acaulis	8.2	18.8	11.0	20.9
Salsola collina	3.5	15.9	10.0	11.5
Artemisia scoparia	—	26.9	19.2	21.5
Astragalus mongolicus	—	21.6	—	—
Saussurea maximowiczii	—	15.3	17.2	17.9
Caragana stenophylla	—	—	—	13.1
number of species	27	17	24	22

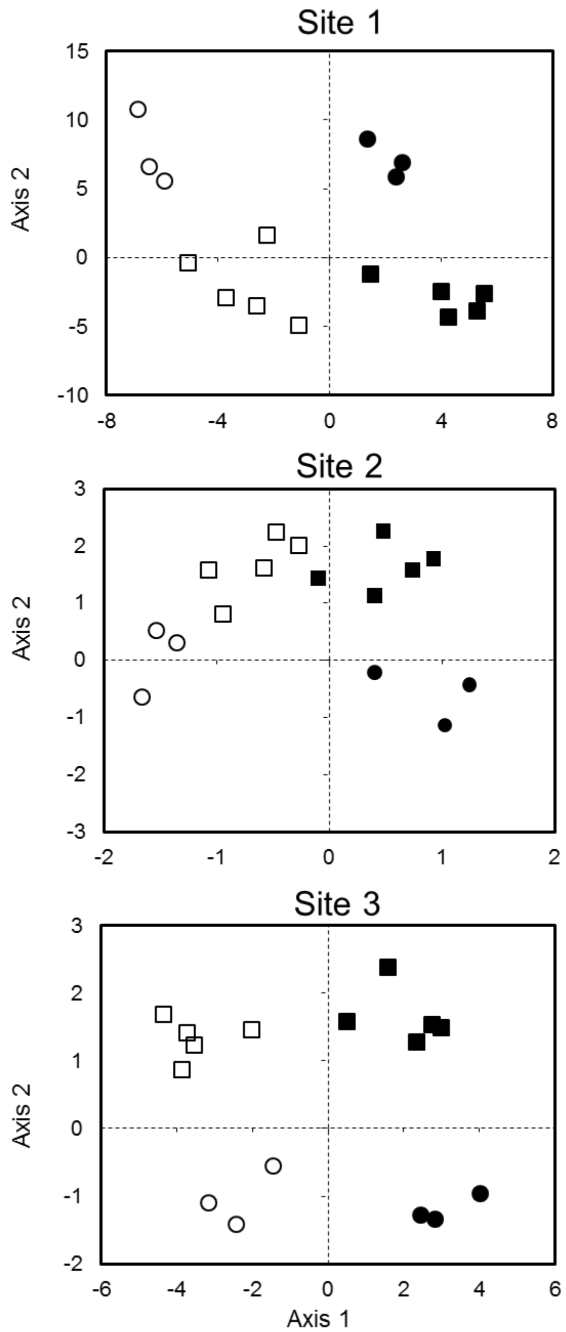

Fig. 6-2 Detrended correspondence analyses (DCA) based on species composition at the all sites along the grazing gradients. ○: CG-2006 ●: DSG-2006 □: CG-2010 ■: DSG-2010; DSG: Deferred Spring Grazing area; CG: Continuous Grazing area

Table 6-5 Results of the two-way ANOVA for the effects of SDR_3 values of the common species in all sites and all the dominant species (species take up for more than 50% of proportion of SDR_3) in each area. (Sampling numbers of CG and DSG are 3 and 3 for 2006; 5 and 5 for 2010, respectively.)

Species	Source	Site 1			Site 2			Site 3		
		df	*F*	*P*	df	*F*	*P*	df	*F*	*P*
Stipa gobica	G	1	3.50	ns						
	Y	1	5.81	*						
	G x Y	1	1.99	ns						
	E	12	-	-						
Astragalus scaberrimus	G	1	1.92	ns						
	Y	1	174.12	***						
	G x Y	1	3.47	ns						
	E	12	-	-						
Echinops gmelini	G	1	258.55	***						
	Y	1	4.91	ns						
	G x Y	1	4.44	ns						
	E	12	-	-						
Stipa grandis	G				1	13.79	**	1	14.13	**
	Y				1	11.27	**	1	3.63	ns
	G x Y				1	5.76	*	1	0.10	ns
	E				12	-	-	12	-	-
Leymus chinensis	G				1	16.92	**	1	50.50	***
	Y				1	14.00	**	1	3.38	ns
	G x Y				1	60.11	***	1	7.57	*
	E				12	-	-	12	-	-
Salsola collina	G	1	85.53	***	1	50.86	***	1	13.64	**
	Y	1	351.17	***	1	3.04	ns	1	0.67	ns
	G x Y	1	10.41	*	1	0.03	ns	1	9.13	*
	E	12	-	-	12	-	-	12	-	-
Artemisia frigida	G	1	43.74	***	1	14.65	**	1	10.96	**
	Y	1	142.38	***	1	4.55	ns	1	1.37	ns
	G x Y	1	0.33	ns	1	4.68	ns	1	7.04	*
	E	12	-	-	12	-	-	12	-	-

G, Grazing pressure; Y, Year; G x Y, Grazing pressure x Year, E, Error.
ns, not significant; *, P < 0.05; **, P < 0.01; ***, P < 0.001.

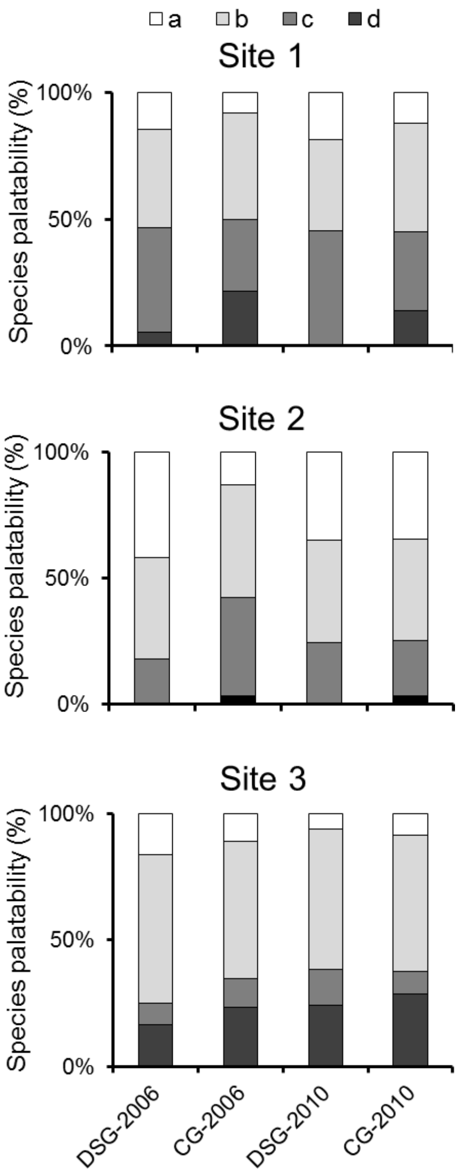

Fig. 6-3 The palatability levels of all species and their SDR values. We allocated the plants to one of four categories: a – species eaten by five different types of livestock; b – species eaten by three or four types of livestock; c – species eaten by one or two types of livestock; and d – species rarely eaten by any livestock. DSG: Deferred Spring Grazing area, CG: Grazing area.

Table 6-6 Equations describing the relationship between measured biomass and *v*-value at each site

Sites		Equations	r	*p*
1（2006）	DSG	MB=0.02v'+2.41	0.96	***
	CG	MB=0.01v'+0.49	0.97	***
2（2006）	DSG	MB=0.01v'+2.67	0.95	***
	CG	MB=0.01v'+1.15	0.85	***
3（2006）	DSG	MB=0.02v'+4.19	0.9	***
	CG	MB=0.02v'+2.91	0.94	***
1（2010）	DSG	MB=0.02v'+2.79	0.95	***
	CG	MB=0.03v'+0.83	0.93	***
2（2010）	DSG	MB=0.04v'+2.11	0.88	***
	CG	MB=0.02v'+3.22	0.95	***
3（2010）	DSG	MB=0.02v'+1.70	0.99	***
	CG	MB=0.05v'+1.96	0.98	***

MB, measured biomass; v', v'-value(X100cm^3); ***, significant at p<0.001

Table 6-7 Results of the two-way ANOVA for the effects of treatments (present grazing pressure and year) on species estimated biomass, richness, diversity and evenness at each site. (Sampling numbers of CG and DSG are 3 and 3 for 2006; 5 and 5 for 2010, respectively.)

Variable	Source	Site 1			Site 2			Site 3		
		df	*F*	*P*	df	*F*	*P*	df	*F*	*P*
Biomass	G	1	0.3	ns	1	64.7	***	1	65.5	***
	Y	1	67.2	***	1	58.6	***	1	7.6	ns
	G x Y	1	1.6	ns	1	18.9	**	1	6.6	ns
	E	12	–	–	12	–	–	12	–	–
Richness	G	1	0.43	ns	1	2.84	ns	1	3.1	ns
	Y	1	1.55	ns	1	0.66	ns	1	4.7	ns
	G x Y	1	0.43	ns	1	4.1	ns	1	26.6	***
	E	12	–	–	12	–	–	12	–	–
Diversity (*H'*)	G	1	3.7	ns	1	126.7	***	1	97.4	***
	Y	1	96.2	***	1	149.2	***	1	3.9	ns
	G x Y	1	6.9	ns	1	28.3	***	1	5.8	ns
	E	12	–	–	12	–	–	12	–	–
Evenness (*J'*)	G	1	1.1	ns	1	14.7	**	1	190.4	***
	Y	1	28.1	***	1	23.6	***	1	3.4	ns
	G x Y	1	2.4	ns	1	17.8	**	1	9.6	*
	E	12	–	–	12	–	–	12	–	–

G, Grazing pressure; Y, Year; G x Y, Grazing pressure x Year, E, Error.
ns, not significant; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$.

Chapter 7

General discussion

The work presented in this thesis was conducted to investigate the effects of farming and grazing in Kazakhstan and the steppe of Inner Mongolia. This chapter summarizes the results obtained and then discusses how the recovery of vegetation and soil at former crop and grazing sites can be promoted as well as the design of protective policies for the future.

7-1 Cultivation effects

Wind erosion of croplands and reductions in the soil’s nutrient content make it difficult to sustain farming in steppe regions (Chuluun and Ojima, 2002). Moreover, this region cannot sustain intensive mechanized or irrigated agriculture (Mainguet, 1994). Consequently, croplands are abandoned over time on the assumption that their vegetation will recover naturally via processes of secondary succession (Pueyo and Alados, 2007). In this study, the characteristics of the vegetation and soil properties were investigated along with the grassland degradation process for natural grassland sites that had previously been used as cropland. The results showed that the aboveground biomass and the diversity and density of vegetation decreased significantly both in Kazakhstan and the Inner Mongolian steppe following cropland abandonment. New species became dominant in the former croplands shortly after their abandonment, such as *Artemisia vulgaris* and *Agropyron repens* in Kazakhstan and *Artemisia eriopoda*, *Elymus duhuricus* and *Salsola collina* in Inner Mongolia. These are not the same species that are dominant on natural grassland, which implies that the natural vegetation community of the steppes is initially replaced by hardier annual plant species. The invasion of drought-, sand-, and wind-tolerant species reduced the competitive capacity and dominance of native plants, increasing the likelihood of their ultimate disappearance.

Salt tolerant plants, such as *Saussurea laciniata*, *Leymus angustus* and *Glycyrrhiza uralensis* were identified in croplands that had been abandoned for 50 years in Kazakhstan, and *Eragrostis pilosa*, *Potentilla anserine* and *Salsola collina* appeared in sites that had been abandoned for 21 to 35 years in Inner Mongolia. None of these species were found in natural grassland sites in the same regions. In addition, carbon, nitrogen and phosphorus were lost from the soil of abandoned croplands as a result of clay removal and the breakdown of water-stable aggregates, whereas their pH and EC content increased. These results suggest that farming may cause salt accumulation.

The line dividing regions where the precipitation is above 250 mm per year and those where it is below this value is regarded as a transfer zone separating typical steppe and desert steppe. Within this zone, vegetation coverage decreases in line with precipitation (He *et al*., 2012). The results presented herein suggest that cultivation is generally abandoned in areas where the precipitation is below 200 mm. Additionally, the recovery of soil and vegetation after site abandonment is difficult in regions with annual precipitation of less than 250 mm. This suggests that farming activities on weakening soil in arid regions may accelerate soil erosion. The most severe adverse effects of wind erosion were observed in semi-arid regions. The effects of cultivation on soil erosion by wind have been documented in a number of studies. For example, it has been demonstrated that the erosion potential of some soils can be greatly increased by cultivation (Aguilar *et al*., 1988), and that newly broken dry land soils become highly susceptible to erosion by wind (Chepil *et al*., 1952). With the increase in the human population, the disturbance of grassland by cultivation has become more extensive (Liu *et al*, 2003). It is therefore essential to have reliable estimates of soil loss from cultivated lands in order to understand and assess the acceleration of wind erosion by human activities (Dong *et al*., 1987). Wind erosion is a dynamic process that is affected by many natural and social factors (Zobeck, 1991). Of the relevant climatic factors, the influence of wind strength is extremely difficult to evaluate because of its variability in magnitude, time and space. The equation used to estimate the impact of wind erosion on annual potential soil loss is based on a yearly time scale and uses data on long term climatic conditions (Woodruff and Siddoway, 1965). Erosive wind energy distribution has been used to determine when the most erosive or hazardous periods occur. In this study, soil loss correlated positively with time since abandonment, increasing from the most recently abandoned plots to the oldest ones in each location. Similar trends were observed for aboveground vegetation coverage and biomass. On the other hand, the microbial diversity in cultivation sites that had been abandoned for 25 years site was identical to that in untouched plots, although the 16s rRNA gene copy number was lower than at the natural site. This may be due to a lack of soil nutrients. The microbial communities of the studied abandoned croplands were strongly affected by the plant biomass within the plot, soil moisture and NH_4-N content.

It can be concluded from this study that farming on semi-arid land induces a high risk of soil erosion. Moreover, this study suggests that the soil erosion resulting from farming is caused by the destruction of vegetation coverage, which sensitizes the land to wind erosion in environments with strong winds together with other factors such as grazing on abandoned farmland.

7-2 Grazing effect

Grazing by livestock has been identified as a major cause of steppe degradation in grassland (Sneath 1998; Tong *et al.*, 2004). It plays an important role in determining grassland productivity, forage quality, species composition and plant species diversity (Lane *et al.*, 1998; Sala *et al.*, 1988; Schonbach *et al.*, 2012; Wondzell *et al.*, 1996). Previous studies have indicated that long-term grazing can affect the species diversity of plant communities in grasslands either positively or negatively (Bullock *et al.*, 2001; Grace *et al*., 2007), but that grazing at moderate intensity tends to increase species diversity (Huston, 1979; Sasaki *et al*., 2009).

In addition, in the arid and semi-arid regions, both the precipitation pattern and grazing affect ecosystem functions (Austin *et al.*, 2004; Heisler-White *et al.*, 2008). However, the widely debated theories of equilibrium and non-equilibrium rangeland ecology emphasize different main drivers in grassland: grazing pressure and precipitation fluctuations, respectively. According to the results of my study, deferred spring grazing (45 days annual deferral) clearly had different effects at different sites. Deferral had no effect in the desert steppe (where the annual precipitation was below 200 mm). However, it had a noticeable effect on the typical steppe. Moreover, the vegetation parameters also varied significantly from year to year at sites with relatively low precipitation (i.e. where the annual precipitation is between 200 and 300 mm).

Herbivores on grassland are highly selective and will preferentially feed on highly digestible and palatable species while avoiding unpalatable plants. Grazing disturbance accelerates the regrowth of new tissues and thereby reduces the mean age of plant tissue in the grassland. Therefore younger plant material remains but has less fibre in its regrowth after grazing. The differential responses of *S. grandis* and *L. chinensis* to grazing indicate that the slow-growing *S. grandis* is more avoidant and fast growing *L. chinensis* is more tolerant. Under grazed conditions, sheep selectively consume the tolerant species and avoid the avoidant species. Grazing therefore has negative effects on *L. chinensis* but little influence on *S. grandis*. This conclusion is supported by Wang (2004). This suggests that *S. grandis* will be favored as grazing pressure intensifies and that similarly unpalatable species such as *Echinops gmelini*, *Salsola collina* and *Potentilla acaulis* will also tend to persist in grazed sites.

7-3 Recovery process of steppe

7-3-1 Recovery process after cultivation and grazing

The processes of vegetation and soil recovery in abandoned agricultural sites in Kazakhstan and Inner Mongolia are summarized in **Figs. 7-1** and **7-2**.

The recovery of the vegetation communities in each case was clearly related to the amount of time that had elapsed since abandonment. In Kazakhstan, vegetation and soil took at least 13 years to start recovering. In abandoned grazing areas, only partial recovery of soil and vegetation was achieved even 27 years after abandonment. In abandoned croplands, vegetation recovered after 50 years of abandonment but the soil did not fully recover. In the wet regions of Inner Mongolia, vegetation partially recovered at sites that had been abandoned for 20 years but the soil microorganisms did not and the original physicochemical properties of the soil were not restored. In the dry regions where cultivation had been abandoned for 20 years, neither vegetation nor soil recovered.

There has been an increased emphasis on the sustainability of human soil use in recent years due to concerns that soil quality may be declining as a result of cultivation (Dang, 2007; Liu *et al*., 2006). Two commonly used variables for assessing soil quality are pH and EC (De Clerck *et al*., 2003). In this work, it was found that the soils of abandoned croplands had higher pH and EC values than reference sites. In Kazakhstan, the average soil pH was 6.68 in natural grassland and 8.31 in abandoned cropland. In Inner Mongolia, the average pH in natural grasslands was 7.45 compared to 8.56 in abandoned cropland. In both cases, the pH at the natural sites was within the optimal range for plant growth whereas that at the abandoned agricultural sites was somewhat higher, meaning that their soils were slightly alkaline. This may contribute significantly to the disturbance of the land on the steppe (Shinchilelt *et al*., 2013). Additionally, salt leaching does not occur in steppe regions with low precipitation, so the concentration of soluble salts is high in untouched soils (Haase, 1983). It has been observed that in such regions, the rate of evaporation from the soil increases following disturbances such as cultivation, and that increases in the soil pH and EC are accompanied by increases in the salt concentration in the topmost soil layers (Yamamura *et al*., 2013). On the other hand, the C/N ratio in abandoned croplands was higher than in reference sites. In Kazakhstan, the C/N ratio for natural grasslands was 10.32 compared to 14.54 in abandoned cropland. In Inner Mongolia, the corresponding values were 9.48 for natural grassland and 12.35 in abandoned cropland. The C/N ratio increases in abandoned areas due to a decrease in the soil's N content as microorganisms consume large quantities of nitrogen to drive their growth.

My results showed that the biotic and abiotic ecosystem functions were not fully recovered in croplands that had been abandoned for a quarter of a century. The degree of recovery was higher for sites that had been abandoned for 25 years than for those that had only been abandoned 1 or 5 years ago. Notably, the microbial community in the long-term abandoned sites recovered to the "natural" state in terms of diversity but had a lower 16s rRNA gene copy number than the untouched sites. Robust information on the mechanisms that regulate the diversity, structure, and composition of natural communities is urgently needed to help conserve ecosystem function and mitigate biodiversity losses due to current and future environmental changes. In addition, the results of this study suggest that advances in desertification may be prevented by adjusting environmental properties of the abandoned cropland such as the soil's moisture and organic matter content. This will enhance the function of the soil's microbial communities and thereby promote plant biomass production, accelerating ecosystem recovery.

The trends in the recovery of soil carbon and microorganisms were generally similar to those for vegetation. However, at the Sonid-you-qi survey site in Inner Mongolia there were no signs of vegetation or soil recovery despite 21 years of abandonment. This may have been due to soil erosion, the rate of which was above 80 percent. This suggests that the ecological restoration of the steppe will be difficult in locations where the soil erosion rate is above this threshold.

In general, the speeds of vegetation and soil recovery depend on the duration of abandonment, historical land use patterns, and regional precipitation levels. Another factor that has profound effects on recovery is the rate of soil erosion.

7-3-2 Recovery process after fenced

To address the problem of desertification, since 1990 Inner Mongolia has been implementing policies such as the "land contract", "ecological migrants", "enclosing transferring", "banned grazing" and "deferred grazing" systems (Alatansha and Chitose, 2012). These protection policies have resulted in a transition from pastoral management to fenced livestock rearing, both full-time and partial (Alatansha and Chitose, 2012).

The process of vegetation recovery in deferred spring grazing areas is summarized in **Fig. 7-3**. According to two-way ANOVA results, vegetation recovery will occur in wetter regions but not dry ones. This means that the capacity for ecosystem recovery after deferred grazing depends on the precipitation in the region.

Banning grazing is an effective and widely known method of preventing steppe degradation due to overgrazing. It causes increases in vegetation biomass, diversity and coverage regardless of the area’s level of precipitation (Gegentu *et al*., 2006; Yan *et al*., 2007; Xu *et al*., 2008). However, soil recovery still takes around 20 years in sandy steppe sites even when grazing is banned (Ohkuro and Nemoto, 1997); six years are required at typical steppe sites (Li *et al*., 2013). It has been suggested that the recovery speed depends primarily on the area’s precipitation and vegetation type. However, species diversity begins decreasing 4 or 5 years after abandonment (Xu *et al*., 2008; Wang et al., 2010). Additionally, banning grazing in certain areas may destroy the livelihoods of local people by limiting the amount of livestock they can keep. To address this issue, deferred spring grazing was introduced in Xilingol in 2002. It was initially applied to around 9% of the region's grassland and subsequently expanded to 83.3% and then 90% in 2005 and 2009, respectively (Guo *et al*., 2006; Tumenulji, 2010). Results obtained in this work indicate that deferred grazing leads to vegetation recovery in areas with sufficient precipitation not in those with lower levels of rainfall. Therefore, deferred grazing is not effective in the Xilingol area, around 30% of which is desert steppe. An improved grazing policy will therefore be required to enable ecosystem recovery in low precipitation areas.

7-4 Grassland protection policies and future issues

The recovery capacity of grasslands is summarized in **Fig 7-4**. It is essential to account for over-grazing and cultivation in low precipitation regions because they have such low recovery capacities. Cultivation in particular causes severe soil erosion and reductions in recovery capability. While wet regions are more capable of recovery, it is important to note that in some areas of Kazakhstan, recovery was incomplete even 50 years after site abandonment.

This study indicate that farming activity had profound effects on the steppe ecosystems, causing changes in the vegetation community while also promoting soil erosion and accelerating desertification. Careful control of land use patterns is therefore required. The results reported herein demonstrate that while steppe vegetation can recover within 10 to 20 years of site abandonment, soil recovery is slower. In addition, vegetation recovery is difficult to achieve in regions subject to heavy erosion. These results indicate that farming on steppes should be banded to protect their grasslands. In Inner Mongolia, the Cropland Conversion to Forest and Grassland (CCFG) program was recently introduced to prevent desertification. However, at the same time, the area of agricultural land increased. The local government will therefore have to emphasize the unsuitability of the steppe for farming to local people and vigorously enforce a complete ban on illegal cultivation.

The diverse range of steppe protection policies that have been introduced recently in Inner Mongolia, the increasing numbers of livestock in the region, and the ongoing progress of desertification all present important challenges. To solve these problems, it will be necessary to review existing policies for improving the environment and reorganize the joint control system of stock-farming. This could potentially address the labor shortage problem of the pastoral system and enable more time to be spent on the protection of grasslands. It is possible that breed improvements might allow nomadic peoples to increase their incomes without harming local ecosystems. Cultivation in dry regions is extremely dangerous for grassland recovery, so it is extremely important to promote sustainable grazing policies.

In future, it will be necessary to gather more data on soil erosion in Inner Mongolia. Comparative studies of sites subject to banned and deferred grazing policies should be conducted. In order to prevent further grassland desertification, we will need a database of vegetation and soil properties under a wide range of site conditions in order to properly evaluate and predict the likely impacts of suggested policies.

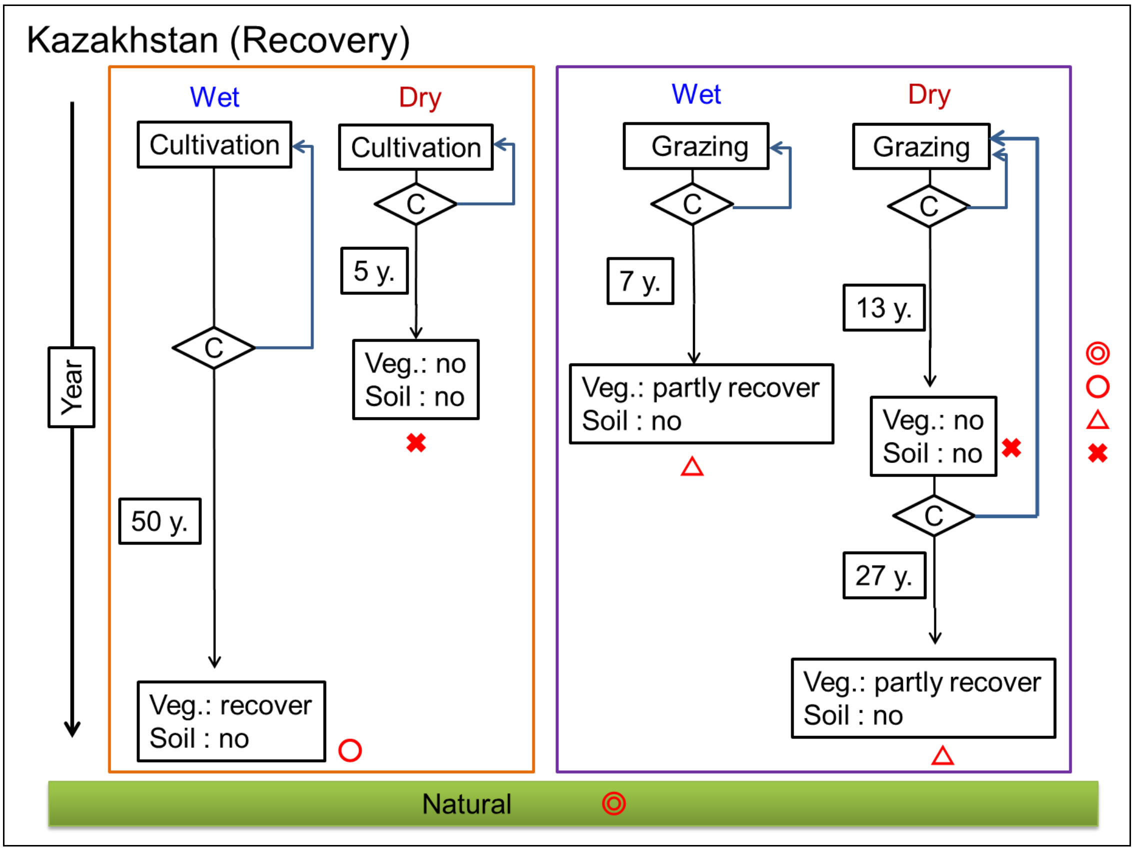

Fig. 7-1 Recovery process of vegetation and soil in Kazakhstan abandoned areas. The numbers indicate the number of years that cultivation and grazing abandoned.

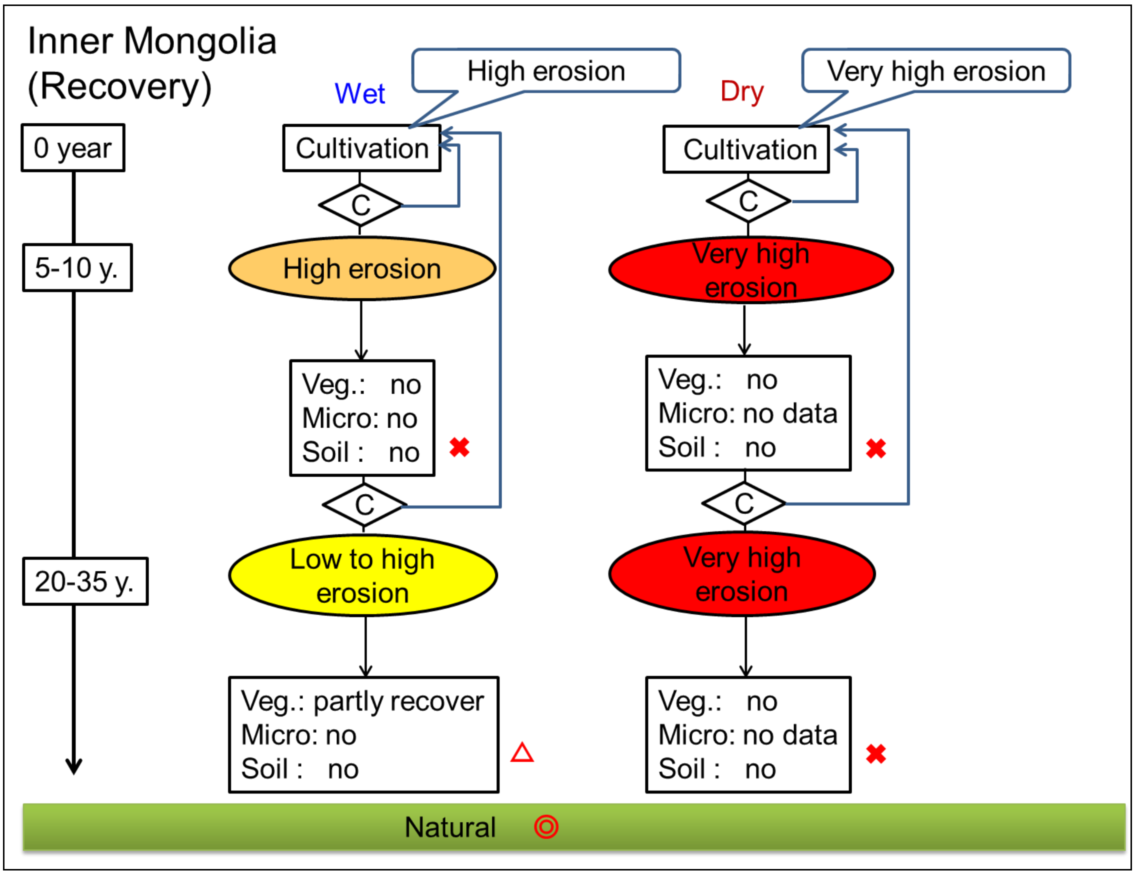

Fig. 7-2 Recovery process of vegetation, soil and microorganisms in Inner Mongolian abandoned areas. The numbers indicate the number of years that cultivation abandoned.

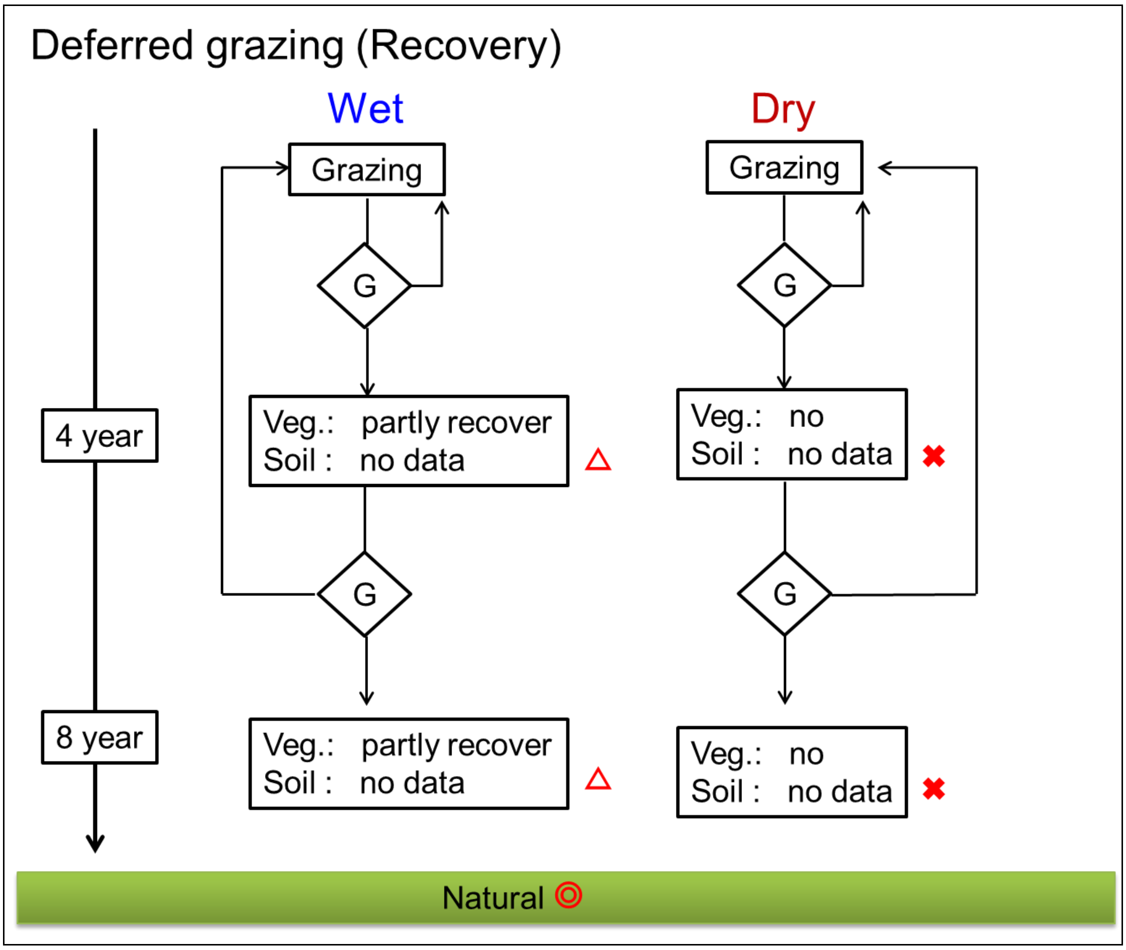

Fig. 7-3 Recovery process of vegetation and soil in deferred spring grazing area. The numbers indicate the number of years that implemented deferred grazing systems.

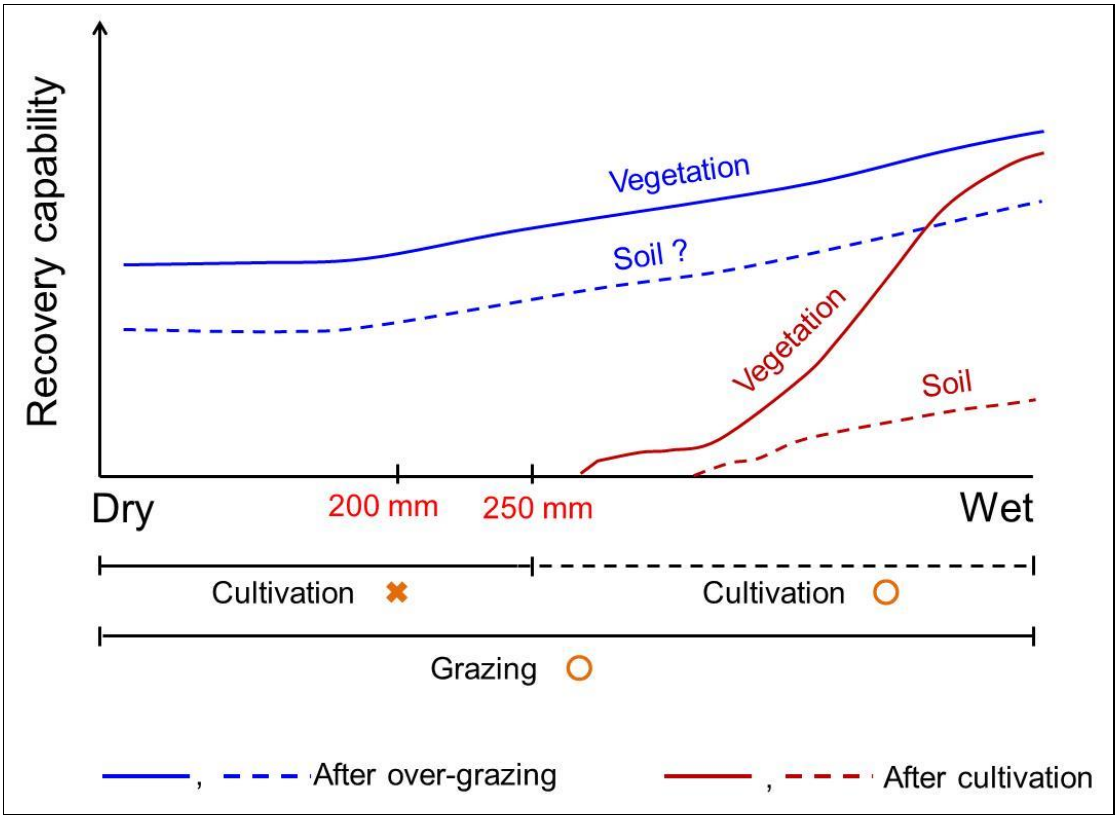

Fig. 7-4 Recovery capability of steppe

Summary

The grasslands in the semi-arid regions of the Eurasian Continent are known as the steppe. In recent years there has been great concern about increases in steppe soil degradation, vegetation degeneration, and soil erosion due to the effects of grazing and increasing cropland coverage driven by population growth. A range of measures have been introduced in different countries experiencing these pressures in order to protect grassland ecosystems, with varying levels of application and enforcement. These include banning or deferring grazing to control overgrazing and forcing the abandonment of croplands to mitigate the effects of excessive cultivation. These measures are expected to promote the recovery of grassland ecosystems. However, they do not always lead to the recovery of vegetation and soil at the targeted sites. We therefore need a better understanding of the recovery of vegetation and soil at sites where these policies are enforced.

This thesis describes field studies conducted in abandoned grazing areas and croplands in Kazakhstan, and abandoned croplands of Inner Mongolia to characterize their vegetation and soil properties. The objectives were to investigate the recovery patterns for soil and vegetation in the area and to clarify the effects of deferred grazing, which has been one of the preferred policies for promoting vegetation recovery in steppe ecosystems.

In the steppe regions of Kazakhstan, data were collected from a total of 35 plots across six study sites using phytosociological methods. It was found that the vegetation and soil at abandoned grazing sites and croplands took at least 13 years to start recovering. In some abandoned grazing areas, the recovery of the vegetation was incomplete even after 27 years, and there was no evidence of soil recovery. In abandoned croplands, vegetation recovery was observed in sites abandoned for 50 years but soil recovery was not achieved. The recovery of vegetation was found to depend on precipitation, and soil recovery was poor in dry regions. This demonstrates that the speed of vegetation and soil recovery depend on land use patterns and precipitation levels. The main factors that hinder grassland recovery are the slow soil recovery observed across the study area as a whole and the particularly slow recovery of soil and

vegetation in the dry regions.

It has been demonstrated that the level of soil erosion in abandoned croplands is much higher than in natural grassland, which will have significant effects on vegetation and soil recovery at these sites. To quantify this erosion, ^{137}Cs levels were measured at sites in Inner Mongolia. In wet regions where cultivation had been abandoned for 20 years, partial recovery of vegetation was observed but there was no such recovery of soil microorganisms or soil physicochemical properties. In dry regions where cultivation had been abandoned for 20 years, no recovery of vegetation or soil was observed. Soil erosion was identified as a key limiting factor in the inhibition of recovery. The rate of soil erosion increased with the length of time for which the plot was cultivated and was higher at recently abandoned sites and those with low precipitation.

One key aim of this work was to determine whether deferred grazing policies are effective at promoting grassland recovery under different precipitation conditions in over-grazed areas of Inner Mongolia. Based on a two-way ANOVA, vegetation recovery will deferred grazing will promote vegetation recovery in wetter regions but not dry ones. That is to say, the capacity for recovery after deferred grazing depends on precipitation. The results obtained suggest that recovery in dry regions will not occur for at least 8 years after site abandonment.

Overall, the results obtained clearly show that over-grazing and cultivation are the main reasons for steppe desertification. The rates of soil erosion in abandoned croplands were higher than in untouched steppe sites. It was clear that the extent of recovery for both vegetation and soil nutrient content increased with the time since abandonment, but no clear trend towards recovery was observed at low precipitation sites. This demonstrates the impact of the desertification crisis. In order to protect grassland ecosystems, it will be necessary to completely prohibit inappropriate cultivation, and to impose the use of grazing systems that are appropriate for the prevailing conditions, particularly the region’s level of precipitation.

Acknowledgements

I would like to express my deepest appreciation to Professor Takashi Kamijo, Professor Emeritus Toru Nakamura, Professor Tatsuhito Fujimura, Professor Kenji Tamura, Associate Professor Tatsuyuki Seino and Dr. Kiyokazu Kawada, Faculty of Life and Environmental Sciences, University of Tsukuba, for their giving me so much brilliant suggestions and patient guidance in my research career.

For assistance in the field, I sincere thank Dr. Sitbaeva Gulnar for her help in identifying the species in Kazakhstan. Prof. Subdaa, Dr. Burinchichig, Dr. Gova and Dr. Jangabaatar, who helped with the field work in Inner Mongolia.

Specially thanks to Prof. Dr. Wuyunna. Thank her for recommending me to study in Hulunbuir grassland and providing of excellent platforms for my research. I also specially thanks to Dr. Hideshi Fujiwara of National Institute for Agro-Environmental Sciences for his kind comments and support on this study.

Especially thank the owners of Hugjilt, Adiya and Altanbulag in Xilingol grassland who provided support and field facilities.

Many thanks are due to Dr. Yunxiang Cheng. She gave me many valuable advices and guidance in my research activities. Her great efforts in revising all my manuscripts are highly indebted.

Thanks Mr. Takashi Kanda and Mr. Huhe for his help in conducting the soil physicochemical and microorganism analysis. Thanks Ms. Harumi Morita for her help in conducting the analysis of radioactivity concentration in soil. Many thanks are also due to Kazuo Obata, Mayu Kurosu and Takumi Kurokawa who conducted field work together and offered valuable comments on the study.

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Appendix Community table of steppe vegetation in Kazakhstan

Ⅰ. *Festuca sulcata-Stipa capillata* community

A. *Helictotrichon desertorum* first lower unit

a. *Pulsatilla patens* second lower unit

b. *Stipa lessingiana* second lower unit

c. Typical second lower unit

B. *Helichrysum arenarium* first lower unit

d. *Euphorbia virgata* second lower unit

e. *Silene wolgensis* second lower unit

Ⅱ. *Artemisia vurgaris-Agropyron repens* community

	I																														Ⅱ				
	A																		B																
	a					b						c							d							e									
Running No.	1	2	3	4	5	7	8	9	10	11	12	13	6	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35
Stand No.	K2	K1	k4	k6	k5	k15	k16	k18	k19	k20	k17	k8	K3	k9	k7	k10	k13	k14	k21	k22	k23	k24	k25	k26	k27	k31	k32	k33	k34	k35	k30	k38	k29	k37	k36
Altitude(m)	394	398	391	397	395	425	427	428	428	427	427	346	396	347	346	346	342	342	323	326	325	325	324	323	320	423	424	421	419	428	252	253	253	256	257
Slope direction	E25S	E10S	-	-	N	-	-	-	-	-	S10E	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	N85W	-	N5W	N35W	N	N30W	N10W
Inclination(°)	3	4	-	-	2	-	-	-	-	-	7	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.5	-	0.5	1	0.5	1.5	1
Research area	Ereymentau	Ereymentau	Ereymentau	Ereymentau	Ereymentau	Klyuchi	Klyuchi	Klyuchi	Klyuchi	Klyuchi	Klyuchi	Ivanovskoe	Ereymentau	Ivanovskoe	Ivanovskoe	Ivanovskoe	Ivanovskoe	Ivanovskoe	Alga	Alga	Alga	Alga	Alga	Alga	Alga	Novorossiyskoy	Novorossiyskoy	Novorossiyskoy	Novorossiyskoy	Novorossiyskoy	Olke	Olke	Olke	Olke	Olke
Soil type	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	chernozems	kastanozems	kastanozems	kastanozems	kastanozems	kastanozems	kastanozems	kastanozems	umbrisols	umbrisols	umbrisols	umbrisols	umbrisols	chernozems	chernozems	chernozems	chernozems	chernozems
Survey area (m^2)	9	9	16	36	25	9	4	16	25	25	25	12	12	4	4	12	4	9	25	16	9	20	25	100	225	25	25	16	25	225	50	100	16	100	100
Height of 1st herb layer (m)	0.45	0.50	0.60	0.70	0.25	0.50	0.40	0.44	0.65	0.70	0.70	0.65	0.55	0.65	0.70	0.75	0.80	0.64	0.75	1.00	0.90	0.80	1.05	1.10	0.90	0.90	0.70	0.80	0.70	0.80	1.00	0.40	0.57	0.45	0.90
Height of 2nd herb layer (m)	0.15	0.20	0.12	0.14	-	0.14	0.14	0.07	0.25	0.30	0.45	0.25	0.20	0.40	0.50	0.40	0.40	0.27	0.20	0.30	0.40	0.45	0.60	0.50	0.40	0.37	0.35	0.27	0.27	0.30	0.15	0.08	0.10	0.08	0.12
Height of 3rd herb layer (m)	-	-	-	-	-	-	-	-	-	-	0.17	-	0.05	-	0.10	0.10	0.20	-	-	0.10	0.30	0.06	0.08	0.10	0.10	0.07	0.06	0.05	0.06	0.06	-	-	-	-	-
Coverage of 1st herb layer (%)	80	70	70	75	60	80	70	90	20	30	60	40	80	50	40	60	40	30	40	50	20	20	20	40	50	10	10	10	5	30	25	70	25	70	10
Coverage of 2nd herb layer (%)	50	40	50	40	-	40	40	30	90	90	70	95	30	70	60	50	60	100	50	40	40	40	40	60	50	50	50	50	40	60	60	50	50	30	60
Coverage of 3rd herb layer (%)	-	-	-	-	-	-	-	-	-	-	10	-	10	-	60	50	50	-	-	30	50	10	10	30	50	60	50	50	40	60	-	-	-	-	-
Number of species	32	26	32	22	23	18	21	26	28	34	29	17	18	21	30	20	12	22	22	18	15	15	14	19	20	16	14	12	19	18	19	26	26	24	26
Differential species of *Festuca sulcata-Stipa capillata* community																																			
Festuca sulcata	2	2	3	2	4	+	2	2	2	2	1	4	3	2	3	2	.	2	3	2	3	2	2	3	3	3	3	3	2	3	.	+	.	.	.
Stipa capillata	2	.	2	4	1	2	2	4	4	4	+	3	1	3	3	4	3	.	3	3	2	2	2	2	2	2	2	2	1	2	.	.	.	.	.
Artemisia frigida	1	+	1	+	1	.	2	+	+	+	.	2	1	1	3	2	1	.	1	2	1	1	2	2	1	1	2	+	2	2	.	+	.	+	+
Carex physodes	+	2	1	1	1	2	1	+	+	1	1	3	.	1	.	2	3	2	1	.	.	.	.	.	2	.	2	.	+	.	.	.	.	.	.
Galium verum	1	1	+	1	1	+	1	1	1	1	+	+	1	+	+	1	2	+	.	.	.	.	.	+	.	.	+	.	.	.	.	.	.	.	.
Dianthus campestris	+	+	+	+	+	.	.	+	+	.	.	+	.	.	.	+	.	.	+	+	.	.	2	.	.	+	+	+	+	+	.	.	.	.	.
Koeleria cristata	+	+	+	+	1	.	+	1	.	+	.	+	.	.	1	+	.	.	+	1	1	1	1	2	2	.	.	.	.	.	.	.	.	.	.
Poa pratensis	.	.	.	.	.	+	.	+	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	2	.	3	.	3	.	3	.	.	.	.	.
Differential species of *Helictotrichon desertorum* first lower unit																																			
Helictotrichon desertorum	5	4	4	2	.	3	3	3	3	3	.	.	4	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Phlomis tuberosa	+	.	2	.	+	+	+	2	+	1	.	.	+	2	+	+	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Aster alpinus	+	+	+	.	+	+	+	+	+	+	+	+	1	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Plantago lanceolata	+	.	+	.	.	+	2	+	+	1	+	.	.	2	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Artemisia schrenkiana	.	.	2	.	+	+	.	.	.	+	+	1	.	3	3	1	2	.	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Thymus stepposus	2	2	.	.	.	.	.	+	2	+	1	.	.	2	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	+	.	.	.	.	.
Pulsatilla patens second lower unit																																			
Pulsatilla patens	3	2	1	2	2	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Veronica incana	1	+	+	+	+	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.
Acroptilon repens	2	1	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1
Artemisia nitrosa	1	2	1	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Hieracium umbelliferum	+	+	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.

Appendix Continued

Ⅰ. *Festuca sulcata-Stipa capillata* community
A. *Helictotrichon desertorum* first lower unit
a. *Pulsatilla patens* second lower unit
b. *Stipa lessingiana* second lower unit
c. Typical second lower unit
B. *Helichrysum arenarium* first lower unit
d. *Euphorbia virgata* second lower unit
e. *Silene wolgensis* second lower unit
Ⅱ. Artemisia vurgaris-Agropyron repens community

	I																														Ⅱ				
	A																		B																
	a					b						c							d							e									
Differential species of *Stipa lessingiana* second lower unit																																			
Stipa lessingiana	.	.	.	.	.	2	2	2	3	3	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	+	.	.	.	.	.
Ferula soongarica	.	.	.	.	.	2	1	2	+	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Inula hirta	.	.	.	.	.	+	.	+	+	2	3	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Differential species of *Helichrysum arenarium* first lower unit																																			
Helichrysum arenarium	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	2	+	1	1	2	1	1	1	1	+	.	.	.	.	.
Potentilla acaulis	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	2	2	.	+	+	3	3	3	2	1	.	.	.	.	.
Poa stepposa	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	.	1	2	2	.	2	.	3	.	2	.	.	.	.	.	.
Astragalus physodes	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	+	+	.	.	+	1	1	+	+	1	.	.	.	.	.
Artemisia scoparia	.	.	+	.	.	.	.	.	.	.	+	.	.	.	+	.	.	.	2	1	1	2	2	2	2	.	.	.	1	.	.	.	.	.	.
Differential species of *Euphorbia virgata* second lower unit																																			
Euphorbia virgata	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	+	1	+	+	+	.	.	.	.	.	.	.	.	.	.
Silene odoratissima	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	+	1	+	1	.	.	.	.	.	.	.	.	.	.
Differential species of *Silene wolgensis* second lower unit																																			
Silene wolgensis	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	+	+	+	.	.	+	.	.
Koeleria glauca	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	.	2	+	.	.	.	.	+
Artemisia arenaria	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	1	+	+	.	.	.	.	.	.
Centaurea ruthenica	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	1	.	+	.	.	.	.	.	.
Differential species of *caragana* facies																																			
Caragana pumila	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Differential species of *Sanguisorba officinalis-Vicia cracca* community																																			
Sanguisorba officinalis	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Vicia cracca	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Trifolium lupinaster	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Chamaenerium angustifolium	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Senecio erucifolius	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Filipendula ulmaria	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Calamagrostis sp	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Viola sp	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Scutellaria krylovii	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Gentiana pneumonanthe	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Lathyrus pratensis	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Sonchus arvensis	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Differential species of *Artemisia vurgaris-Agropyron repens* community																																			
Artemisia vurgaris	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	4	3	3	2	3
Agropyron repens	.	.	.	.	.	.	+	.	+	.	.	.	2	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	2	4	2	4	1
Cichorium intybus	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	2	1	1	2	2
Helianthus annuus	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	2	1	+	+
Kochia prostrata	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	+	.	.	.	.	.	.	+	.	2	+	+	+	+
Carduus stenocephalus	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	2	+	+	+	+
Atriplex aucheri	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	+	1	+	+	+
Ceratocarpus arenarius	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	2	.	1	1	3
Orobanche arenaria	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	.	2	1
Fagopyrum sagittatum	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	+	+	1
Convolvulus arvensis	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	+	+
Cuscuta campestris	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	2	+	2	+
Raphanus sativus	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	+	+
Matricaria recutita	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	+	.

Appendix Continued

Ⅰ. *Festuca sulcata-Stipa capillata* community
A. *Helictotrichon desertorum* first lower unit
a. *Pulsatilla patens* second lower unit
b. *Stipa lessingiana* second lower unit
c. Typical second lower unit
B. *Helichrysum arenarium* first lower unit
d. *Euphorbia virgata* second lower unit
e. *Silene wolgensis* second lower unit
Ⅱ. Artemisia vurgaris-Agropyron repens community

	I																														Ⅱ				
	A																		B																
	a					b						c							d							e									
Companions																																			
Achillea millefolium	+	.	.	.	.	.	.	+	.	+	+	+	.	+	+	1	1	.	+	+	.	.	.	+	.	+	.	.	1	+	2	3	2	1	+
Medicago falcata	+	+	+	.	+	2	.	2	1	+	1	.	.	2	2	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	+	1	+	.	+
Polygonum aviculare	+	+	.	.	.	.	.	.	.	.	.	+	.	+	2	+	.	+	+	+	.	.	.	+	+	.	.	.	.	.	2	+	1	+	+
Artemisia lercheana	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	1	+	+	+	+	+
Erysimum marschallianum	.	.	.	.	.	.	.	.	.	.	.	.	+	.	+	.	.	.	.	+	.	.	2	.	.	.	.	.	.	.	+	+	+	+	+
Potentilla erecta	+	1	+	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	+	.	+	+	.	+	+	1	.	.	.	.	.
Scaligeria setacea	+	.	.	.	.	.	2	.	.	+	.	2	.	.	.	.	.	.	.	+	+	+	.	+	1	.	.	.	.	.	.	.	.	.	.
Potentilla impolita	.	.	.	.	.	+	+	+	+	+	+	.	.	2	+	+	+	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Veronica longifolia	+	.	+	.	+	+	.	+	+	+	+	.	1	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Linaria vulgaris	.	.	.	.	.	.	.	1	+	+	.	.	.	.	.	.	.	.	+	+	+	.	.	.	.	.	.	+	.	.	+	+	+	+	.
Potentilla bifurca	.	+	+	.	+	+	.		+	+	.	.	+	.	.	.	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Eryngium planum	.	.	+	.	.	.	.	.	+	+	1	+	.	+	+	+	+	+	.	.	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.
Filipendula hexapetala	+	+	.	.	+	.	.	.	.	.	1	.	2	.	.	+	.	3	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Caragana frutex	2	+	.	1	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	3	3	.	.	.	.	.	.	.	.	.	.
Onosma tinctorum	+	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Tanacetum vulgare	+	+	.	.	+	.	.	.	.	.	+	.	1	.	.	.	.	2	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Scabiosa ochroleuca	.	+	+	.	.	+	.	.	.	1	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Limonium gmelinii	.	.	.	.	.	.	+	.	.	+	.	.	.	+	2	1	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Spiraea hypericifolia	.	.	.	.	.	.	.	.	.	.	.	+	2	.	.	.	2	2	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.
Silene gebleriana	.	.	.	.	.	.	.	.	.	+	.	.	.	+	+	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Linosyris tatarica	.	.	.	.	.	.	+	+	.	+	.	+	.	.	.	.	.	+	.	.	.	.	.	.	+	.	.	.	+	.	.	.	.	.	.
Bromus inermis	.	.	1	.	.	.	.	.	.	.	3	.	+	.	.	.	.	.	.	.	.	.	.	2	.	.	.	.	.	.	.	.	.	.	.
Plantago major	.	.	+	.	.	.	.	.	.	.	.	.	.	.	1	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Tragopogon tatarica	.	.	+	.	+	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Inula britannica	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Leymus angustus	.	.	.	.	.	.	.	.	.	+	.	1	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Asperula humifusa	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Phragmites australis	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Teucrium scordium	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Appendix Continued

Ⅰ. *Festuca sulcata-Stipa capillata* community
A. *Helictotrichon desertorum* first lower unit
a. *Pulsatilla patens* second lower unit
b. *Stipa lessingiana* second lower unit
c. Typical second lower unit
B. *Helichrysum arenarium* first lower unit
d. *Euphorbia virgata* second lower unit
e. *Silene wolgensis* second lower unit
Ⅱ. Artemisia vurgaris-Agropyron repens community

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Pedicularis achilleifolia	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Tripleurospermum inodorum	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Phlomis sberosa	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Taraxacum officinale	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
Seratula coronata	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
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Astragalus tauricus	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.	.	.	.	.	.	.	.	.	.
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Polygonum convolvulus	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.

Cover Class; 5: any number, coverage (%) between 75-100; 4: any number, coverage (%) between 50-75; 3: any number, coverage (%) between 25-50; 2: any number, coverage (%) between 10-25; 1: any number coverage (%) between 1-10; +: any number, coverage (%) between 0-1 (Braun-Blanquet 1964; Mueller-Dombois and Ellenberg 1974).