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Effects of grassland restoration programs on ecosystems in arid and semiarid China

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ABSTRACT

We explored the ecological effects of grassland restoration programs using satellite imagery and field plots sampling data and analyzing the patterns and mechanisms of land cover change and vegetation activities in arid and semiarid China during the period from 1982 to 2008. The grassland cover in the 1980s, 2000 and 2005 was compared before and after the restoration programs. The variability of net primary production (NPP) and rain use efficiency (RUE) were analyzed as indicators of vegetation productivity. Our study showed that changes in grassland cover were closely related to the relative area of farmland, with increases in grassland being caused by returning farmland to grassland and decreases being caused by reclamation for agriculture. The results of NPP and RUE measurements over the past 30 years showed systematic increases in the area of grassland in most regions, especially from 2000 to 2008. This fact was reflected by intensified vegetation activity and cannot be completely explained by the warmer and wetter climate, which suggested a contribution from restored, ungrazed grasslands. Our analysis indicates that both vegetation activity and grassland cover increased in regions in which grassland and rangeland restoration programs were implemented.

1. Introduction

Grassland covers more than 40% of China's total territory (Ni, 2002; Fan et al., 2008), and nearly 80% of the grassland occurs in arid and semiarid regions (NBSC, 2009). Its ecological and social value in western China is higher than that of forests or wetlands (Jiang et al., 2007). However, sand storms, desertification and eco-refugees caused by the degradation of grassland vegetation (Unkovich and Nan, 2008) have had huge impacts on grassland animal husbandry and ecological security in China. Some studies show that this is a serious problem, while others indicate that grasslands have largely been restored. A range of studies have examined grassland degradation and desertification in arid and semiarid areas (Zha and Gao, 1997; Wu and Ci, 1998; Runnström, 2000; Long, 2000; Zhong and Qu, 2003; Liu et al., 2005; Yang et al., 2005; Huang and Siegert, 2006; Yang et al., 2007; X.M. Wang et al., 2007). Most studies concluded that parts of northern China have undergone a reversion of the desertification since the 1980s owing to increasing vegetation growth, or vegetation "greenness" indexes (Fang et al., 2005; Piao et al., 2005; Liu et al., 2005), but have decreased after the 1990s due to drought stress strengthened by warming and reduced precipitation (Peng et al., 2011). This pattern cannot be fully explained by the increasing rainfall in most of the arid and semiarid regions (Runnström, 2000; Zhang et al., 2003; Fang et al., 2005; Piao et al., 2005) and may be attributed to climatic change coupled with ecological conservation efforts. Some studies of the ecological restoration programs themselves showed positive effects on vegetation coverage (Xu et al., 2006; Liu et al., 2008), while others indicated negative effects on biodiversity (Cao, 2008).

During the past several decades, the Chinese government has made significant efforts to combat and prevent grassland desertification by increasing vegetation coverage and repairing shifting sand dunes and soil erosion (Runnström, 2000; Yang et al., 2005) through ecological programs such as the Three-North Shelter Forest Program starting in 1978 and the Combating of Sand Desertification Project beginning in 1991. However, extensive efforts to plant trees in many

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arid and semiarid regions have caused environmental deterioration because trees consume excessive amounts of the limited soil moisture, reduce overall vegetation cover, and lead to more severe wind erosion (Liu and Diamond, 2005; Cao, 2008). Since 2000 and 2003, to prevent water and soil erosion and land desertification and to protect and restore the grasslands, the Returning Farmland to Grassland and Returning Rangeland to Grassland programs were promoted, respectively. Rather than focusing on a forestation approach to ecosystem restoration, it would be more effective to focus on recreating the natural ecosystems that are more suitable to the arid and semiarid regions (Cao, 2008).

All large-scale programs should undergo "reality checks" on their effects by assessing their success or failure. Therefore, appropriate monitoring is necessary for assessing the successful implementation of conservation projects (Tallis et al., 2008; Daily and Matson, 2008). This paper asks the following questions: (1) Is there long-term, increased vegetation activity in arid and semi-arid China? (2) How have climate change and ecological restoration efforts influenced the ecosystem? (3) Have the grassland restoration programs that have been promulgated at national and local levels been effective for grassland ecosystem conservation? We address these questions by examining the patterns and mechanisms of grassland cover change and vegetation activity using satellite imagery, field investigation plots and meteorological datasets. The grassland cover data in the 1980s, 2000 and 2005 were used to compare conditions before and after the restoration programs. The vegetation activity was assessed based on an analysis of a time series of net primary production (NPP) and rain use efficiency (RUE) presented as an index available from 1982 to 2008.

2. Materials and methods

2.1. Study area

Arid and semi-arid regions in China cover a very large area, of which 35.6% is grassland, 23.75% is harsh desert, 11.24% is sandy desert and 5.77% is bare rock (Fig. 1). In this study, the average Thornthwaite moisture index (Im) between 1960 and 2008 was applied to define the boundaries of arid and semi-arid regions, which could effectively indicate climatic water conditions and has been used by UNESCO. Regions with an Im below −40 were classified as arid, and an Im between −40 and −20 were classified as semi-arid. The areas affected by the Returning Farmland to Grassland Program and the Returning Rangeland to Grassland Program account for more than two-thirds of the total arid and semi-arid territory.

2.2. Meteorological datasets and their analysis

The data from the 198 meteorological stations for the years between 1960 and 2008 were assembled by the China Bureau of Meteorological Administration and included daily maximum and minimum temperatures, daily total water equivalent precipitation, canopy net radiation, wind velocity, saturation vapor pressure and actual water vapor pressure. For each meteorological station, we calculated a moisture index by combining the improved Penman–Monteith equation and the Thornthwaite Moisture Index for each year, as follows,

$$\text{PET} = \frac{0.408\Delta(R_n - G) + \gamma \dfrac{900}{T+273} u_2(e_s - e_a)}{\Delta + \gamma(1 + 0.34u_2)} \quad (1)$$

$$\text{Im} = 100\% \times \left(\frac{\text{precipitation}}{\text{PET}} - 1\right) \quad (2)$$

In Equations (1) and (2), PET is the potential evapotranspiration (mm day^{-1}), R_n is the canopy net radiation (MJ m^{-2} day^{-1}), G is the soil heat flux (MJ m^{-2} day^{-1}), T is the air temperature at 2 m height (°C), u_2 is the wind velocity at 2 m height (m s^{-1}), e_s and e_a are the saturation vapor pressure and actual water vapor pressure, respectively (kPa), Δ is the curve slope of the saturation vapor

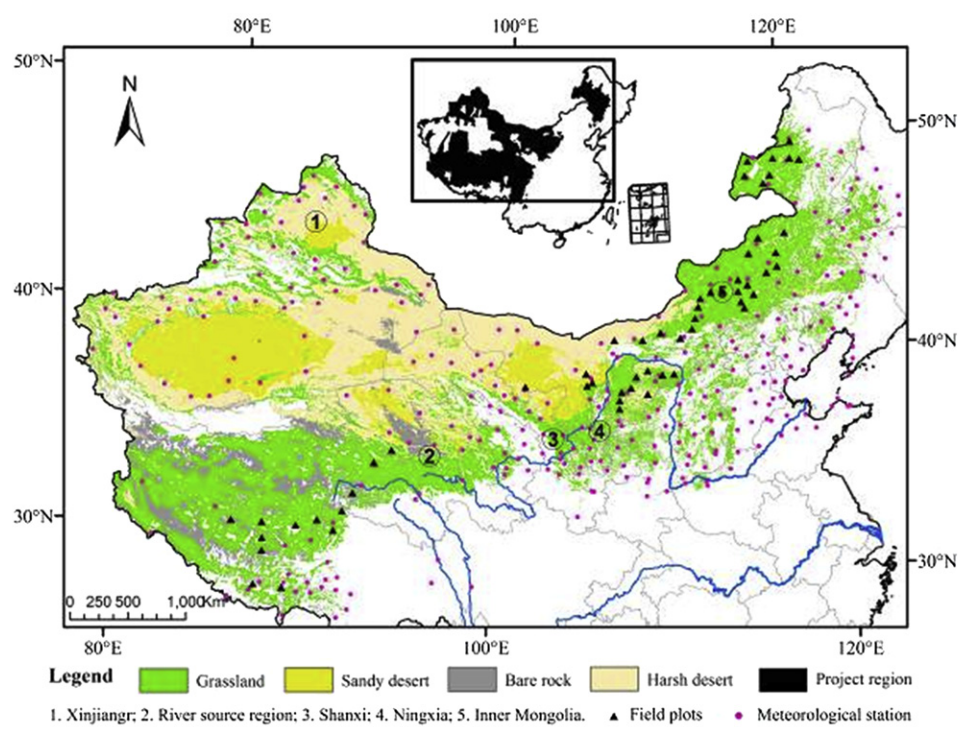

Fig. 1. Distribution map of ecosystems, project regions, meteorological stations, and field plots in arid and semiarid China.

pressure (kPa °C^{-1}), and γ is the constant of the psychrometer (kPa °C^{-1}).

The rainfall datasets for well-distributed sites from 1980 to 2008 were interpolated using ANUSPLIN software, which considered the impacts of terrain factors and was suitable for long-term data observation. We used a spatial resolution of 1 km for all of China and a clipped Moisture Index to analyze the arid and semiarid regions. Some errors might result from the interpolation of the Tibetan plateau data owing to the limited number of available stations. The average annual precipitation and temperature from 1980 to 2008 was compared with the long-term average trends at the stations (1960–2008) to analyze the potential existence of trends in rainfall and temperature data.

2.3. Land cover change datasets

For the historical transitions in grassland cover, data and methods were taken from and summarized by Liu et al. (2005). Human–machine interaction interpretations were performed on remote sensing images using ESRI's ArcGIS 9.3 platform. Land use and land cover datasets were produced for 1980, 2000 and 2005. The hierarchical classification system of 25 land-cover classes (Liu et al., 2005) was further grouped into six aggregated classes: croplands (i.e., land cultivated for crops), woodlands (i.e., land growing trees, including arbors, shrubs, bamboo and trees for forestry), grasslands (i.e., land covered by herbaceous plants), water bodies (i.e., land covered by natural water bodies or land with facilities for irrigation and water containment), unused land (i.e., land that is not put into practical use or that is difficult to use) and built environment areas (i.e., land used for urban and rural settlements, factories and transportation facilities). Grasslands comprise three vegetation density-dependent types: dense, moderate and sparse grasslands. Finally, we detected and recorded the transition areas between grasslands and other ecosystems during the 1980s until 2000 and 2000–2005.

2.4. Grassland vegetation growth estimation

The remote sensing data used to estimate vegetation productivity in this study were a time-series of 16-day resampling from the AVHRR (The Advanced Very High Resolution Radiometer, 1 km) from 1982 to 2003 and from MODIS (Moderate Resolution Imaging Spectroradiometer, 1 km) from 2000 to 2008. The data during the overlap period of 2000–2003 were calibrated and matched by pixels, based on field plot information. The Global Production Efficiency Model (GLO-PEM) that has been developed uses satellite-derived variables, including the fraction of photosynthetically active radiation (FPAR), and meteorological variables for estimating the NPP. GLO-PEM consists of linked components that describe the processes of canopy radiation absorption, utilization, autotrophic respiration, and the regulation of these processes by environmental factors (Prince and Goward, 1995; Goetz et al., 1999; Cao et al., 2003). The estimation of NPP in the GLO-PEM model can be described as follows:

$$\mathrm{NPP} = [(S*\mathrm{FPAR}*\mathrm{LUE}) - R] \quad (3)$$

where S is the incident photosynthetically active radiation (PAR). FPAR is calculated as a linear function of the Normalized Difference Vegetation Index. LUE is the light utilization efficiency of the absorbed PAR by vegetation in terms of gross primary production and is determined based on the biochemical process of photosynthesis and the stomatal conductance, which are in turn affected by the atmospheric CO_2 concentration, temperature, water vapor pressure deficit, and soil moisture. R is autotrophic respiration calculated as a function of the standing aboveground biomass, the air temperature, and the rate of photosynthesis.

NPP was affected by rainfall more strongly than were any of the other factors, yet rainfall in arid and semiarid regions typically varies significantly between years. Therefore, without taking the rainfall into account, NPP values alone might be an unwarranted extrapolation from the currently available data (Prince et al., 1998; Huxman et al., 2004; Hein and Ridder, 2006; Prince et al., 2007). RUE equals the ratio of NPP to rainfall and has been implicated as a practical and objective indicator in many NPP analyses with respect to rainfall, using satellite-derived data to analyze the degradation in arid and semiarid ecosystems (Le Houérou et al., 1988; Prince et al., 1998; Prince, 2002; Hein and Ridder, 2006; Prince et al., 2007; Bai et al., 2008; Hu et al., 2010).

To investigate the geographical differences, the linear slopes of precipitation, NPP and RUE over time were calculated using the least-squares method for each pixel during the 1980s, 1990s and from 2000 to 2008. The slope of the linear function was used as the indicator of an average year-to-year trend for each pixel: a positive slope indicated an overall increase and vice versa. The slopes were calculated as follows:

$$S = \frac{\sum_{i=1}^{n} m_i X_i \frac{1}{n} * \sum_{i=1}^{n} m_i * \sum_{i=1}^{n} X_i}{\sum_{i=1}^{n} m_i^2 \frac{1}{n} * \left(\sum_{i=1}^{n} m_i\right)^2} \quad (4)$$

where X_i is the value of the precipitation, NPP or RUE for year i, $i = 1, 2, 3, \ldots n$ and m_i is the sequence number of the year, $m_1 = 1$, $m_2 = 2$, $m_3 = 3, \ldots m_n = n$.

2.5. Field plot data

To collect the localized key parameters for the model and evaluate the simulated NPP and RUE, the above-ground NPP (ANPP) and the below-ground NPP (BNPP) were collected at 78 sites along the China Grassland Transect (a 4500 km grassland transect) (Fan et al., 2008) and at 113 representative grassland field plots in the Returning Rangeland to Grassland project region in the Sanjiangyuan Region, located on the northeastern Tibetan Plateau. The average trends for the simulated NPP and RUE for the 113 project plots were extracted to compare them with those from the non-project region to assess the effectiveness of the grassland restoration programs. For each field plot, 5 1 m × 1 m samples were placed in a 40 m × 50 m area, the community-level means for vegetation cover, numbers, height and species were recorded, and the below-ground and above-ground biomass were measured. Above-ground biomass was harvested to ground level and dried for weighing. Below-ground biomass was sampled mostly by taking either 9–36 soil cores 3.1 cm in diameter at 10–30 cm-deep layers within each quadrant or obtaining samples by digging out 25 × 25 cm pits. Roots contained within the excavated soil were separated with water through a 0.3-mm mesh sieve and dried in an oven at 80 °C for 24 h.

3. Results

3.1. Grassland cover change before and after 2000

From the 1980s to 2000, grassland decreased by 2.59 million hm^2 in arid and semiarid China at a rate of 0.22 million-hm^2 yr^{-1}. The primary driving factor for this decrease was that grassland was brought under cultivation, which accounts for 60% of

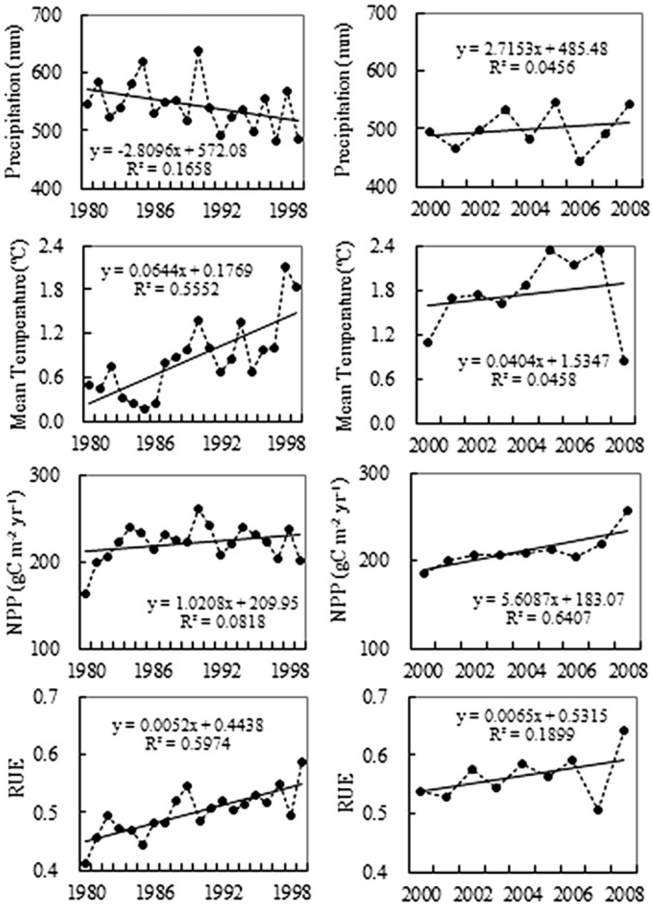

Fig. 2. Interannual changes in annual precipitation, annual mean temperature, NPP, and RUE in arid and semiarid China from 1980 to 2008.

the total decrease and occurred mainly on pasture lands in eastern Inner Mongolia, on the margin of every oasis in Xinjiang and along agro-pastoral ecotones on the Loess Plateau and in northern China (see Supplementary Material).

From 2000 to 2005, grassland cover change showed a net decrease of 0.69 million hm^2 at a rate of 0.14 million hm^2 yr^{-1}. Decreasing grassland was primarily converted to farmland, which accounts for more than 48% of the grassland transformation along the margins of oases in Xinjiang and the central section of Inner Mongolia. In contrast, increases in grassland area resulted primarily from farmland, especially along the agro-pastoral ecotones of the Loess Plateau, due to the policy of returning farmland to grassland (see Supplementary Material).

3.2. Climate change over the past 30 years

Precipitation trends for each meteorological station were calculated and presented as cycles of different sizes, which were classified into eight categories. Statistical analysis showed that among all of the meteorological stations in the arid and semiarid regions, the annual precipitation at 61% (186 of 306 stations) of the total stations had decreased (Fig. 3). Average annual precipitation showed a slightly declining trend from 1980 to 2000 (-2.810 mm yr^{-1}) and an increasing trend from 2000 to 2008 (2.715 mm yr^{-1}). Nearly all of the stations showed an increased annual mean temperature. The trend was obviously rising from 1980 to 2000 (0.064 °C yr^{-1}) and from 2000 to 2008 (0.040 °C yr^{-1}) (Fig. 2).

3.3. Net primary production variations from 1982 to 2008

Year-to-year variations in grassland NPP in arid and semiarid China showed an increasing trend over the past 30 years (Fig. 2). The average rates were 1.021 g C m^{-2} yr^{-1} from 1982 to 2000 and 5.609 g C m^{-2} yr^{-1} ($R^2 = 0.641$, $p < 0.001$) from 2000 to 2008.

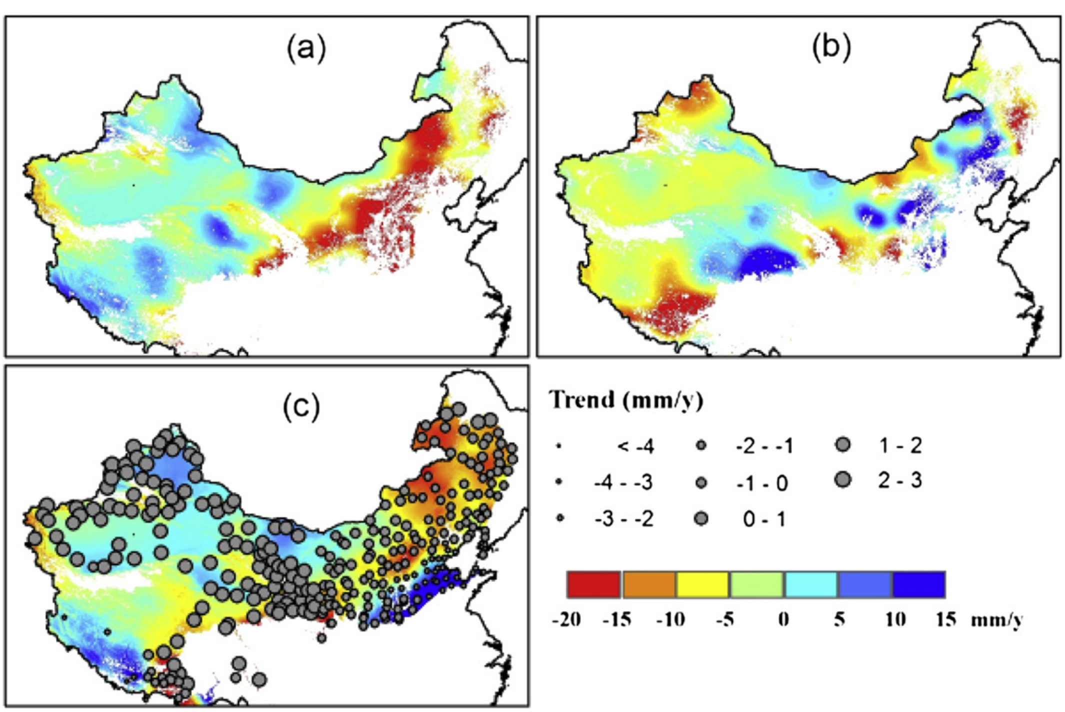

Fig. 3. Annual precipitation trends from (a) 1980 to 2000, (b) 2000 to 2008, and (c) 1980 to 2008 in arid and semiarid regions; trends of stations from 1960 to 2008 are presented as cycles.

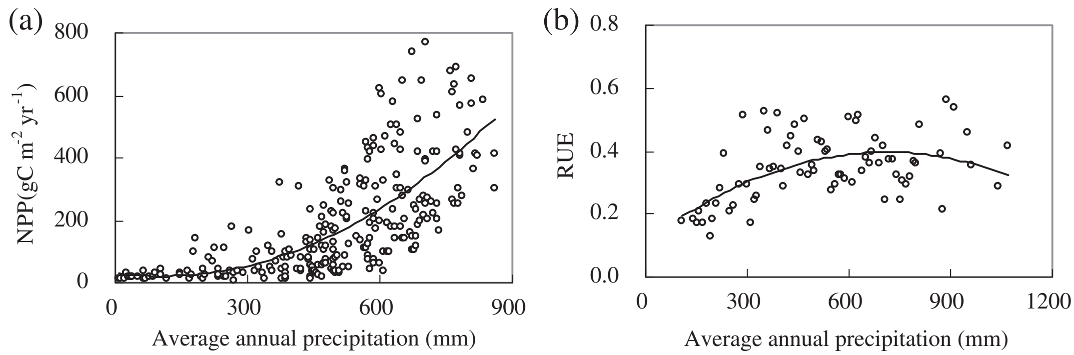

Fig. 4. Relationships between average annual precipitation and (a) NPP and (b) RUE.

Interannual variations in NPP corresponded closely with changes in climate, especially in precipitation and the length of the growing season. The observed data for annual precipitation at the weather stations and the simulated NPP have confirmed that the annual NPP was significantly positively correlated with the annual precipitation for grasslands in arid and semiarid China (Fig. 4a). From 1982 to 2000, NPP showed a significant decline, which was associated with the declining precipitation and increasing temperature. Fig. 5 shows that NPP decreased along the agro-pasture ecotone on the Loess Plateau, in eastern Inner Mongolia and along the margins of oases that underwent rapid over-reclamation from farmland and overgrazing. Since 2000, NPP increased over most of the arid and semiarid regions, along with increasing temperature and precipitation. Significant NPP increases occurred in eastern Inner Mongolia and on the Loess Plateau, especially in southern Gansu Province and northern Shanxi Province, where farmland was largely transferred to grassland. In contrast, the margins of oases in Xinjiang experienced a continued decrease in NPP, as did the central section of Inner Mongolia.

3.4. Patterns and trends of RUE during 1980–2008

Year-to-year variations of RUE in the arid and semiarid regions exhibited a slightly upward trend over the past 30 years (Fig. 2). The annual mean RUE of grasslands was 0.499 g C mm^{-1} and 0.564 g C mm^{-1}, and the average rate of increase was 0.005 and 0.007 g C mm^{-1} yr^{-1} from 1982 to 2000 and 2000 to 2008, respectively. From 1982 to 2008, the average RUE of grassland increased by 56.69%, with an annual mean increase rate of 18.14%. The RUE trends indicated a significant geographical heterogeneity during various periods. High values of RUE were generally associated with sites close to lakes, rivers, wetlands, and oases. The spatial distribution of RUE trends per pixel supported the findings, shown in Fig. 6, that the average annual RUE for most pixels in the arid and semiarid regions exhibited a positive trend since 1982. An increasing RUE accounted for 74.76% and 61.75% of the total in the arid and semiarid regions from 1982 to 2000 and 2000 to 2008, respectively. Significant increases in RUE occurred mainly along agro-pastoral ecotones on the Loess Plateau and in northern Inner Mongolia, such as in the Mu Us Sandland. However, a significantly decreased RUE accounted for 7.93% and 1.85% of the total, respectively. In most of the marginal areas near oases, the RUE sharply decreased continuously owing to over-farming and the destruction of desert vegetation over the past 30 years. The RUE in western Inner Mongolia, as well as along the lower reaches of the Heihe River, showed a decreasing trend over the past 30 years due to declining water supplies arriving from higher elevations. For some alpine grasslands on the Tibetan Plateau, the RUE showed fluctuations controlled by climate change (Fig. 6).

The relationship between annual rainfall and RUE was also examined (Fig. 4b). RUE is highest in regions where the annual rainfall is close to the average rainfall, and it is relatively low for periods of both low and high rainfall. During periods of low rainfall, relatively more water is lost through evaporation, leaving less water available for plants. Consequently, the RUE decreased. During periods of high rainfall, the RUE decreases because ecosystem productivity becomes limited by nutrients, as mentioned by Hein and Ridder (2006) and Bai et al. (2008). Plant biomass production remained high because biomass production is a function of both rainfall levels and RUE. The decrease in RUE at high rainfall levels

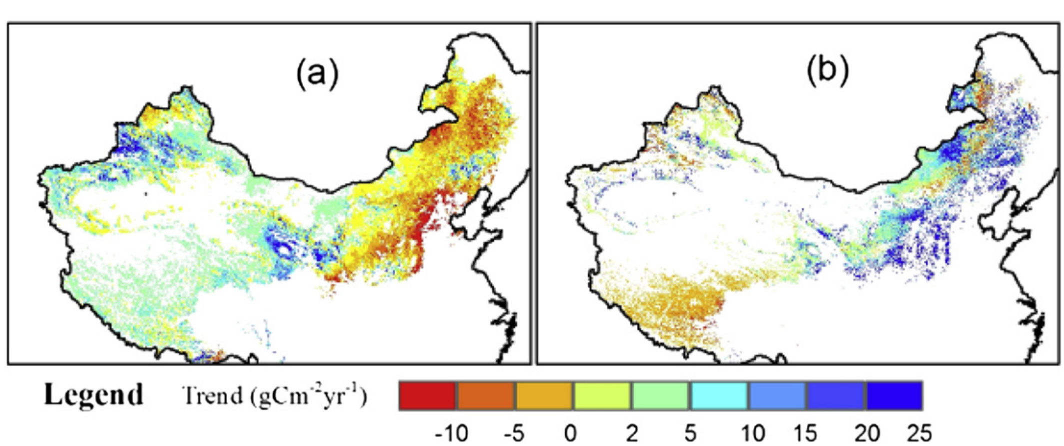

Fig. 5. Spatial distribution of annual NPP trends from (a) 1982 to 2000 and (b) 2000 to 2008.

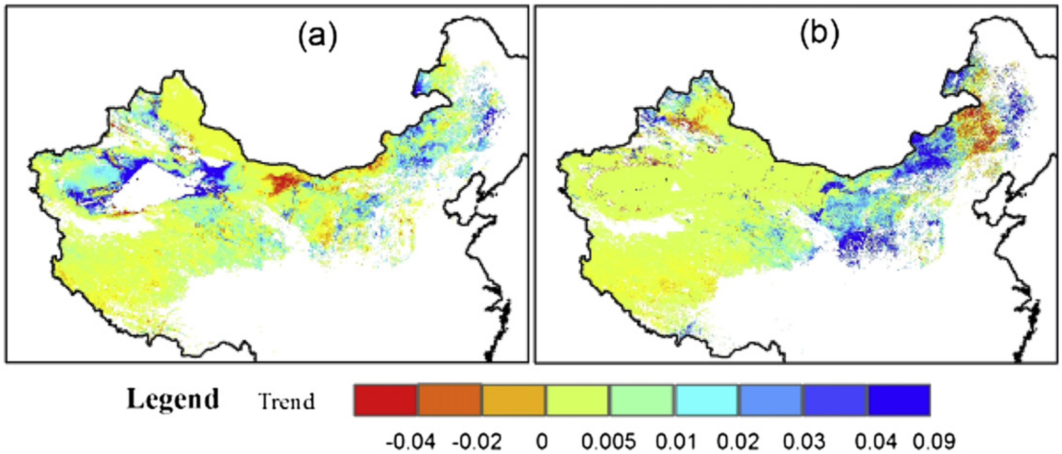

Fig. 6. Spatial variations of annual RUE trends from (a) 1982 to 2000 and (b) 2000 to 2008.

simply indicated that the available rain was used less efficiently. The discrepancy range at similar rainfall levels would indicate variations in warming and extended growing seasons.

4. Discussion

4.1. Has there been long-term strengthened vegetation activity in arid and semiarid China?

In remote sensing studies of vegetation degradation, it is assumed that RUE is constant over time if there has been an absence of degradation at a given site (Prince et al., 1998; Prince, 2002). Therefore, considering the relationship between NPP and rainfall, the relatively slight increase in RUE indicates that a gradual restoration of the vegetation is likely. We should consider several processes that might affect the relationship between rainfall and NPP. Both vegetation and soil condition have been shown to be the most important factors affecting RUE and its variability (Prince, 2002; Hu et al., 2010). For any given site, a decline in RUE over time is generally acknowledged to indicate ecosystem degradation. The decline may be caused by changes in soil moisture and evapotranspiration, drainage from the rooting zone, excessive gaps between rainfall events, loss of vegetation cover, a reduced capacity of the vegetation to transform water and nutrients to biomass, or by a decrease in the availability of plant nutrients. Increases in RUE may be caused by run-on, fertilizer use, changes in species composition or any microclimatic effect on water use efficiency, such as humid air near water bodies (Prince et al., 1998). RUE was often highest in the years of lowest rainfall and lowest in the years of highest rainfall, as shown in Fig. 2, which had been documented by Prince et al. (1998, 2007), Prince (2002). Climatic warming can influence productivity in two ways. One way is through its direct impact on plant photosynthesis and growth. The other way is through its indirect impact due to changing phenology (Piao et al., 2007) by extending the growing season (Linderholm, 2006), which strongly correlates with annual productivity.

We concluded in our study that the relatively increasing RUE found throughout most of arid and semiarid China, especially along the agro-pastoral ecotones, indicates a process of intensified vegetation activity. This also implies that there has been “greening” from the impacts of increasing rainfall. The lack of large declines in RUE over the past 30 years strongly suggested that no extensive degradation could be detected using remote sensing methods. In fact, there was a small but significant increase in the NPP/rainfall relationship throughout the period. This was also indicated by a number of studies in northern China demonstrating that the growing season has lengthened significantly, either through an earlier onset of turning green, or through a later termination of its wilting (Zheng et al., 2002; H. Wang et al., 2007). The evidence from changes in RUE does not justify fears of extensive and progressive degradation in arid and semiarid China. The dynamics of grassland vegetation activity were classified into five types for China: regions showing continuous restoration of the vegetation due to gradually decreasing degradation and increasing of vegetation coverage; regions with expansive degradation in the past which have recovered owing to obvious vegetation restoration and control of desertification; regions that were intensively degraded before but are less so now; regions that have shown degradation continuously; and regions with newly occurring degradation that are considered to be at high risk of further desertification.

4.2. Have grassland restoration programs yielded progress toward conserving ecosystems?

Before 2000, decreases in grassland in northern China were a consequence of climate conditions associated with economic development (Liu et al., 2005). Initially, grasslands with light and heat resources that were optimal for crop growth were often transformed into farmland, which resulted in continually reduced areas of grassland. Furthermore, expanding population and the associated demand for animal products led to overgrazing in traditional rangelands. As a result, reduced natural pasture, intensified grassland degradation and even desertification occurred (Yang et al., 2005, 2007). Since 2000 and 2003, the Returning Farmland to Grassland and Rangeland to Grassland programs have provided food, increased infrastructure, and subsidized incomes to farmers and livestock herders who transform their farmland or rangeland into grasslands within three to five years. We can conclude from our results that grassland conservation programs, especially the two projects mentioned above, significantly restored and protected grassland ecosystem structure and functions. In particular, vegetation activity was obviously intensified on the Loess Plateau and along the northern agro-pastoral ecotone regions, compared with other regions that are primarily under the pressure climate change.

The tradeoffs between multiple ecosystem services and economic feedback, their respective causes and possible interventions should all be identified for different landscapes and at different scales (Raudsepp-Hearne et al., 2010). Reclamation for agriculture is the primary transformation factor for decreasing grassland, which means that ecological conservation strategies that appear to be desirable from the point of view of society are often unattractive to people who manage ecosystems directly (Wunder et al., 2008; S.X. Cao et al., 2009). Enhancing ecosystem conservation programs often leads to tradeoffs between food security and livelihood improvement. We were also reminded that environmental goals cannot be achieved without some type of economic development that can provide a livelihood for the people affected by the program. Additionally, these restoration programs have often failed to address significant social problems that are tightly integrated with environmental degradation problems (Liu et al., 2008; Chen et al., 2009). Currently, participants do not receive sufficient employment training or relocation assistance that would enable local farmers or herders to seek a sustainable form of employment when the project ends, and many of the farmers and livestock grazers in the project areas have no alternative other than to return to their former way of life as soon as the project ends, leading to a resumption of the activities that were responsible for the original environmental degradation (S.X. Cao et al., 2009; S. Cao et al., 2009). For policies and projects to be effective, they must be scientifically appropriate, but such projects also require long-term support from the participants and from others who are affected by the policies. In addition to investing funding and labor resources in protecting the improvements achieved by previous projects, more efforts should be directed toward providing income or reducing costs for local communities, especially after government subsidies end, which will greatly increase the likelihood that the environmental improvements will be sustainable (S.X. Cao et al., 2009; Chen et al., 2009).

Some problems we need to consider in future work are as follows. First, for combating desertification and restoring the environment, tree plantings on grasslands or deserts have been promoted since the 1970s in most of the arid and semiarid regions. Are trees more suitable than grass in this region for conserving the environment? Second, how should sustainable and effective ecological programs be promoted? Is payment an effective method of support? Past and current payments for ecosystem services were primarily governmental investments made by financial transfers. This is a “blood transfusion” payment, not a substantive, long-term payment. Therefore, credible, replicable, scalable, and sustainable payment strategies should be established for the long term, and subsequent policies will be very important for the sustainability of conservation benefits. Thirdly, fencing was constructed to conserve grassland. Is fencing grassland and forbidding grazing useful for conserving resources? The migration of large mammals was restricted, and natural habitats were fragmented by fences, which would be a significant threat for these species. Without grazing livestock, the grassland food chain is incomplete. Furthermore, our satellite data provide the basis for asking questions about the long-term permanence of grassland restoration programs and a survey or questionnaire would be needed to draw conclusions about the social and cultural strengths and weaknesses of the programs.

5. Conclusions

This study revealed a change in the overall grassland cover and variations in vegetation activity in the arid and semiarid regions of China before and after the implementation of two primary grassland restoration programs, using satellite observation and ground-based data. The observed increases in vegetation productivity and RUE in most areas likely resulted from climate change and ecological restoration. Whatever the explanation, vegetation degradation as a large-scale phenomenon cannot be demonstrated, at least in terms of the vegetation’s response to rainfall, which disagrees with the opinion of many contemporary observers. The satellite data did not show widespread, long-term grassland degradation during the period between 1982 and 2008. We could also conclude that ecological restoration programs are winning the battle against degradation in some arid and semiarid regions, especially on the Loess Plateau, where the returning farmland or rangeland to grassland programs was promoted by government programs. However, more efforts are required for long-term ecological management in the future. The results have helped to raise public awareness of the need to address the continued livelihood of local communities affected by ecological projects and have provided decision makers with scientific data to more effectively control grassland degradation.

Acknowledgments

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2012.12.040.

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