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# Severe depletion of soil moisture following land-use changes for ecological restoration: Evidence from northern China

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ABSTRACT

Soil moisture is fundamental to ecosystem sustainability in arid and semi-arid regions, and characterizing temporal variations in soil moisture levels in response to changes in land use is important in assessing whether vegetation that has been restored as part of ecological restoration can be sustained. Such an assessment is presented here based on 78 recent publications focused on China's initiatives at ecological restoration including the 'Grain for Green' programme and the 'Three Norths Shelter Forest System' project. The study analysed 1740 observations at 83 sites in eight provinces in northern China to determine temporal and spatial variations in soil moisture and the causes of those variations. Changes in land use for restoration of ecosystems led to severe depletion in soil moisture levels – as low as 9%, determined gravimetrically – in the 0–100 cm layer of soil. The extent of depletion was influenced significantly by the choice of species for restoration (trees, shrubs, or grasses) and land use before the restoration. Deliberate restoration of vegetation may have the largest negative impact on soil moisture at sites that receive less than 600 mm of annual precipitation and may be practical only when it exceeds 600 mm. Afforestation decreased the levels of soil moisture significantly, whereas natural restoration had no significant effect on soil moisture. Therefore, natural restoration is the better option for maintaining the stability of water resources in arid and semi-arid regions. Afforestation would be a poor choice for places in which annual precipitation is close to or less than potential evapotranspiration but a better choice if annual precipitation is adequate. In planning revegetation initiatives, planners must understand that different environments support different vegetation types, and therefore require different solutions.



## 1. Introduction

Changes in land use and land cover and the dynamics of water use are important issues in studying global environmental change and must be clearly understood to sustain certain ecosystems (Fischer and Sun, 2001). Over the past century, afforestation and reforestation (artificial or deliberate forestation) have been implemented extensively (Del Lungo et al., 2006; IPCC, 2014), and increasing attention has been paid to their ecological benefits such as increased sequestration of soil carbon (Deng et al., 2014), reduction in water loss and control of soil erosion (Brown et al., 2007), and control of desertification and conservation of biodiversity (Porto et al., 2009). However, a major concern at present related to deliberate restoration of vegetation is its effect on water resources (Jin et al., 2011).

Soil moisture is a significant component of the overall terrestrial water resources, particularly in arid and semi-arid regions. Precipitation in these regions is unevenly distributed: most of it is received only in the rainy season that lasts for a few months, a great deal of water is therefore lost to run-off (Tsunekawa et al., 2013). Characterizing the variations in soil moisture across a range of spatial and temporal scales has important applications in both theory and practice (Ivanov et al., 2010) and provides a basis for optimal allocation of space for restoring lost vegetation (Yang et al., 2015). Spatio-temporal variations in soil moisture are affected by many factors including climate, particularly precipitation (Longobardi, 2008; Jin et al., 2011), topography (Wilson et al., 2005), soil depth (Legates et al., 2011; Venkatesh et al., 2011; Jia and Shao, 2014), vegetation type (Brown et al., 2005; Ursino and Contarini, 2006; Vivoni et al., 2008), and land use and land cover (Fu et al., 2003; Jackson et al., 2005). Vegetation and

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land use may have a major influence on soil moisture content in arid and semi-arid regions (Chen et al., 2007; Sanchez-Mejia et al., 2014). Vegetation can mediate the effect of precipitation on soil moisture and change its *spatial* distribution (Vivoni et al., 2008) and can change the pattern of distribution of soil moisture between shallow and deep layers of soil in semi-arid regions (Yang et al., 2012). Such effects also vary depending on the plant species and lead to *temporal* variations in soil moisture (Aranda et al., 2012; Jost et al., 2012). Lastly, variations in soil moisture levels can be both temporal and spatial, changing with depth (Yang et al., 2014). Therefore, managing soil water resources efficiently both in time and in space is a major challenge in the efforts to restore vegetation in arid and semi-arid areas.

Changes in land use or land cover can strongly affect the dynamics of soil moisture in arid and semi-arid regions (Chen et al., 2007; Xiao et al., 2011). However, evaluating the effects of land use and its pattern on soil moisture is difficult because differences in land use, which change soil properties and the rate of evapotranspiration, inevitably increase soil moisture variability across the landscape (Wang et al., 2001). Xiao et al. (2011) found that shrubs (*Caragana korshinskii*) and forests (*Robinia pseudoacacia*) raised on sites that had been farmlands deplete soil moisture. Yang et al. (2014) also reported that in Gansu, China, soil moisture content decreased drastically in farmlands that were converted into perennial vegetation cover, and Du et al. (2007) found a steady decrease in soil moisture when an abandoned farmland was taken over by other forms of vegetation. Newly introduced vegetation usually consumes more soil moisture than native plants do and thus rapidly depletes local soil moisture resources (Cao et al., 2011; Yang et al., 2012). One consequence of large-scale afforestation is increasingly severe water shortages (Cao et al., 2009). It is possible that large-scale vegetation restoration projects are limited by soil moisture resources in arid and semi-arid regions (Chen et al., 2008; Cao et al., 2011). Therefore, comparing the distinctive effects of land use on soil moisture is critical to successful restoration of vegetation in the arid and semi-arid regions.

In China, widespread ecological degradation has constrained sustainable socio-economic development in recent decades, particularly before the end of the 20th century (Lü et al., 2012). Poor vegetation as a result of severe human interference is considered to be one of the major reasons (Chen et al., 2007). Since the 1950s, the Chinese government has initiated many large-scale efforts to check soil erosion and to restore damaged ecosystems (Fu et al., 2002). More than 9.27 million ha of farmlands on hill slopes, abandoned lands, and desert lands in China has been afforested or planted with grasses through such ecological restoration initiatives (Lü et al., 2012) as the 'Grain for Green' programme and the 'Three Norths Shelter Forest System' project. Although the initial goal was to control soil erosion (Cao et al., 2011; Deng et al., 2012), the interventions may indeed have strongly affected soil moisture (Cao et al., 2009, 2011). Although such programmes confirmed that poor or exploitative land-use practices dry the soil out (Cao et al., 2009), and despite many observations made at the local level, comprehensive assessments of changes in soil moisture in ecological restoration zones have been limited so far (Fu et al., 2003; Chen et al., 2007; Cao et al., 2009; Yang et al., 2014, 2015). In addition, potential land uses for ecological restoration are many and diverse, and it is necessary to study the long-term effects of previous land use on soil moisture on sites that were transformed into forests, shrub lands, or grasslands *before* restoring more farmland, grassland, abandoned land, desert land, and so on with new vegetation types.

Northern China, dominated by arid and semi-arid areas, is currently undergoing tremendous changes in land use and land cover because of the implementation of the 'Grain for Green' programme and the 'Three Norths Shelter Forest System' project. The region has a temperate, monsoonal climate with hot and dry summers and cold and dry winters. Soil moisture is lost mainly during the season of vegetation growth and cannot be completely replenished by precipitation during the rainy season because it is limited to a short period and a great deal of precipitation is lost in the form of run-off. Inadequate moisture thus inhibits sustainable growth of vegetation in such zones (Chen et al., 2007), which makes water a key factor in restoration of vegetation. It was against this background that the present study examined 78 research papers comprising 1740 observations at 83 sites in eight provinces in northern China (1) to determine temporal variations in soil moisture under seven land uses; (2) to analyse the differences in soil moisture profiles under those seven land uses; and (3) to study spatial variations in soil moisture and the causes of such temporal and spatial variations in soil moisture.

## 2. Materials and methods

### 2.1. Data compilation

All of the available peer-reviewed publications published during 2000–2015 and concerning changes in soil moisture (gravimetric water content) were collected. These publications dealt with soils from forests (trees), shrub lands, and grasslands that were once farmlands, grasslands (abandoned land), or desert lands and were converted as part of such restoration initiatives as the 'Grain for Green' programme and the 'Three Norths Shelter Forest System' project in northern China. The following criteria were used to select publications for analysis:

- inclusion of data on both the current and the past land use
- soil moisture (gravimetric) determined from various depths within the 0–100 cm layer (0–20 cm, 20–40 cm, 40–60 cm, 60–80 cm, and 80–100 cm)
- soil moisture determined during the period of maximum biomass (every year in August) and in the field (laboratory experiments excluded)
- paired-site, chronological sequence, or retrospective design
- comparable conditions (in terms of soil types and elevation)
- afforestation confined to the first rotation
- number of years since land use conversion was either clearly mentioned or directly ascertainable
- location, temperature (°C), and precipitation (mm) clearly given
- adequate replications and uniform soils (studies were excluded if the experiments were not adequately replicated or if the paired sites or sites in chronological sequence were confounded by different soil types.)

Of the 1740 observations (Appendix S1) in eight provinces in northern China (Fig. 1), 1317 showed a clear chronological sequence and 423 came from sites that had once been farmlands, grasslands, or desert lands.

The raw data were either obtained from tables or extracted by digitizing graphs using the GetData Graph Digitizer (ver. 2.24, Russian Federation). For each paper, the following information was compiled: sources of data, location (longitude and latitude), weather parameters (mean annual temperature and mean annual precipitation), current land use (grassland, restored deliberately or naturally), shrub land (conifers or broad-leaved shrubs), forest (conifers or broad-leaved trees), years since land-use conversion (afforestation, planted grass, or natural restoration), soil depth, and soil moisture in the five layers in the 0–100 cm depth. In studies with many replicates, data for the plots of the same age, edaphic conditions, and land use were pooled. Where a particular chronological sequence or retrospective study had recorded

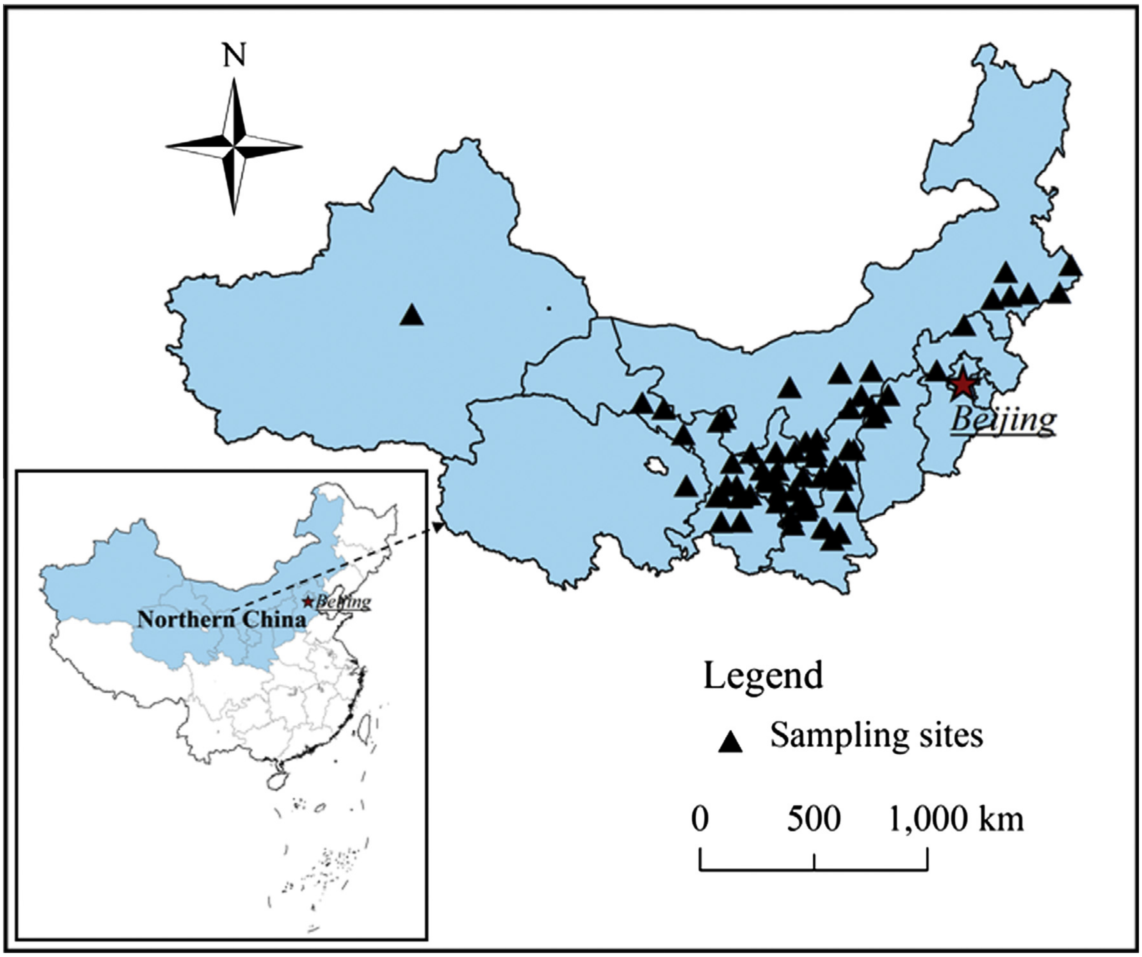


**Fig. 1.** Distribution of sampling sites in the dataset.

observations over a number of years after planting, plants of different ages were taken as different and independent units for the analysis.

Depending on the stage of restoration (early, middle, or late), trees at the afforested sites were divided into three subgroups, as follows: 1–20 years, 21–40 years, and older than 40 years. Similarly, shrubs and grasslands (whether restored naturally or deliberately) were divided into three subgroups, as follows: 1–10 years, 11–20 years, and older than 20 years. In addition, northern China was roughly divided into four regions based on mean annual precipitation – less than 250 mm, 250–450 mm, 451–600 mm, and more than 600 mm – which corresponded closely to the degree of climatic drought.

### 2.2. Meta analysis

The size of the effect in each investigation was calculated as the response ratio $r = x_e/x_c$, where $x_e$ is the mean soil moisture under current land use and $x_c$ is the mean soil moisture in the associated control plots (farmland, grassland, or desert land). As is typical in meta analyses, most of the published papers reported only mean values for treatments and control plots without reporting standard deviations or standard errors. To maximize the number of observations included in the present analysis, we used unweighted meta analysis, as described in earlier studies (Powers et al., 2011; Deng et al., 2016). The mean response size ($R = r - 1$) for each categorical subdivision was calculated, and a 95% confidence interval (CI) was ascertained by using METAWIN ver. 2.0.

A method reported earlier (Luo et al., 2006, 2009) was used for calculating 95% CI of the means for soil moisture, as shown in Eqs. (1) and (2):

$$\mathrm{SE}_{total} = \sqrt{\frac{V_S}{N}} \tag{1}$$

$$95\%\mathrm{CI} = 1.96 \times \mathrm{SE}_{total} \tag{2}$$

where $\mathrm{SE}_{total}$ denotes the standard error of the response size for soil moisture and $V_S$ and $N$ are the variances of response size for soil moisture and the number of observations, respectively. In this study, 95% confidence interval (CI) was calculated for the overall data and for each category: the observed response sizes were considered statistically different from zero if the 95% CI did not include zero, and the grouping factors were considered significantly different from one another if their 95% CIs did not overlap.

### 2.3. Data analysis

Results of the analysis of variance (ANOVA) for the effects of age, prior land use (PLU), land-use change (LUT) or trees or shrubs or grassland, mean annual precipitation (MAP), and soil layers (SL) and their interactions on soil moisture in different land-use conversion types were tested with a general linear model (GLM). The total heterogeneity among groups ($Q_t$) was partitioned into within-group heterogeneity ($Q_w$) and between-group heterogeneity ($Q_b$). The $Q_b$ of each categorical variable was determined for the response variable. A significant $Q_b$ value indicated that the size of the effect differed between different categorical subdivisions. The differences were evaluated at 0.05 significance level. If significant at $P < 0.05$, the least significant difference (LSD) post-hoc test was used for multiple comparisons. Stepwise regression analysis was used for analysing the relationship between response size of soil moisture after land use conversion and mean annual temperature (MAT), MAP, years since land conversion (Age), or initial soil moisture (I) in each soil layer in each land use type. Linear regression analysis was carried out between response size of the soil moisture and the four other factors (Age, I, MAT, and MAP) following land-use conversions. All statistical analyses were performed using SPSS ver. 17.0 (SPSS Inc., Chicago, Illinois, USA).

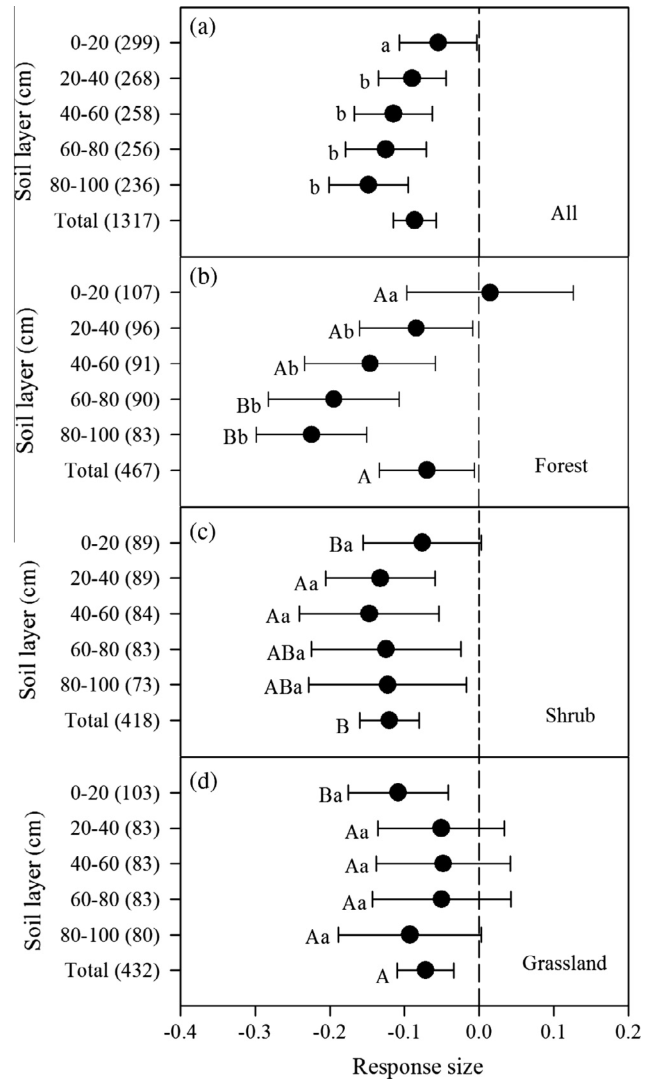


**Fig. 2.** The impact of vegetation restoration under the GGP with different vegetation types (forest, shrub, grassland) on soil water of different soil layers in Northern China. Note: a, all the whole GGP; b, forest; c, shrub; d, grassland. Dots with error bars denote the overall mean values and the 95% CI, and numbers of observations are in the parenthesis. Different upper-case letters left the bars mean significant differences in different vegetation types ($P < 0.05$), and different lower-case letters left the bars mean significant differences in different soil layers ($P < 0.05$). Total includes the five soil layers data. The dash line indicates $x = 0$.

## 3. Results

### 3.1. Decrease in soil moisture due to changes in land use

When land was brought under perennial vegetation as part of the entire 'Grain for Green' programme, soil moisture in the 0–100 cm layer of soil was depleted irrespective of the type of vegetation, whether trees (forests), shrubs, or grasses (Fig. 2 and Supplementary Table S1). Overall, soil moisture decreased by 9%, a significant reduction. Across the soil profile, the reduction in moisture was greater in the deeper layers than in the shallower layers (Fig. 2a). In the case of trees, the decrease in the surface layer (0–20 cm) was not significant, whereas that in the subsoil (>20 cm) was significant (Fig. 2b). The size-effect of these changes also differed significantly *between* the two layers but not *within* different layers of subsoil (Fig. 2b). In the case of shrubs, the pattern of changes was similar to that in the case of trees except that the largest decrease occurred in the middle layer (40–60 cm, Fig. 2c). In the case of grasses, the decrease was significant only in the top layer (0–20 cm, Fig. 2d). The type of vegetation changed the amount of soil moisture significantly in the surface layer (0–20 cm) and in the deepest layer (60–100 cm). Overall, across the entire profile, the decrease in soil moisture was the largest (−12%) in the case of shrubs but equal (−7%) in the case of trees and grasses (Fig. 2b–d).

In lands brought under forests, the extent of overall decrease was also influenced significantly by the species, although the differences between different layers were not significant (Table 1): *R. pseudoacacia* decreased soil moisture significantly but *Pinus tabuliformis* did not (Table 1). Shrub species also had a significant effect (Table 1), the decrease being significantly greater in the case of *C. korshinskii* but not in the case any other shrub species (Table 1). In the case of grassland, although the overall decrease was significant, that in the 20–40 cm layer was not (Table 1). Secondly, the decrease was significant in naturally restored grasslands but not in those restored deliberately (Table 1).

### 3.2. Temporal changes in soil moisture due to changes in land use

Overall, the number of years that had elapsed after restoration had no significant effect on soil moisture (Supplementary Table S1), although in specific cases the changes were significant (Table 2). For example, in the case of trees, soil moisture was significantly decreased in the intermediate years (20–40 years following restoration) but neither in the early years (<20 years) nor in the later years (>40 years). In the case of shrubs, a slight decrease in the early years (<10 years) was followed by status quo in the intermediate years (11–20 years) and a significant decrease in the later years (>20 years, Table 2). In the case of grasslands, the patterns was as follows: in the early years (<10 years), soil moisture in the surface layer (0–20 cm) decreased significantly, whereas that in subsoil (20–100 cm) showed no change; in the intermediate years (11–20 years), none of the layers showed a change; in the later years (>20 years), soil moisture in the top layers (0–60 cm) showed no change but that in the deeper layers (60–100 cm) decreased significantly (Table 2).

### 3.3. Changes in soil moisture due to prior land use

Prior land use affected the levels of soil moisture significantly even after the land was put to a different use (Supplementary Table S1, Fig. 3): the levels under the new land use were 20% lower in the former farmlands but remained unchanged in former grasslands and desert lands (Table 2). Some combinations of prior and current land use also affected moisture levels significantly but others did not (Fig. 3): in former grasslands, a switch to forests

**Table 1**
The impact of tree/shrub species, grassland types on soil water in different soil layers. Note: RP, *Robinia pseudoacacia*, PT, *Pinus tabuliformis*, OB, other breadleaf, OC, other conifer; CK, *Caragana korshinskii*, OS, other shrub; MG, man-made grassland, NG, natural grassland. The values were mean ± 95% CI, and numbers of observations are in the parenthesis. Different lower-case letters mean significant differences in different planting ages ($P < 0.05$), and the same lower-case letters mean no significant ($P > 0.05$). Total includes the five soil layers data.

| Land use types | Tree/shrub species & grassland types | Soil layer (cm) | | | | | Total |
|---|---|---|---|---|---|---|---|
| | | 0–20 | 20–40 | 40–60 | 60–80 | 80–100 | |
| Forest | RP | −0.06 ± 0.20*a* (*n* = 16) | −0.16 ± 0.11*a* (*n* = 12) | −0.23 ± 0.11*a* (*n* = 12) | −0.28 ± 0.12*a* (*n* = 11) | −0.29 ± 0.12*a* (*n* = 12) | −0.20 ± 0.06*b* (*n* = 63) |
| | PT | 0.25 ± 0.29*a* (*n* = 37) | 0.06 ± 0.19*a* (*n* = 37) | −0.10 ± 0.17*a* (*n* = 36) | −0.17 ± 0.13*a* (*n* = 36) | −0.22 ± 0.15*a* (*n* = 32) | −0.01 ± 0.10*a* (*n* = 178) |
| | OB | 0.02 ± 0.20*a* (*n* = 33) | −0.08 ± 0.14*a* (*n* = 27) | −0.08 ± 0.19*a* (*n* = 27) | −0.13 ± 0.19*a* (*n* = 27) | −0.14 ± 0.15*a* (*n* = 27) | −0.08 ± 0.08*a* (*n* = 141) |
| | OC | −0.14 ± 0.12*a* (*n* = 21) | −0.17 ± 0.10*a* (*n* = 20) | −0.22 ± 0.09*a* (*n* = 16) | −0.26 ± 0.08*a* (*n* = 16) | −0.30 ± 0.11*a* (*n* = 12) | −0.21 ± 0.05*b* (*n* = 85) |
| Shrub | CK | −0.14 ± 0.07*b* (*n* = 55) | −0.22 ± 0.07*b* (*n* = 55) | −0.28 ± 0.08*b* (*n* = 53) | −0.26 ± 0.07*b* (*n* = 52) | −0.24 ± 0.09*b* (*n* = 48) | −0.22 ± 0.03*b* (*n* = 263) |
| | OS | 0.02 ± 0.17*a* (*n* = 34) | 0.00 ± 0.15*a* (*n* = 34) | 0.08 ± 0.19*a* (*n* = 31) | 0.11 ± 0.22*a* (*n* = 31) | 0.09 ± 0.24*a* (*n* = 25) | 0.06 ± 0.08*a* (*n* = 155) |
| Grassland | MG | −0.03 ± 0.09*a* (*n* = 52) | 0.01 ± 0.10*a* (*n* = 51) | 0.06 ± 0.12*a* (*n* = 51) | 0.05 ± 0.13*a* (*n* = 51) | −0.01 ± 0.13*a* (*n* = 49) | 0.02 ± 0.05*a* (*n* = 254) |
| | NG | −0.18 ± 0.09*b* (*n* = 51) | −0.15 ± 0.14*a* (*n* = 32) | −0.22 ± 0.10*b* (*n* = 32) | −0.21 ± 0.11*b* (*n* = 32) | −0.22 ± 0.14*b* (*n* = 31) | −0.20 ± 0.05*b* (*n* = 178) |

**Table 2**
The impact of planting ages on soil water in different soil layers. Note: The values were mean ± 95% CI, and numbers of observations are in the parenthesis. Different lower-case letters mean significant differences in different planting ages ($P < 0.05$), and the same lower-case letters mean no significant ($P > 0.05$). Total includes the five soil layers data.

| Land use types | Restoration age (yr) | Soil layer (cm) | | | | | Total |
|---|---|---|---|---|---|---|---|
| | | 0–20 | 20–40 | 40–60 | 60–80 | 80–100 | |
| Forest | 1–20 | −0.01 ± 0.12*b* (*n* = 55) | −0.04 ± 0.11*a* (*n* = 47) | −0.10 ± 0.15*a* (*n* = 44) | −0.12 ± 0.14*a* (*n* = 44) | −0.16 ± 0.10*a* (*n* = 42) | −0.08 ± 0.16*a* (*n* = 232) |
| | 21–40 | −0.10 ± 0.16*b* (*n* = 34) | −0.15 ± 0.13*a* (*n* = 32) | −0.18 ± 0.13*a* (*n* = 31) | −0.25 ± 0.15*a* (*n* = 30) | −0.28 ± 0.15*a* (26) | −0.19 ± 0.06*a* (153) |
| | >40 | 0.31 ± 0.45*a* (*n* = 18) | −0.07 ± 0.17*a* (*n* = 17) | −0.20 ± 0.14*a* (*n* = 16) | −0.30 ± 0.14*a* (*n* = 16) | −0.30 ± 0.14*a* (*n* = 15) | −0.10 ± 0.12*a* (*n* = 82) |
| Shrub | 1–10 | 0.00 ± 0.09*a* (*n* = 31) | −0.03 ± 0.1*a* (*n* = 31) | −0.13 ± 0.12*a* (*n* = 28) | −0.07 ± 0.10*a* (*n* = 27) | −0.03 ± 0.14*a* (*n* = 23) | −0.05 ± 0.05*a* (*n* = 140) |
| | 11–20 | 0.13 ± 0.23*a* (*n* = 20) | −0.05 ± 0.14*a* (*n* = 20) | −0.10 ± 0.23*a* (*n* = 20) | −0.08 ± 0.29*a* (*n* = 20) | −0.13 ± 0.30*a* (*n* = 17) | −0.04 ± 0.11*a* (*n* = 97) |
| | >20 | −0.25 ± 0.10*a* (*n* = 38) | −0.26 ± 0.12*a* (*n* = 38) | −0.18 ± 0.16*a* (*n* = 36) | −0.19 ± 0.15*a* (*n* = 36) | −0.18 ± 0.15*a* (*n* = 33) | −0.21 ± 0.06*a* (*n* = 181) |
| Grassland | 1–10 | −0.10 ± 0.09*a* (*n* = 54) | −0.06 ± 0.10*a* (*n* = 44) | −0.02 ± 0.13*a* (*n* = 44) | 0.00 ± 0.13*a* (*n* = 44) | −0.02 ± 0.14*a* (*n* = 43) | −0.04 ± 0.05*a* (*n* = 229) |
| | 11–20 | −0.08 ± 0.11*a* (*n* = 25) | 0.04 ± 0.16*a* (*n* = 20) | 0.00 ± 0.17*a* (*n* = 20) | −0.01 ± 0.17*a* (*n* = 20) | −0.06 ± 0.17*a* (*n* = 18) | −0.02 ± 0.07*a* (*n* = 103) |
| | >20 | −0.16 ± 0.17*a* (*n* = 24) | −0.12 ± 0.23*a* (*n* = 19) | −0.16 ± 0.18*a* (*n* = 19) | −0.22 ± 0.18*a* (*n* = 19) | −0.29 ± 0.16*a* (*n* = 19) | −0.19 ± 0.08*a* (*n* = 100) |

decreased soil moisture by 7% and that to shrubs, by 14%; on the other hand, when former grasslands were restored but grasses continued to be the dominant species, soil moisture increased significantly, by as much as 10% (Fig. 3).

### 3.4. Changes in soil moisture in different climatic zones

The four climatic zones categorized on the basis of MAP (less than 250 mm, 250–450 mm, 450–600 mm, and more than 600 mm) differed in the extent of change in soil moisture after land-use conversions (Supplementary Table S1). Soil moisture decreased the most (18%) in the zone with the lowest precipitation and in the 450–600 mm zone and by 8% in the 250–450 mm zone; on the other hand, soil moisture increased by 5% in the zone with the highest precipitation (Fig. 4).

In each zone, soil moisture was significantly affected by the current land use (Fig. 4). In the zone with the lowest precipitation, the decrease in soil moisture was greater in grasslands than in shrub lands; in the 250–450 mm zone, on the other hand, the decrease was greater in forest lands and shrub lands than in grasslands. In the 450–600 mm zone, land use had no effect on soil moisture (Fig. 4).

### 3.5. Relationship between soil moisture and age, initial soil moisture, precipitation, and temperature

Overall, soil moisture content changes refer to prior land use was significantly ($P < 0.01$) and negatively correlated with the number of years elapsed since restoration and with the initial soil moisture content (when the conversion began) (Fig. 5d and h). The changes of soil moisture content was also significantly and negatively correlated with MAP in the case of shrubs (Fig. 5j) but significantly and positively correlated with MAP in the case of grasslands (Fig. 5k). However, MAT had no effect on soil moisture (Fig. 5).

Stepwise regression analysis revealed that, overall, initial soil moisture, MAP, and the number of years since restoration

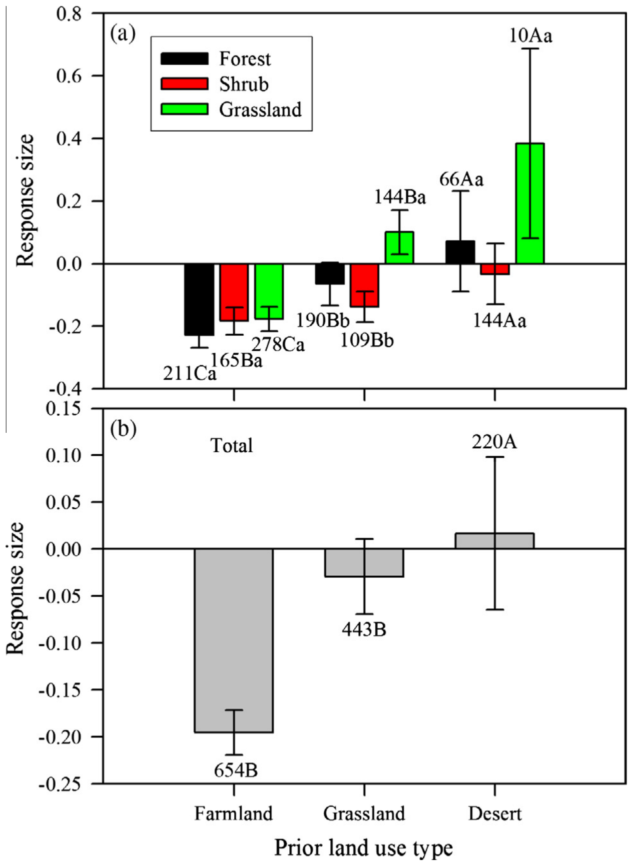


**Fig. 3.** The response size of soil moisture in different prior land use types. Note: The chards with error bars denote the overall mean values and the 95% CI, and numbers of observations are above the bars. Different upper-case letters mean significant differences among different prior land use types for the same land use types ($P < 0.05$), and different lower-case letters mean significant differences among different land use types for the same prior land use type ($P < 0.05$).

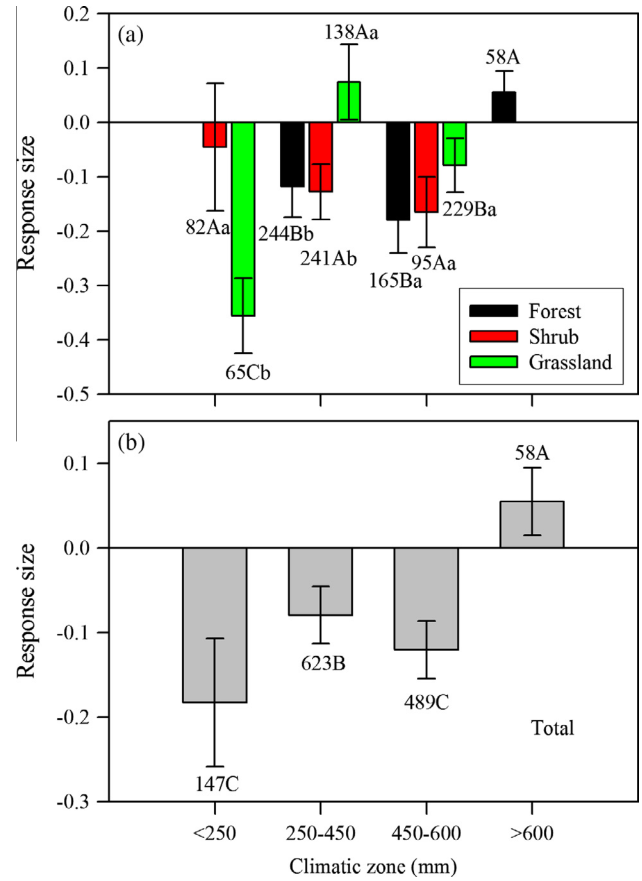


**Fig. 4.** The response size of soil moisture in different climatic zones. Note: The chards with error bars denote the overall mean values and the 95% CI, and numbers of observations are above the bars. Different upper-case letters mean significant differences among different climatic zones for the same land use types ($P < 0.05$), and different lower-case letters mean significant differences among different land use types for the same climatic zone ($P < 0.05$).

exercised a combined effect on soil moisture (Table 3). Although initial soil moisture and MAP were the more important factors for all land uses (Table 3), forests and grasslands were affected more by initial soil moisture and shrub lands, by MAP (Table 3).

## 4. Discussion

### 4.1. Dynamics of soil moisture as affected by land use

Land use affects soil moisture levels primarily because of the water consumption characteristics of vegetation (Xiao et al., 2011) and mainly in three ways: moisture levels can increase, decrease, or fluctuate (Fu et al., 2003). In the present study, soil moisture decreased significantly when land use was changed to growing trees, shrubs, or grasses, but the changes were also influenced by soil depth. Earlier studies of the effects of trees on moisture levels in the topsoil (<20 cm) have reported the effects to be either significantly negative (Breshears et al., 1997), significantly positive (Joffre and Rambal, 1998), or negligible (Maestre et al., 2001). We found the levels unchanged in the 0–20 cm layer in lands converted to forest lands or shrub lands, a result that matches the report by Maestre et al. (2001). In addition, afforestation is known to decrease soil moisture compared to the levels before afforestation, irrespective of prior land use, because of the interception of rainfall by leaves, increased uptake by roots, and losses through evapotranspiration (Jin et al., 2011; Yang et al., 2014). When grasslands are either invaded by or replaced deliberately with trees or shrubs, large quantities of water are lost through transpiration from the deeply rooted woody vegetation, lowering the water table and making it harder for native grasses and other species to survive (Asner et al., 2008). However, when lands of any type are converted to grasslands, soil moisture decreases significantly only in the surface layer (0–20 cm), probably because the roots of grasses are mainly confined to the surface layer (Hoshino et al., 2009; Mueller et al., 2013).

In the present study, the number of years elapsed since restoration had no effect on soil moisture (Table 2), a finding consistent with that of Jin et al. (2011). Yang et al. (2014) also found no obvious temporal changes in soil moisture in the sub-surface layers and deeper layers as a result of introduced vegetation during restoration. However, over time, soil moisture continued to decrease, probably as a result of increasing transpiration as the plants grew larger (Chen et al., 2007). Cao et al. (2009) found a near-continuous decrease in soil moisture in afforested plots that had once been farmlands after an initial increase in the first 2 years—a decrease that, after 7 years, resulted in the afforested plants being severely arid, with moisture levels only 36.8% ($P < 0.05$) of the initial level in the abandoned farmland. The trees (*Pinus tabulaeformis*, *Platycladus orientalis*, *Populus canadensis*, and *Prunus armeniaca*) and shrubs (mainly *Hippophae rhamnoide*) grew well at the beginning but were affected adversely once the initial water supply was more or less exhausted (Li, 2001). Therefore, in the long term, such injudicious land use practices can dry the soil out.

As to the significant decrease in soil moisture associated only with the intermediate years (20–40 years) of tree plantations

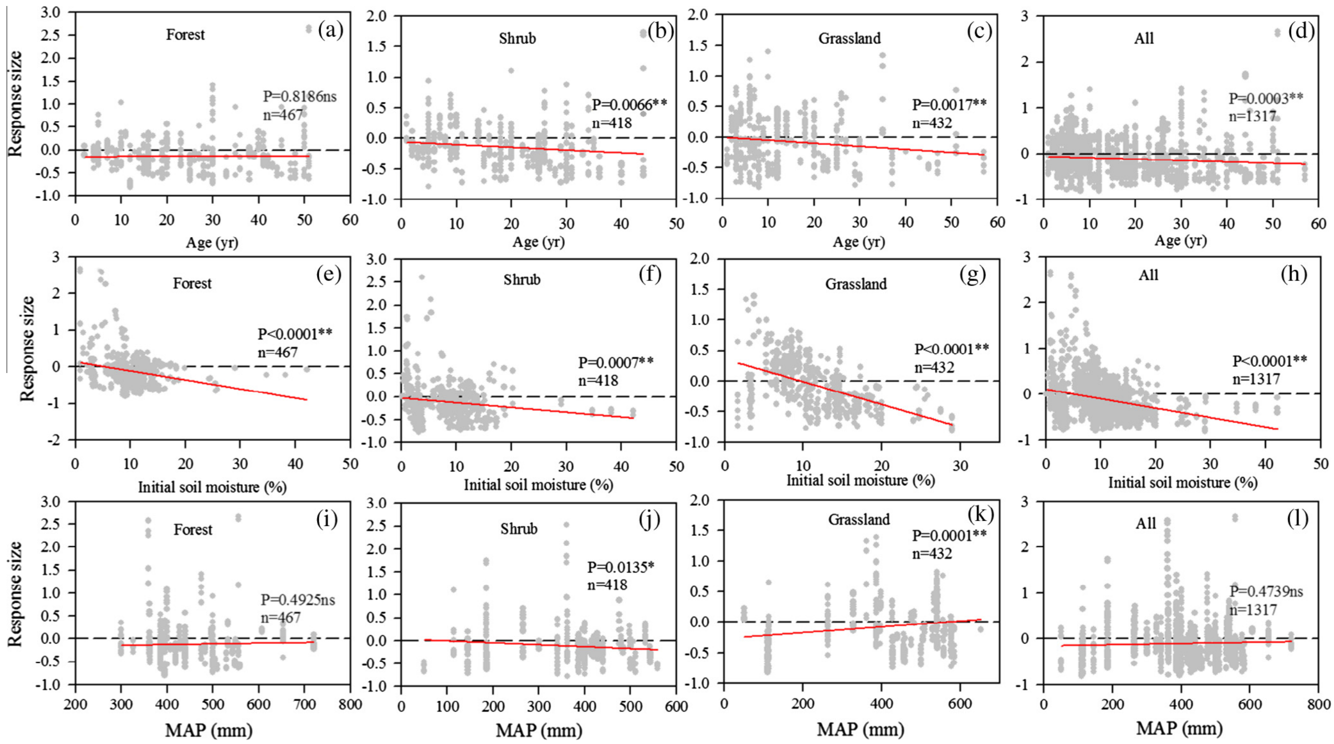


**Fig. 5.** The relationship between response size and age, initial soil moisture, mean annual precipitation. Mean annual temperature (MAT) had non-significant ($P > 0.05$) effect on soil water changes, so the results was not showed in the figure. Note: * indicates significant at $P < 0.05$, ** indicates significant at $P < 0.01$.

**Table 3**
Stepwise regression to detect factors (MAP, MAT, Age and I) determining response size of soil moisture following land-use conversions.

| Types | Models | $R^2$ | *Sig. (P)* | *n* |
|---|---|---|---|---|
| Forest | $R = -0.031 \times I + 0.001 \times MAP - 0.031MAT - 0.135$ | 0.315 | 0.000*** | 467 |
| Shrub | $R = -0.01 \times MAP - 0.006 \times I + 0.071$ | 0.218 | 0.000*** | 418 |
| Grassland | $R = -0.043 \times I + 0.001 \times MAP - 0.032 \times MAT + 0.034$ | 0.595 | 0.000*** | 432 |
| All | $R = -0.024 \times I + 0.000 \times MAP - 0.004 \times Age - 0.373$ | 0.331 | 0.000*** | 1317 |

Note: R is response size of soil moisture; MAP (mm) is the mean annual precipitation; MAT (°C) is the mean annual temperature; Age (yr) is the restoration age; I (%) is the initial soil moisture.
*** Indicates significant at $P < 0.001$.

(Table 2), the effect was probably due to (1) the initial rapid growth supported by the initial stocks of soil moisture, (2) the decline once the initial stocks were used up (Chen et al., 2007) and greater losses through transpiration from the 20–40-year-old trees, and (3) the cessation of growth on reaching maturity. In the case of shrubs, the unchanged levels of soil moisture across the soil profile in the intermediate years (11–20 years), may be attributed to the weaker 'transpirational pull' or the hydraulic pressure exerted by root systems of shrubs than by trees (Prieto and Ryel, 2014), as explained below. Shrubs (*C. korshinskii*, for example) need more water to sustain their rapid growth; when water is in short supply in the shallow layer (<1 m), they tap into the water resources of the deeper layer and then release the absorbed water into the shallow layer (Prieto and Ryel, 2014; Huang and Zhang, 2015)—in effect, there is no *net* change in the moisture levels across the entire soil profile (0–100 cm).

### *4.2. Factors affecting soil moisture after changes in land use*

Trees and shrubs affected soil moisture levels in the entire soil profile (0–100 cm) significantly but differently after the land conversions (Table 1). The differences can be attributed to the differences in root distribution and, in turn, to the differences in water uptake. The arid and semi-arid regions of northern China are dominated by *R. pseudoacacia* and *C. korshinskii* (Appendix S1). *R. pseudoacacia* is a shallow-rooted species; although its roots can go as deep as 190 cm, its effective roots are concentrated in the 0–60 cm layer, especially in the 20–60 cm layer, indicating that this species will absorb soil moisture mostly from the 0–60 cm layer (Jin et al., 2011). *C. korshinskii* has a strong taproot and its root system can extend to a depth of 10 m (Yang et al., 2015), although Cheng et al. (2009) found that most (70%) of its fine roots were distributed in the upper 1 m layer. Thus, the shallow layer (<1 m) is the main layer providing water for plant growth, and soil moisture levels in this layer change with water uptake. The negative correlations between the attributes of the above-ground vegetation with soil moisture in the deeper layers are related to patterns of water uptake by plants, as was also shown by Ferreira et al. (2007) in Brazilian savannah ecosystems. Moreover, trees affect soil moisture through interception of rainfall by leaves, buffering by the layer of leaf litter, and changes in water-holding capacity of soil (Jin et al., 2011). Litter increases water retention by increasing the hydraulic conductivity of the duff layer (Robichaud, 2000), but afforestation can also decrease soil moisture because of leaf interception and root uptake. Moreover, increasing biomass leads to a decrease in net rainfall, since rain interception and evapotranspiration increase, resulting in less water in the soil (Jackson et al., 2005). According to Chirino et al. (2001), 23–35% of the total annual rainfall is intercepted by the canopy of *Pinus halepensis* in the Ventos–Agost catchment area in Alicante, Spain. Therefore, choosing suitable species with respect to soil water balance is crucial to vegetation restoration if water shortage is the key factor.

In the present study, soil moisture was reduced significantly in grasslands that had been restored naturally but not in grasslands that had been restored deliberately (Table 1). In the study regions, alfalfa (*Medicago sativa*) is the dominant species used for grassland restoration (Appendix S1). Alfalfa, like some shrubs, has a strong taproot and its root system can not only extend to a depth of 10 m (Li and Huang, 2008) but also generates strong hydraulic pressure (Li et al., 2008). In fact, such pressure in root cells is essential for releasing water into the dry soil under high water potential (Wan et al., 2000). Low water potential in shallow soils and significant differences in soil water potential between deep and shallow layers of soil builds up hydraulic pressure in plant roots (Wan et al., 2000; Li et al., 2008). Therefore, due to the strong hydraulic pressure in the root systems in deliberately restored grasslands, soil moisture in the entire profile (0–100 cm) was not affected significantly. In afforested sites that had once been farmland but since abandoned, soil moisture was significantly decreased ($P < 0.01$) but not in those in which the restoration had been a natural process ($P > 0.05$) (Fig. 6), indicating that natural restoration is more conducive to the stability of water resources in arid and semi-arid regions, probably because forests have higher evapotranspiration rates than grasslands (Zhang et al., 2001).

Prior land use also affected soil moisture levels significantly (Supplementary Table S1, Fig. 3) because the initial levels (at the beginning of the restoration) were different, depending on land use (Supplementary Fig. S1). When such prior land use was farming, soil moisture decreased by 20% after conversion; however, sites that had been either grassland or desert land showed no such change (Table 2). Grasslands and deserts are relatively stable ecosystems under natural conditions; farmland ecosystems, however, are relatively unstable because of such interventions as irrigation and application of fertilizers over the long term. When perennial plants, whether trees or shrubs, are introduced into farmland ecosystems, the levels of soil moisture are greatly affected (Cao et al., 2009; Xiao et al., 2011; Yang et al., 2014). Joffre and Rambal (1998) observed an improvement in the status of soil moisture after trees had replaced grasslands in southern Spain. Thus, prior land use does affect the dynamics of soil moisture. The absence of any difference in such effect between sites that had once been farmlands or grasslands (Fig. 3) was probably due to similar initial levels in both (Supplementary Fig. S1).

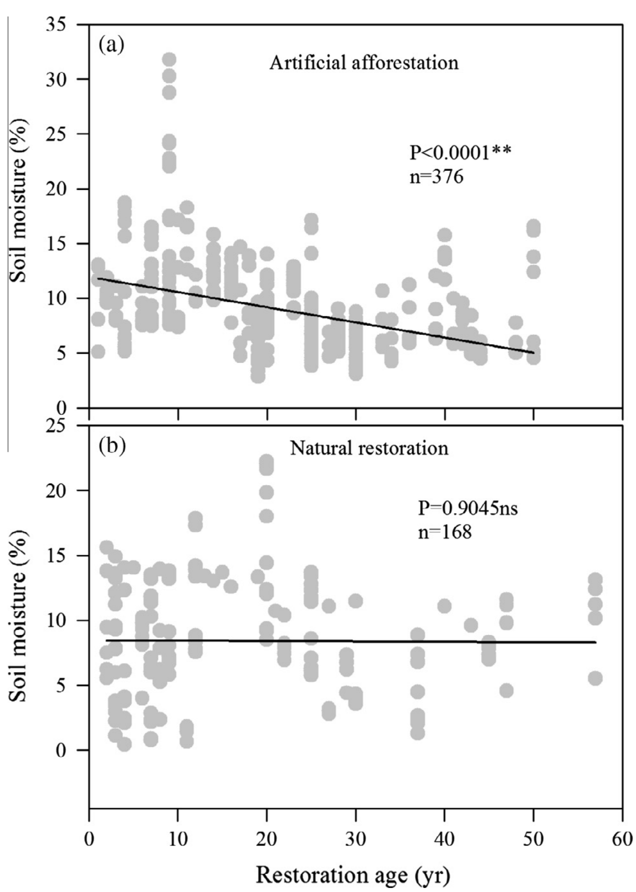


**Fig. 6.** Dynamics of soil moisture since farmland abandonment in different vegetation restoration models. Note: a, artificial afforestation; b, natural restoration. ** indicates significant at $P < 0.01$, ns indicates non-significant ($P > 0.05$).

Overall, soil moisture decreased over time following the conversions despite the initial increase (Fig. 5h). The higher the initial levels of soil moisture, the smaller the changes after conversion. In addition, initial levels were also higher in the zones that had higher MAP (Supplementary Fig. S1) and the quantum of change was smaller in the zones that had higher precipitation. After the conversions, soil moisture in zones with precipitation below 600 mm decreased significantly, by 8–18%, but increased significantly, by 5%, in the zone with precipitation greater than 600 mm (Fig. 4). Thus the extent of variation in soil moisture was smaller in the high-precipitation zone. Jin et al. (2011) found that variations in soil moisture after planting were largely dependent on local precipitation. Longobardi (2008) also reported that climate, precipitation in particular, has a major influence on soil moisture content in arid and semi-arid regions. In the Loess Plateau, for example, Jin et al. (2011) reported that afforestation may be practical only if precipitation is adequate (a MAP of more than 617 mm); trees planted in some areas with water shortages (a MAP of 509 mm) might grow well initially; however, they draw water in large quantities and eventually dry the soil out, making the ecosystem unsustainable in the long term. In extremely dry areas (a MAP of less than 352 mm), soil moisture levels are too low to support the growth of saplings. Moreover, the significant correlation between soil moisture and MAP – negative in the case of shrubs (Fig. 5j) and positive in the case of grasslands (Fig. 5k) – indicates that shrubs depleted soil moisture more than grasslands did in high-MAP zones. Thus, the analyses suggest that changes in soil moisture after restoration are influenced not by a single factor but by a combination of multiple factors. Stepwise regressions analysis and interaction analysis also revealed that initial soil moisture, MAP, and the number of years after restoration affect the level of soil moisture, but the effect varies with current land use (Supplementary Table S1, Table 3). Therefore, any study of the dynamics of soil moisture following restoration of vegetation on a regional scale should take into account the time elapsed since restoration began, MAP, vegetation type or land cover, and prior land use. Annual evapotranspiration could also be another factor, because the pattern of variation in soil moisture related to MAP in the present analysis is consistent with the model developed by Zhang et al. (2001), who reported that annual evapotranspiration is generally greater in afforested sites than in grasslands, especially in high-rainfall areas. It is therefore possible that the relationship between annual evapotranspiration and precipitation can explain the differences in soil moisture levels between grasslands and shrub lands in humid areas.

### 4.3. Implications for management

The initial goal of the 'Grain for Green' programme and the 'Three Norths Shelter Forest System' project was to control soil erosion in northern China. However, these initiatives have also played a significant role in the dynamics of soil moisture (Cao et al., 2009, 2011). In arid and semi-arid regions, availability of water is the central consideration in afforestation (Cao et al., 2009; Jin et al., 2011). In the present study, changes in land use driven by ecological restoration programmes severely depleted soil moisture in northern China. Forest soil is generally deficient in moisture as a result of low annual precipitation, the use of unsuitable tree species, and overly high planting density (Xu, 2006). For examples, planting trees (e.g. *R. pseudoacacia*, *P. tabulaeformis*, *P. orientalis*, *P. canadensis*, and *P. armeniaca*) and shrubs (e.g. *H. rhamnoide* and *C. korshinskii*) is a popular approach to restoring degraded sites but can fail if the choice of species is inappropriate or the management of stands in early stages is inadequate or inappropriate (Lamb et al., 2005). Therefore, the government's overemphasis on afforestation using non-native species appears likely to increase the risk of ecological degradation in this region.

Our study also indicated that natural restoration is better than deliberate afforestation in maintaining the stability of water resources in arid and semi-arid regions. Xiao et al. (2011) also reported that converting abandoned farmland into grassland using native species may be the best option for rehabilitation of vegetation in the Loess Plateau in areas with a MAP of 510 mm. Deliberate afforestation – unlike native vegetation – usually has greater water demands, and annual rainfall levels in arid and semi-arid regions cannot meet those demands (Chen et al., 2008; Wang et al., 2010). Cao et al. (2006) reported that every ecosystem has a finite carrying capacity: when that capacity is exceeded, the result is degradation of the ecosystem. However, ecosystems that are not damaged too badly show a remarkable ability to restore themselves rapidly and economically through natural processes (Mitchell and Ricardo, 2004).

In northern China, deliberate efforts to restore vegetation may have the largest adverse effect on soil moisture in areas with a MAP of less than 600 mm. In such areas, human intervention by way of restoring vegetation with introduced species may lead to drought stress for the young plants and depletion of soil moisture, which is critical to restoration. Afforestation may be practical only if MAP is greater than 600 mm. Therefore, the adaptability of tree species and the ecological significance of establishing plantations should be comprehensively considered before undertaking any afforestation programme (Jin et al., 2011). Forests are clearly not a suitable choice for all areas (Cao et al., 2009) and a particularly inappropriate choice for areas with a MAP close to or below the potential evapotranspiration: afforestation is a more suitable choice for areas where precipitation is adequate, and it would take considerable research to identify suitable species for the vulnerable arid and semi-arid agricultural regions of northern China. Planners must understand that different environments support different communities of vegetation and therefore require different solutions.

## 5. Conclusions

Changes in land use driven by initiatives for ecological restoration severely depleted soil moisture in northern China. And along with the time increasing, soil water decreased getting more and more. Both prior and current land use affected soil moisture significantly. Deliberate restoration of vegetation may have the largest negative effect on soil moisture in areas with a MAP below 600 mm. In such areas, intervention in the form introduced species may lead to drought stress for the young plants and depletion of soil moisture, which is critical to restoration. Afforestation may be practical only if the MAP exceeds 600 mm. Afforestation of abandoned farmlands decreased soil moisture significantly ($P < 0.01$) but natural restoration had no significant effect on soil moisture ($P > 0.05$) (Fig. 6), indicating that natural restoration is better for maintaining the stability of water resources in arid and semi-arid regions. Therefore, any study on the dynamics of soil water after vegetation restoration on a regional scale should take into account the time elapsed since restoration, precipitation, vegetation type or land cover, and prior land use. In particular, afforestation is an inappropriate choice where MAP is close to or below the potential evapotranspiration. Afforestation is a more suitable choice in areas where precipitation is adequate, but it would take considerable research to identify suitable species for the vulnerable arid and semi-arid agricultural regions of northern China. Planners must understand that different environments support different communities of vegetation and therefore require different solutions.

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## Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2016.01.026.

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