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Responses of runoff, sedimentation, and induced nutrient loss to vegetation change in the Tengger Desert, northern China

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Abstract. Runoff and nutrient loss in drylands are closely related to vegetation cover. Simulated rainfall experiments with an intensity of 80 mm/h were conducted in sandy grassland and shrubland in the Tengger Desert, China, to investigate the responses of runoff and associated carbon (C) and nitrogen (N) losses to the replacement of grassland by shrubland. Times to ponding and to generate runoff, and the amount of rainfall for runoff commencement in bare, inter-shrub plots were significantly smaller than in shrub (ST) and grassland (GT) plots; no statistical differences were found for these parameters between ST and GT. Overall, this indicated a higher soil water infiltration rate in grassland than in shrubland. The volume-weighted concentrations of organic C (OC) and total N (TN) in runoff from shrubland (0.083 and 0.011 g/L, respectively) were lower than those from grassland (0.103 and 0.012 g/L, respectively). The cover-weighted runoff coefficients, and sediment, OC, and TN losses from shrubland (34.46%, and 44.95 1.72, and 0.23 g/m^2, respectively) were greater than from grassland (15.22%, and 15.91, 0.94, and 0.11 g/m^2). Vegetation degradation was accompanied by reduced nutrient retention capacity; both soil OC and TN of grassland (8.97 and 0.62 g/kg, respectively) were greater than those weighted values for shrubland (4.18 and 0.26 g/kg). Understanding of these processes suggests that decline or loss of vegetation cover, with the appearance of biological soil crust patches, inevitably leads to increases in runoff and induced soil loss, further accelerating desertification.

Additional keywords: biological soil crust, nutrient loss, runoff, Tengger Desert, vegetation degradation.

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Introduction

Vegetation change is a crucial issue globally for desert grasslands in arid and semi-arid regions. One of the most serious challenges is the increase of shrub-dominated communities in regions where herbage had formerly prevailed (Biedenbender *et al.* 2004). The invasion of desert grasslands by shrubs has been observed worldwide throughout the past 200 years (e.g. Graetz 1994; Schlesinger *et al.* 1999). Various factors are believed to be responsible for this phenomenon. Some authors stated that the replacement of grasslands by shrubs indicates that limited water cannot support a dense and homogeneous vegetation cover (e.g. Tongway and Ludwig 1994). Some proposed that shrub invasion can be the result of overgrazing (e.g. Schlesinger *et al.* 1990). Research has shown that potentially responsible factors might include changing climate, increasing atmospheric carbon dioxide, livestock overgrazing, altered fire regimes, and periodic drought, as well as the variability of precipitation patterns (e.g. Polley *et al.* 1997; Bates *et al.* 2006; Scott *et al.* 2006). Although it is impossible to establish a specific causal relationship between any of these factors and actual vegetation change (Bhark and Small 2003), it has been widely reported that shrub encroachments have resulted in higher spatial–temporal heterogeneity of soil resources and hydraulic conductivity (e.g. Glendening 1952; Schlesinger *et al.* 1990), shallower distribution of soil moisture (e.g. Glendening 1952; Li *et al.* 2008), and variation in hydrothermal conditions (e.g. Moran *et al.* 2009; Kidron 2010). The direct effects of these alterations are to reduce availability of soil water and nutrients, and as a consequence to induce widespread changes to the structure and function of ecosystems (Cross and Schlesinger 1999; Bhark and Small 2003; Howes and Abrahams 2003; Barger *et al.* 2006), especially decreases in species richness and productivity, changes in species composition and biomass allocation, and decline or loss of biotic processes (e.g. Báez and Collins 2008).

Invasion of grassland by shrubs in arid and semi-arid ecosystems is associated with the loss and redistribution of resources (soil material, water, and nutrients) by wind and water erosion (Schlesinger *et al.* 1990, 1996, 1999; Li *et al.* 2008), bringing about pronounced changes in the biogeochemical cycles of such systems (Cross and Schlesinger 1999; Li *et al.* 2008). On the one hand, the resulting patchy distribution of vegetation improves soil physicochemical characteristics and biotic and ecohydrological processes as well as microclimate under plant canopies compared with the bare patches, resulting in the formation of so-called 'islands of fertility' (Schlesinger *et al.* 1990; Rostagno *et al.* 1991; Bochet *et al.* 1999, 2000; Kidron 2010). On the other hand, this transition induces a significant change in the nature of overland flow (Howes and Abrahams 2003; Wilcox *et al.* 2003). In general, grasslands have higher hydraulic conductivity than shrubland, due to superior soil structure, particularly the development of macropores from biotic processes, e.g. root channels, animal burrows, and worm holes, which facilitate infiltration. This, combined with dense vegetation cover, provides resistance to overland flow and results in low-velocity flow and decreased generation of runoff (Abrahams *et al.* 1994; Guebert and Gardner 2001; Ludwig *et al.* 2005). After degradation into patchy shrubland habitats, however, the efficiency of runoff flow concentration on hillslopes is reduced (Wilcox *et al.* 2003). Infiltration rates are higher beneath canopies than between them, causing runoff in the inter-shrub areas to be partly captured by shrub patches (Abrahams *et al.* 1995; Cross and Schlesinger 1999; Barger *et al.* 2006). The runoff that is not trapped by shrubby areas flows downslope around the shrubs at greater flow velocities, causing more erosion than on grasslands, so that overland flow in shrubland habitats develops a complex reticular pattern (Howes and Abrahams 2003; Wilcox *et al.* 2003).

Although shrub invasion of grasslands in dryland areas has been extensively described, previous studies have focussed on the spatial structure of the vegetation in order to gain an understanding of the way that different factors affect shrub invasion (Bhark and Small 2003), and on the resultant 'fertile islands' (e.g. Rostagno *et al.* 1991; Bochet *et al.* 1998, 2000; Cross and Schlesinger 1999). The only published studies have been conducted in North America, for example in the Chihuahuan Desert of New Mexico, and their focus was on changes in inter-rill runoff, erosion, and soluble nutrients loss following vegetation conversion (Abrahams *et al.* 1995; Schlesinger *et al.* 1999, 2000). Consequently, limited information is available in temperate desert areas, especially those in China, where the natural conditions, particularly vegetation composition and precipitation pattern, are different from the regions that have been studied. In the south-eastern fringe of the Tengger Desert, large areas of desert steppe have been invaded by shrubs during the past 200 years, and desertification processes are a major concern for management of the degraded region (Li and Jia 2005). In order to control desertification and to better manage ecosystems of this area, it is important to understand the changes that take place in the biogeochemical processes. Li *et al.* (2008), for example, reported smaller runoff, sediment, and nutrient losses from shrub patches than from inter-shrub patches in shrub-dominated areas of the Tengger Desert; however, few studies have investigated the consequent resource losses (water, soil material, and nutrients) following vegetation replacement.

The objective of the present study was to compare the contents of water, sediment, carbon (C), and nitrogen (N) in the runoff from grass-dominated and shrub-dominated areas at the south-eastern fringe of the Tengger Desert, with the additional purpose of contributing to an understanding of the effects of resource losses due to desertification and the management of degraded ecosystem in this region.

Methods

Study area

This study was conducted at the long-term observation field of the Shapotou Desert Research and Experiment Station of the Chinese Academy of Sciences, on the south-eastern fringe area of the Tengger Desert in northern China (Fig. 1). The mean elevation of the area is 1 339 m. Meteorological records show a mean annual air temperature of 10.0°C, with daily minimum and maximum values of –25.1°C and 38.1°C, and mean annual precipitation of 180.2 mm, ~80% of which falls between May and September. Potential evapotranspiration during the growing season was 2300–2500 mm.

The grassland selected for study was 26 km west of the Shapotou Desert Research and Experimental Station. Vegetation cover in this grassland was ~58%, dominated by *Stipa glareosa* P. Smirn., *S. breviflora* Griseb., and *Cleistogenes squarrosa* (Trin.) Keng. Cover of sporadic biological soil crusts (BSCs) was ~7%, which comprised algae (e.g. *Euglena* sp.), lichen (e.g. *Collema tenax* Ach.), and cyanobacteria (e.g. *Microcoleus vaginatus* Gom). The topography drained

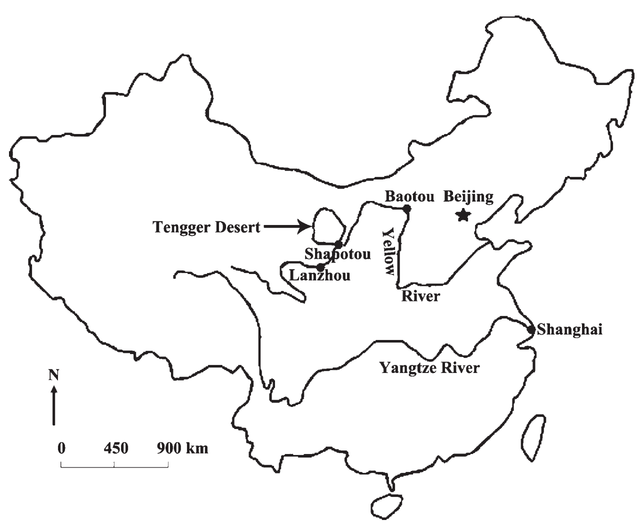

Fig. 1. Map of China showing the extent of the Tengger Desert and the site of the Shapotou Desert Research and Experiment Station of the Chinese Academy of Sciences. Grassland and shrub experimental sites of the study are 26 and 40 km west of Shapotou, respectively.

southward with a slope of ~13° (Fig. 2*a*). The shrub-dominated site was on a north-facing hillside 14 km west of the grassland, dominated by *Salsola passerina* Bunge and *Reaumuria soongorica* (Pall.) Maxim., scattered on soil mounds with a coverage of ~11%. About 6% of the soil under canopies and 74% of the soil between them was covered by BSCs, dominated by algae (e.g. *Euglena* sp., *Hantzschia amphioxys* var. *capitata* Grum), lichen (e.g. *Collema tenax* Ach.), mosses (e.g. *Bryum argenteum* Hedw, *Didymodon constrictus* (Mitt)), and cyanobacteria (e.g. *Microcoleus vaginatus* Gom, *Hydrocoleum violaceum* Marten) (Li *et al.* 2008). The average slope in the shrubland area was 10° (Fig. 2*b*). The zonal soils in both grass- and shrub-dominated areas were grey desert soil derived from colluvial–eluvial material according to Chinese Soil Taxonomy, which is equivalent to the Camborthid in terms of USDA Soil Taxonomy classification (Shapotou Desert Research Station 1980; Qiu *et al.* 2000; Soil Survey Staff 2010). The bulk density, clay and silt contents, and amounts of organic C (OC) and total N (TN) indicated that soil quality has been significantly degraded following the vegetation replacement (Table 1). Soil in the inter-shrub patch was sandy, whereas both the shrub patch and grassland habitat had loamy sand soil, according to the USDA Soil Taxonomy. The entire grassland and shrubland site has been fenced for >30 years to exclude grazing.

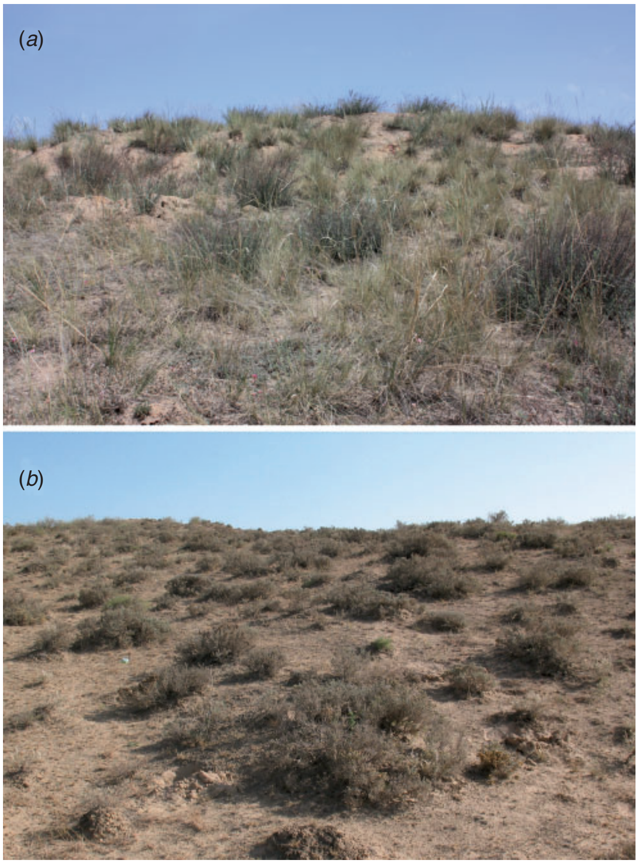

Fig. 2. General view of (*a*) grassland and (*b*) shrubland at the experimental sites.

Experimental design

Ten plots 1 m by 1 m were set up in grassland (grassland treatment, GT). Twenty experimental plots, each measuring 1 m by 1 m, were set up in shrubland: 10 plots on the BSC-covered bare soil between shrubs (bare soil treatment, BT); and 10 plots with a natural mosaic of both shrubs and bare soil (shrub treatment, ST), each with 20% of the surface covered by shrubs and 80% BSC-covered bare soil. In order to minimise differences in runoff, sediment, and C and N losses among replicates in each treatment, and to ensure the comparability of the data between treatments, the plots selected for each group were kept with identical slope angle. The shrub patches in ST, measuring 0.4 m by 0.5 m, were in the downslope part of the 1-m^2 plots. A 15-cm-high steel sheet was carefully hammered 10 cm into the soil to enclose each plot, and a flume and measuring cylinder were attached at the lower end of each plot for runoff and sediment collection and measurement. Runoff samples were stored in separate polypropylene bottles for each plot.

Simulated rain at an intensity of 80 mm/h was applied once to each plot, with a portable sprinkler rainfall simulator. A high intensity was adopted to highlight the differences between these two habitats despite its infrequency in the Tengger Desert. The simulator delivered water from a height of 2.0 m; droplet average diameter was 2.0 mm and each drop produced a kinetic energy of 1.35 J/m^2.s. Each experiment lasted for 45 min, by which time it was assumed that the values of both infiltration rate and runoff outflow would have stabilised. The actual simulated rainfall intensity on each plot was measured using four rain gauges located at each corner of each plot.

Measurements

Once runoff started, samples of outflow were initially collected at 1-min intervals, and then at 3-min intervals from 15 min after the rainfall started. Samples were immediately frozen for storage in the laboratory. When thawed, each sample was centrifuged and filtered through a 0.45-μm fibreglass filter. Remaining sediments were oven-dried at 60°C. The filtered runoff water and sediments were analysed for TN with a Kjeltec System 1026 distilling unit (Tecator AB, Sweden), and for OC with the Walkley and Black dichromate oxidation method (Nelson and Sommers 1982). Yields of TN and OC were recorded as the sum of their respective values in runoff and sediments; N and C background values in the simulated rainfall water were subtracted from their content in runoff water.

Table 1. Comparison of physiochemical properties of topsoil (0–10 cm) in the study area

Within rows, means followed by the same letter are not significantly different at $P = 0.05$ ($n = 10$)

	Bare soil patch	Shrub patch	Grassland
Bulk density (g/cm^3)	1.43 ± 0.21a	1.18 ± 0.11b	1.17 ± 0.14b
Clay content (%)	5.68 ± 1.13a	11.81 ± 2.06b	12.26 ± 1.87b
Silt content (%)	15.17 ± 2.49a	22.61 ± 3.74b	23.69 ± 2.88b
Organic carbon (g/kg)	3.78 ± 1.03a	8.14 ± 1.12b	8.97 ± 0.94b
Total nitrogen (g/kg)	0.23 ± 0.08a	0.58 ± 0.14b	0.62 ± 0.12b

The main parameters calculated for each simulated rainfall event were as follows: time to ponding, which is the time for water to accumulate over 60% of the plot surface; time to runoff; amount of rainfall for runoff commencement; and runoff coefficient (i.e. runoff as a proportion of total applied water).

Statistical analyses

Differences in runoff, sediment, and C and N losses between grassland and shrubland were quantified by analysis of variance (ANOVA) in conjunction with Tukey's range test to distinguish between the means for the different plot types, using SPSS Statistics 10th edition software (SPSS, Chicago, USA).

Results

Hydrological response

For the BT plots, time to ponding and time to runoff occurred 3.3 and 4.6 min after the rainfall started, respectively. In comparison, time to ponding and time to runoff for both ST (6.1 and 20.1 min) and GT (6.6 and 22.9 min) typically took much longer ($P < 0.05$); no significant difference was found between ST and GT ($P > 0.05$; Table 2). The amount of rainfall for runoff commencement for BT was 6.1 mm, which was significantly smaller than for both ST (26.9 mm) and GT (30.0 mm); however, we did not find any statistical difference in this parameter between ST and GT, all which confirmed that shrub patches and grasses both delay or decrease runoff generation, and more rainwater infiltrated directly into the soil than in the BT plots. On average, water took twice as long to pond and four times longer to commence runoff for ST and GT compared with BT, and the ST and GT plots required more than four times as much rainwater as BT to produce runoff (Table 2). All of these observations indicated lower infiltration rates in shrubland than in grassland.

Table 2. Mean values (standard deviations in parentheses) of selected hydrological parameters for different treatment plots (BT, inter-shrub bare soil; ST, shrub; GT, grassland)
Within rows, means followed by the same letter are not significantly different at $P = 0.05$ ($n = 10$)

	BT	ST	GT
Initial soil moisture (%)	2.94 (0.18)a	2.82 (0.37)a	2.97 (0.31)a
Time to ponding (min)	3.32 (0.72)a	6.12 (0.61)b	6.60 (0.48)b
Time to runoff (min)	4.58 (1.88)a	20.14 (0.64)b	22.92 (1.96)b
Rainfall to produce runoff (mm)	6.11 (2.51)a	26.86 (3.04)b	30.03 (3.74)b

Runoff discharge and sediment loss

Runoff discharge in the BT plots generally peaked at ~10 min after the simulated rainfall began, during which time the plots received 13 mm of rain; the ST and GT maxima were reached some 15–20 min later (Fig. 3*a*). In almost all cases, the discharge rate was maintained for the remainder of the experiment for all three treatments. Fluctuations of runoff discharge in the BT plots were more pronounced than for either the ST or GT plots. Mean runoff coefficients were 53.33%, 15.22%, and 19.01% for BT, GT, and ST, respectively. It was assumed that BSC-covered soil patches in the ST and BT plots produced the same amount of runoff per unit area, weighting the volume of runoff from these two kinds of plots according to the proportion of shrub patches (11%) and inter-shrub patches (89%), this gave a runoff coefficient of 34.46% for the shrubland habitat (Table 3), indicating that runoff generated from shrubland would be significantly greater than from grasslands.

The maximum sediment discharge for BT usually occurred ~12 min after the start of the experiment, during which 16 mm of rain was applied to the plots, whereas the sediment discharge in the ST and GT plots peaked 18–20 min later (Fig. 3*b*). In most cases, the rates of sediment discharge for ST and GT were unchanged during the rest of the simulated rainfall, and in a minority of cases, they decreased slightly during the last part of the experiment. For almost the whole process, sediment

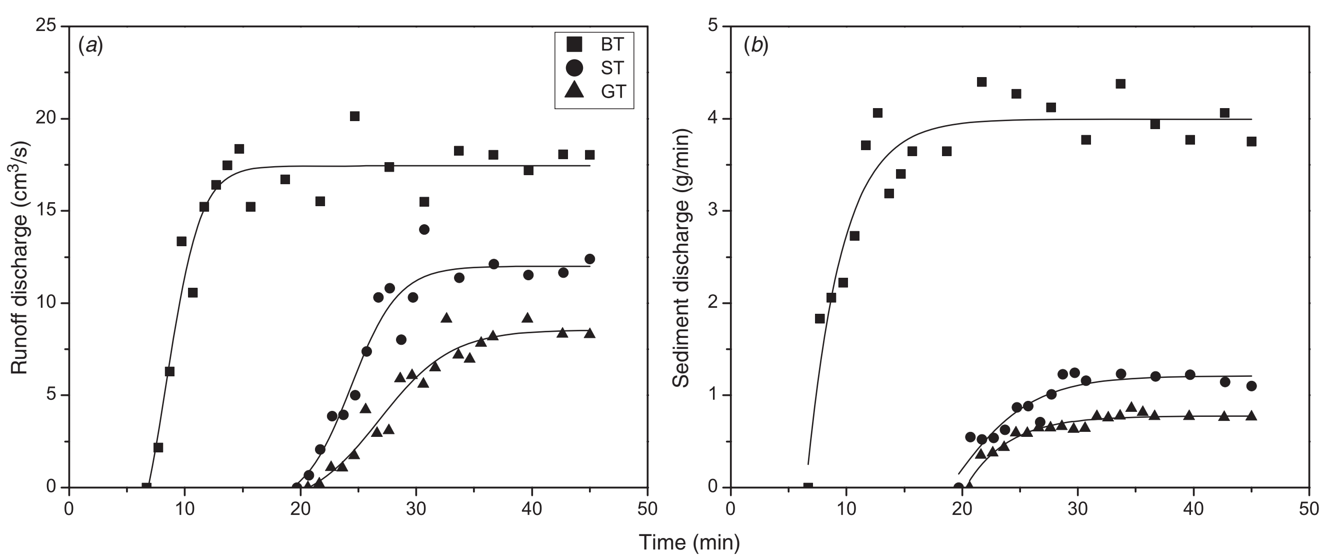

Fig. 3. Changes in (*a*) runoff and (*b*) sediment discharge during simulated rainfall experiments on grassland (GT), shrub (ST), and inter-shrub bare soil (BT) plots. Plots GT2, ST5, and BT9 are presented as examples.

Table 3. Mean values (standard deviations in parentheses) of total resource yields and concentrations from different types of treatment plots (BT, inter-shrub bare soil; ST, shrub; GT, grassland)/habitats

Within rows, means followed by the same letter are not significantly different at $P = 0.05$ ($n = 10$)

	GT	BT	ST	Shrubland
Runoff coefficient (%)	15.22 (0.37)a	53.33 (1.95)b	19.01 (3.60)a	34.46 (2.86)c
Sediment loss (g/m^2)	15.91 (3.86)a	80.34 (17.89)b	16.00 (7.98)c	44.95 (13.89)d
Organic carbon (OC) loss (g/m^2)	0.94 (0.03)a	2.82 (0.79)b	0.82 (0.34)c	2.06 (0.49)d
Total nitrogen (TN) loss (g/m^2)	0.11 (0.04)a	0.29 (0.09)b	0.09 (0.03)c	0.23 (0.07)b
Volume-weighted conc. of OC (g/L)	0.103	0.088	0.072	0.083
Volume-weighted conc. of TN (g/L)	0.012	0.009	0.007	0.011

discharge for the BT plots showed greater fluctuation than the other two plot types. Sediments collected from different plots represented total soil material eroded by rain splash and runoff-driven processes. Average sediment yields in BT, GT, and ST were 80.34, 15.91, and 16 g/m^2, respectively (Table 3). Weighted by the average cover of shrub patches and inter-shrub patches, sediment yield in runoff from shrubland was 44.95 g/m^2, which was 2.8 times more than from grassland.

In most cases, time-dependent sediment concentrations of all three treatments increased quickly in the initial period of runoff generation, and peaked for BT plots ~10 min after the simulated rainfall began and for both ST and GT ~15 min later. In all cases, sediment concentrations declined sharply soon after their maxima were reached (Fig. 4).

Carbon and nitrogen yields and concentrations

The maximum discharge of OC for BT plots generally occurred ~15 min after the start of the experiment, during which ~20 mm of simulated rain was applied to the plots. For the GT and ST plots, however, the peak was observed 25 min later, during which these plots have received >53 mm of simulated precipitation (Fig. 5*a*), and in most cases, the OC discharge remained more-or-less constant for the remainder of the simulated rainfall. Mean OC yields for BT, GT, and ST were 2.82, 0.94, and 0.82 g/m^2, respectively. The cover-weighted

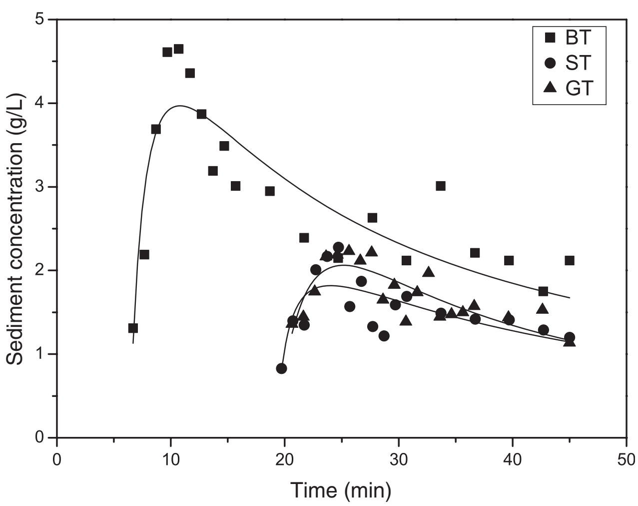

Fig. 4. Changes in sediment concentration during simulated rainfall experiments on grassland (GT), shrub (ST), and inter-shrub bare soil (BT) plots. Plots GT2, ST5, and BT9 are presented as examples.

value for shrubland was 2.06 g/m^2. Volume-weighted concentrations of OC (total OC loss divided by the total amount of runoff) were 0.088, 0.103, and 0.072 g/L for BT, ST, and GT, respectively. By the same estimation, the weighted OC concentration for the shrubland was 0.083 g/L, which meant that the concentration of OC in the runoff from grassland was 1.24 times higher than from shrubland (Table 3).

The patterns of TN discharge were similar to those for OC (Fig. 5*b*). Average TN losses were 0.29, 0.11, and 0.09 g/m^2 for BT, GT, and ST, respectively. The cover-weighted value for shrubland was 0.23 g/m^2, which was 2.09 times higher than from grassland. Volume-weighted concentrations of TN (TN loss divided by the total runoff volume) were 0.009, 0.012, and 0.007 g/L for BT, GT, and ST, respectively. The cover-weighted N concentration was 0.011 g/L for shrubland, which was slightly less than for grassland (Table 3).

For most of the BT and ST plots, the time-dependent concentrations of OC rose briefly and peaked shortly after the start of runoff, and then declined immediately. For the GT plots, OC concentrations decreased dramatically with time for the duration of runoff (Fig. 6*a*). Concentrations of OC in runoff were highest in GT and lowest in BT for the duration of each simulated experiment. The data for GT plots also showed greater fluctuation than for the other two plot types. A similar pattern was found for time-dependent concentrations of TN; that is, in all plots of BT, ST, and GT, the N concentration steadily decreased with time throughout almost the entire experiment, except for a brief rise in the ST and GT plots shortly after runoff began (Fig. 6*b*).

For all the samples collected from each treatment, negative linear correlations were found between OC concentrations and runoff discharge (Fig. 7*a–c*), and similar patterns were found for TN concentrations (Fig. 7*d–f*).

Discussion

Degradation of grasslands in arid and semi-arid desert regions is a worldwide phenomenon (e.g. Humphrey 1958; Polley *et al.* 1997; Scott *et al.* 2006). It is broadly believed that degradation will eventually result in vegetation cover loss and severe alteration of ecohydrological processes (Milton *et al.* 1994). Since the early 19th Century, the occurrence of various shrubs and subshrubs have significantly increased in former grasslands on the south-eastern fringe of the Tengger Desert, due mainly to overgrazing by livestock and the expansion of the desert (Li and Jia 2005). This has led to a sparse patchy distribution of vegetation, characterised by a mosaic of BSC-covered patches

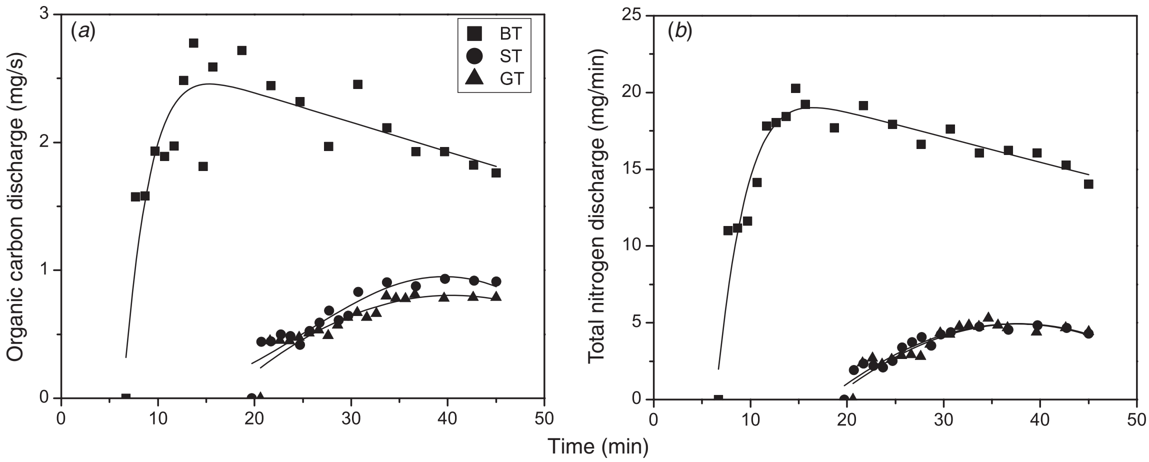

Fig. 5. Changes in the discharge of (*a*) organic carbon and (*b*) total nitrogen during simulated rainfall on grassland (GT), shrub (ST), and inter-shrub bare soil (BT) plots. Plots GT2, ST5, and BT9 are presented as examples.

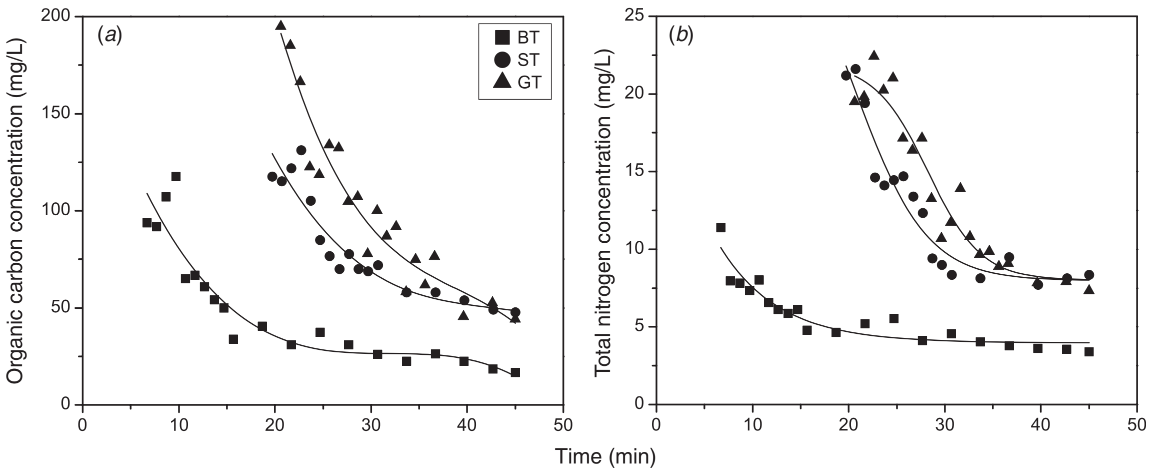

Fig. 6. Changes of (*a*) organic carbon and (*b*) total nitrogen concentration during simulated rainfall experiments on grassland (GT), shrub (ST), and inter-shrub bare soil (BT) plots. Plots GT2, ST5, and BT9 are presented as examples.

and distinct patches of perennial vegetation, typically shrubs, and a decrease in vegetation coverage. This has inevitably modified runoff and induced erosion and nutrient losses (Li *et al.* 2008).

In the rainfall simulation experiments, it took much longer for the plots of both ST and GT to pond and to generate runoff than the BT plots, and the amounts of simulated rain required to produce runoff were significantly greater for ST and GT (Table 2). Times to ponding and to runoff and amount of rainfall for runoff commencement were slightly greater for GT than for ST, although the measured differences were not significant. However, the actual values for ST would be much smaller than the experimental data indicated once the actual proportions of shrub patches (11%) and inter-shrub, crust-occupied patches (89%) were accounted for, and would be much smaller than those for the GT plots, indicating lower soil-water infiltration in shrubland than in grassland. The GT plots also produced about the same amount of runoff as ST plots, but less than BT plots and also less than the estimated value for shrubland (Fig. 3*a*, Table 3), confirming changes to the hydrological processes induced by vegetation replacement. Many similar results have been reported in the literature, including greater runoff on shrubland habitats and greater hydraulic conductivity in grasslands (e.g. Abrahams *et al.* 1995; Castillo *et al.* 1997; Schlesinger *et al.* 1999; Braud *et al.* 2001; Gimeno-García *et al.* 2007).

For a given rainfall event, runoff generation time and amount are determined to a large degree by soil infiltration capacity, which is influenced by soil porosity and the duration of rain water on the soil surface (Barger *et al.* 2006). Compared with original grasslands, the appearance of large areas of bare soil following woody plant invasion results in the decrease or even loss of many biotic processes in these areas, such as root growth and animal digging, that create macropores and facilitate soil water infiltration in saturated conditions (Guebert and Gardner 2001; Ludwig *et al.* 2005). Consequently, this kind of vegetation change is associated with decreased hydraulic conductivity and increased runoff (e.g. Castillo *et al.* 1997). More importantly, previous research has shown that the colonisation of biological soil crusts on the surface of inter-shrub areas results in reduced

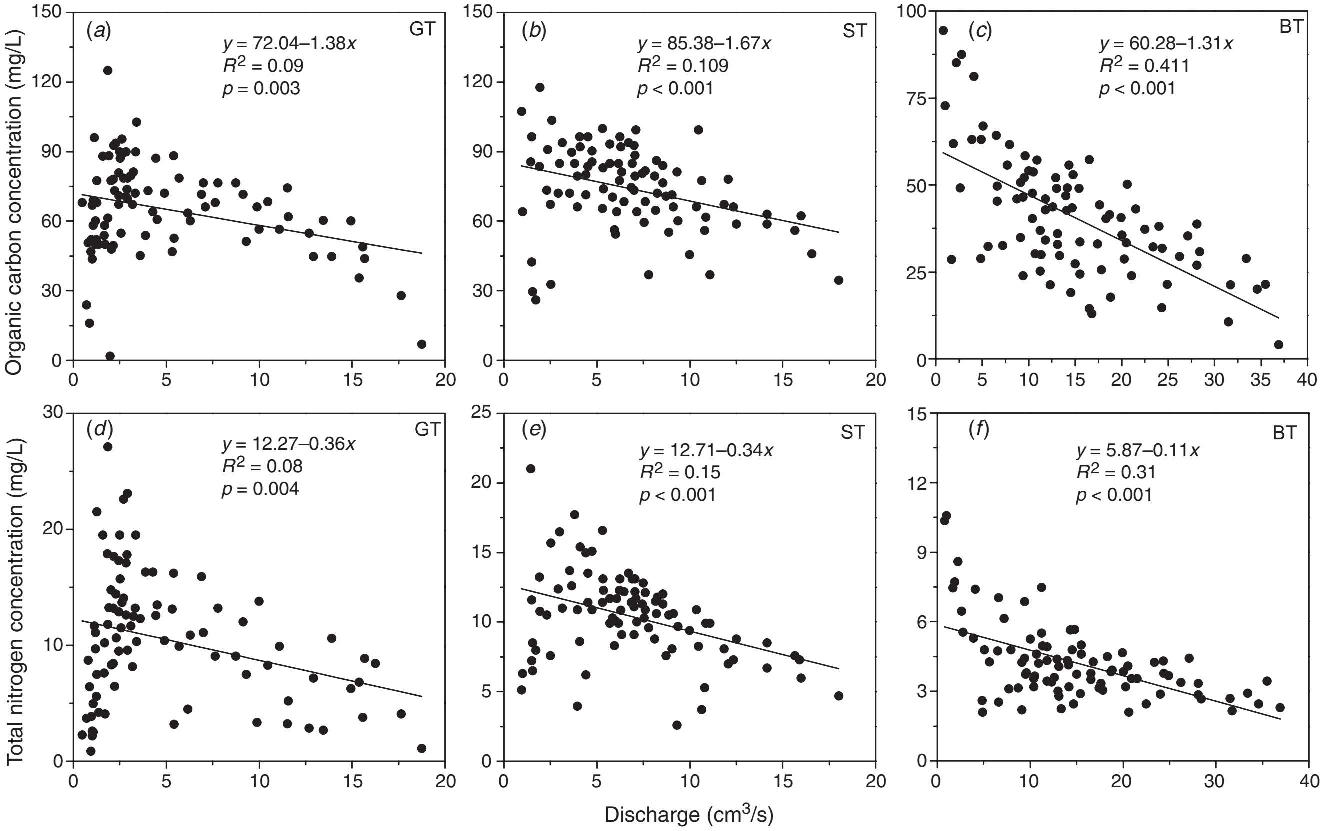

Fig. 7. Relationship between the concentration of (*a*, *b*, *c*) organic carbon and (*d*, *e*, *f*) total nitrogen, and runoff discharge volume at the time of each sampling. Treatments: GT, grassland; ST, shrub; BT, inter-shrub bare soil. Circles represent paired values for each sample; solid lines are fitted curves, assuming that concentration is a function of the discharge rate.

infiltration rate (Eldridge *et al.* 2002; Li *et al.* 2008), mainly due to the cementing of soil particles by polysaccharides secreted by cyanobacteria (Mazor *et al.* 1996; Mager and Thomas 2011), as well as dust falling on the crust and organisms comprising BSCs expanding when exposed to rainwater (e.g. George *et al.* 2003; Chamizo *et al.* 2012). Other researchers have shown that BSCs have a negative impact on infiltration in coarse-textured soils, to control runoff, and thereby affect runoff-erosion and sedimentation (Yair 1990; Kidron and Yair 1997); this was first demonstrated by Yair (1990) in the western Negev Desert, Israel. Kidron and Yair (1997, 2001) found that no runoff was generated from plots devoid of biological soil crust in the Negev Desert, due to the high porosity and infiltration rate of the soil, whereas significant runoff was produced from plots with >95% crust cover. Hence, it is not surprising that greater runoff and sediment were produced from shrubland than from grassland in our study (Figs 3 and 4, Table 3).

In our simulated rainfall experiments, both time-dependent concentrations and volume-weighted concentrations of OC and TN in runoff were greater in GT plots than in BT and ST plots (Fig. 6, Table 3), and the cover-weighted concentration of GT plots was also greater than in BT and ST (Table 3). Despite this, C and N discharges and yields in runoff were smaller from GT than from BT and ST (Fig. 5, Table 3), and total C and N losses from grassland were lower than the cover-weighted values for shrubland (Table 3). These results do not agree with those of Schlesinger *et al.* (1999), who, using higher intensity rainfall simulation than was adopted here, reported that nutrient losses from grassland were greater than those from shrubland. They attributed this to the lower nutrient content in shrubland (Schlesinger *et al.* 1996). However, in our work, runoff coefficients for ST and BT, as well as the weighted values for shrubland habitat, were larger than those from GT plots (Table 3), indicating that the higher yields of OC and N from shrubland largely stemmed from the greater quantity of runoff from this habitat (Fig. 3). On the other hand, the higher OC and N yields from inter-shrub plots and shrubland were mainly derived from well-developed BSCs, which have been recognised as a vital contributor of soil C and N in arid and semi-arid ecosystems (Housman *et al.* 2006; Su *et al.* 2011). Further, greater sediment yield from shrubland than from grassland was one of the important causes for higher C and N loss, owing to the higher erosivity of rain resulting from the decline or even loss of the physical protection provided by shrub canopies against splash erosion and overland flow (Bochet *et al.* 1998).

In our rainfall simulation experiments, the concentrations of OC and TN show a negative linear correlation with the amount of runoff discharge (Fig. 7), suggesting similarity in C and N production with the increasing runoff. The discharge of C and N in BT declined steadily with time, whereas the peak values were maintained for the remainder of the experiment in ST and GT (Fig. 5). These results imply that the decreases in C and N concentrations in runoff with time in ST and GT plots were due to dilution by the increasing runoff discharge, while the decreases in concentration in runoff from BT can be attributed to the combined effects of increases in runoff amount and decreasing capacity in C and N supply.

In the present study, the differences in runoff, sediments, and C and N loss between these two habitats are compromised somewhat by the slightly steeper slope of the grassland plots; nevertheless, it was still found that the runoff from shrubland, and from grasslands that have suffered long-term degradation, carried large amounts of sediment and resulted in the loss of both C and N. It also showed the expected increased nutrient losses following vegetation degradation. If this general trend is allowed to continue without intervention, then natural rainwater, sediment, and nutrients will no longer be efficiently captured and stored within the ecosystem and the landscape will become degraded, non-conserving, and dysfunctional (Eldridge *et al.* 2002; Ludwig *et al.* 2005), leading to continuous desertification (Avni *et al.* 2006).

Conclusions

Grassland degradation, typically characterised by invasions of woody plants, is a widely recognised phenomenon in the Tengger Desert. Understanding the effects of vegetation change on hydrological processes is helpful for devising strategies or practices for the prevention of desertification in these areas. Simulated rainfall experiments indicated that ponding and runoff occurred more quickly in inter-shrub bare soil than in shrub plots and desert grassland, while the volume of rainfall for runoff commencement in both shrub and grassland plots was greater than in inter-shrub plots, with greater runoff and sediments being produced from shrubland than from grasslands. Likewise, runoff-induced OC and TN losses from shrubland were both greater than from grasslands, despite both lower time-dependent concentrations in inter-shrub and shrub plots and lower weighted mean concentrations of C and N for shrubland than grassland. The increased runoff and nutrient losses following degradation of grasslands into patchy shrublands will inevitably result in a dysfunctional ecosystem and an increased vulnerability to further desertification.

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