Ecological Engineering 95 (2016) 10–18

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Short-term grazing exclusion from heavy livestock rangelands affects vegetation cover and soil properties in natural ecosystems of southeastern Iran

Mahdieh Ebrahimi*, Hamideh Khosravi, Masood Rigi

University of Zabol, Department of Range and Watershed Management, Mofateh Street, Jahad Square, Zabol 9861335856, Iran

ARTICLE INFO

Article history:
Received 24 March 2016
Received in revised form 30 May 2016
Accepted 16 June 2016

Keywords:
Grazing impacts
Soil physical and chemical properties
Plant species diversity
Arid rangelands

ABSTRACT

Grazing exclusion is an effective rangeland management practice used to achieve sustainability of natural ecosystems worldwide. To clarify the effects of short-term grazing exclusion on the plant community and soil characteristics, we investigated the plant and soil properties by comparing overgrazing and short-term grazing exclusion (underwent exclusion for 2, 4, 6 years) sites in an arid rangeland of southeastern, Iran. Soil samples were extracted at depth of 0–30 cm. In total, 22 species from 9 families and 18 genera were observed along the plant communities. Results showed that the livestock exclusion significantly affected the community composition for species, genera, and families. The numbers of species, genera, and families increased slowly during exclusion, reaching their maximum value in the 6 years' exclusion, while the minimum number of species, genera, and families were observed in the overgrazed site. The numbers of species and the proportion of annual and perennial species were significantly affected by the exclusion. The 6 years' exclusion exhibited the highest numbers of plant species, of which approximately 63.63% were perennials. The soil nutrient values gradually increased during exclusion. Organic carbon, total nitrogen, available potassium, and available phosphorus attained significantly greater values under the 6 years' exclusion. The pH level was significantly higher in the overgrazed soils compared to the grazing exclusions soils. The EC value was statistically similar under the four treatments. The particle size distribution showed more silt and clay and less sand in the soils of grazing exclusion sites compared with the soil of overgrazed site. The silt and clay values were the highest in the soils under 6 years' exclusion. Totally, the results imply that short-term exclusion had a great influence on the vegetation restoration and soil conservation of degraded ecosystems in arid regions.

© 2016 Elsevier B.V. All rights reserved.

1. Introduction

Unreasonable human management of the soil resources is resulting in land degradation due to soil erosion, soil organic matter exhaustion, loss of soil structure, pollution, forest fires or deforestation (Zhao et al., 2013; Keesstra et al., 2014; Lu et al., 2015). This is why there is a need to restore and rehabilitate soils as a source of nutrients and services to humankind (Lu et al., 2015; Roa-Fuentes et al., 2015). Grazing by domestic ungulates is one of those human uses of the land that will effect on many ecosystem processes and functions, such as nutrient pool and cycling, soil moisture and structure, soil degradation, net primary productivity, vegetation composition, belowground biomass productivity, as well as the associated changes in the soil microbial community (Wang et al., 2014; Costa et al., 2015; Tarhouni et al., 2015; Lu et al., 2015). It is critical to obtain a better understanding of how grazing influences the key properties of ecosystem function and sustainability and, thereby, to provide guidelines for improving rangeland management practices (Wang et al., 2014).

In Iran, rangelands are important natural resources with great ecological, economic and social importance due to their crucial role in the development of rural areas. Generally, they support forage for herbivores; offer the opportunity for outdoor recreational activities and enjoyment of nature (Amiri, 2009). In addition, they play great ecological role in conserving biodiversity. However, continuous overgrazing has led to declining biodiversity and vegetation cover, along with accelerated soil erosion (Prieur-Richard and Lavorel, 2000). It has also decreased biodiversity with widespread species loss and the replacement of endemic and specialist species by exotic species (Stanners and Bordeau, 1995).

* Corresponding author.
E-mail address: maebrahimi2007@uoz.ac.ir (M. Ebrahimi).

http://dx.doi.org/10.1016/j.ecoleng.2016.06.069
0925-8574/© 2016 Elsevier B.V. All rights reserved.

Grazing exclusion from the creation of large-scale enclosures has become a common management strategy to prevent rangeland degradation and sustain rangeland ecosystem function by the restoration of degraded vegetation and improvement of soil quality throughout the world in recent decades (Wang et al., 2014; Strahan et al., 2015; Lu et al., 2015). However, few studies have evaluated such management practices in the rangelands of Iran. For instance, soil organic carbon in the surface soil under grazing exclusion conditions was reportedly increased in a semiarid woody rangeland (22 years of grazing exclusion) in the Zagros Mountains, central Iran (Raiesi and Riahi, 2014). An assessment of species composition of exclusion in Kerman provinces of Iran showed that species composition appeared to reach its maximum towards the middle of the succession (Ebrahimi et al., 2014).

Previous studies examining the effect of grazing exclusion on rangeland have primarily investigated the vegetation productivity, plant species and communities (Gonzales and Clements, 2010; Schultz et al., 2011). Nevertheless, soil also plays an important role in supplying organic matter and cycling nutrients, such as nitrogen and carbon; it could also directly affect vegetation productivity, community composition and plant species richness during the rangeland restoration succession process (Mekuria and Aynekulu, 2013). Comparison of vegetation composition and diversity, including species richness and abundance, plant functional groups and soil properties in fenced areas could reflect the system stability and resilience of the rangelands (Metzger et al., 2005; Al-Rowaily et al., 2015). Such approach can help to guide sustainable management strategies for conserving natural ecosystem goods and services (Wang et al., 2014; Al-Rowaily et al., 2015).

However, in Iran, little research has been conducted to determine the effects of grazing exclusion on biodiversity conservation and soil properties in the natural ecosystems. This study was therefore conducted to determine the influence of short-term grazing exclusion with different ages on the vegetation cover and soil properties of the vegetation communities in a steppe rangeland of Iran to determine whether the ecosystem has been restored comparison with open grazing.

We contrast grazing exclusion and open grazing treatments to address the following questions: (1) how does grazing exclusion affect the vascular plants richness and diversity in arid region of eastern Sistan and Baloochestan, Iran? and (2) do the soil properties responses to short-term grazing exclusion differ among different ages? Our hypothesis posits that (1) short-term grazing exclusion helps the establishment of plant species and (2) physico-chemical properties of the soil in the short-term grazing exclusions differ from open grazing.

2. Materials and methods

2.1. Study area

Taftan rangeland is located in Sistan and Baloochestan province in Iran, between latitude 28°30′41″–28°39′00″N and between longitude 60°51′35″–61°00′09″E (Fig. 1). The experimental area is characterized by dry summers, a rainy season, and warm autumn and the cool winter weather. According to data available for the period 2006–2014 at the study site from the National Meteorological Information Center of Iran, the mean annual rainfall levels reach 160 mm. The mean annual evaporation reaches approximately 60.10 mm, denoting a high water deficit in the region. The minimum and maximum elevations are 1382 m in the south and 4042 m in the north. The mean maximum temperatures reach 30 °C in May and June. The mean minimum temperatures range from 8.7 °C in December and January, and occasional periods of subfreezing surface temperature occur. The site was within an area where the topography was characteristic of plains, mounds and ridges. The vegetation types are dominated by arid land vegetation (e.g. *Hammada salicornia*, *Zygophyllum eurypterum*, *Artemisia santolina*, *Salsola tomentosa*). The growing season is from March to May. Vegetation in the area has changed considerably over the past several decades, primarily due to overgrazing by goats.

2.2. Sampling method

Since the restore-rangeland ecological program started in 2010, more than 2000 ha of arid rangelands in Taftan have been fenced to exclude livestock grazing. We conducted a multi-site survey during the growing season from March to June in 2014. We selected three fenced sites that underwent succession for 2, 4 and 6 years. An overgrazed site was conducted as a control. There were no differences between topography, soil type, and spatial heterogeneity among the selected sites. The fenced areas covered a total area of about 10 ha. The mean stocking rate of the grazed rangeland was 60–70 AU ha^{-1} from April to September, and 30–40 AU ha^{-1} from October to November. The data collected for the year of 2014. At each site, we conducted a comprehensive investigation of the vegetation types. The age of enclosures was determined from data provided by Agriculture and Natural Resources Research Center of Sistan and Baloochestan province, Iran. Species identification and nomenclature were carried out in the laboratory, University of Zabol, according to Rechinger (1968, 1970, 1972, 1997, 1984), Rechinger and Schiman-Czeika (1964) and Zielinski (1982). Chorotype of the recorded species were identified according to Zohary (1963). A total of 20 sampling stands (50 m × 50 m) were selected to represent the prevailing habitat and community variations in the sites (5 stands for each). Within each stand, vegetation properties were measured using the simple transect line (100 m) method within quadrats (7 m × 7 m) with a systematically-randomized method. In total, 60 transects were sampled in the sites. Data on vegetation/canopy cover was obtained using the quadrat estimation methods (Hanley, 1978). The plant density was measured by counting the number of individuals of a species in a plot (Coulloudon et al., 1999). Proportion of bare soil and litter in each site was measured using the quadrat estimation methods (Hanley, 1978). Plant species were classified as class I (High) II (Medium) and III (Low) according their palatability. Palatability is a plant characteristic that refers to the relish with which plants or its parts or feed is consumed as stimulated by the sensory impulses of grazing animal (Heath et al., 1985). In the present study, palatability determined using reference texts (Baghestani et al., 2001; Arzani et al., 2004; Bagheri et al., 2007). The importance value (IV) for the plant species was calculated using the following formula (Zhang et al., 2006):

$$IV = {RD + RC + RF}/{3}$$

where RD is the relative density (the ratio of the number of individuals of a species to the total number of individuals of all species, %); RC is the relative cover (the ratio of the cover of a species to the total cover of all species, %); and RF is the relative frequency (the ratio of the percentage frequency of a species to the total frequency of all species, %) (Jiang et al., 2006). In total, 150 vegetation plots were sampled in each site. Shannon species diversity index [$H' = -\sum$ pi. lnpi] (Magurran, 1988) were determined by calculating the frequency of each plant species (pi = proportion of points along each transect at which species i was recorded). Plant species richness (S = number of species sampled per transect) and evenness of species abundances (Pielou's J index = H′/lnS) were also calculated for each transect.

The experimental sites consist of soils of silt loam texture that are taxonomically characterized as moderate, loamy, mixed, and

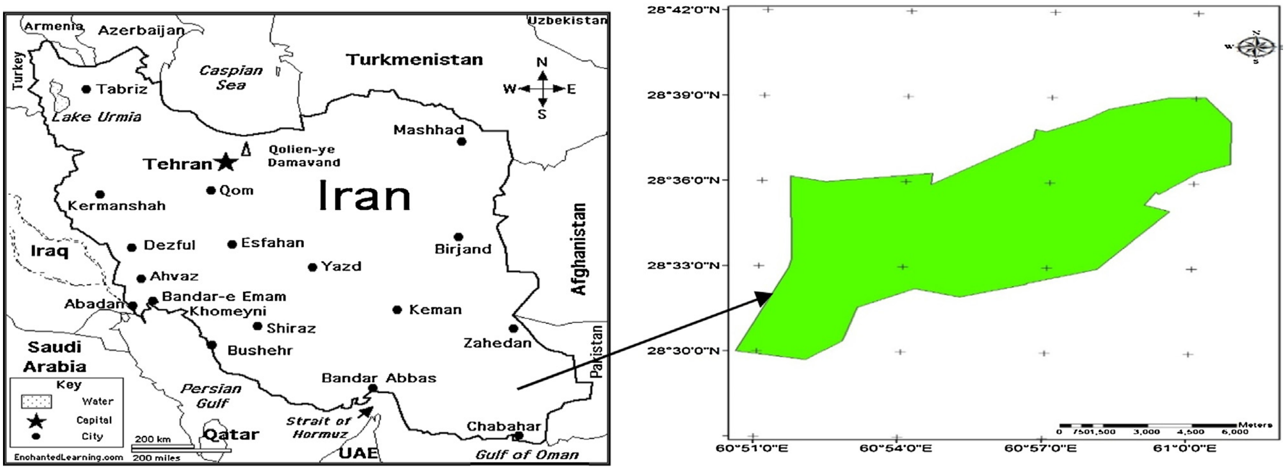

Fig. 1. Study area in Sistan and Baloochestan province, southeastern Iran.

Table 1
Numbers of taxa (species, genera and families), annuals and perennial plant sampled at grazed and excluded sites.

Site	Number of species	Number of Genera	Number of Families	Annuals	Proportion of total (%)	Perennials	Proportion of total (%)
Overgrazed site	9.00 ± 0.30[c]	9.01 ± 0.30[c]	5.00 ± 0.10[b]	5.00 ± 0.01[b]	61.24 ± 3.21[a]	4.00 ± 0.10[b]	11.42 ± 4.02[d]
2 years' exclusion	11.00 ± 0.30[c]	9.00 ± 0.30[c]	7.00 ± 0.10[a]	5.00 ± 0.01[b]	34.19 ± 2.16[b]	6.00 ± 0.10[b]	56.16 ± 3.00[c]
4 years' exclusion	18.00 ± 1.23[b]	14.00 ± 0.4[b]	8.00 ± 0.10[a]	4.00 ± 0.01[b]	15.42 ± 0.11[c]	14.00 ± 0.30[a]	72.23 ± 5.10[b]
6 years' exclusion	21.00 ± 1.44[a]	17.00 ± 0.40[a]	8.00 ± 0.10[a]	7.00 ± 0.01[a]	18.61 ± 0.11[c]	14.00 ± 0.30[a]	86.12 ± 5.10[a]

Values shown are the means ± SE. Values within a column followed by different letters are significantly different ($p < 0.05$, post hoc Duncan test)

hyper-thermic family of typic Cambothrid according to the US soil taxonomy classification (Table 4). Soil samples were taken at three points along the transects in each stand (within quadrats at three points) using a soil auger from the surface layers (0–30 cm). The soil samples in each quadrat were then mixed together to make one composite sample. A total of 180 soil samples were selected (45 soil samples at each site). The soils were put in plastic bags with label; they were thereafter air dried and taken to the laboratory at the Department of Range and Watershed Management, University of Zabol, for analysis of soil physical and chemical properties. The soil's texture was determined using laser diffractometry (Wang et al., 2012); the soil pH was determined in a 1:5 soil to distilled water slurry after one hour of agitation using a digital pH-meter (Model 691, Metrohm AG Herisau Switzerland) (Thomas, 1996); electrical conductivity of saturated soil paste extract (ECe) using an EC-meter (DDS-307, Shanghai, China) (Rhoades, 1996); total soil N (N_{tot}) was analyzed calorimetrically with a continuous flow ion analyzer following wet digestion in sulfuric acid (Bremner, 1996); Calcium carbonate ($CaCO_3$) was determined volumetrically by a calcimeter (Allison and Moodie, 1965). Organic carbon content (OC) was determined using the methods described by Lo et al. (2011). Available phosphorus (AP) was determined by the method of Bray and Kurtz (1954). Available potassium (AK) was measured by flame photometry method (Knudsen et al., 1982).

2.3. Statistical analysis

Statistical analyses of the experimental data were performed using the SPSS 18.0. All reported results are the means of five replicates and deviations were calculated as the standard error of the mean (SEM). The statistical processing was mainly conducted by analysis of variance (after testing for homogeneity of variance and confirming a normal distribution). A paired T test was used to test for significant differences in vegetation importance values between grazed and excluded sites. Duncan test post hoc analysis was performed to define which specific mean pairs were significantly different. A probability of 0.05 or lower was considered as significant.

3. Results

3.1. Grazing exclusion effects on floristic composition

We collected a total of 22 plant species in the study areas (Table 1) from 9 families and 18 genera. The livestock exclusion significantly affected the community composition for species, ($P < 0.05$), genera ($P < 0.05$), and families ($P < 0.05$, Table 1). The numbers of species, genera, and families increased slowly during exclusion, reaching their maximum value in the 6 years' exclusion, while the minimum number of species, genera, and families were observed in the overgrazed site (Table 1). The differences in the number of species ($P < 0.05$) and genera ($P < 0.05$), among the 4 and 6 years' fenced sites and overgrazed site were significant but not between the overgrazed and 2 years' fenced sites (Table 1). Moreover, the changes in the families number were not significant among the fenced rangelands (Table 1). The numbers of species and the proportion of annual and perennial species (Table 1) were significantly affected by the exclusion ($P < 0.05$). The 6 years' exclusion exhibited the highest numbers of plant species, of which approximately 63.63% were perennials.

The areas were dominated by species in the Compositae (30%), followed by the Papilionaceae (22%), and Graminae (16%) (Table 2). There were four dominant species based on the importance value: *Ar. santolina, S. tomentosa, Z. eurypterum and H. salicornia* (Table 2). Results in Fig. 2 show that livestock exclusion influenced the structural characteristics of the plant communities. The plant species of *Ar. santolina, S. tomentosa, Z. eurypterum and H. salicornia* were more abundant with higher frequency in the excluded sites than in the overgrazed rangeland. However, these plant species were not found in the overgrazed site. The noxious species *Euphorbia helioscopia* was not found in the 4 and 6 years' fenced sites (Fig. 2). However, this plant species was the more abundant with higher impor-

Table 2
Variation in importance values of plants at grazed and excluded sites. The index of change shows differences between grazed and excluded site. Bold types indicate differences are significantly different ($p < 0.05$, paired T test).

Species	Family	Growth form	Life history	Chorotype	Overgrazed site IV (%)	2 years' exclusion IV (%)	Index of change	4 years' exclusion IV (%)	Index of change	6 years' exclusion IV (%)	Index of change
Bromus gracillimus Bunge	*Graminae*	G	A	IT	0	23.11	**23.11**	21.12	**21.12**	24.55	**24.55**
Bromus tectorum L.	*Graminae*	G	A	IT	24.30	17.11	−7.19	17.32	**−6.98**	21.13	−3.17
Cardaria draba (L.) Desv.	*Brassicaceae*	F	A	Cosm	24.10	0	**−24.10**	12.18	**−11.92**	10.00	**−14.10**
Cicer spiroceras Jaub. & Spach	*Papilionaceae*	F	P	IT	0	0	–	0.90	0.90	4.23	**4.23**
Cousinia gedrosiaca Bornm. & Gauba	*Compositae*	F	A	IT	21.19	24.05	2.86	0	**−21.19**	9.75	**−11.44**
Erodium cicutarium (L.) L'Hér. ex Aiton	*Geraniaceae*	F	A	IT-ES-SS	35.25	18.11	**−17.14**	0	**−35.25**	0.70	**−34.55**
Descurainia sophia (L.) Webb. ex Prantl.	*Brassicaceae*	F	A	IT	0	0	–	0	–	15.62	**15.62**
Zygophyllum eurypterum Boiss. & Buhse.	*Zygophyllaceae*	Sh	P	IT-ES	0	7.77	**7.77**	55.20	**55.20**	61.23	**61.23**
Hammada salicornia (Moq.) Iljin	*Amaranthaceae*	Sh	P	IT-SA	0	7.06	**7.06**	50.12	**50.12**	54.56	**54.56**
Artemisia santolina Schrenk.	*Compositae*	B	P	IT	0	8.56	**8.56**	61.52	**61.52**	68.12	**68.12**
Salsola tomentosa (Moq.)	*Amaranthaceae*	F	P	IT	0	7.35	**7.35**	57.86	**57.86**	60.86	**60.86**
Artemisia lehmanniana Bunge.	*Compositae*	B	P	IT	0	9.21	**9.21**	37.10	**37.10**	42.10	**42.10**
Artemisia sieberi Besser.	*Compositae*	B	P	IT	8.48	0	**−8.48**	42.20	**33.72**	45.20	**−36.72**
Amygdalus lycioides Spach.	*Rosaceae*	Sh	P	IT-ES	0	0	–	4.20	**4.20**	7.89	**7.89**
Cymbopogon olivieri (Boiss) Bor.	*Graminae*	F	P	IT	0	0	–	32.15	**32.15**	39.21	**39.21**
Astragalus mucronifolius Boiss.	*Papilionaceae*	B	P	IT	0	0	–	9.73	**9.73**	21.86	**21.86**
Amygdalus scoparia L.	*Papilionaceae*	Sh	P	IT	0	0	–	2.20	**2.20**	19.33	**19.33**
Alhagi camelorum Fisch.	*Papilionaceae*	F	P	IT	27.19	12.78	**−14.41**	1.15	**−26.04**	6.21	**−20.98**
Cousinia stocksil C.Winkl.	*Graminae*	B	P	IT	29.42	0	**−29.42**	1.22	**−28.20**	14.77	**−14.65**
Scariola orientale (Boiss.) Sojak.	*Compositae*	F	P	IT-ES	7.22	0	**−7.22**	0.90	**−6.32**	13.21	**5.99**
Euphorbia helioscopia L.	*Euphorbiaceae*	F	A	M-ES	68.15	23.12	**−45.03**	0	**−68.15**	0	**−68.15**
Isatis minima Bunge.	*Brassicaceae*	F	A	IT	0	0	–	31.14	**31.14**	39.21	**39.21**

Sh: shrub, T: tree, F: forb, G: grass, B: bush, P: perennial, A: annual.
IV: importance value.
Chorotype: IT: Irano–Turanian, Cosm: Cosmopolitan, ES: Europe–Siberian, SS: Saharo–Sindian, SA: Saharo–Arabian, M: Mediterranean.

Table 3
Plant properties in rangeland communities at grazed and excluded sites.

Site	Density (nm^{-2})			Canopy cover (%)	Litter (%)	Bare soil (%)	Diversity (H′)	Richness	Evenness
	I	II	III						
Overgrazed site	4.11 ± 0.20^c	4.09 ± 1.21^c	9.02 ± 0.20^a	17.33 ± 2.02^c	10.03 ± 3.43^c	59.98 ± 3.43^a	0.21 ± 0.00^c	9.00 ± 0.10^c	0.50 ± 0.10^a
2 years' exclusion	6.21 ± 0.20^c	4.56 ± 1.21^c	6.33 ± 0.20^b	19.33 ± 2.13^c	17.34 ± 3.21^b	41.26 ± 3.21^b	1.17 ± 0.10^b	11.44 ± 0.20^b	0.50 ± 0.10^a
4 years' exclusion	14.35 ± 0.50^b	7.52 ± 2.33^b	3.85 ± 0.10^c	41.25 ± 3.05^b	22.90 ± 0.50^a	7.11 ± 0.50^c	2.42 ± 0.10^a	18.32 ± 0.20^a	0.70 ± 0.10^a
6 years' exclusion	17.69 ± 0.53^a	11.24 ± 2.16^a	3.27 ± 0.10^c	54.39 ± 3.21^a	31.13 ± 0.50^a	6.13 ± 0.50^c	3.18 ± 0.10^a	21.00 ± 0.11^a	0.70 ± 0.10^a

Values shown are the means ± SE. Values within a column followed by different letters are significantly different ($p < 0.05$, post hoc Duncan test).

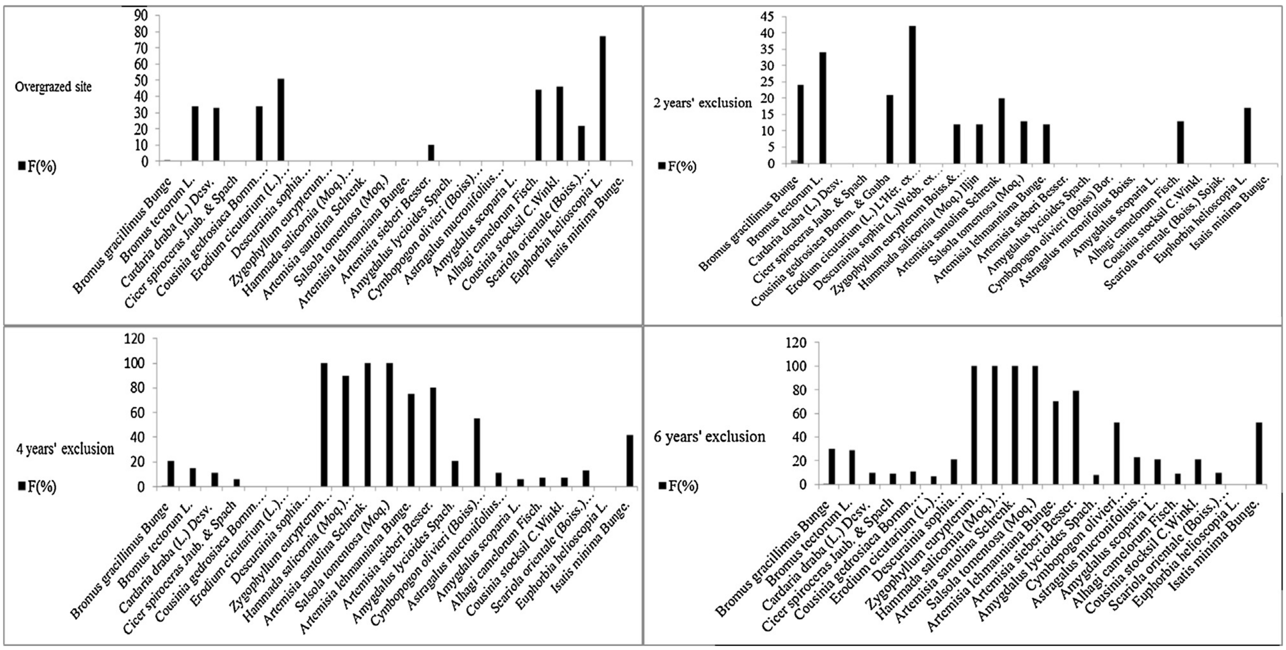

Fig. 2. Variation in frequency of plant species at grazed and excluded sites.

tance value in the overgrazed site. Chorological study (Table 2) showed that the largest proportion of the flora belongs to the Irano–Turanian elements (90.75%) followed by Europe–Siberian (5.17%).

3.2. Grazing exclusion effects on vegetation cover, species diversity and richness

The values of plant density of class I and II were 17.69 and 11.24 plant m^{-2} respectively, in the 6 years' fenced site, and were significantly greater than those in the overgrazed site (4.11 and 4.09 plant m^{-2}) at the peak growing season harvest ($P < 0.01$; Table 3). Moreover, the changes in the plant density were not significant between 2 year' fenced rangeland and the overgrazed site (Table 3). The plant density of class III decreased under grazing exclusion, from 6.33 in 2 years' fenced site to 3.27 plant m^{-2} in 6 years' fenced site ($P < 0.05$; Table 3). The litter and canopy cover were also notably higher under 6 years' fenced rangeland than those under overgrazed and 2 years' fenced site. Accordingly, the bare soil decreased from 59.89% in overgrazed site to 6.13% in 6 years' fenced site ($P < 0.01$, Table 3) during the study period.

The diversity of plant species and richness were significantly affected by grazing exclusion ($P < 0.01$, Table 3). Both shannon diversity index (H′) and richness of the rangeland communities increased gradually with the increase of exclusion age, and were maximum for the 6 years' exclusion, while overgrazing site had the minimum diversity and richness of plant species. Results indicated that there were no significant differences in the plant species evenness of the sites (Table 3). Totally, the results of the present study showed that livestock grazing exclusion in arid rangelands significantly altered the vegetation properties, which supports our first hypothesis.

3.3. Grazing exclusion effects on soil physical and chemical properties

The variations of the physical and chemical characteristics of the soils are shown in Table 4. An ANOVA proved that soils under 6 year's fenced site had significantly lower pH values in the 0–30 cm soil layer ($P < 0.05$), when compared with those under overgrazed site, but it was not significantly different between the overgrazed site and 2 years' fenced sites. The EC was not significantly changed after grazing exclusion (Table 4). Short-term grazing exclusion resulted in notable enrichment of organic carbon, nitrogen, potassium, and phosphorus reserves in the 0–30 cm soil layer (Table 4).

In comparison with the overgrazed site, organic carbon levels were 1.23, 3.09 and 3.78 times higher in the 2, 4 and 6 short-term exclusions, respectively. The levels of nitrogen, potassium, and phosphorus followed the same pattern. The chemical content was found to be greater in grazing exclusion site compared with the overgrazed site. Short-term grazing exclusion did not result in increases in the calcium carbonate content of the investigated soil layer (Table 4).

The particle size distribution showed more silt and clay and less sand in the soils of grazing exclusion sites compared with the soils of overgrazed site (Table 4). The silt and clay values were the highest in the soils under 6 year's fenced site. Sand content in the

Table 4
Characteristics of soil sampled at excluded and grazed sites.

Soil properties	Overgrazed site	2 years' exclusion	4 years' exclusion	6 years' exclusion
pH	8.99 ± 0.20^{a}	8.74 ± 0.20^{a}	8.30 ± 0.20^{b}	8.00 ± 0.20^{C}
ECe ($dS\,m^{-1}$)	4.00 ± 0.01^{a}	4.13 ± 0.01^{a}	4.10 ± 0.01^{a}	4.20 ± 0.01^{a}
Organic carbon (%)	0.56 ± 0.01^{c}	1.79 ± 0.01^{b}	3.65 ± 0.01^{a}	4.34 ± 0.01^{a}
Nitrogen (%)	0.03 ± 0.01^{c}	0.06 ± 0.01^{c}	1.21 ± 0.0^{b}	1.44 ± 0.01^{a}
Potassium (ppm)	115.60 ± 6.20^{c}	212.27 ± 11.00^{b}	361.52 ± 11.09^{a}	421.12 ± 11.21^{a}
Phosphorus (ppm)	3.12 ± 0.10^{d}	7.21 ± 0.20^{c}	15.09 ± 0.30^{b}	18.03 ± 0.30^{a}
$CaCO_3$ (%)	33.86 ± 3.50^{a}	33.00 ± 3.43^{a}	33.21 ± 3.37^{a}	34.50 ± 3.30^{a}
Silt (%)	50.20 ± 3.11^{c}	53.50 ± 3.16^{bc}	58.00 ± 4.00^{ab}	60.50 ± 2.00^{a}
Sand (%)	42.20 ± 2.00^{a}	35.50 ± 1.00^{b}	25.00 ± 1.00^{c}	17.50 ± 1.00^{d}
Clay (%)	7.60 ± 0.70^{d}	11.00 ± 0.70^{c}	17.00 ± 0.70^{b}	22.00 ± 0.70^{a}
Soil texture	Silt loam	Silt loam	Silt loam	Silt loam

Values shown are the means ± SE. Values within a row followed by different letters are significantly different ($p < 0.05$, post hoc Duncan test).

overgrazed site soil has increased by 44.20%, whereas it slightly decreased following livestock exclusion.

4. Discussion

4.1. Influence of grazing exclusions on vegetation properties

In degraded rangelands, livestock exclusion has been suggested as a simple and effective method for restoring and conserving vegetation productivity and diversity (Rutherford and Powrie, 2010; Al-Rowaily et al., 2015). The results of the present study showed that short-term grazing exclusion has influenced different measured diversity attributes. The diversity of the plants showed differences between grazed and fenced sites at the levels of species, genera and families. Short-term grazing exclusion had a positive effect on the total number of species and species richness. The results showed that the grazing exclusion enhances the cover, density and richness of perennial plant which facilitate the establishment and growth of herbaceous and annual species under their canopies (Al-Rowaily et al., 2015). Livestock exclusion practices on desertified rangeland were shown to be good alternatives to recover vegetation and attenuate soil loss by wind erosion in these erodible rangelands (Su et al., 2004; Yong-Zhong et al., 2005).

Changes in the plant community in ecological processes through succession likely provide a high diversity of niches, attracting seed dispersers and, as a consequence, facilitating colonization by new plant species (Jules et al., 2008; Suganuma et al., 2014), thereby facilitating the establishment of desirable species in later successional stages (Liu et al., 2015). Plant establishment provides suitable micro-habitats for the growth of plant species in arid lands (Amici et al., 2012). The habitat-modifying capacity of a plant can alter its environment, both above and below-ground. Understory microclimate is characterized by lower irradiance and air temperature, and consequently lower evapotranspiration demands, than the areas without vegetation (Maestre et al., 2003). In addition, plant establishment can help vegetation communities in the arid lands by trapping seeds (Rathore et al., 2015). In this way, perennials act as seed accumulators by shielding wind-dispersed seeds on the plant species, thereby enhancing possibilities for recruitment. Seedling establishment is often possible under the shade of existing perennials, allowing for the colonization and long-term persistence of herbaceous species (Shumway, 2000). Finally, reduced soil erosion and improved soil properties associated with shrub development create a nutrient-rich, water-retaining substrate, thus providing a better environment for germination, seedling growth, and productivity in water and nutrient poor environments (Su et al., 2002).

The fenced sites had the lowest proportions of the bare soil, probably because the dominant plants (*Ar. santolina*, *S. tomentosa*, *Z. eurypterum and H. salicornia*) grow in large clumps and hinder erosive processes (Ninot et al., 2007). The reason might explain the greater canopy cover observed in the areas. Many reports have confirmed that the positive effects of livestock exclusion used in the arid lands include an increase in the number of plant species and biodiversity (Landsberg et al., 2002; Brooks et al., 2006; Al-Rowaily et al., 2015). For example, Wang et al. (2014), in investigation of the effects of livestock exclusion on C and N pools of typical grassland on the Loess Plateau (China) concluded that livestock exclusion significantly increased plant biomass and diversity. Jeddi and Chaieb (2010), in the study of the effect of grazing exclusion in a degraded arid environment of South Tunisia, reported that grazing exclusion greatly altered vegetation cover and provided suitable micro-habitats for the growth of herbaceous plants in degraded arid lands.

Species composition and diversity are fundamental characteristics of ecosystems and vegetation diversity should be considered in the course of vegetation restoration (Rodrigues et al., 2009). Results of the present study showed that grazing exclusion helped in vegetation enrichment. The short-term exclusion increased the diversity and abundance of plant species. In the present study, the population of the perennial species generally increased and up to the maximum value at the area where short-term exclusion was performed for 6 years. Livestock exclusion can be of benefit to landscape composition by increasing ecotone density and landscape diversity (Abdallah et al., 2008; Al-Rowaily et al., 2015). Therefore, shrubs and trees establishment during livestock exclusion may cause an accumulation of mineral nutrients and water, leading to a local increase in soil fertility (El-Keblawy and Ksiksi, 2005), and may protect the understory species against high irradiance and temperature (Vetaas, 1992). Favorable soil and micro-climatic conditions underneath plant canopies act as "resource islands" for understory herbaceous plants (Stinca et al., 2015). These "resource islands" are critical to arid land rehabilitation, because they may spur natural succession by facilitating the growth of other plants (Gómez-Aparicio et al., 2004; Rathore et al., 2015).

The lower species diversity and richness in overgrazed site indicated that the plant species such as *Ar. santolina*, *S. tomentosa*, *Z. eurypterum and H. salicornia* are as sensitive and intolerant grazing, and consequently they are dominant under protected condition. Among the human activities that degrade rangelands, overgrazing by livestock is perhaps one of the most significant (Mainguet, 1994). The effects of overgrazing on the plant community and soils are considered destructive because of the reduction of canopy cover, the destruction of topsoil structure, and compaction of soil as a result of trampling (Manzano and Návar, 2000). In turn, these processes increase soil crusting, reduce soil infiltration, and enhance soil erosion susceptibility (Manzano and Návar, 2000; Yong-Zhong et al., 2005). Al-Rowaily et al. (2012) showed a deterioration of perennial grasses under heavy grazing in the rangelands of central Saudi Arabia, and such deterioration accelerates erosion and desertification (Barth, 1999). Li et al. (2005) stated that overgrazing in the

Inner Mongolian desert steppe has led to a significant reduction in palatable species. This could be attributed to the selective grazing of highly palatable species that are not tolerant to heavy grazing and trampling (Ksiksi et al., 2007; Al-Rowaily et al., 2015).

4.2. Influence of grazing exclusions on soil properties

Results revealed that the 4 and 6 years' grazing exclusions had significant lower pH value comparison with 2 years' exclusion and overgrazed sites. The difference was probably related to the plant coverage, root systems, and soil organic carbon content, because extensive secretion of organic acids from the roots and amounts of CO_2 released from roots and microorganisms could lead to the decrease in pH (Zhao et al., 1999; Juo and Manu, 1996; Yong-Zhong et al., 2005). Also, the addition of livestock urine increases soil pH largely due to the hydrolysis of urine urea in grazed rangeland (Raiesi and Riahi, 2014).

This phenomenon may be related to several mechanisms that release H^+ ions, such as cation uptake by biomass, decomposition of organic matter to organic acids and CO_2, root respiration, and nitrification. These processes are counter balanced to some extent by several sinks for H^+, the weathering of soil minerals, anion uptake by biomass, and release of cations from soil organic matter (Binkley and Richter, 1987). The increased accumulation of aboveground biomass and associated cation uptake by the plant component is possibly one of the causes for decreased pH in these soils (Tornquist et al., 1999).

The result of the present study showed that short-term grazing exclusion increased soil amelioration, while overgrazing resulted in less vegetation cover, soil coarseness, and very low soil enrichment. Previous research has shown a link between livestock and soil degradation, looking at animal grazing as one of the most important causes of desertification (Yang et al., 2010) due to inappropriate management practice such as a prolonged overgrazing, high time of permanence of animals in the same place and over-lumbering (Zhu and Wang, 1992). The quality of soil could be compromised from an inadequate assessment of ecosystem sustainability due to the role of animal grazing (Pierce and Larson, 1993).

Due to continuous grazing and frequent trampling by the goat, the ground surface at the overgrazed site became bare and exposed to wind erosion under strong winds. Accelerated wind erosion due to the decreased vegetation cover and litter accumulation resulted in soil coarsening and loss of soil organic matter (Yong-Zhong et al., 2005). Soil organic carbon and total nitrogen concentrations following 6 years of livestock exclusion showed a significant increase, compared to the overgrazed site. Bauer et al. (1987) noted that grazed grasslands contained less organic carbon than adjacent fenced grasslands. A similar increase in soil organic carbon concentrations following fencing was reported in Inner Mongolia in northern China by Gao et al. (2011). The amount of soil organic carbon associated with silt and clay due to their higher capacity for holding water and nutrients compared to sand (Plante et al., 2006; Lu et al., 2015). Thus, soil particle size distributions play an important role in regulating the capacity of a soil to preserve organic matter; for instance, soil organic carbon content significantly increased due to grazing exclusion with both higher clay and silt contents and lower sand content in a desert steppe in northwestern China (Wen et al., 2013; Lu et al., 2015).

Muñoz et al. (2013) reported a negative influence of domestic camelid grazing on the decreasing of soil organic carbon stocks. The increase in organic carbon and total nitrogen concentrations resulted mostly from the increase in organic matter returned to the soil and reduced wind erosion due to vegetation recovery and litter accumulation. Also, changes in species composition could also have affected organic matter and nutrient contents (Yong-Zhong et al., 2005). The greater species diversity and abundance of the perennial plants in the fenced area could be attributed partly to the higher fertility of the soil. Increases in the soil nutrient levels reflect accumulation of litter and roots, animal fragmentation of particulate organic matter and soil aggregates, and high root activity (Chen, 2003; Rathore et al., 2015). This mechanism occurs in response to two factors. (I): improved vegetation cover is likely to reduce soil erosion while trapping wind-blown, nutrient-enriched, fine materials from surrounding open areas (Wezel et al., 2000; Rathore et al., 2015). (II): nutrient enhancement is largely attributable to the plant litter and root mass additions to the soil (Zhang et al., 2006).

The soil nutrient values (N, P, K) increased gradually during the exclusion and were maximum for the area with 6 years' exclusion, which supports our second hypothesis that we have followed in the Introduction. The higher soil nutrition in the 6 years' exclusion can be partially explained by the fact that the canopy cover of a plant is positively correlated with the amount of litter accumulation under its canopy (Reynolds et al., 1999; Li et al., 2008). The large canopies of the mature plants, thus may have trapped more nutrients than the small canopies of the younger individuals. It has also been reported that perennials with a long life span may form fertile soils and those with a short one may not, because nutrients accumulated by short-lived shrubs, especially those with small canopies, will fade away quickly (Li et al., 2008). In addition, lower radiation under plant species reduces soil temperature and evaporative water losses, whereas higher organic matter content improves soil water retention, thereby causing soil moisture and rates of litter decomposition to be higher under plant canopies than in the areas without vegetation (Aguilera et al., 1999). Thus, more litter accumulation for a longer time may have contributed to the higher fertility of the soils.

The values of silt and clay were found to be greater in grazing exclusions compared with the overgrazed site. This result was consistent with the results from the Imam Kandi Rangelands, Iran (Mofidi et al., 2013), and in the sandy rangeland of Inner Mongolia, northern China (Li et al., 2011), in which grazing exclusion led to greater fine soil particle content and lower coarse sand content due to an increased ability of vegetation to prevent soil erosion and trap windblown fine particles (Chen et al., 2012; Wen et al., 2013). Higher concentrations of branches and denser canopies may have trapped more transported soil, which may have caused more significant changes in physical properties (higher silt content). Singh et al. (2005) reported that denser canopies show the strongest positive correlation with soil deposition levels beneath shrub canopies. The distance over which the soil surface materials are transported by the wind decreases. Moreover, large quantities of wind-blown, fine-soil materials are collected in the vicinity of shrubs by stemflow and through-fall processes of dust entrapment and deposition (Wezel et al., 2000; Rathore et al., 2015).

5. Conclusion

Fencing with grazing exclusion is an effective rangeland restoration and management practice used to achieve sustainability of natural ecosystems. Our findings suggest that excluding grazing from rangeland is likely to reap benefits for habitat characteristics and vegetation enrichment in the short-term on the steppe rangeland of Taftan. The numbers of species increased slowly during exclusion. Species richness was increased as well as plant diversity in the fenced sites. While, overgrazing gives rise to a considerable decrease in species richness and diversity and ground cover, which accelerates soil erosion by wind, resulting in a further coarseness in surface soil, loss of soil nutrient. The results indicate that the short-term grazing exclusion alter physical soil traits and enhance soil nutrients. In conclusion, grazing exclusion can be an effective tool to enhance diversity and improve soil quality. However, more study are needed to assess the effect of livestock exclusion on ecosystem process in the rangelands of arid ecosystems.

Acknowledgments

The work was conducted as part of scientific research. The assistance of University of Zabol, Department of Range and Watershed Management and the laboratory of the Agriculture and Natural Resources Research Center, Zahedan, Iran for analysis of the soil samples is gratefully acknowledged.

References

Abdallah, F., Noumi, Z., Touzard, B., Ouled Belgacem, A., Neffati, M., Chaieb, M., 2008. The influence of *Acacia tortilis* (Forssk.) subsp. raddiana (Savi) and livestock grazing on grass species composition, yield and soil nutrients in arid environments of South Tunisia. Flora 203, 116–125.

Aguilera, L.E., Gutiérrez, J.R., Meserve, P.L., 1999. Variation in soil micro-organisms and nutrients underneath and outside the canopy of *Adesmia bedwellii* (Papilionaceae) shrubs in arid coastal Chile following drought and above average rainfall. J. Arid Environ. 42, 61–70.

Al-Rowaily, S.L., El-Bana, M.I., Al-Dujain, F.A., 2012. Changes in vegetation composition and diversity in relation to morphometry, soil and grazing on a hyper-arid watershed in the central Saudi Arabia. Catena 97, 41–49.

Al-Rowaily, S.L., El-Bana, M.I., Al-Bakre, D.A., Assaeed, A.M., Hegazy, A.K., Ali, M.B., 2015. Effects of open grazing and livestock exclusion on floristic composition and diversity in natural ecosystem of Western Saudi Arabia. Saudi J. Biol. Sci. 22, 430–437.

Allison, L.E., Moodie, C.D., 1965. Carbonates. In: Black, C.A., Evans, D.D., White, J.L., Ensminger, L.E., Clark, F.E., Dinauer, R.C. (Eds.), Methods of Soil Analysis. American Society of Agronomy and Soil Science Society of America, Madison, Wisconsin, pp. 1379–1396.

Amici, V., Rocchini, D., Geri, F., Bacaro, G., Marcantonio, M., Chiarucci, A., 2012. Effects of an afforestation process on plant species richness: a retrogressive analysis. Ecol. Complex. 9, 55–62.

Amiri, F., 2009. A GIS model for classification of rangeland suitability for sheep grazing in arid and semi-arid regions of Iran. Livest. Res. Rural Dev. 21, 68–82.

Arzani, H., Zohdi, M., Fish, E., Zahedi, A., Nikkhah, G.H.A., Wester, D., 2004. Phenolojical effects on forage quality of five grass species. J. Range Manage. 57, 624–630 (in Persian).

Bagheri, H., Adnani, M., Tavili, A., 2007. Studying the relationship between livestock and plant composition case study: semi steppic ranges of Vesf-Qom province. Pajouhesh Sazandegi 74, 155–162 (in Persian).

Baghestani, N., Arzani, H., Zare, T., Abdollahi, J., 2001. Study of forage quality the important species of stepic rangeland of Poshtkhoeh Yazd. Iran. J. Range Desert Res. 11, 137–162 (in Persian).

Barth, H.J., 1999. Desertification in the Eastern Province of Saudi Arabia. J. Arid Environ. 43, 399–410.

Bauer, A., Cole, C.V., Black, A.L., 1987. Soil property comparison in virgin grasslands between grazed and non-grazed management systems. Soil Sci. Soc. Am. J. 51, 176–182.

Binkley, D., Richter, D., 1987. Nutrient cycles and H budgets of forest ecosystems. Adv. Ecol. Res. 16, 1–51.

Bray, R.H., Kurtz, L.T., 1954. Determination of total, organic and available forms of phosphorus in soils. Eur. J. Soil Sci., 39–45.

Bremner, J.M., 1996. Nitrogen total. In: Bartels, J.M. (Ed.), Methods of Soil Analysis. Soil Science Society of America, Madison, Wisconsin, pp. 1085–1122.

Brooks, M.L., Matchett, J.R., Berry, K., 2006. Effects of livestock watering sites on plant communities in the Mojave Desert, USA. J. Arid Environ. 67, 125–147.

Chen, Y., Li, Y., Zhao, X., Awada, T., Shang, W., Han, J., 2012. Effects of grazing exclusion on soil properties and on ecosystem carbon and nitrogen storage in a sandy rangeland of Inner Mongolia, northern China. Environ. Manage. 50, 622–632.

Chen, H., 2003. Phosphatase activity and P fractions in soils of an 18-year-old Chinese fir (*Cunninghamia lanceolata*) plantation. For. Ecol. Manage. 178, 301–310.

Costa, C., Papatheodorou, E.M., Monokrousos, N., Stamou, G.P., 2015. Spatial variability of soil organic C, inorganic N and extractable P in a Mediterranean grazed area. Land Degrad. Dev. 26, 103–109.

Coulloudon, B., Eshelman, K., Gianola, J., Habich, N., Hughes, L., Johnson, C., Pellant, M., Podborny, P., Rasmussen, A., Robles, B., Shaver, P., Spehar, J., Willoughby, J., 1999. Sampling Vegetation Attributes Technical Reference. Grazing Land Technology Institute, Denver, Colorado.

Ebrahimi, M., Arab, M., Ajorloo, M., 2014. Effects of enclosure on ecological indexes of rangeland health using Landscape Function Analysis Method (Case Study: Jiroft Jbalbarez Rangeland). Rangeland 8, 261–271 (in Persian).

El-Keblawy, A., Ksiksi, T., 2005. Artificial forests as conservation sites for the native flora of the UAE. For. Ecol. Manage. 213, 288–296.

Gómez-Aparicio, L., Zamora, R., Gómez, J.M., Hódar, J.A., Castro, J., Baraz, E., 2004. Applying plant facilitation to forest restoration: a meta-analysis of the use of shrubs as nurse plants. Ecol. Appl. 14, 1128–1138.

Gao, Y., Zeng, X., Schumann, M., Chen, H., 2011. Effectiveness of exclosures on restoration of degraded alpine meadow in the eastern Tibetan Plateau. Arid Land Res. Manage. 25, 164–175.

Gonzales, E.K., Clements, D.R., 2010. Plant community biomass shifts in response to mowing and fencing in invaded oak meadows with non-native grasses and abundant ungulates. Restor. Ecol. 18, 753–761.

Hanley, T.A., 1978. A comparison of the line-interception and quadrat estimation methods of determining shrub canopy coverage. J. Range Manage. 31, 60–62.

Heath, M.E., Barnes, R.F., Metcalfe, D.S., 1985. Forages: The Science of Grassland Agriculture. Iowa State University, Iowa, Aimes.

Jeddi, K., Chaieb, M., 2010. Changes in soil properties and vegetation following livestock grazing exclusion in degraded arid environment of south Tunisia. Flora 205, 184–189.

Jiang, G.M., Han, X.G., Wu, J.G., 2006. Restoration and management of the Inner Mongolia grassland require a sustainable strategy. Ambio 35, 269–270.

Jules, M.J., Sawyer, J.O., Jules, E.S., 2008. Assessing the relationship between stand development and understory vegetation using a 420 year chronosequence. For. Ecol. Manage. 255, 2384–2393.

Juo, A.S.R., Manu, A., 1996. Nutrient effects on modification of shifting cultivation in West Africa. Agric. Ecosyst. Environ. 58, 49–60.

Keesstra, S.D., Maroulis, J., Argaman, E., Voogt, A., Wittenberg, L., 2014. Effects of controlled fire on hydrology and erosion under simulated rainfall. Cuad. Invest. Geogr. 40, 269–293.

Knudsen, D., Peterson, G.A., Pratt, P., 1982. Lithium, sodium and potassium. In: Page, A.L. (Ed.), Methods of Soil Analysis. American Society of Agronomy, Madison, Wisconsin, pp. 225–246.

Ksiksi, T., El-Keblawy, A., Al-Ansari, F., Elhadramy, G., 2007. Desert ecosystems vegetation and potential uses as feed sources for camels and wildlife. In: Proceeding of the 8th Annual Conference for Research Funded by UAE University, Al-Ain, United Arab Emirates, pp. 71–81.

Landsberg, J., James, C.D., Maconochie, J., Nicholls, A.O., Stol, J., Tynan, R., 2002. Scale-related effects of grazing on native plant communities in an arid rangeland region of South Australia. J. Appl. Ecol. 39, 427–444.

Li, J.H., Li, Z.Q., Ren, J.Z., 2005. Effect of grazing intensity on clonal morphological plasticity and biomass allocation patterns of Artemisia frigida and Potentilla acaulis in the Inner Mongolia steppe. N. Z. J. Agric. Res. 48, 57–61.

Li, X.R., He, M.Z., Jia, R.L., 2008. The response of desert plant species diversity to the changes in soil water content in the middle-lower reaches of the Heihe River (in Chinese). Adv. Earth Sci. 23, 685–696.

Li, Y., Zhao, H., Zhao, X., Zhang, T., Li, Y., Cui, J., 2011. Effects of grazing and livestock exclusion on soil physical and chemical properties in desertified sandy grassland Inner Mongolia, northern China. Environ. Earth Sci. 63, 771–783.

Liu, B., Zhao, W., Liu, Z., Yang, Y., Luo, W., Zhou, H., Zhang, Y., 2015. Changes in species diversity, aboveground biomass, and vegetation cover along an afforestation successional gradient in a semiarid desert steppe of China. Ecol. Eng. 81, 301–311.

Lo, I., Tsang, D., Yip, T., Wang, F., Zhang, W.H., 2011. Influence of injection conditions on EDDS-flushing of metal-contaminated soil. J. Hazard. Mater. 192, 667–675.

Lu, X., Yan, Y., Sun, J., Zhang, X., Chen, Y., Wang, X., Cheng, G., 2015. Short-term grazing exclusion has no impact on soil properties and nutrients of degraded alpine grassland in Tibet, China. Solid Earth 6, 1195–1205.

Maestre, F.T., Bautista, S., Cortina, J., 2003. Positive, negative, and net effects in grass-shrub interactions in Mediterranean semiarid grasslands. Ecology 84, 3186–3197.

Magurran, A.E., 1988. Ecological Diversity and its Measurement. Princeton University Press, Princeton, NJ.

Mainguet, M., 1994. Desertification: Natural Background and Human Mismanagement. Speringer-Verlag, Berlin, Germany.

Manzano, M.G., Návar, J., 2000. Processes of desertification by goats overgrazing in the *Tamaulipan thrnscrub* (matorral) in north-eastern Mexico. J. Arid Environ. 44, 1–17.

Mekuria, W., Aynekulu, E., 2013. Exclosure land management for restoration of the soils in degraded communal grazing lands in northern Ethiopia. Land Degrad. Dev. 24, 528–538.

Metzger, K.L., Coughenour, M.B., Reich, R.M., Boone, R.B., 2005. Effects of seasonal grazing on plant species diversity and vegetation structure in a semi-arid ecosystem. J. Arid Environ. 61, 147–160.

Mofidi, M., Jafari, M., Tavili, A., Rashtbari, M., Alijanpour, A., 2013. Grazing exclusion effect on soil and vegetation properties in Imam Kandi Rangelands, Iran. Arid Land Res. Manage. 27, 32–40.

Muñoz, M.A., Faz, A., Zornoza, R., 2013. Carbon stocks and dynamics in grazing highlands from the Andean Plateau. Catena 104, 136–143.

Ninot, J., Carrillo, E., Font, X., Carreras, J., Ferré, A., Masalles, R., Soriano, I., Vigo, J., 2007. Altitude zonation in the Pyrenees. A geobotanic interpretation. Phytocoenologia 37, 371–398.

Pierce, F.J., Larson, W.E., 1993. Developing criteria to evaluate sustainable land managements. In: Kimble, J.M. (Ed.), Proceedings of the VIII International Soil Management Workshop: Utilization of Soil Survey Information for Sustainable Land Use Sacramento. CA, pp. 7–14.

Plante, A.F., Conant, R.T., Stewart, C.E., Paustian, K., Six, J., 2006. Impact of soil texture on the distribution of soil organic matter in physical and chemical fractions. Soil Sci. Soc. Am. J. 70, 287–296.

Prieur-Richard, A.H., Lavorel, S., 2000. Invasions: the perspective of diverse plant communities. Austral Ecol. 25, 1–7.

Raiesi, F., Riahi, M., 2014. The influence of grazing exclosure on soil C stocks and dynamics and ecological indicators in upland arid and semi-arid rangelands. Ecol. Indic. 41, 145–154.

Rathore, V.S., Singh, J.P., Bhardwaj, S., Nathawat, N.S., Mahesh Kumar, M., Roy, M., 2015. Potential of native shrubs *Haloxylon salicornicum* and *Calligonum Polygonoides* for restoration of degraded lands in arid western Rajasthan, India. Environ. Manage. 55, 205–216.

Rechinger, K.H., Schiman-Czeika, H., 1964. *Flora Iranica*. Euphorbiaceae. Akademische Druck U Verlagsanstalt, Graz, Austria.

Rechinger, K.H., 1968. *Flora Iranica*. Cruciferae. Akademische Druck U Verlagsanstalt, Graz, Austria.

Rechinger, K.H., 1970. *Flora Iranica*. Gramineae. Akademische Druck U Verlagsanstalt, Graz, Austria.

Rechinger, K.H., 1972. *Flora Iranica*. *Zygophyllaceae*. Akademische Druck U Verlagsanstalt, Graz, Austria.

Rechinger, K.H., 1984. *Flora Iranica*. Papilionaceae. Akademische Druck U Verlagsanstalt, Graz, Austria.

Rechinger, K.H., 1997. *Flora Iranica*. *Chenopodiaceae*. Akademische Druck U Verlagsanstalt, Graz, Austria.

Reynolds, J.F., Virginia, R.A., Kemp, P.R., de Soyza, A.G., Tremmel, D.C., 1999. Impact of drought on desert shrubs: effects of seasonality and degree of resource island development. Ecol. Monogr. 69, 69–106.

Rhoades, J.D., 1996. Salinity: electrical conductivity and total dissolved solids. In: Page, A.L. (Ed.), Methods of Soil Analysis. American Society of Agronomy, Madison, Wisconsin, pp. 417–435.

Roa-Fuentes, L.L., Martínez-Garza, C., Etchevers, J., Campo, J., 2015. Recovery of soil C and N in a tropical pasture: passive and active restoration. Land Degrad. Dev. 26, 201–210.

Rodrigues, R.R., Lima, R.A.F., Gandolfi, S., Nave, A.G., 2009. On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. Biol. Conserv. 142, 1242–1251.

Rutherford, M.C., Powrie, L.W., 2010. Severely degraded rangeland: implications for plant diversity from a case study in Succulent Karoo South Africa. J. Arid Environ. 74, 692–701.

Schultz, N.L., Morgan, J.W., Lunt, I.D., 2011. Effects of grazing exclusion on plant species richness and phytomass accumulation vary across a regional productivity gradient. J. Veg. Sci. 22, 130–142.

Shumway, S.W., 2000. Facilitative effects of a sand dune shrub on species growing beneath the shrub canopy. Oecologia 124, 138–148.

Singh, J.P., Soni, M.L., Rathore, V.S., 2005. Role of shrubs in rehabilitation of degraded lands in hot arid region. In: Prasad, S.N., Singh, R.K., Kumar, A., Parandiyal, A.K., Ali, S., Somasundaram, J., Sethy, B.K., Sharda, V.N. (Eds.), Natural Resource Management for Sustainable Development in Western India. Allied Publishers, India, pp. 169–170.

Stanners, D., Bordeau, P., 1995. Europe's Environment. The Dobris Assessment. European Environmental Agency, Copenhagen.

Stinca, A., Battista Chirico, G., Incerti, G., Bonanomi, G., 2015. Regime shift by an exotic nitrogen-fixing shrub mediates plant facilitation in primary succession. PLoS One 10, e0123128.

Strahan, R.T., Laughlin, D.C., Bakker, J.D., Moore, M.M., 2015. Long-term protection from heavy livestock grazing affects ponderosa pine understory composition and functional traits. Rangel. Ecol. Manage. 68, 257–265.

Su, Y.Z., Zhao, H.L., Zhang, T.H., 2002. Influencing mechanism of several shrubs and subshrubs on soil fertility in Keerqin sandy land. Chin. J. Appl. Ecol. 13, 802–.

Su, Y.Z., Zhao, H.L., Zhang, T.H., Zhao, X.Y., 2004. Soil properties following cultivation and non-grazing of a semiarid sandy grassland in northern China. Soil Tillage Res. 75, 27–36.

Suganuma, M.S., de Assis, G.B., Durigan, G., 2014. Changes in plant species composition and functional traits along the successional trajectory of a restored patch of Atlantic Forest. Community Ecol. 15, 27–36.

Tarhouni, M., Ben Hmida, W., Neffati, M., 2015. Long-term changes in plant life forms as a consequence of grazing exclusion under arid climatic conditions. Land Degrad. Dev., http://dx.doi.org/10.1002/ldr.2407.

Thomas, G.W., 1996. Soil pH and soil acidity. In: Sparks, D.L. (Ed.), Methods of Soil Analysis. American Society of Agronomy and Soil Science Society of America, Madison, Wisconsin, pp. 475–490.

Tornquist, C.G., Hons, F.M., Feagley, S.E., Haggar, J., 1999. Agroforestry system effects on soil characteristics of the Sarapiqui region of Costa Rica. Agric. Ecosyst. Environ. 73, 19–28.

Vetaas, O.R., 1992. Micro-site effects of trees and shrubs in dry savannas. J. Veg. Sci. 3, 337–344.

Wang, A., Luo, C., Yang, R., Chen, Y., Shen, Z., Li, X., 2012. Metal leaching along soil profiles after the EDDS application—a field study. Environ. Pollut. 164, 204–210.

Wang, D., Wu, G.L., Zhu, Y.J., Shi, Z.H., 2014. Grazing exclusion effects on above- and below-ground C and N pools of typical grassland on the Loess Plateau (China). Catena 123, 113–120.

Wen, H., Niu, D., Fu, H., Kang, J., 2013. Experimental investigation on soil carbon, nitrogen, and their components under grazing and livestock exclusion in steppe and desert steppe grasslands, Northwestern China. Environ. Earth Sci. 70, 3131–3141.

Wezel, A., Rajot, J.L., Herbrig, C., 2000. Influence of shrubs on soil characteristics and their function in Sahelian agro-ecosystems in semi-arid Niger. J. Arid Environ. 44, 383–398.

Yang, M., Nelson, F.E., Shiklomanov, N.I., Guo, D., Wan, G., 2010. Permafrost degradation and its environmental effects on the Tibetan Plateau: a review of recent research. Earth Sci. Rev. 103, 31–44.

Yong-Zhong, S., Yu-Lin, L., Jian-Yuan, C., Wen-Zhi, Z., 2005. Influences of continuous grazing and livestock exclusion on soil properties in a degraded sandy grassland Inner Mongolia, northern China. Catena 59, 267–278.

Zhang, T.H., Su, Y.Z., Cui, J.Y., Zhang, Z.H., Chang, X.X., 2006. A leguminous shrub (*Caragana microphylla*) in semi-arid sandy plain of north China. Pedosphere 16, 319–325.

Zhao, H.L., Zhang, T.H., Chang, X.L., 1999. Cluster analysis on change laws of the vegetation under different grazing intensities in Horqin sandy pasture. J. Desert Res. 19, 40–44 (in Chinese).

Zhao, G., Mu, X., Wen, Z., Wang, F., Gao, P., 2013. Soil erosion, conservation, and eco-environment changes in the loess plateau of China. Land Degrad. Dev. 24, 499–510.

Zhu, Z.D., Wang, T., 1992. Theory and practice on sandy desertification in China. J. Quat. Sci. 2, 97–106.

Zielinski, J., 1982. *Flora Iranica*. Rosaceae. Akademische Druck U Verlagsanstalt, Graz, Austria.

Zohary, M., 1963. On the geobotanical structure of Iran. Bull. Res. Counc. Isr. Sect. D Bot. Suppl.