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Nutrient losses in runoff from grassland and shrubland habitats in southern New Mexico: II. Field plots

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Abstract. Losses of dissolved nutrients (N, P, K, Ca, Mg, Na, Cl, and SO_4) in runoff were measured on grassland and shrubland plots in the Chihuahuan desert of southern New Mexico. Runoff began at a lower threshold of rainfall in shrublands than in grasslands, and the runoff coefficient averaged 18.6% in shrubland plots over a 7-year period. In contrast, grassland plots lost 5.0 to 6.3% of incident precipitation in runoff during a 5.5-year period. Nutrient losses from shrubland plots were greater than from grassland plots, with nitrogen losses averaging 0.33 kg ha^{-1} yr^{-1} vs 0.15 kg ha^{-1} yr^{-1}, respectively, during a 3-year period. The greater nutrient losses in shrublands were due to higher runoff, rather than higher nutrient concentrations in runoff. In spite of these nutrient losses in runoff, all plots showed net accumulations of most elements due to inputs from atmospheric deposition. Therefore, loss of soil nutrients by hillslope runoff cannot, by itself, account for the depletion of soil fertility associated with desertification in the Chihuahuan desert.

Introduction

Since its initial presentation more than 30 years ago (Bormann & Likens 1967), the watershed ecosystem concept has been used to study the gains and losses of plant nutrients on well-defined units of landscape in many areas of the world. These mass-balance studies have given special insight to the nutrient losses that accompany forest harvest (Hornbeck et al. 1986; Johnson et al. 1988), forest fires (Wright 1976; Chorover et al. 1994), exposure of soils to acid rain (Wright et al. 1994; Likens et al. 1996) and excess deposition of fixed nitrogen from the atmosphere (Peterjohn et al. 1996; Swank & Vose 1997). Mass-balance studies also show nutrient accumulations in regenerating forests (Vitousek 1977; Likens et al. 1978). Few such studies have been conducted in arid and semiarid lands, but losses and spatial redistributions of soil nutrients are thought to accompany the desertification of semiarid grasslands that are invaded by desert shrubs (Schlesinger et al. 1990, 1996; Kieft et al. 1998).

In part I of this series, we reported on the transport of soil N and P in runoff during short-term rainfall simulation experiments in areas of black grama grassland (*Bouteloua eriopoda*) and creosotebush shrubland (*Larrea tridentata*) in the Chihuahuan desert of New Mexico (Schlesinger et al. 1999). Estimated losses of N were 0.25 kg ha^{-1} yr^{-1} in the grassland and 0.43 kg ha^{-1} yr^{-1} in the shrubland, consistent with the depletion of soil nutrients that appears to accompany shrub invasion in this region. The highest nitrogen concentrations in runoff were found in grassland; the greater overall N losses in the shrubland were derived from a greater volume of runoff. In both habitats, losses of dissolved forms of phosphorus were negligible.

In addition to the rainfall simulation experiments, we have measured the yield of dissolved solutes in the runoff generated by natural precipitation events falling on small plots located in grassland and shrubland habitats in the Chihuahuan desert. Unlike the rainfall simulation experiments, which were conducted at a constant rainfall intensity, the discharge from these field plots is derived from natural storms of varying duration and intensity. In this paper, our objectives are: (1) to compare the annual discharge of water and nutrients in grassland and shrubland habitats over a 3-year period, (2) to compare these values to those derived from the rainfall simulation experiments, and (3) to assess the nutrient losses in runoff for their role in the desertification of semiarid grasslands in this region.

Study area and methods

This study was conducted on lands managed by the Chihuahuan Desert Rangeland Research Center and the USDA Jornada Experimental Range, 40 km northeast of Las Cruces, in Doña Ana County, New Mexico, as part of the Jornada Basin Long-Term Ecological Research (LTER) program. Vegetation of the study area has been described by Buffington and Herbel (1965); widespread invasion of grasslands by arid-land shrubs, including mesquite (*Prosopis glandulosa*) and creosotebush, has occurred during the past century. Mean annual precipitation at the study site is 230 mm/yr, with about 60% derived from convectional, monsoonal thunderstorms during the late summer. These summer storms, in contrast to lower intensity synoptic winter storms, often generate surface runoff. During a 3-year period, 48 summer storms, delivering > 3.6 mm/each, generated 24 runoff events (Tromble 1988).

Table 1. Soil characteristics of the runoff plots.

Area	Horizon	Depth Interval (cm)	Sand	Silt	Clay	Organic C	$CaCO_3$	Reference
			%					
CC	A	0–8	79	11	10	0.38	2.1	Rojas (1995) Pit 93.1
GS	A	0–8	71	21	7	0.52	0.19	Lajtha (1986) Pit 82.3
GI	C	0–11	89	7	4	0.15	Trace	Herbel et al. (1994) Pit H2

Runoff plots were located in two areas of black grama (*Bouteloua eriopoda*) grassland. One area (GS) was located on the alluvial piedmont of Mount Summerford, a quartz monzonite batholith that forms the northernmost peak of the Doña Ana Mountains (Seager et al. 1976). This grassland is found on soils of late Holocene age, which are classified as Ustic Haplargids (Rojas 1995). The topography drains eastward with a slope of approximately 9.3%. The second area of grassland (GI) was located approximately 10 km to the north of Mount Summerford, where soils are derived from eolian materials deposited on top of floodplain alluvium of the ancient Rio Grande. These soils are classified as Argic Petrocalcids (Monger, pers. comm.), and the topography of the site drains eastward with a slope of approximately 5.1%.

The shrubland plots (CC) were located in an area dominated by creosotebush (*Larrea tridentata*), approximately 600 m southeast and downslope from the grassland plots at Mount Summerford. This area has Typic Haplargid soils of mid-Holocene age derived from Mount Summerford alluvium (Marion et al. 1990; Rojas 1995). The average slope in the area of the shrubland plots is 4.7%. Basic characteristics of the soils in all plots are given in Table 1.

Four runoff plots were established in each area – two with relatively high (33–73%) and two with relatively low (4–28%) plant cover. In the shrubland, the low cover plots contained herbaceous and subshrub vegetation, but no creosotebush. Each plot was 2×2 m square and surrounded on 3 sides by a metal frame that prevented overland flow from crossing the area of study (Figure 1). Overland flow generated by precipitation falling on the plot itself was collected along the downslope edge using a 10-cm diameter PVC pipe, split lengthwise, to channel the discharge into collection buckets. Runoff from small events was collected in a 19-liter bucket; runoff from larger events overflowed this bucket into a subtending 120-liter tub.

Runoff volume derived from individual precipitation events was measured for 7 years (1988–1994) in the shrubland and 5.5 years (1989–1994) in the grasslands. Throughout this paper we refer to these data as the "Hydrology

Figure 1. Runoff plot located in creosotebush shrubland of the Jornada Basin, New Mexico. The edges of the plot are 2 meters in length. Photograph by David Hamilton.

Dataset." During 1988–1991 (3 years) in the shrubland and 1989–1991 (2 years) in the grasslands, subsamples of the runoff were analyzed for dissolved constituents. Throughout this paper, we refer to these data as the "Chemistry Dataset." To generate this dataset, if the runoff, normally measured one day after each precipitation event, exceeded 50 ml, a subsample was taken, centrifuged, and preserved with phenylmercuric acetate. These samples were then filtered through pre-rinsed 0.45-μ Millipore HA filters, and analyzed for NH_4, NO_3, and PO_4 using standard methods on a Traacs 800 Autoanalyzer. Total inorganic N is taken as the sum of NH_4-N + NO_3-N. Each sample was then subjected to a persulfate digestion (D'Elia et al. 1977) and reanalyzed. The difference between the digested and undigested concentrations is assumed to represent dissolved organic forms of N and P. Concentrations of cations were measured by atomic absorption (Ca and Mg) or flame emission (Na and K) spectrophotometry (Perkin Elmer 3100), and Cl and SO_4 were measured by ion chromatography (Dionex 2010i). We excluded from analysis samples collected from field buckets that were obviously contaminated by the activities of animals or samples potentially derived from more than one precipitation event.

At each site, an 8″ (20.32-cm) diameter tipping bucket raingage was used to measure the incident rainfall associated with each runoff event. The content

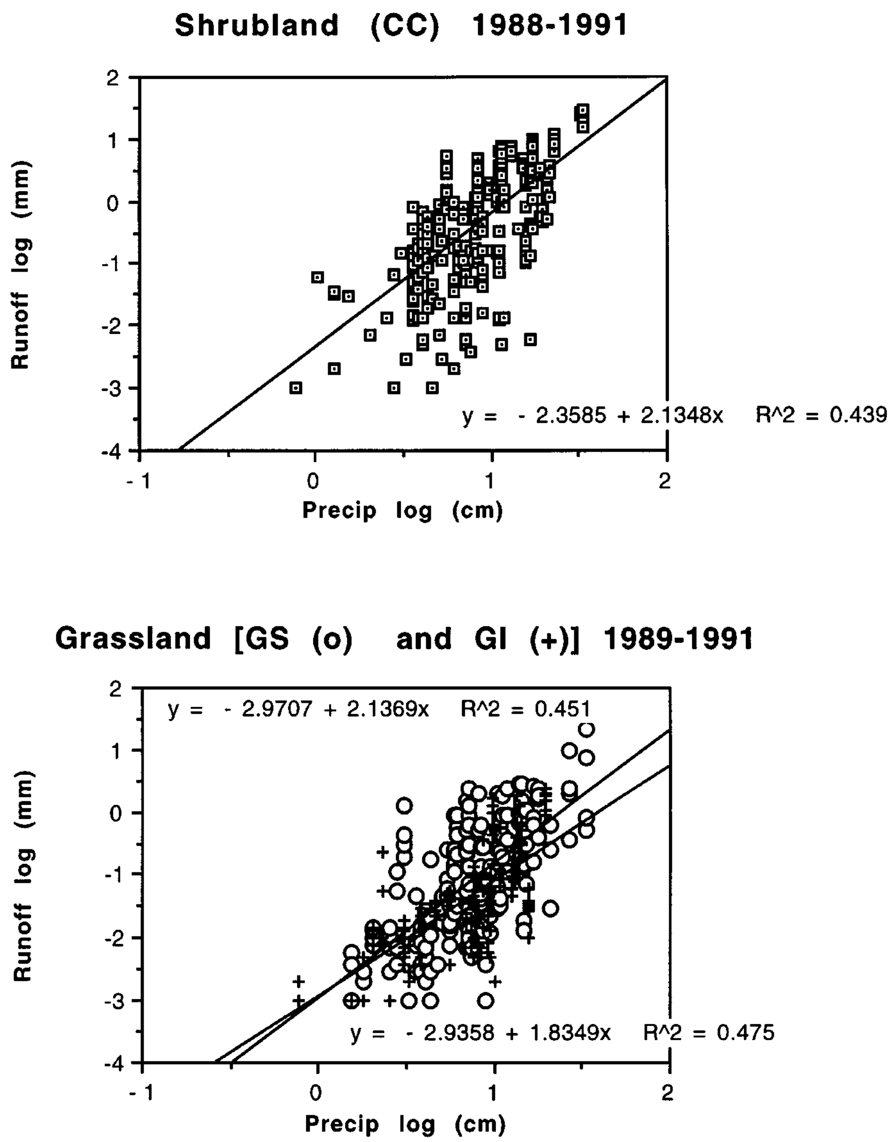

Figure 2. Runoff from plots in shrubland and grassland habitats as a function of the total precipitation defining the event. Both axes are logarithmic.

of dissolved constituents received in atmospheric deposition was measured in a single wetfall/dryfall collector (Aerochem Metrics Inc.) with 28-cm diameter buckets. This collector was located at the LTER weather station, approximately 600 m north of the Mt. Summerford plots. Wetfall was collected after each event, whereas dryfall was collected monthly. The dryfall bucket was rinsed each month with 500 ml of distilled water, and the content of constituents dissolved in this water was analyzed.

The data associated with this publication are accessible via the World Wide Web at http://jornada.nmsu.edu.

Results

During the 3-year period of the Chemistry Dataset, 276 runoff events were recorded amongst the four plots located in the creosotebush shrubland. In the grassland sites, monitored for 2 years, there were 198 runoff events among the GS plots and 140 among the GI plots. Across all plots at each site, runoff was logarithmically related to precipitation volume, with coefficients of variation (R^2) ranging from 0.44 to 0.48 (Figure 2). An analysis of variance showed that the mean slope of this relationship, calculated from the slope of the regressions for the individual runoff plots in each area (n = 4), was not significantly different among grassland (GS and GI) and shrubland (CC) plots ($P < 0.43$). However, the analysis of variance showed that the intercepts of the relationship were significantly ($P < 0.0023$) lower in both grasslands (–2.97 and –2.94) than in the shrubland plots (–2.36), suggesting that runoff commenced at a lower threshold of storm size in the shrubland than in the grasslands. In the longer record of the Hydrology Dataset, the annual runoff coefficient (total depth of runoff expressed as a percentage of the total depth of precipitation received) averaged 18.6% in the creosotebush shrubland and 5.0 to 6.3% in the grassland plots (Table 2).

Within each area, plant cover did not significantly affect the slope of the relationship between runoff and precipitation, although the difference between high and low cover plots is nearly significant in the GS grassland ($P < 0.1033$; Figure 3). In the GS grassland, the intercept was significantly higher on low cover plots ($P < 0.000l$). In the Hydrology Dataset, the volumetric runoff coefficient averaged 8.5% in low-cover plots and 4.1% in high-cover plots during the 5.5-yr period of collections at the GS grassland. During the 7-year record in the shrubland, the annual runoff coefficient was 16.4% in high-cover plots versus 20.8% in low-cover plots, which contained no creosotebush.

The concentrations of dissolved constituents declined with increasing total runoff volume in all habitats. The best fit relationships between concen-

Table 2. Total annual precipitation and total annual discharge from runoff plots in the Chihuahuan desert.

Plot	1988	1989	1990	1991	1992	1993	1994†	Mean
Precipitation (mm)								
Shrubland (CC)	271	268	243	421	393	274	166	
Grassland (GS)		209*	229	406	379	273	170	
Grassland (GI)		112*	208	390	313	249	96	
Runoff (mm)								
CC	60	62	21	104	94	27	29	
GS		18*	5	34	21	6	18	
GI		4*	3	35	5	21	5	
Runoff coefficient (%)								
CC	22.1	23.2	8.8	24.6	24.0	10.0	17.6	18.6
GS		8.8*	2.2	8.4	5.5	2.4	10.6	6.3
GI		3.9*	1.3	9.1	1.6	8.4	5.5	5.0

* 1 July–31 December 1989.
† 1 January–31 October 1994.

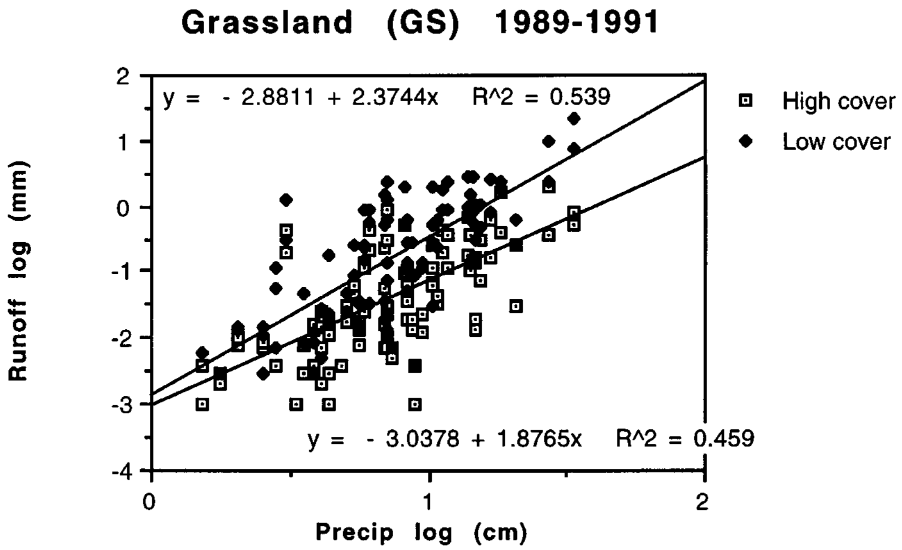

Figure 3. Runoff from high and low cover plots in the grassland habitat at Mount Summerford (GS).

tration and volume were always logarithmic (Table 3), reflecting a rapid dilution of dissolved constituents with increasing discharge. The coefficients of determination (R^2) for these relationships were nearly always higher in the grassland habitats than in the shrubland, and the R^2 for total dissolved phosphorus in the shrubland was very low. Typically, the slope of these relationships was lower (more negative) in grassland sites than in the shrubland. For Cl, SO_4, Ca, and K in the shrubland and Cl and K in the GS grassland, the slope and/or intercept was significantly different between separate regressions for high and low cover plots. Lower intercepts indicate lower concentrations of these constituents in the runoff from low-cover plots (Table 3).

The yield of dissolved constituents in each runoff event was calculated by multiplying the concentration (mg/l) by the runoff volume (liters). However, the total annual yield of dissolved constituents could not be calculated simply from the sum of the yield from individual storms, because we were unable to analyze all runoff samples. Upon collection, some samples were discarded because the collection buckets were obviously contaminated by the activity of animals; for others the volume was too small for a complete analysis. We used the logarithmic relationships between concentration and discharge (Table 3) to estimate the concentration of dissolved constituents in discharge samples that were otherwise not available. These derived concentrations were then multiplied by measured runoff to calculate the yield of dissolved constituents in those events. The number of times this procedure was employed can be ascertained by comparing the number of available analyses (Table 3) to the total number of runoff events (276 at CC, 198 at GS, and 140 at GI).

Annual losses of dissolved forms of plant nutrients in runoff were always higher in the creosotebush shrubland than in either of the grassland habitats (Table 4). Losses of TDN ranged from 0.28 to 0.41 kg ha^{-1} yr^{-1} in the shrubland versus 0.11 to 0.21 kg ha^{-1} yr^{-1} in the grassland plots. In all cases, the losses of TDP were very small (0.01 to 0.06 kg ha^{-1} yr^{-1}). The greater loss of N and P in the shrubland was related to the greater discharge in those plots. The weighted mean concentration of total dissolved nitrogen (TDN) was 0.71 mg N/l in the creosotebush shrubland, 1.39 mg N/l in the GS grassland, and 3.24 mg N/l in the GI grassland plots. Dissolved organic nitrogen comprised between 10 to 30% of the loss of TDN in each habitat. In the shrubland, where Cl, SO_4, Ca and K showed different concentration-runoff relationships between high and low cover plots (Table 3), losses were always greater from high cover plots. In the GS grassland, losses of Cl and K were greater from low cover plots, as a result of their greater volume of runoff.

Atmospheric deposition in wetfall and dryfall showed considerable year-to-year variation during the 3-year period of the Chemistry Dataset (Table 5).

Table 3. Slope and intercept for logarithmic regressions between the concentration of dissolved constituents (Y in mg/l) and runoff volume (X in mm), in the form $\log Y = m \log X + b$. For those cases when the slope or intercept was significantly different between these regressions, separate regressions are given for the data from high and low cover plots in each area.

Area Constituent/cover		Slope (m)	Intercept (b)	R^2	n^*	P
Creosotebush Shrubland (CC)						
Cl	High Cover	–0.303	0.0764	0.268	83	<0.0001
	Low Cover	–0.307	–0.0663	0.288	79	<0.0001
	Total				162	
SO_4	High Cover	–0.345	0.6429	0.408	83	<0.0001
	Low Cover	–0.320	0.5358	0.350	79	<0.0001
	Total				162	
Ca	High Cover	–0.272	0.7830	0.262	83	<0.0001
	Low Cover	–0.246	0.6653	0.254	79	<0.0001
	Total				162	
Mg		–0.158	–0.0781	0.200	162	<0.0001
Na		–0.179	–0.3904	0.113	162	<0.0001
K	High Cover	–0.103	0.5537	0.087	83	0.0067
	Low Cover	–0.138	0.4354	0.235	79	<0.0001
	Total				162	
TDN[1]		–0.305	–0.0505	0.286	161	<0.0001
	NH_4	–0.344	–0.5968	0.142	153	<0.0001
	NO_3	–0.203	0.2540	0.127	159	<0.0001
	DON[2]	–0.124	–0.5418	0.030	120	0.0577
TDP[3]		–0.042	–1.0700	0.004	37	0.7214
Grassland (GS)						
Cl	High Cover	–0.311	0.2302	0.225	46	0.0009
	Low Cover	–0.395	–0.0985	0.462	53	<0.0001
	Total				99	
SO_4		–0.468	0.5534	0.594	99	<0.0001
Ca		–0.341	0.6155	0.635	99	<0.0001
Mg		–0.319	–0.0261	0.639	99	<0.0001

Table 3. Continued.

Area Constituent/cover		Slope (m)	Intercept (b)	R^2	n*	P
Na		–0.336	–0.3055	0.328	99	<0.0001
K	High Cover	–0.198	0.7624	0.212	46	0.0013
	Low Cover	–0.226	0.6408	0.520	53	<0.0001
	Total				99	
TDN		–0.467	0.1325	0.402	99	<0.0001
	NH_4	–0.512	–0.2271	0.249	95	<0.0001
	NO_3	–0.399	–0.3742	0.522	93	<0.0001
	DON	–0.222	–0.4810	0.088	73	0.0108
TDP		–0.273	–0.7112	0.088	50	0.0361
Grassland (GI)						
Cl		–0.610	–0.1050	0.460	57	<0.0001
SO_4		–0.654	0.4802	0.603	57	<0.0001
Ca		–0.424	0.6241	0.525	57	<0.0001
Mg		–0.417	0.0441	0.567	57	<0.0001
Na		–0.562	–0.4084	0.454	57	<0.0001
K		–0.531	0.5292	0.419	57	<0.0001
TDN		–0.568	0.1934	0.460	57	<0.0001
	NH_4	–0.579	0.0395	0.338	57	<0.0001
	NO_3	–0.468	0.1896	0.465	55	<0.0001
	DON	–0.295	–0.3016	0.165	43	0.0068
TDP		–0.751	–0.8629	0.268	26	0.0067

* The number of available analyses is reduced when samples had no measurable content of some constituents, because the logarithmic value of zero is undefined.
[1] Total dissolved nitrogen.
[2] Dissolved organic nitrogen.
[3] Total dissolved phosphorus.

Table 4. Yield of runoff (mm) and dissolved constituents in runoff (kg ha^{-1} yr^{-1}) for habitats of the Jornada Basin, New Mexico For cases when the loss from high and low cover plots was significantly different, the overall estimate of nutrient loss in shrublands was calculated by assuming 30% shrub cover and 70% bare soil.

	Creosotebush (CC)			Grassland (GS)		Grassland (GI)	
Constituent	1988–89*	1989–90	1990–91	1989–90	1990–91	1989–90	1990–91
Precipitation (mm)	274.9	284.5	257.8	269.2	241.8	155.0	223.3
Runoff (mm)	62.6	63.6	25.3	18.9	7.8	4.6	2.8
Runoff coefficient (%)	22.8	22.4	9.8	7.0	3.2	3.0	1.3
Yield, dissolved forms (kg/ha)							
Chloride	0.46	0.40	0.21	0.15	0.12	0.06	0.07
High Cover	0.58	0.47	0.26	0.10	0.12		
Low Cover	0.41	0.37	0.19	0.19	0.12		
Sulfate	1.28	1.48	0.92	0.50	0.42	0.26	0.27
High Cover	1.45	1.66	1.12				
Low Cover	1.21	1.40	0.84				
Calcium	1.64	3.26	1.53	0.60	0.37	0.28	0.19
High Cover	2.11	3.43	1.76				
Low Cover	1.44	3.19	1.43				
Magnesium	0.34	0.46	0.23	0.13	0.09	0.06	0.06
Sodium	0.36	0.19	0.14	0.08	0.05	0.03	0.03
Potassium	1.39	1.59	0.79	0.69	0.43	0.21	0.38
High Cover	1.70	1.92	0.94	0.31	0.56		
Low Cover	1.25	1.45	0.72	1.06	0.56		
Total Nitrogen	0.29	0.41	0.28	0.21	0.16	0.13	0.11
NH_4-N	0.10	0.13	0.07	0.11	0.06	0.09	0.07
NO_3-N	0.15	0.22	0.13	0.08	0.06	0.02	0.02
DON	0.04	0.05	0.08	0.02	0.04	0.02	0.02
Total Phosphorus	0.02	0.06	0.01	0.03	0.01	0.01	0.02

* Hydrologic year; e.g., 1 July 1988–30 June 1989.

Table 5. Atmospheric deposition by wetfall and dryfall of dissolved constituents at the Jornada Basin LTER, for the 3-year period 1 July 1988–30 June 1991 For wetfall, comparative data are given for the National Atmospheric Deposition site near Mayhill, New Mexico. All data are kg ha^{-1} yr^{-1} ! 1 S.E. for *n* = 3 years.

Constituent	NADP Wetfall*	Jornada Basin		
		Wetfall	Dryfall	Total
Cl	0.42	0.59 ± 0.07	0.24 ± 0.04	0.83 ± 0.11
SO_4	4.35	4.32 ± 0.36	2.03 ± 0.07	6.35 ± 0.42
Ca	1.26	1.35 ± 0.29	1.36 ± 0.17	2.71 ±0.28
Mg	0.09	0.07 ± 0.03	0.10 ± 0.01	0.17 ± 0.03
Na	0.30	0.44 ± 0.15	0.22 ± 0.02	0.65 ± 0.16
K	0.06	0.20 ± 0.04	0.27 ± 0.03	0.47 ± 0.07
TDN		1.72 ± 0.32	0.81 ± 0.17	2.52 ± 0.47
NH_4-N	0.63	0.93 ± 0.14	0.21 ± 0.07	1.13 ± 0.19
NO_3-N	0.82	0.83 ± 0.17	0.44 ± 0.04	1.26 ± 0.21
DON		0	0.16 ± 0.06	0.13± 0.08
TDP		0.06 ± 0.06	0.02 ± 0.02	0.08 ± 0.05

* 4 years; 1988–1991; National Atmospheric Deposition Program, http://nadp.sws.uiuc.edu/nadpdata.

For Cl, SO_4, Na and N, wetfall deposition exceeded dryfall deposition by a factor of 2 or more. For most constituents, the mean annual deposition in wetfall showed close agreement with that measured at the nearest station of the National Atmospheric Deposition Program (NADP), located in Mayhill, New Mexico – 140 km to the east (Table 5).

An estimate of the *net* gain or loss of constituents in the ecosystem is obtained by subtracting the mean annual loss of dissolved forms in runoff from the mean annual input derived from the sum of wet and dry deposition (Table 6). With one exception, K at GS, both grassland habitats showed net accumulations of all constituents, including an average net accumulation of 2.38 kg ha^{-1} yr^{-1} of nitrogen. For Cl, SO_4, Ca, Na and N in the grasslands, the difference between deposition and runoff is so large that the apparent net gain of these elements is probably robust against annual variations in these parameters. This is also true for Cl, SO_4, Na, K, and N in the shrubland, which appeared to accumulate nitrogen at a rate of 2.19 kg ha^{-1} yr^{-1}. Despite the large loss of Ca in runoff from the shrubland (2.14 kg ha^{-1} yr^{-1}), there was a potential net accumulation of 0.57 kg ha^{-1} yr^{-1} of Ca in these soils from atmospheric deposition.

Table 6. Mass balance for ions in Jornada Basin habitats, New Mexico. All data are kg ha^{-1} yr^{-1}.

Area/Constituent	Atmospheric Deposition*	Runoff Loss*	Net Loss (–) or gain (+)
Creosotebush (CC) Shrubland			
Cl	0.83	0.36	+0.47
SO4	6.35	1.22	+5.13
Ca	2.71	2.14	+0.57
Mg	0.17	0.39	–0.22
Na	0.65	0.23	+0.42
K	0.47	1.26	–0.79
TDN	2.52	0.33	+2.19
TDP	0.08	0.03	+0.05
Grassland (GS)			
Cl	0.83	0.13	+0.70
SO4	6.35	0.46	+5.89
Ca	2.71	0.49	+2.22
Mg	0.17	0.11	+0.06
Na	0.65	0.07	+0.58
K	0.47	0.56	–0.21
TDN	2.52	0.19	+2.33
TDP	0.08	0.02	+0.06
Grassland (GI)			
Cl	0.83	0.07	+0.76
SO4	6.35	0.27	+6.08
Ca	2.71	0.24	+2.47
Mg	0.17	0.06	+0.11
Na	0.65	0.03	+0.62
K	0.47	0.30	+0.17
TDN	2.52	0.12	+2.40
TDP	0.08	0.02	+0.06

* Three-year mean for atmospheric deposition (Table 5) and runoff (Table 4) in the shrubland; 2-year mean for runoff in grasslands (Table 4).

Discussion

The element mass-balances reported in Table 6 consider only the receipt of nutrients from atmospheric deposition and the losses of dissolved forms in hillslope runoff. Additional nutrient losses from the ecosystem may occur in the transport of suspended sediments and bedload sediment carried in stream channels. These sediments are known to carry the majority of phosphorus, in adsorbed forms, lost from terrestrial ecosystems (Avnimelech & McHenry 1984). Indeed, Bolton et al. (1991) found a strong power relationship between the concentration of total P and the concentration of suspended sediments lost during rainfall simulation experiments in the Chihuahuan desert.

In desert rivers, the transport of suspended and bedload sediments is episodic (e.g., Gifford & Busby 1973; Fisher & Grimm 1985), with a large proportion of these sediments carried by rare events (Laronne & Reid 1993). Most of the total elemental content of the suspended load is not immediately available for plant uptake, so budgets for dissolved nutrients may be the best indication of any changes in soil fertility that are associated with changes in vegetation. Thus, our budgets are conservative estimates of annual losses of nutrients in runoff.

In the rainfall simulation experiments reported by Schlesinger et al. (1999), the mean runoff coefficient from grassland plots (24.2%) was lower than that found in plots with creosotebush (29.9%) or in bare plots located between creosotebush (55.4%). In the present study, the corresponding runoff coefficients were lower – 5.0 to 6.3% in grasslands, 16.4% in plots with creosotebush, and 20.8% in the low-cover plots, without creosotebush, in the shrubland. Tromble (1988) reported a mean runoff coefficient of 20.1% from natural precipitation events received over 3 years on four 81-m^2 plots dominated by creosotebush in the Jornada basin, similar to values found in the present study. Fisher and Grimm (1985) report runoff coefficients ranging from 10.3 to 25% in a shrub-covered Sonoran desert watershed receiving rainfall at intensities of 0.28 to 0.46 mm/min in 3 late-summer storms. The relatively high runoff coefficients recorded in the rainfall simulation experiments stem from the high intensity of rainfall (2.33 mm/min) applied in that work (Schlesinger et al. 1999).

As in the rainfall simulation experiments, the data reported here show that a loss of grass cover and invasion by arid-land shrubs leads to greater runoff from dryland ecosystems (cf. Abrahams et al. 1995; Gutierrez & Hernandez 1996; Castillo et al. 1997). Discharge commences at a lower threshold of storm size in shrublands and on low-cover grassland plots. Hawkins and Ward (unpublished) report initial discharge after 5.6 mm of precipitation in shrublands and 8.9 mm of precipitation on grasslands at the Jornada. The total discharge increases as a function of the volume of precipitation received, but the slope of these relationships differs little between grass vs shrub and between high vs lower cover plots (Figures 2 and 3). Higher overall discharge, rather than higher nutrient concentrations in runoff waters, accounts for the greater losses of nutrients from shrublands.

A comparison of nutrient losses in the creosotebush shrubland and the Mount Summerford grassland (GS) is particularly instructive, because both are located on the same quartz monzonite parent materials. During the 3-year period of study, there were apparent small net accumulations of Na and Cl in the creosotebush shrubland, perhaps leading to the deposition of halite (NaCl) in a lower soil horizon. The molar ratio of Na to Cl in the runoff from the creosotebush plots (0.98) is very close to the ratio in the ratio in halite (1.00). The higher losses of K from the high cover plots in the creosotebush shrubland may be related to the uptake and cycling of K by the shrubs (Table 4). The net accumulation of Ca and SO_4 in the creosotebush plots may be associated with the formation of gypsic ($CaSO_4 \cdot 2H_2O$) horizons in the soil profile, although Marion et al. (1990) reported that the upper horizons of these soil profiles were undersaturated with respect to gypsum. The molar ratio of Ca to Cl in the runoff from the creosotebush plots (5.26) is much higher than that in atmospheric deposition (2.89), suggesting that $CaCO_3$ is being lost from the surface horizon of these soils which are supersaturated with respect to calcite (cf. Marion et al. 1990). Greater losses of Ca from the high cover plots (Table 4) may be related to the dissolution of soil $CaCO_3$ that is sometimes seen under creosotebush (Wallace & Romney 1972, p. 308).

The molar ratio of Na to Cl in the runoff from grasslands (0.83 in GS and 0.66 in GI) is much lower than that in atmospheric deposition (1.21) and halite, suggesting that Na must be retained on the cation exchange capacity of the grassland soils. Grassland soils also show strong net accumulations of Ca and SO_4, perhaps related to the formation of gypsic soil horizons. The molar ratio of Ca and SO_4 in the net retention of these elements is 0.90 in the GS grassland and 0.98 in the GI grassland; both values are close to the ratio in gypsum (1.00). Ca may also be retained in calcic horizons. Indeed, the net accumulation of Ca in the grassland plots is much less than the measured rate of $CaCO_3$ accumulation in soils of the Jornada basin (Marion 1989). There are also small accumulations of Cl in the grassland plots, so the molar ratio for Ca/Cl in runoff (3.34 at GS and 3.04 at GI) remains close to that in atmospheric deposition (2.89). Fisher and Grimm (1985) also report net Cl accumulations in a Sonoran desert watershed.

This study and the rainfall simulation experiments are consistent in showing greater losses of total dissolved nitrogen from shrublands than from grasslands. The greater loss of N from the shrubland may be attributed in part to the greater precipitation received in the shrubland during 1989–1991 (Table 4), but the runoff coefficients for the shrubland plots were always higher than those for grassland plots during the 6-year comparison (Table 2). The GS grassland lost more N than the GI grassland, in which the surface soil is sandy with low organic content (Table 1).

In both shrubland and grassland plots, the losses of N found here were less than those estimated from the rainfall simulation experiments (0.43 and 0.25 kg ha^{-1} yr^{-1}, respectively, Schlesinger et al. 1999). The mean runoff for 1988–1991 was below the long-term average (1989–1994, Table 2) that we used to make a long-term extrapolation of N yield from the rainfall simulation experiments. Our values for DON are also lower (<20% of TDN) than those measured during the rainfall simulation experiments (ca. 60% of TDN). The rainfall simulation experiments were performed at the end of the dry season, when a large quantity of soluble organic nitrogen compounds may have accumulated, undecomposed, in the soil. Bolton et al. (1991) report concentrations of 0.84 mg/l of total nitrogen in runoff from rainfall simulation experiments in creosotebush shrublands at the Jornada, suggesting that the concentration of total dissolved nitrogen reported here (0.71 mg/l) accounts for 84% of the total loss of nitrogen in runoff. The nitrogen losses reported here are similar to those reported in forests in western North Carolina [<0.25 kgN ha^{-1} yr^{-1} (Swank & Vose 1997)], but lower than those reported for the Hubbard Brook forest in New Hampshire [2.68 kgN ha^{-1} yr^{-1} (Likens & Bormann 1995)] – an area that receives an excess deposition of atmospheric nitrogen.

Although nitrogen losses in runoff are greater in the shrubland (0.33 kg ha^{-1} yr^{-1}) than in the grasslands (0.12 and 0.19 kg ha^{-1} ha^{-1}), the transport of dissolved nitrogen compounds in runoff cannot by itself account for the depletion of soil N that is associated with desertification in this region (Schlesinger et al. 1996; Kieft et al. 1998). Suspended and bedload sediments may account for additional losses (ca. 16%) of nitrogen, although Bolton et al. (1991) report no significant relationship between total N concentration and the concentration of suspended sediments in runoff from the Jornada. The N losses in runoff are lower than the inputs in atmospheric deposition, so all habitats show a net gain of soil N if one compares only atmospheric deposition and runoff (cf. Fisher & Grimm 1985). Recent work suggests that the emission of N gases (NH_3, NO, N_2O and N_2) and soil dust may contribute to the total loss of N from these ecosystems. For example, Guilbault and Matthias (1998) report a loss of 0.4 kg N ha^{-1} yr^{-1} as N_2O from a Sonoran desert shrubland, and Hartley (1997) suggests that total gaseous emissions of N may be as high as 3.9 and 6.2 kg ha^{-1} yr^{-1} in grassland and shrubland soils, respectively, at the Jornada. Losses of soil fertility to the atmosphere (gases + eolian transport) should be considered in any assessment of changes in soils that are associated with desertification.

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