CSIRO PUBLISHING

The Rangeland Journal, 2013, **35**, 285–297
http://dx.doi.org/10.1071/RJ12095

Can livestock and fires convert the sub-tropical mountain rangelands of central Argentina into a rocky desert?

A. M. Cingolani[A,C], *M. V. Vaieretti*[A], *M. A. Giorgis*[A], *N. La Torre*[B], *J. I. Whitworth-Hulse*[A] *and D. Renison*[B]

[A]Instituto Multidisciplinario de Biología Vegetal (CONICET – Universidad Nacional de Córdoba), CC 495, X5000JJC, Córdoba, Argentina.
[B]Centro de Ecología y Recursos Naturales Renovables – Dr Ricardo Luti, Instituto de Investigaciones Biológicas y Tecnológicas (CONICET – Universidad Nacional de Córdoba), Av. Vélez Sarsfield 1611, X5016GCA Córdoba, Argentina.
[C]Corresponding author. Email: acingola@com.uncor.edu

Abstract. Soil erosion, as a result of livestock grazing, has been widely reported for arid and semiarid ecosystems, but information is lacking in more mesic ecosystems where erosion is generally studied in relation to agriculture. To test the hypothesis that, in the high-mountain rangelands of Córdoba (Argentina), grazing by livestock can drive the system into a rocky desert, 200 4 × 4 m plots under different livestock stocking rates and timings of grazing were monitored for 5 years. Four indicators of soil erosion: change rate of rock surface and of total bare surface, advance rate of erosion edges, and their activity persistence were estimated for each plot. Erosion edges are steps with a vertical bare soil surface, whose advance usually leaves behind an exposed rock area. For each plot, the average annual stocking rate for the 5-year period, and an index of seasonality, were calculated. Multiple regressions were used to analyse the data. Under high stocking rates, rock and bare surface increased, edges advanced faster and persisted more actively, while under low or nil stocking rates, rock and bare surface decreased and edges tended to stabilise. From these results, it was estimated that under high stocking rates, 18% of the whole area could be transformed into rocky surface in 400 years. As fire is a usual tool for this rangeland management, surface soil loss during 1 year in 77 burned and unburned plots, with and without post-fire livestock grazing, were compared. Burned plots lost 0.6 cm of surface soil when grazed, and 0.4 cm when ungrazed, while unburned plots lost less than 0.05 cm when grazed, and gained 0.07 cm when ungrazed. It was concluded that the present-day combination of livestock and fire management has the potential to convert this rangeland into a rocky desert. It is suggested that commercial livestock production, as it is carried on at present, is not sustainable, and some suggestions on changes necessary for a future sustainable grazing industry are made.

Additional keywords: desertification, land degradation, overgrazing, restoration, soil erosion.

Received 26 November 2012, accepted 10 April 2013, published online 28 May 2013

Introduction

Increased rates of soil loss due to human activities are a major worldwide environmental concern. In the case of croplands, it has been estimated that ~10 million ha year^{-1} of arable land are abandoned due to erosion (Pimentel *et al*. 1995). In pastures and rangelands used for livestock production, erosion is less accentuated but also dramatic. In the USA, in the order of 6 tons ha^{-1} year^{-1} is being lost from rangelands due to overgrazing (Pimentel *et al*. 1995). Erosion rates, however, can be much higher in less affluent countries (Singh *et al*. 1992; Pimentel *et al*. 1995; Mwendera *et al*. 1997; Pimentel 2006; Chartier *et al*. 2009). The average global rate of soil formation is far lower than the soil depletion rate, indicating that the livestock industry is producing a net loss of natural capital in many rangelands of the world (Pimentel *et al*. 1995; Ares 2007). Fire, which is often used to facilitate land conversion to pastures and to manage rangelands, is also an important driver of soil erosion (Wondzell and King 2003; Asner *et al*. 2004). Several studies have shown that erosion rates immediately after fire can be one or more orders of magnitude higher than in unburned sites (e.g. Morris and Moses 1987; Inbar *et al*. 1998; Johansen *et al*. 2001; Pausas *et al*. 2008). If fire is combined with post-fire grazing, erosion processes can be intensified (Pfeiffer and Steuter 1994; Ludwig *et al*. 2005).

Almost all ecosystems used for livestock production are prone to high soil erosion rates when overgrazed or burned, but susceptibility to erosion varies according to topography, soil characteristics, lithology, climate, ecosystem productivity and the evolutionary history of grazing (Davenport *et al*. 1998; Perevolotsky and Seligman 1998; Lal 2001; Cingolani *et al*. 2005, 2008*a*; Diamond 2005). In arid or semiarid ecosystems, overgrazing can eliminate plant and biological soil crust cover, triggers erosion processes, and can drive the system into desertification, with the consequent economic and social losses (Descroix *et al*. 2001; Lal 2001; Tongway *et al*. 2003; Asner *et al*. 2004; Ludwig *et al*. 2005; Neff *et al*. 2005; Yong-Zhong *et al*. 2005; Ares 2007; Pimentel 2006; Chartier *et al*. 2009). In more mesic ecosystems, soil erosion is largely reported in association with agricultural practices (Reganold *et al*. 1987; O'Hara *et al*. 1993; Lang 2003; Pimentel *et al*. 2005; Pimentel 2006), and there are a surprisingly low number of studies measuring soil erosion as a result of livestock pressure (e.g. Mwendera *et al*. 1997; Renison *et al*. 2010). Particularly lacking are studies focusing on soil erosion produced by livestock in tropical and sub-tropical mountain environments (Reinhardt *et al*. 2010).

The upper belt of the Córdoba Mountains in central Argentina has been traditionally used for livestock production. This is a sub-humid ecosystem (900 mm annual rainfall, with an extended dry season) dominated by a mosaic of grasslands and woodlands, with granite rocky patches occupying a large portion of the area (42% of 124 700 ha mapped by Cingolani *et al*. 2004). This large surface of bare rock has been interpreted by landowners and scientists as natural features of the landscape, a result of a long history of geo-morphological processes, only slightly modified by anthropogenic activities (Cabido *et al*. 1987; Funes and Cabido 1995; C. Cuello and G. Altamirano (landowners), pers. comm.). However, through Landsat satellite images, two types of bare rock: the 'natural outcrops' and the 'pavements and stony ground exposed by recent erosion' ('exposed rock') can be spectrally discriminated, and these can be recognised also on the terrain (Fig. 1). According to Cingolani *et al*. (2004), about half of the rocky area in the upper mountain belt (i.e. 20% of the whole area) is recently exposed rock.

In later studies (Cingolani *et al*. 2005, 2008*b*; Renison *et al*. 2010), it was suggested that most of this recently exposed rocky area was covered by soil and vegetation before the introduction of European livestock four centuries ago. It can be hypothesised that livestock has the potential to convert the upper mountain belt of central Argentina into a rocky desert. It is suggested that the introduction of livestock strongly increased the grazing and trampling pressure on the system, previously grazed only by camelids and smaller herbivores (Díaz *et al*. 1994; Pastor 2000; Medina 2006; Flores *et al*. 2012). As a result, soil erosion was triggered through the creation of numerous active erosion edges, which are steps with a vertical surface of bare soil usually perpendicular to the slope (Fig. 1). Active erosion edges are common features in the present landscape. On shallow soils (i.e. less than 70 cm), which are dominant in the area, edge height is usually equivalent to the depth of the soil profile ('soil-rock' edges, Fig. 1). Edges advance when soil particles in the vertical surface are detached and transported by weather agents (including frosts and thaw cycles, raindrop impacts, wind and running water), leaving behind large surfaces of exposed rock (Cingolani *et al*. 2003, 2004; Renison *et al*. 2010). On deeper soils, erosion edges can contact the rock substrate but often are shallower than the soil profile, leaving behind bare soil, as they advance, which can later be colonised by plants ('soil-soil' edges, Fig. 1). In addition, fires seem to have increased with the introduction of livestock, because of the need to clear woodlands for pastures and to produce grass regrowth in the dry season. It is suggested that, after vegetation is burned, soil remains unprotected and more exposed to detachment and transport by weather agents, particularly if livestock are present (Morris and Moses 1987; Lal 2001; Cingolani *et al*. 2008*b*; Renison *et al*. 2010). As deep soils became progressively shallower due to post-fire surface erosion or the advance of soil-soil edges, the creation of new soil-rock erosion edges is facilitated. According to our hypothesis, once created, soil-rock edges advance faster under heavy rather than low livestock pressures. Moreover, some evidence suggests that under livestock exclusion or low grazing pressure, edge activity declines and eventually stops (Cingolani *et al*. 2003; Renison *et al*. 2010). This would accentuate the present role of livestock as a geomorphologic agent (Trimble and Mendel 1995), by progressively converting the rangeland into a rocky desert. Due to shallower soils and coarser soil texture, this conversion is

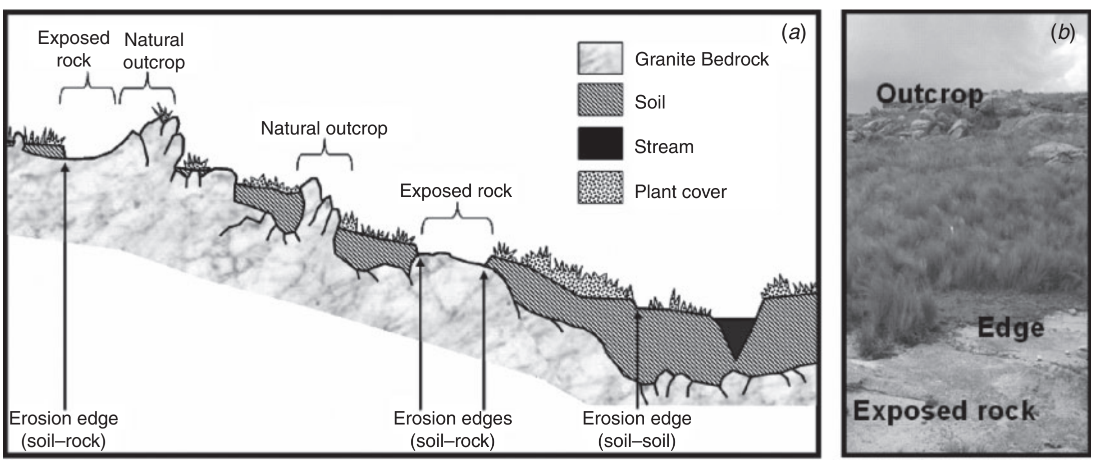

Fig. 1. (*a*) Schematic topographical gradient showing the soil above the granite bedrock, and the two types of rocky surfaces (recently exposed rock and natural outcrops), and the soil erosion edges (modified from Cabido *et al.* 1987). (*b*) Picture showing both types of bare rock: natural outcrops and recently exposed rock. The erosion edge can be observed in the limit between the recently exposed rock and the soil with vegetation.

more prone to occur in higher than in lower topographic positions (Cingolani *et al*. 2003, 2008*b*). Away from edges, the most extended plant communities have less than 5% of exposed horizontal bare soil; thus, surface erosion is probably very low in the majority of the area, except after burning.

To test some of the mechanisms hypothesised above, soil erosion under different stocking rates and at different locations in relation to a fire event were compared. Specifically, our objectives were: (1) to analyse how the rock surface and the total bare surface (rock surface plus bare soil) changed in sites with nil to high stocking rates during a 5-year period; (2) to analyse the advance and activity persistence of soil-rock erosion edges in the same period and under the same stocking rates as before; (3) to determine whether season of grazing, or landscape and site variables influence the process; and (4) to compare the surface soil loss during a 1-year period, in burned and unburned areas with and without a post-fire low to moderate livestock pressure.

It was predicted that edges will advance faster and persist more actively under high stocking rates than under low or nil stocking rates. In consequence, rock and bare surface in sites with high stocking rates will increase while in sites with low or nil stocking rates they will decrease or remain stable. Additionally, it was predicted that surface soil erosion in burned sites will be much higher than in unburned sites, particularly if the burned sites are stocked after a fire.

Materials and methods

Study area

The ‘Sierras Grandes’, in Córdoba province, is the highest mountain range of Central Argentina, rising to 2880 m a.s.l. This range runs north to south and has the tectonic structure of a horst. Most of the upper belt above 1700 m a.s.l. is occupied by a relatively flat plateau, the ‘Pampa de Achala’, remnant of an ancient crystalline peneplain (Cabido *et al*. 1987). A National Park (Quebrada del Condorito) and a Provincial Water Reserve (Pampa de Achala) were designated in 1997 and 1999, respectively, in the area. While the National Park is under state control, the Reserve is privately owned and relatively few effective conservation actions have been carried out. On the plateau, at 2200 m a.s.l., mean temperature of the coldest and warmest months are 5.0 and 11.4°C, respectively, with no frost-free period. Mean annual precipitation is 900 mm, with most rainfall concentrated in the warmest months, from October to April (Cabido 1985; Colladon 2010). The area has been grazed by domestic livestock for ~400 years, with almost no other agricultural use. At present, most livestock consists of cattle, while sheep, horses and goats are less abundant. Some decades ago sheep were more important.

Most soils are Mollisols derived from the weathering of the granite substrate and fine-textured eolian deposits. Due to the cold climate, soils are dark, with high organic matter content, and a thin peat layer may form in the valley floors (Cabido *et al*. 1987). The toposequence, with many disruptions due to the complex characteristics of the granite substrate, is shown in Fig. 1. At upper topographic levels, soils tend to be shallow and well drained because of a coarser texture, while at middle levels soils tend to be somewhat deeper, with a lower sand content. At lower slopes and on the valley floors, soil depth may reach several metres, texture is finer, and drainage poorer (Cabido *et al*. 1987; Cingolani *et al*. 2003; Enrico *et al*. 2004).

At upper to middle topographic levels, active erosion is mainly observed in edges or steps usually less than 70 cm high and generally perpendicular to the slope gradient (Fig. 1). In flat uplands, they can form circular cavities where water temporally accumulates (Cabido and Acosta 1986). At lower slopes and valley floors, erosion edges often form gullies some metres deep and parallel to the slope gradient. Gully floors are usually densely vegetated but can also have exposed rock. The detachment of particles along edges is produced by raindrops during the wet summer season, and by diurnal frosts and thaw cycles during the autumn season, when soil water content is still high and temperatures begin to decline. Runoff water after summer rainstorms also contributes to particle detachment, especially in deep gullies at low topographic levels. Ice and/or water first produce a concavity in the lower portion of the vertical surface of the edges, and then the upper soil with vegetation collapses, aided by the trampling of animals. Later, this loose soil is exported by wind or running water, unless it is colonised by vegetation (A. M. Cingolani, pers. obs.). Away from edges, plant cover is high, except when vegetation is burned or during extremely dry years (A. M. Cingolani, pers. obs.) when soils may be exposed to surface erosion. However, even in recently burned sites, rill erosion in the study area has not observed except in very local situations.

The landscape is a mosaic of woodlands and grasslands that alternate with rocky surfaces in the form of natural outcrops or rock exposed by recent soil erosion (Cingolani *et al*. 2004). Natural outcrops are usually above the soil level, covered by lichens and often with many crevices hosting various plant species. Recently exposed rock surfaces are usually below or at the same level of adjacent soil, and have the appearance of a pavement or stony ground. Often, erosion edges in the proximity indicate the previous soil level (Cingolani *et al*. 2003). This rock is covered by less lichen, has fewer crevices with vegetation, and it is less rough and of a lighter colour than the natural outcrops (Fig. 1). Woodlands are generally small patches dominated by *Polylepis australis* Bitter. Grasslands can be dominated by tussock grasses [*Poa stuckertii* (Hack.) Parodi, *Deyeuxia hieronymi* (Hack.) Türpe and *Festuca* spp.], or by short grasses, sedges, rushes and forbs (‘grazing lawns’ *sensu* McNaughton 1984; Cingolani *et al*. 2003, 2010).

In the National Park, domestic livestock were maintained in some areas to preserve local biodiversity and prevent excessive accumulation of dead biomass (Cingolani *et al*. 2010). Grazing management schemes have been implemented, which involve various stocking rates and timing of grazing. In the surrounding Reserve area, stocking rates are generally higher than in the Park, and fire is often used to reduce tussock cover and stimulate regrowth.

Sampling design

Stocking rates and soil erosion

Effects of livestock on soil erosion were assessed from 200 permanent plots of 4 × 4 m distributed in a large portion of the upper altitudinal belt, within the National Park and the Provincial Reserve. Plots occupied a range of 50 km in the north–south direction and 20 km in the east–west direction (approximate central point of the study area: 64°46′W, 31°36′S), varying in altitude from 1800 to 2300 m a.s.l. Annual rainfall is very similar across the area (A. M. Cingolani, unpubl. data). Plots were located in the upper and medium portions of five eastward catchments of 39–109 km^2, which range from 1500 to 2200–2350 m a.s.l. They were distributed in fenced paddocks or in grazing areas limited by natural boundaries (hereafter also named 'paddocks' for simplicity) under variable grazing regimes and stocking rates, which included grazing exclusion of various lengths (less than 1-year exclusions to 6-year exclusions, $n = 36$ plots); continuous grazing ($n = 107$); and seasonal grazing ($n = 57$). According to detailed data provided by National Parks administration and livestock owners, annual stocking rates during the 5-year period of the study varied from 0 to 0.34 cattle units (CU) per ha of non-rock surface (CU ha^{-1}) among paddocks, but instantaneous stocking rates may have been as high as 2 CU ha^{-1}. Plots were chosen to represent all the plant cover units and topographic conditions present in the paddocks, except that closed woodlands (i.e. more than 50% woody vegetation cover), areas with more than 75% of bare rock, and very steep slopes and rugged outcrops, were not selected. Additionally, 92 out of the 200 plots were selected to include soil-rock erosion edges. Selected edges varied from less than 1 to 41 cm high. Stocking rates and timing of grazing in each paddock were fairly constant during the 5-year period of the study.

Post-fire erosion assessment

To assess the combined effects of fire and livestock on surface-soil erosion, a two-factor design was set up as follows. In September 2009, 77 plots were selected after a wildfire occurred in July 2009 over part of the study area. Forty-two of the selected plots were part of the initial set of 200 plots, while 35 were new. Burned areas included 41 grazed plots and 15 ungrazed plots, and unburned areas included 15 grazed plots and 6 ungrazed plots. The grazed plots, both in burned and unburned areas, had a similar range of annual stocking rates for the 5-year period previous to the post-fire erosion assessment (from 0.20 to 0.26 CU ha^{-1} depending on the paddock where the plot was located), and included annual and seasonal grazing. In the case of grazed and burned plots, livestock was excluded from the paddocks immediately after the fire for 6 months, until January 2010, after the onset of the rain season. Thus, annual stocking rates during the year of post-fire erosion assessment (from September 2009 to September 2010) varied between 0.18 and 0.23 CU ha^{-1} for unburned plots and from 0.10 and 0.19 CU ha^{-1} for burned plots.

Measurements

Stocking rate and soil erosion

The 200 sampling plots were established in different years (142 plots in 2004, 40 plots in 2005 and 18 plots in 2006) but always in September to standardise the estimation of the different variables at the driest and most inactive period of the year. When establishing the plots, slope aspect (degrees from the north), slope inclination (%), and soil depth (cm) were recorded. Soil depth was estimated by hammering an iron pin into the soil at each corner of the plot or the nearest available site within the plot if bare rock was encountered at the corners. The four pin depths were averaged per plot. Location and altitude (m a.s.l.) were recorded using a Global Positioning System. Additionally, for each plot the following variables at the landscape scale were obtained from a Geographic Information System and a Digital Elevation Model with a resolution of 30 × 30 m (Jain and Kothyari 2000; Cingolani *et al*. 2004, 2008*b*): bare rock cover (%), slope inclination (%), roughness (a dimensionless index), the run-on contributing area (expressed as the number of 30 × 30-m cells), and the relative position of the site in the landscape, measured in relation to the total flow path between the drainage divide and the closest drainage channel (%). Finally, in the 92 sites selected to include a soil-rock edge, a line was painted on the rock at the rock-soil transition and repainted when necessary at later dates to avoid losing the trace due to weathering (paint collar method, Hudson 1993; Fig. 2). In some cases, because of physical difficulties (e.g. sand deposition on rock), the line could not be painted exactly at the contact between the edge or the rock. Sometimes it was necessary to paint more than one line to better represent the plot. Due to the different characteristics of the plots, the total (cumulative) length of the line(s) was very variable between plots.

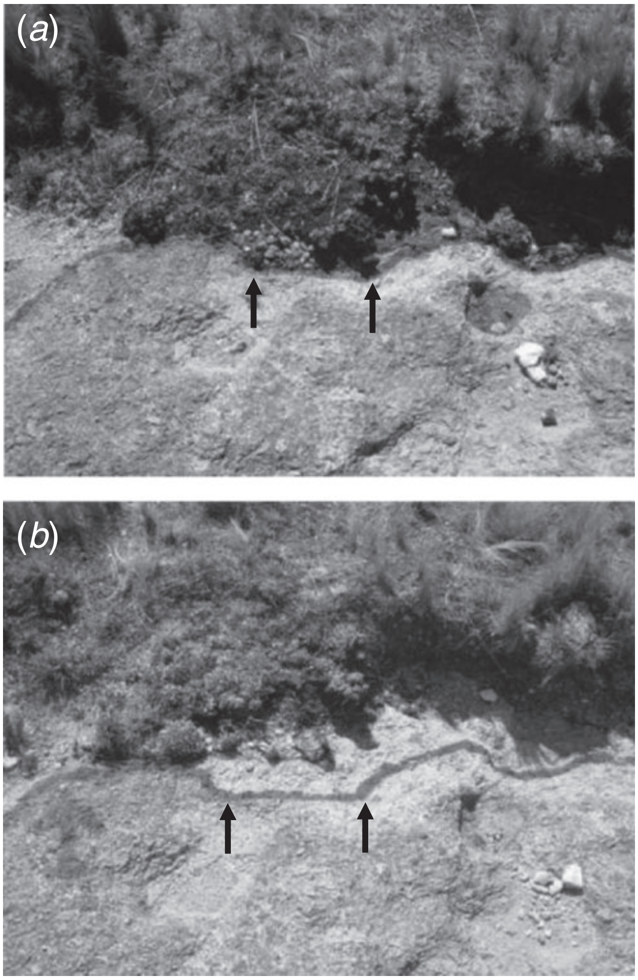

Fig. 2. (*a*) Soil-rock edge with the painted line at the initial date (2004) and (*b*) 5 years later (2009). Distance between arrows: 23 cm.

At the initial date, and every September thereafter until 2009, four variables were estimated in each plot for later calculation of soil erosion indicators: (1) rock surface, (2) bare surface, (3) distance from the soil erosion edge to the visible painted line and (4) length of the visible painted line.

Rock surface (and bare surface) was defined as the proportion of the plot with rock (or rock plus bare soil), which had no litter or any kind of plant canopy above. Both variables were visually estimated as per cent cover values to the nearest 5%, except for low values (<10%), which were estimated to the nearest 1%. When cover was far less than 1%, a value of 0.1% was registered.

The distance from the soil erosion edge to the painted line (cm), and the length of the painted line (cm), were measured considering only the portions of the painted line which could be observed (i.e. not completely covered by fallen soil, litter, algae or mosses). To measure the average distance from the soil erosion edge to the visible painted line, the lines of each plot were partitioned into 25-cm segments. For each segment the distance was measured at a representative point; and at the first date only, the height of the edge was also measured.

Post-fire erosion assessment

In the 77 sites selected for the post-fire erosion assessment, erosion pins were used to estimate loss of surface soil (Hudson 1993). In September 2009 (2 months after the fire but before the onset of the rain season) 12–16 iron pins, each of 0.4 cm in diameter and 40–60 cm in length, were hammered into the soil surrounding each plot, leaving exposed 10 cm of the pin. Pins were 1–2 m apart. In September 2010, the length above the soil surface of all iron pins at each plot was measured, and then these values were averaged per plot.

Data analyses

Stocking rate and soil erosion

From the variables measured every year, four erosion indicators were calculated for each plot: (1) change rate of the rock surface, (2) change rate of the bare surface, (3) the advance rate of erosion edges and (4) the persistence of activity in erosion edges. For simplicity these indicators will hereafter be named as ‘rock surface change’, ‘bare surface change’, ‘edge advance’ and ‘activity persistence’, respectively.

For all the 200 plots, the rock surface change was calculated as the slope of a linear regression between cover percentage of rock surface and time (% year^{-1}). The same procedure was applied to estimate the bare surface change (% year^{-1}). The data of 2009 for 22 plots, which were burned during that year, were discarded. Because the initial date was also variable among plots, for 121 plots the slopes were calculated from six dates (i.e. a 5-year period from 2004 to 2009), for 58 plots the slopes were calculated for a 4-year period (from 2005–09 or from 2004–08) and for the remaining 21 plots the slopes were calculated for a 3-year period (from 2005–08 or from 2006–09). Rock surface and bare surface change are two closely related indicators and, in both cases, positive values indicate soil or plant degradation (i.e. increase in rock surface or bare surface), while negative values indicate soil or plant recovery (i.e. decrease in rock surface or bare surface).

Edge advance was estimated by calculating the rate of change of the distance between the visible painted line and the soil edge as the slope of the linear regression between distance and time (cm year^{-1}, $n = 92$). For this indicator, values were mostly positive, because at first date the average distance values were close to zero for most plots, and only positive or nil distances were possible to measure in the following years. However, some small negative slopes were obtained, due to fluctuations in temporary soil deposition on the rock and to measurement errors. After calculating edge advance (cm year^{-1}), to satisfy statistical assumptions, the data were ln-transformed after adding 0.5 to eliminate negative values. For this indicator, high values mean high soil erosion, and low values, low soil erosion.

Edge activity persistence was estimated by first calculating the change rate in the length of the visible painted line as the slope between time and the line length (cm year^{-1}). As an edge becomes less active, algae and mosses colonise the bare rock. Fallen soil, from the erosion edge and plant litter, tends to stay for long periods on the rock, and can be later colonised by vegetation. Because of these processes, the line is progressively covered, and the visible length decreases. Conversely, in fully active edges, fallen soil and litter remain only for a short time on the rock, and the visible length of the line tends to remain constant. In consequence, most slopes were negative or close to zero, although, due to similar reasons as above, some positive values were obtained. Because the initial length of the line was very variable among plots (35–1213 cm), the slope values, expressed in cm year^{-1}, were expressed as a proportion of the initial length and expressed as a percentage (% year^{-1}). For this indicator, more negative values indicate less activity, while close to zero or slightly positive values indicate that the edge remains fully active (i.e. higher activity persistence). The co-variation of the four soil erosion indicators (rock surface change, bare surface change, and edge advance and activity persistence) were described by performing pair-wise correlations.

For each paddock, the stocking rate (CU ha^{-1}) of each month from September 2004 to August 2009 was calculated from detailed data provided by National Park administration and land owners. From the monthly data for each plot (according to the paddock in which it was situated), the annual stocking rate, the winter stocking rate (from July to September, the driest period of the year) and the summer stocking rate (from January to March, the wettest period of the year) were calculated considering the monitoring period of each plot. Thus, plots in the same paddock may have slightly different stocking rates if monitored during different time periods. From data on winter and summer stocking rates a ‘seasonality index’ for each plot was calculated as:

$$\mathrm{SI} = (\mathrm{WSR} - \mathrm{SSR})/(\mathrm{WSR} + \mathrm{SSR} + 0.01)$$

where SI is the seasonality index, WSR is the winter stocking rate (CU ha^{-1}), and SSR is the summer stocking rate (CU ha^{-1}). In this way, plots in paddocks which were used only in the winter months, had a positive value close to 1, plots under a continuous stocking rate or permanent exclusion had a value close to 0, and plots in paddocks used only in summer, had a negative value close to –1.

The influence of stocking rate, season of grazing, and other independent variables on the four erosion indicators were determined using multiple regression. One multiple regression was performed for each of the four erosion indicators. For rock surface change and bare surface change, the independent variables used were: stocking rate, seasonality index, the initial value of the parameter (rock surface or bare surface), and the interaction term between the initial value and stocking rate. These variables/terms were included after a preliminary visual examination of scatter-plots. Additionally, other possible explanatory landscape and site variables were considered. Included as landscape variables were: altitude (m a.s.l.), rock surrounding the plot (%), landscape roughness (dimensionless), landscape slope (%), run-on contributing area (number of 30×30-m cells, square-root transformed), and relative position along the flow path (% in relation to the total flow path between the drainage divide and the closest drainage channel). As site variables, a local insolation dimensionless index of the plot (obtained from plot slope aspect and inclination, Cingolani *et al*. 2010), local slope inclination (%) and soil depth (cm) were used. For the other two indicators, viz. edge advance and edge activity persistence, stocking rate, the seasonality index, the edge initial height the initial bare surface of the plot (or alternatively the initial rock surface), and the site and landscape variables referred to previously, were used as independent variables.

In all cases, the best set of significant variables or terms were selected by means of a forward stepwise regression. After each regression model was obtained, the normality of residuals and their independence in relation to the paddock where they were situated were tested. All statistical analyses were performed by the InfoStat program (Di Rienzo *et al*. 2010).

Post-fire erosion assessment

Post-fire surface erosion was estimated for each of the 77 plots by calculating the difference in the average length of the iron pin above the soil between the final and the initial date (September 2010 and September 2009, respectively). A two-factor ANOVA (first confirming statistical assumptions), was used to test the combined effect of fire (two treatments: burned and unburned) and post-fire livestock grazing (two treatments: grazed and ungrazed) on loss of soil.

Results

Stocking rate and soil erosion

All pair-wise correlations among the four erosion indicators were positive (Table 1). As expected, rock surface change and bare surface change were highly and significantly correlated, and most of the other pair-wise correlations were also significant. Plots where bare surface and bare rock increased, had edges with more

Table 1. Pair-wise Pearson correlation coefficients (r) among the four erosion indicators

** $P \leq 0.01$; * $P \leq 0.05$

	Rock surface change (% year^{-1})	Bare surface change (% year^{-1})	Edge advance (cm year^{-1})^{A}
Bare surface change (% year^{-1})	0.61**	–	–
Edge advance (cm year^{-1})^{A}	0.28*	0.09	–
Activity persistence (% year^{-1})	0.31*	0.43**	0.15

^{A}Ln-transformed after adding 0.5 to the original values.

persistent activity and which advanced faster than plots where the proportion of bare rock and bare surface declined.

Change in rock surface varied from –4.8% year^{-1} to 4.2% year^{-1}. Variables explaining these changes were stocking rate, the initial rock surface cover and the relative position along the flow path. Plots with high initial rock surface recovered vegetation and/or soil when stocking rate was low or nil, but increased the rock surface when stocking rate was high. Plots with low initial rock surface remained almost constant (Fig. 3*a*). Additionally, rock surface tended to increase in upper positions (i.e. closer to catchment divides) and to decrease at lower positions (i.e. close to drainage channels). The regression model explained only 20% of the variance (Table 2).

Changes in the bare surface varied from –6.7% year^{-1} to 3.4% year^{-1}. The stocking rate of the dry season, the initial cover of bare surface, altitude above sea level, the proportion of rock in the surrounding landscape, and the relative position along the flow path explained these changes (Fig. 3*b*, Table 2). Plots with a high initial bare surface recovered vegetation when stocking rate was low or nil, while under high stocking rates bare surfaces tended to increase. Plots with a low initial bare surface showed no changes (Fig. 3*b*). Additionally, the results showed that, at the lowest altitudes, close to drainage channels and on non-rocky landscapes, vegetation recovered faster than at high altitudes, close to catchment divides and in rocky landscapes. The model explained 22% of variance (Table 2).

Edge advance varied between –0.25 and 7.1 cm year^{-1} (and between –1.42 and 2.03 for the ln-transformed variable). The advance was explained by stocking rate, the average initial height of the edge, and the initial bare surface of the plot. Edges in plots under high stocking rates advanced faster than edges in plots under exclusion or low stocking rates. Additionally, the highest edges in plots with a high initial bare surface advanced faster than shallower edges with a low initial bare surface (Fig. 3*c*, Table 2). This model explained 17% of the variance.

The persistence of edge activity varied between –37% year^{-1} and 13% year^{-1}. Activity was less persistent under exclusion from grazing or low stocking rates than under high stocking rates (Fig. 3*d*). Additionally, at the highest altitudes on rocky landscapes and in sites close to catchment divides, activity persistence was greater than at low altitudes on less rocky sites, and in sites close to drainage channels. This model explained 32% of the variance (Table 2).

The seasonality index, the roughness index, the run-on contributing area, landscape slope, localised slope, the insolation index and soil depth were not selected for any model. For all regression models, residuals were independent of paddock, and exhibited normal or near-normal distributions.

Post-fire erosion assessment

Loss of surface soil was significantly higher in burned than in unburned plots ($F = 49.3$, $P \leq 0.001$) and in grazed than in ungrazed plots ($F = 5.3$, $P \leq 0.05$). No interaction between fire and livestock grazing was detected. The effect of burning was stronger than the effect of grazing (Fig. 4). Sites with no burning and no grazing gained some soil (i.e. the length of the pin was shorter at the end than at the start of the measurement period, evidenced in a negative soil loss of 0.07 cm, Fig. 4), while

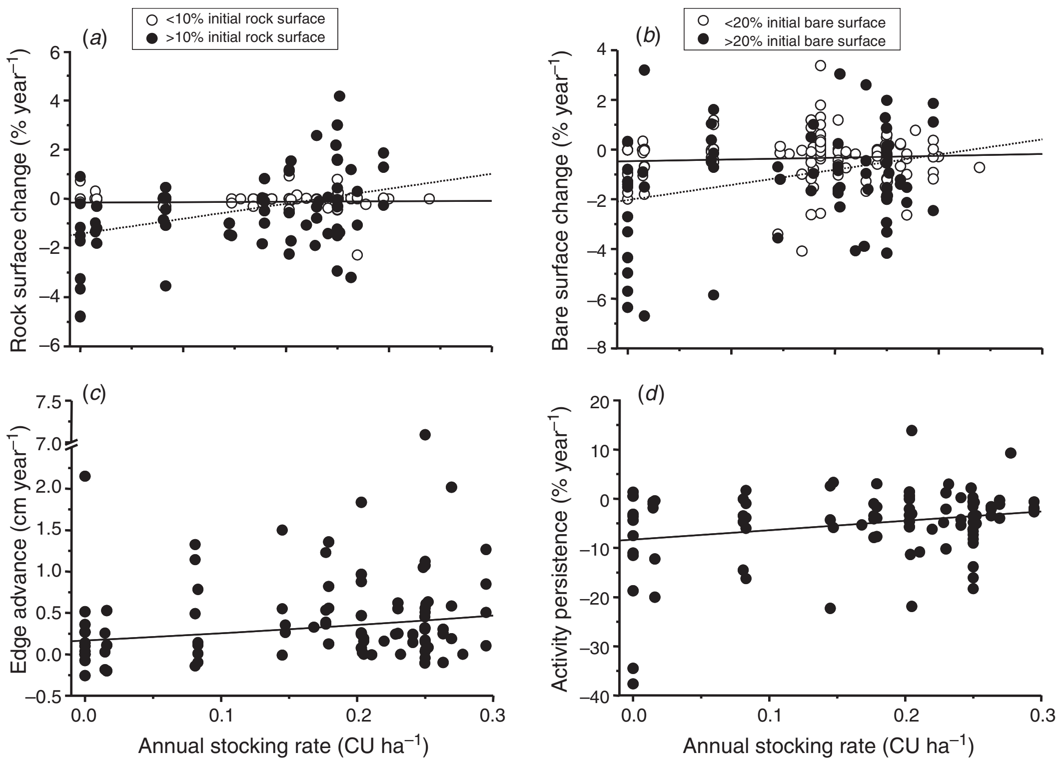

Fig. 3. Scatter-plots of the four erosion indicators in relation to annual stocking rate, and regression lines (solid and dotted). (*a*) Rock surface, (*b*) bare surface, (*c*) edge advance, using the original, non-transformed values, and (*d*) edge activity persistence. For (*a*) and (*b*), to illustrate the interaction between stocking rate and initial rock (or bare) surface, plots were arbitrarily classified in those with low and high initial rock (or bare) surface (black and white circles, respectively). Regression lines were drawn based on the regression equations of Table 2, considering non-plotted variables as constant values. For (*a*) and (*b*), solid lines represent the regression for plots with low initial cover, and dotted lines the regression for plots with high initial cover. For (*c*) the equation was de-transformed to show the relationship considering the original values.

Table 2. Selected regression models for the four erosion indicators

All variables included in the models were statistically significant in the regression analyses ($P \leq 0.05$)

Erosion indicator	Equation[A]	r^2
Rock surface change	–0.506 – 0.046 × (initial rock surface) + 0.214 × (initial rock surface) × (stocking rate) + 0.0052 × (relative position along the flow path)	0.20
Bare surface change	–7.307 – 0.046 × (initial bare surface) + 0.159 × (initial bare surface) × (stocking rate) + 0.0027 × (altitude) + 0.016 × (landscape rock) + 0.0096 × (relative position along the flow path)	0.22
Edge advance[B]	–0.77 + 1.25 × (stocking rate) + 0.025 × (edge height) + 0.0080 × (initial bare surface)	0.17
Activity persistence	–70.93 + 19.15 × (stocking rate) + 0.026 × (altitude) + 0.117 × (landscape rock) + 0.048 × (relative position along the flow path)	0.32

[A]$P \leq 0.05$ for all variables and terms in the models.

[B]Ln-transformed after adding 0.5 cm to the original values.

unburned grazed sites lost very little soil (less than 0.05 cm on average). Burned sites lost ~0.6 and 0.4 cm for grazed and ungrazed sites, respectively.

Discussion

The results support the hypothesis that livestock grazing and fire accelerate soil loss in the upper altitudinal belt of the Córdoba mountains. These results are in line with several studies in different ecosystems, which have found rates of increased soil erosion as a consequence of high stocking rates of livestock (Belsky and Blumenthal 1997; Mwendera *et al.* 1997; Ludwig *et al.* 2005; Neff *et al.* 2005) or after the total or partial elimination of vegetation by fire (Inbar *et al.* 1998; Vermeire *et al.* 2005), and suggest that livestock rearing practices may convert a sub-tropical mountain rangeland into a rocky desert.

Stocking rate and soil erosion

In this study, soil erosion caused the replacement of vegetated surfaces by rocky surfaces at rates which could be roughly estimated for the entire study region. Applying the regression

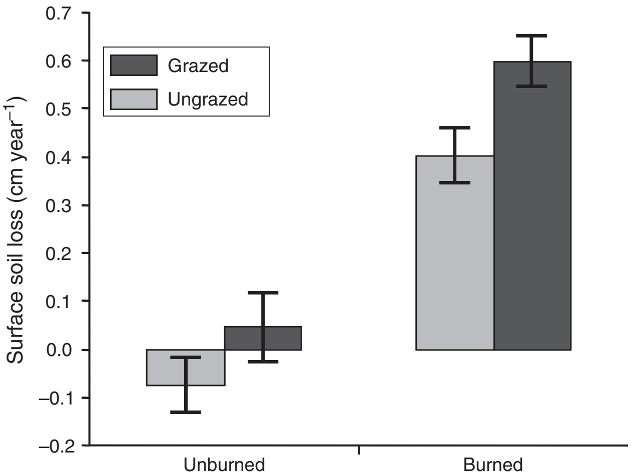

Fig. 4. Surface soil loss, estimated, using erosion iron pins, as the difference between the final pin height and the initial exposed pin height, for burned and unburned plots, with and without post-fire livestock grazing.

equation for edge advance rate, and considering a stocking rate of 0.34 CU ha^{-1}, it can be estimated that edge advance rates were 0.41–1.87 cm year^{-1} for the eight vegetation units described for the area, on the basis of unit's average values of height and bare surface (Cingolani *et al*. 2004; and A. M. Cingolani, unpubl. data, Appendix 1). Stocking rates of 0.34 CU ha^{-1} are fairly common in privately owned properties outside the National Park, and were the maximum stocking rates included in this study. The estimated edge advance rate of each unit was multiplied by the average length of their soil-rock edges (from Cingolani *et al*. 2004; and A. M. Cingolani, unpubl. data, Appendix 1) to calculate the average replacement rate of vegetated surface by bare rock, (which varied from 0.3 to 8.6 m^2 ha^{-1} year^{-1}). Finally, using a weighting factor of the proportional area of each unit in the landscape (Cingolani *et al*. 2004), an average replacement rate of 4.4 m^2 ha^{-1} year^{-1} was calculated for the whole region. Since domestic European livestock have been present in the area for around four centuries, this value could be scaled to ~1700 m^2 ha^{-1} in 400 years, or 18% of the rangeland surface. This figure is consistent with the presumption made previously (Cingolani *et al.* 2008*b*) that 20% of the area has lost its soil after the introduction of European livestock, even though the assumptions of scaling-up are to some extent questionable because the parameters used to estimate the replacement rate (e.g. the length of erosion edges in the landscape) are not static, and the process might not be linear. Visually estimated changes in rock and bare surfaces are also consistent with this degree of loss, showing a trend towards the loss of plant and soil cover as stocking rates increase. The loss of soil and plant cover is a common outcome of excessive livestock pressure (e.g. Chartier and Rostagno 2006), but the exposition of large surfaces of bedrock after the total loss of the soil profile seems to be a particular characteristic of the study area.

The replacement of large vegetated surfaces by bare rock evidences how biotic processes, such as herbivory by livestock, can exert a strong influence on landscape configuration. Traditionally, physical factors have been assumed as independent factors influencing the distribution and abundance of organisms (Reinhardt *et al*. 2010). For example, some studies have reported how plants and/or animals respond to the proportion and size of rock fragments on the surface, which were assumed to be the result of geomorphologic processes independent of life (e.g. Funes and Cabido 1995; Ferreyra *et al*. 1998; Anchorena and Cingolani 2002; Milchunas and Noy-Meir 2002). However, more recent work shows that organisms can modify the physical environment in several ways, generating feedbacks between biotic and abiotic processes (Reinhardt *et al*. 2010). Livestock in particular has been considered as an important agent of geomorphological change (Trimble and Mendel 1995). For example, in arid lands of central Australia, Tongway *et al*. (2003) reported changes in the geomorphic stratum as a result of erosion produced by livestock pressure. These changes in turn influence plant composition, which can alter the behaviour of grazers and change local grazing pressures (Adler *et al*. 2001; Cingolani *et al*. 2008*a*). In this study, livestock is a relatively new factor introduced in an ecosystem where native large herbivores were extinct at the end of the Pleistocene (Pucheta *et al*. 1998*a*) or in low numbers due to pre-historic hunting by local inhabitants during the Holocene (Medina and Rivero 2007). In this context, the introduction of livestock has represented a strong impact, breaking the limits of resilience of the system, causing important structural changes, even when the system is to some extent adapted to herbivory (Díaz *et al*. 1992; Cingolani *et al*. 2005, 2008*b*; Renison *et al*. 2010).

Livestock exclusion or very low stocking rates resulted in a partial inactivation of edges and a much slower advance of the portions which remained active. This suggests that the advance of edges can stop almost completely under lengthy periods of livestock exclusion, at least for edges up to 41 cm high as were the ones studied. Additionally, a fast retraction of rock and bare surface was detected. In some plots, this retraction was caused by the growth of a woody plant canopy, particularly of the tree *Polylepis australis* (A. M. Cingolani, unpubl. data). Individuals of this species are often maintained as small shrubs due to heavy consumption by livestock in the dry season (Giorgis *et al*. 2010). Exclusion of livestock results in a rapid increase in both height and width of the canopy which covers the bare rock (Teich *et al*. 2005; Renison *et al*. 2006; Giorgis *et al*. 2010).

Another cause of rock surface retraction, which was noted visually, was the partial covering of bare rock by pioneer life forms, such as mosses and algae, or by litter and sediment accumulation followed by colonisation of annual plants. Although this does not imply that the soil profile with all its properties will be restored in the short-term, it is an indication that removal of livestock can be an adequate measure to stop and even to partially reverse the erosion process, and favour sediment accumulation and soil formation. Full restoration is almost impossible once the soil has been partially or totally lost and the system has passed a threshold towards a new state (Westoby *et al*. 1989; Cingolani *et al*. 2005; Chartier and Rostagno 2006). This has been demonstrated in different rangelands where soils do not recover all their properties after decades of livestock exclusion (Neff *et al*. 2005; and references in Trimble and Mendel 1995). However, in agreement with the results of this study, various studies in strongly degraded ecosystems have shown that some plant and soil properties can be partially recovered by removing livestock (Yong-Zhong *et al*. 2005; Castellano and Valone 2007; Pei *et al*. 2008).

Deep gullies were not included in this study, as most of them have soil in their base, and only edge advance on soil-rock edges was measured. Nevertheless, soil-soil edges advance with similar mechanisms to soil-rock edges, and the results indicate that edge advance rate increases with edge height. This suggests that deep gullies advance faster than shallow edges, resulting in the loss of large volumes of soil. Edge height was not selected to explain activity persistence but observations in livestock-excluded areas suggest that the removal of livestock is not enough to stop the advance of large gullies which are several metres deep, at least in the short and medium term (Cabido *et al*. 1987; Landi and Renison 2010). In these cases, besides exclusion of livestock, it would be necessary to adopt more active measures to reduce water concentration, such as the construction of stone terraces and planting of native vegetation (Landi and Renison 2010). In the study area, soil erosion has been quickly stopped by transplanting grass tussocks (mainly *Poa stuckertii*) to totally cover the previously smoothed vertical soil edges (D. Renison, pers. comm.). This has proven to be an effective but very labour-intensive technique, which is limited by the availability of tussocks to transplant. As an example, 15 years of livestock exclusion in an area of 45 ha provided enough tussocks to stop erosion only in 180 m of erosion edges, a very low proportion of the edges present in the area (D. Renison, unpubl. data).

The effect of stocking rate on rock and bare surface change was dependent on initial values of these variables. When rock or bare surfaces were low, the plot remained stable for all the studied stocking rates (see Fig. 3*a, b*, dotted lines). When initial rock or bare surface was high, plots responded to livestock removal by a decrease in these parameter values of these variables, and to higher stocking rates with an increase. This means that, when plants cover the entire surface, the system can resist moderate to high stocking rates of up to 0.34 CU ha^{-1} without losing soil stability and/or plant cover, at least in the short term. These results agree with studies in other ecosystems, which highlight the role of vegetation in protecting soil from degradation and erosion (Trimble and Mendel 1995; Davenport *et al*. 1998; Lal 2001; Pimentel 2006). Conversely, when the plot had some portion of uncovered surface, it was more susceptible to further losses under higher stocking rates. The distance to the painted line was a much more precise measure than visual estimation to detect the replacement of vegetated soils by rock, and confirmed that livestock accelerates the advance of edges in sites with a high initial bare surface.

All these results indicate the presence of a positive feedback. When part of the area lacks vegetation cover, either due to degradation or to natural causes (i.e. natural outcrops), the bare soil, or the contact line between soil and rock, are fragile points where, it is suggested, livestock trampling, combined with weathering agents, result in the detachment and transport of soil particles. This process is different from the negative feedback described for some arid or semiarid ecosystems, where selective surface erosion is a self-limiting process because it produces an increasing proportion of surface stones, which protects the soil from further erosion (review in Trimble and Mendel 1995; Descroix *et al*. 2001).

The position along the flow path also influenced the erosion indicators. At upper positions, close to the catchment divides, erosion tended to persist more actively and recovery was slower than at lower positions. This result agrees with previous studies in the area showing higher presence of active rock-soil edges and larger areas of exposed rock at upper topographic positions (Cingolani *et al*. 2003, 2008*b*). The coarser texture of soils at upper positions probably facilitates particle detachment, and the low soil water and nutrient content slows plant growth (Morgan 1979; Cingolani *et al*. 2003). Bare surface change and edge activity persistence were additionally explained by rock in the landscape and altitude. These results indicate that lateral growth of the plant canopy, or colonisation of bare soil and erosion edges by seedlings is easier at lower altitudes, probably due to less limiting temperatures (Marcora *et al*. 2008); and in less rocky landscapes, probably due to higher soil stability in absence of rock-soil limits. Additionally, at lower altitudes, the detachment produced by frosts and thaw cycles is probably less pronounced.

Management of domestic livestock, and not complete exclusion, has been highly recommended for the restoration of degraded rangelands in a variety of ecosystems (Papanastasis 2009). However, the results of this study showed that season of grazing did not have a large effect on the rate of soil erosion, suggesting that erosion processes may be better controlled with exclusion of grazing or very low stocking rates, rather than with any particular type of grazing management. Since only the 28% of the study plots were actually subjected to seasonal stocking, before completely ruling out a possible influence of season of grazing, a more detailed study with a more balanced design would be necessary.

As is usual in erosion studies, a large proportion of variance in erosion indicators remained unexplained (Renschler and Harbor 2002). Different reasons can be contributing, including the heterogeneus distribution of animals within the paddocks (von Müller *et al*. 2012), the different ages of exclosures, the different surrounding vegetation, interactions between stocking rate and topographic variables, which were not modelled, and measurement errors, particularly in the case of visual estimations. Further research would be necessary to better understand these mechanisms.

Post-fire erosion assessment

As predicted, the elimination of plant cover by fire resulted in surface erosion, which was accentuated when the burned sites were later stocked. Burning vegetation is a common management practice in this study area and in other mountain areas of South America (Kessler and Driesch 1993; Renison *et al*. 2002). By burning vegetation at the end of the dry season, livestock owners stimulate green re-growth of tussock grasses to compensate for a shortage of forage. An important long-term consequence of this practice is the generation and maintenance of short swards, or grazing lawns – communities with relatively low productivity but high forage quality – which attract livestock. On the contrary, tussock grasslands, particularly those dominated by *Poa stuckertii*, are highly productive communities with a low nutritive value and strongly avoided by livestock (Pucheta *et al*. 1998*a*, 1998*b*; Cingolani *et al*. 2008*b*; Vaieretti *et al*. 2010; von Müller 2011). Grazing lawns can be partially maintained by livestock, because animals sometimes consume small seedlings of *Poa stuckertii* growing in the lawn. However, livestock at moderate to high stocking rates, such as those studied here, can slow down, but not completely stop, the advance of *Poa* tussocks (A. M. Cingolani, unpubl. data). It seems that, only when combined with burning, can livestock effectively create and maintain grazing lawns. Judging by the large areas of lawns maintained in private lands (Cingolani *et al*. 2003, 2004), and from frequent observations of burning practices, fire frequency was probably relatively high during the whole 400-year history of livestock rearing in the area. Though the studied fire was accidental, it was similar to fires used for management in two senses: first, it burned in the dry season, and second, it also left the soil exposed to livestock trampling and to erosion agents (wind and raindrop impacts). Putting all this evidence together, it seems probable that the cost of maintaining high forage quality lawns is a progressive loss of soil depth after each cycle of tussock burning. This facilitates the formation of soil-rock edges, and finally the replacement of lawns by rocky areas (Cingolani *et al*. 2008*b*).

Management implications

It is concluded that the present-day combination of livestock and fire management has the potential to convert this sub-tropical mountain rangeland into a rocky desert, indicating that commercial livestock production, as it is carried on at present, is not a sustainable activity. A similar conclusion was obtained by Cáceres (2009) who performed a sustainability analysis of the same socio-ecological system during 5 years (2002–06) and observed a decay in the ecological sustainability index. Livestock and fire exclusion are adequate measures to slow down erosion and prevent further losses, or even to partially recover soils and vegetation in the base of shallow edges. The results of this study also indicated that soils in sites without rock cover, with no susceptible border areas, are more resistant to erosion.

A sustainable future of the livestock industry would depend on the reduction of erosion processes. Based in the conclusions of this study, it is suggested that different livestock management strategies according to the characteristics of the landscape need to be applied. For areas with low rock cover, which occupy 28% of the rangeland (Cingolani *et al*. 2004), it is suggested that moderate annual stocking rates (from 0.18 to 0.23 CU ha^{-1}) should be applied. For areas with rocky outcrops but without active erosion processes, which occupy 42% of the area, a recommendation is made for low stocking rates (<0.18 CU ha^{-1}). In both cases, stocking rates should be calculated on the basis of non-rocky surfaces. Permanent monitoring of vegetation and erosion processes in paddocks would be indispensable. Finally, for the 30% of the area, which has a large proportion of exposed rock and shows very active erosion processes, complete livestock exclusion for at least 5 years is recommended. After the exclusion period, paddocks should be lightly stocked and a monitored to avoid a reactivation of the erosion processes. To reduce soil erosion more rapidly, particularly when edges are large and deep, the complementary application of more active restoration techniques is suggested (Landi and Renison 2010). The use of fire as a management tool is not recommended until more detailed studies to evaluate the effects of different burning strategies have been carried out.

As the above suggested management scenarios involve stocking rates similar or below present-day commercial livestock production in areas with no erosion, and total livestock exclusions for at least 5 years in areas with soil erosion, we have to conclude that economically we find no 'win-win' solution managing the area with European-type livestock. Thus to reduce present-day soil erosion and avoiding the area being completely converted into a 'rock desert' we have to accept that profits have to be reduced. Thus implementing these policies will need a large commitment from land managers or state compensation like the 'paying for water' policies adopted in other countries (i.e. in Mexico, Muñoz-Piña *et al*. 2008).

Acknowledgements

We are grateful to CONICET (6196/05, 112-200801-01458) and 'Ministerio de Ciencia y Técnica de Córdoba' who funded this study, and to the Quebrada del Condorito National Park personnel, for logistic support. O. Martinez, V. Saur, I. Barberá, A. von Müller, M. Poca, P. Tecco, D. E. Gurvich, V. Falczuk and L. Hoyos helped in the field work. The Associate Editor, allocated this paper, and three anonymous reviewers helped to improve the manuscript. Authors are researchers and fellows of CONICET.

References

Adler, P., Raff, D., and Lauenroth, W. (2001). The effect of grazing on the spatial heterogeneity of vegetation. *Oecologia* **128**, 465–479. doi:10.1007/s004420100737

Anchorena, J., and Cingolani, A. M. (2002). Identifying habitat types in a disturbed area of the forest-steppe ecotone of Patagonia. *Plant Ecology* **158**, 97–112. doi:10.1023/A:1014768822737

Ares, J. O. (2007). Systems valuing of natural capital and investment in extensive pastoral systems: lessons from the Patagonian case. *Ecological Economics* **62**, 162–173. doi:10.1016/j.ecolecon.2006.06.001

Asner, G. P., Elmore, A. J., Olander, L. P., Martin, R. E., and Harris, A. T. (2004). Grazing systems, ecosystem responses and global change. *Annual Review of Environment and Resources* **29**, 261–299. doi:10.1146/annurev.energy.29.062403.102142

Belsky, A. J., and Blumenthal, D. M. (1997). Effects of livestock grazing on stand dynamics and soils in upland forests of the interior west. *Conservation Biology* **11**, 315–327. doi:10.1046/j.1523-1739.1997.95405.x

Cabido, M. (1985). Las comunidades vegetales de la Pampa de Achala, Sierras de Córdoba, Argentina. (The plant communities of Pampa de Achala, Córdoba Mountains, Argentina.). *Documents Phytosocioliques* **9**, 431–443.

Cabido, M., and Acosta, A. (1986). Variabilidad florística a lo largo de un gradiente de degradación de céspedes de la pampa de Achala, Sierras de Córdoba, Argentina. (Floristic variability along a lawn degradation gradient in Pampa de Achala, Córdoba Mountains, Argentina.). *Documents Phytosociologiques* **10**, 289–304.

Cabido, M., Breimer, R., and Vega, G. (1987). Plant communities and associated soil types in a high plateau of the Córdoba mountains, central Argentina. *Mountain Research and Development* **7**, 25–42. doi:10.2307/3673322

Cáceres, D. M. (2009). Sustainability of peasants' farms located in a natural reserve in central Argentina. *Agrociencia* **43**, 539–550.

Castellano, M. J., and Valone, T. J. (2007). Livestock, soil compaction and water infiltration rate: evaluating a potential desertification recovery mechanism. *Journal of Arid Environments* **71**, 97–108. doi:10.1016/j.jaridenv.2007.03.009

Chartier, M. P., and Rostagno, C. M. (2006). Soil erosion thresholds and alternative states in North-eastern Patagonian rangelands. *Rangeland Ecology and Management* **59**, 616–624. doi:10.2111/06-009R.1

Chartier, M. P., Rostagno, C. M., and Roig, F. A. (2009). Soil erosion rates in rangelands of north-eastern Patagonia: a dendrogeomorphological analysis using exposed shrub roots. *Geomorphology* **106**, 344–351. doi:10.1016/j.geomorph.2008.11.015

Cingolani, A. M., Cabido, M. R., Renison, D., and Solís Neffa, V. (2003). Combined effects of environment and grazing on vegetation structure in Argentine granite grasslands. *Journal of Vegetation Science* **14**, 223–232. doi:10.1111/j.1654-1103.2003.tb02147.x

Cingolani, A. M., Renison, D., Zak, M. R., and Cabido, M. (2004). Mapping vegetation in a heterogeneous mountain rangeland using landsat data: an alternative method to define and classify land-cover units. *Remote Sensing of Environment* **92**, 84–97. doi:10.1016/j.rse.2004.05.008

Cingolani, A. M., Noy-Meir, I., and Díaz, S. (2005). Grazing effects on rangeland diversity: diversity-intensity and state and transition models. *Ecological Applications* **15**, 757–773. doi:10.1890/03-5272

Cingolani, A. M., Noy-Meir, I., Renison, D., and Cabido, M. (2008*a*). La ganadería extensiva, ¿es compatible con la conservación de la biodiversidad y de los suelos? (Is extensive livestock production compatible with biodiversity and soil conservation?). *Ecología Austral* **18**, 253–271.

Cingolani, A. M., Renison, D., Tecco, P., Gurvich, D. E., and Cabido, M. (2008*b*). Predicting cover types in a mountain range with long evolutionary grazing history: a GIS approach. *Journal of Biogeography* **35**, 538–551. doi:10.1111/j.1365-2699.2007.01807.x

Cingolani, A. M., Vaieretti, M. V., Gurvich, D. E., Giorgis, M. A., and Cabido, M. (2010). Predicting alpha, beta and gamma plant diversity from physiognomic and physical indicators as a tool for ecosystem monitoring. *Biological Conservation* **143**, 2570–2577. doi:10.1016/j.biocon.2010.06.026

Colladon, L. (2010). 'Anuario pluviométrico 1992–2010. Cuenca del Río San Antonio. Sistema del Río Suquía – Provincia de Córdoba.' ('Annual ploviometry 1992–2010 – San Antonio river. Suquía river system – Córdoba Province.') (Instituto Nacional del Agua y del Ambiente (INAA) y Centro de Investigaciones de la Región Semiárida (CIRSA): Córdoba, Argentina.)

Davenport, D. W., Breshears, D. O., Wilcox, B. P., and Allen, C. D. (1998). Viewpoint: sustainability of pinon-juniper ecosystems – a unifying perspective of soil erosion thresholds. *Journal of Range Management* **51**, 231–240. doi:10.2307/4003212

Descroix, L., Viramontes, D., Vauclin, M., Gonzalez Barrios, J. L., and Esteves, M. (2001). Influence of soil surface features and vegetation on run-off and erosion in the Western Sierra Madre (Durango, Northwest Mexico). *Catena* **43**, 115–135. doi:10.1016/S0341-8162(00)00124-7

Di Rienzo, J. A., Casanoves, F., Balzarini, M. G., Gonzalez, L., Tablada, M., and Robledo, C. V. (2010). 'InfoStat versión 2010.' (Grupo InfoStat, FCA, Universidad Nacional de Córdoba: Argentina.) Available at: www.infostat.com.ar [Verified 29 April 2013]

Diamond, J. (2005). 'Collapse. How Societies Choose to Fail or Succeed.' (Penguin Group: New York.)

Díaz, S., Acosta, A., and Cabido, M. (1992). Morphological analysis of herbaceous communities under different grazing regime. *Journal of Vegetation Science* **3**, 689–696. doi:10.2307/3235837

Díaz, S., Acosta, A., and Cabido, M. (1994). Community structure in montane grasslands of central Argentina in relation to land use. *Journal of Vegetation Science* **5**, 483–488. doi:10.2307/3235974

Enrico, L., Funes, G., and Cabido, M. (2004). Regeneration of *Polylepis australis* Bitt. in the mountains of central Argentina. *Forest Ecology and Management* **190**, 301–309. doi:10.1016/j.foreco.2003.10.020

Ferreyra, M., Cingolani, A. M., Ezcurra, C., and Bran, D. (1998). High-Andean vegetation and environmental gradients in north-western Patagonia, Argentina. *Journal of Vegetation Science* **9**, 307–316. doi:10.2307/3237095

Flores, C. E., Cingolani, A. M., von Muller, A. R., and Barri, F. R. (2012). Habitat selection by reintroduced guanacos (*Lama guanicoe*) in a heterogeneous mountain rangeland of central Argentina. *The Rangeland Journal* **34**, 439–445. doi:10.1071/RJ12040

Funes, G., and Cabido, M. (1995). Variabilidad local y regional de la vegetación rupícola de las Sierras Grandes de Córdoba, Argentina. (Local and regional variability of outcrop vegetation in the Córdoba Mountains, Argentina.). *Kurtziana* **24**, 173–188.

Giorgis, M. A., Cingolani, A. M., Teich, I., and Renison, R. (2010). Do *Polylepis australis* trees tolerate herbivory? Seasonal patterns of biomass production and its consumption by livestock. *Plant Ecology* **207**, 307–319. doi:10.1007/s11258-009-9674-4

Hudson, N. W. (1993). 'Field Measurement of Soil Erosion and Run-off.' FAO Soils Bulletin. (FAO: Rome.)

Inbar, M., Tamir, M., and Wittenberg, L. (1998). Run-off and erosion processes after a forest fire in Mount Carmel, a Mediterranean area. *Geomorphology* **24**, 17–33. doi:10.1016/S0169-555X(97)00098-6

Jain, M. K., and Kothyari, U. C. (2000). Estimation of soil erosion and sediment yield using GIS. *Hydrological Sciences* **45**, 771–786.

Johansen, M. P., Hakonson, T. E., and Breshears, D. D. (2001). Post-pre run-off and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. *Hydrological Processes* **15**, 2953–2965. doi:10.1002/hyp.384

Kessler, M., and Driesch, P. (1993). Causas e historia de la destrucción de bosques Altoandinos en Bolivia. (Causes and history of the Andean forests destruction in Bolivia.) *Ecología en Bolivia* **21**, 1–18.

Lal, R. (2001). Soil degradation by erosion. *Land Degradation and Development* **12**, 519–539. doi:10.1002/ldr.472

Landi, M. A., and Renison, D. (2010). Forestación con *Polylepis australis* en suelos erosionados de las Sierras Grandes de Córdoba: evaluación del uso de terrazas y vegetación nodriza. (Forestation with *Polylepis australis* in eroded soils of the Sierras Grandes of Córdoba: evaluation of the use of terraces and nurse vegetation.). *Ecología Austral* **20**, 47–55.

Lang, A. (2003). Phases of soil erosion-derived colluviation in the loess hills of South Germany. *Catena* **51**, 209–221. doi:10.1016/S0341-8162(02)00166-2

Ludwig, J. A., Wilcox, B. P., Breshears, D. D., Tongway, D. J., and Imenson, A. C. (2005). Vegetation patches and runoff erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology* **86**, 288–297. doi:10.1890/03-0569

Marcora, P., Hensen, I., Renison, D., Seltmann, P., and Wesche, K. (2008). The performance of *Polylepis australis* trees along their entire altitudinal range: implications of climate change for their conservation. *Diversity & Distributions* **14**, 630–636. doi:10.1111/j.1472-4642.2007.00455.x

McNaughton, S. J. (1984). Grazing lawns: animals in herds, plant form and coevolution. *American Naturalist* **124**, 863–886. doi:10.1086/284321

Medina, M. (2006). Análisis zooarqueológico del sitio agroalfarero Puesto La Esquina 1 (Pampa de Olaen, Córdoba.) [Zooarchaeology of the site Puesto La Esquina 1 (Olaen Pampa, Córdoba.)]. *Anales de Arqueología y Etnología de la Universidad Nacional de Cuyo* **61–62**, 107–121.

Medina, M., and Rivero, D. (2007). Zooarqueología, *Lama guanicoe* y dinámica evolutiva del Chaco Serrano. (Zooarchaeology, *Lama guanicoe* and evolutionary dynamics of Mountain Chaco.) *Mundo de Antes* **5**, 211–234.

Milchunas, D. G., and Noy-Meir, I. (2002). Grazing refuges, external avoidance of herbivory and plant diversity. *Oikos* **99**, 113–130. doi:10.1034/j.1600-0706.2002.990112.x

Morgan R. P. C. (1979) 'Soil Erosion: Topics in Applied Geography.' (Longman Inc.: New York.)

Morris, S. E., and Moses, T. A. (1987). Forest fire and the natural soil erosion regime in the Colorado Front Range. *Annals of the Association of American Geographers* **77**, 245–254. doi:10.1111/j.1467-8306.1987.tb00156.x

Muñoz-Piña, C., Guevara, A., Torres, J. M., and Braña, J. (2008). Paying for the hydrological services of Mexico's forests: analysis, negotiations and results. *Ecological Economics* **65**, 725–736. doi:10.1016/j.ecolecon.2007.07.031

Mwendera, E. J., Saleem, M. A. M., and Dibabe, A. (1997). The effect of livestock grazing on surface run-off and soil erosion from sloping pasture lands in the Ethiopian highlands. *Australian Journal of Experimental Agriculture* **37**, 421–430. doi:10.1071/EA96145

Neff, J. C., Reynolds, R. L., Belnap, J., and Lamothe, P. (2005). Multi-decadal impacts of grazing on soil physical and biogeochemical properties in South-east Utah. *Ecological Applications* **15**, 87–95. doi:10.1890/04-0268

O'Hara, S. L., Street-Perrott, F. A., and Burt, T. P. (1993). Accelerated soil erosion around a Mexican mountain lake caused by pre-hispanic agriculture. *Nature* **362**, 48–51. doi:10.1038/362048a0

Papanastasis, V. P. (2009). Restoration of degraded grazing lands through grazing management: can it work? *Restoration Ecology* **17**, 441–445. doi:10.1111/j.1526-100X.2009.00567.x

Pastor, S. (2000). 'Historia aborigen de las Sierras de Córdoba.' (Indigenous history of the Córdoba Mountains.) (Fondo editorial de la municipalidad de Villa Carlos Paz: Carlos Paz, Argentina.)

Pausas, J. G., Llovet, J., Rodrigo, A., and Vallejo, R. (2008). Are wildfires a disaster in the Mediterranean basin? – A review. *International Journal of Wildland Fire* **17**, 713–723. doi:10.1071/WF07151

Pei, S., Fu, H., and Wan, C. (2008). Changes in soil properties and vegetation following exclosure and grazing in degraded Alxa desert steppe of Inner Mongolia, China. *Agriculture, Ecosystems & Environment* **124**, 33–39. doi:10.1016/j.agee.2007.08.008

Perevolotsky, A., and Seligman, N. G. (1998). Role of grazing in Mediterranean rangeland ecosystems. *Bioscience* **48**, 1007–1017. doi:10.2307/1313457

Pfeiffer, K. E., and Steuter, A. A. (1994). Preliminary response of Sandhills prairie to fire and bison grazing. *Journal of Range Management* **47**, 395–397. doi:10.2307/4002337

Pimentel, D. (2006). Soil erosion: a food and environmental threat. *Environment, Development and Sustainability* **8**, 119–137. doi:10.1007/s10668-005-1262-8

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Sphpritz, L., Fitton, L., Saffouri, R., and Blair, R. (1995). Environmental and economic costs of soil erosion and conservation benefits. *Science* **267**, 1117–1123. doi:10.1126/science.267.5201.1117

Pimentel, D., Hepperly, P., Hanson, J., Douds, D., and Seidel, R. (2005). Environmental, energetic, and economic comparisons of organic and conventional farming systems. *Bioscience* **55**, 573–582. doi:10.1641/0006-3568(2005)055[0573:EEAECO]2.0.CO;2

Pucheta, E., Cabido, M., Díaz, S., and Funes, G. (1998*a*). Floristic composition, biomass, and above-ground net plant production in grazed and protected sites in a mountain grassland of central Argentina. *Acta Oecologica* **19**, 97–105. doi:10.1016/S1146-609X(98)80013-1

Pucheta, E., Vendramini, F., Cabido, M., and Díaz, S. (1998*b*). Estructura y funcionamiento de un pastizal de montaña bajo pastoreo y su respuesta luego de su exclusión. (Structure and functioning of a mountain grassland under grazing and its response to exclusion.). *Revista de Agronomía de La Plata* **103**, 77–92.

Reganold, J. P., Elliott, L. F., and Unger, Y. L. (1987). Long-term effects of organic and conventional farming on soil erosion. *Nature* **330**, 370–372. doi:10.1038/330370a0

Reinhardt, L., Jerolmack, D., Cardinale, B. J., Vanacker, V., and Wright, J. (2010). Dynamic interactions of life and its landscape: feedbacks at the interface of geomorphology and ecology. *Earth Surface Processes and Landforms* **35**, 78–101. doi:10.1002/esp.1912

Renison, D., Cingolani, A. M., and Suarez, R. (2002). Efectos del fuego sobre un bosquecillo de *Polylepis australis* (Rosaceae) en las montañas de Córdoba, Argentina. (Fire effects on a *Polylepis australis* (Rosaceae) woodand in Córdoba Mountains, Argentina.). *Revista Chilena de Historia Natural (Valparaiso, Chile)* **75**, 719–727. doi:10.4067/S0716-078X2002000400007

Renison, D., Hensen, I., Suarez, R., and Cingolani, A. M. (2006). Cover of Polylepis woodlands and shrublands in the mountains of central Argentina: human or environmental influence? *Journal of Biogeography* **33**, 876–887.

Renison, D., Hensen, I., Cingolani, A. M., Marcora, P., and Giorgis, M. A. (2010). Soil conservation in *Polylepis* mountain forests of Central Argentina: is livestock reducing our natural capital? *Austral Ecology* **35**, 435–443. doi:10.1111/j.1442-9993.2009.02055.x

Renschler, C. S., and Harbor, J. (2002). Soil erosion assessment tools from point to regional scales – the role of geomorphologists in land management research and implementation. *Geomorphology* **47**, 189–209. doi:10.1016/S0169-555X(02)00082-X

Singh, G., Babu, R., Narain, P., Bhushan, L. S., and Abrol, I. P. (1992). Soil erosion rates in India. *Journal of Soil and Water Conservation* **41**, 97–99.

Teich, I., Cingolani, A. M., Renison, D., Hensen, I., and Giorgis, M. A. (2005). Do domestic herbivores retard *Polylepis australis* woodland recovery in the mountains of Córdoba, Argentina? *Forest Ecology and Management* **219**, 229–241. doi:10.1016/j.foreco.2005.08.048

Tongway, D. J., Sparrow, A. D., and Friedel, M. H. (2003). Degradation and recovery processes in arid grazing lands in central Australia: Part 1. Soil and land resources. *Journal of Arid Environments* **55**, 301–326. doi:10.1016/S0140-1963(03)00025-9

Trimble, S. W., and Mendel, A. C. (1995). The cow as a geomorphic agent – a critical review. *Geomorphology* **13**, 233–253. doi:10.1016/0169-555X(95)00028-4

Vaieretti, M. V., Cingolani, A. M., Pérez Harguindeguy, N., Gurvich, D. E., and Cabido, M. (2010). Does decomposition of standard materials differ among grassland patches maintained by livestock? *Austral Ecology* **35**, 935–943. doi:10.1111/j.1442-9993.2009.02105.x

Vermeire, L. T., Wester, D. B., Mitchell, R. B., and Fuhlendorf, S. D. (2005). Fire and grazing effects on wind erosion, soil water content, and soil temperature. *Journal of Environmental Quality* **34**, 1559–1565. doi:10.2134/jeq2005.0006

von Müller, A. R. (2011). Selección de hábitat de herbívoros domésticos en las Sierras Grandes de Córdoba. (Habitat selection of domestic herbivores in the Córdoba Mountains.) PhD Thesis, Universidad Nacional de Córdoba, Facultad de Ciencias Exactas, Físicas y Naturales, Argentina.

von Müller, A. R., Cingolani, A. M., Vaieretti, M. V., and Renison, D. (2012). Estimación de carga bovina localizada a partir de frecuencia de deposiciones en un pastizal de montaña. (Estimation of localized cattle stocking rate from dung frequency in a mountain grassland.). *Ecología Austral* **22**, 178–187.

Westoby, M., Walker, B., and Noy-Meir, I. (1989). Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* **42**, 266–274. doi:10.2307/3899492

Wondzell, S. M., and King, J. G. (2003). Post-fire erosional processes in the Pacific North-west and Rocky Mountain regions. *Forest Ecology and Management* **178**, 75–87. doi:10.1016/S0378-1127(03)00054-9

Yong-Zhong, S., Yu-Lin, L., Jian-Yuan, C., and 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. doi:10.1016/j.catena.2004.09.001

Appendix 1. Vegetation units and their proportion in the study area (124 700 ha), average edge height, average bare cover, estimated edge advance rate, average edge length, and estimated replacement rate

In the last file, the replacement rate for the whole area, estimated by averaging the replacement rate for each unit weighed by the proportion of the units in the landscape

Unit[A]	Proportion (%)[B]	Edge height (cm)[C]	Bare cover (%)[B]	Advance rate (cm $year^{-1}$)[D]	Edge length (m ha^{-1})[C]	Replacement rate (m^2 ha^{-1} $year^{-1}$)[E]
1	2.5	18.8	6	0.69	46.4	0.32
2	9.4	18.1	23	0.84	218.2	1.83
3	4.0	11.9	1	0.46	93.2	0.43
4	20.0	7.9	8	0.42	380.6	1.60
5	3.9	11.6	6	0.49	484.3	2.39
6	30.5	17.3	49	1.11	365.5	4.07
7	24.5	21.7	74	1.70	506.9	8.62
8	5.2	19.1	91	1.87	443.5	8.28
				Weighted average		**4.39**

[A]Units: 1 Woodland, 2 Shrubby tussock grassland, 3 Thick tussock grassland, 4 Thin tussock grassland, 5 Lawn, 6 Outcrop with tussock grassland, 7 Outcrop with exposed rock, 8 Rock pavement.

[B]Obtained from Cingolani *et al*. (2004).

[C]From Cingolani *et al*. (unpubl. data).

[D] [Advance rate] = (e^(–0.77 + 1.25 × [stocking rate = 0.34] + 0.025 × [edge height] + 0.0080 × [initial bare surface])) – 0.5.

[E][Replacement rate] = [Advance rate] × [Average edge length]/100.