Soil Erosion Rates and Nutrient Loss in Rangelands of Southern Patagonia

Pablo L Peri[a,b], Romina G Lasagno[a], Marcelo Chartier[c], Fidel Roig[d,e], Yamina M Rosas[f], and Guillermo Martínez Pastur[f], [a]EEA INTA Santa Cruz, Grupo Forestal, Agricultura y Manejo de Agua (FAMA), Río Gallegos, Santa Cruz, Argentina; [b]Universidad Nacional de la Patagonia Austral (UNPA) – CONICET, Río Gallegos, Santa Cruz, Argentina; [c]Centro de Ecología y Recursos Naturales Renovables (CERNAR), Facultad de Ciencias Exactas, Físicas y Naturales, Universidad Nacional de Córdoba, Córdoba, Argentina; [d]Laboratorio de Dendrocronología e Historia Ambiental, IANIGLA-CONICET-Universidad Nacional de Cuyo, Mendoza, Argentina; [e]Hémera Centro de Observación de la Tierra, Escuela de Ingeniería Forestal, Facultad de Ciencias, Universidad Mayor, Santiago, Chile; [f]Centro Austral de Investigaciones Científicas (CADIC), Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET), Ushuaia, Tierra del Fuego, Argentina

Abstract

Soil erosion in rangelands is the main driver of desertification as a result of severe drought events and overgrazing reducing potential land productivity. The objectives of this chapter are to provide an overview of soil erosion as it relates to ecosystem services and to determine soil erosion rates from exposed roots of four shrub and dwarf-shrub species in nine sites of Southern Patagonia rangelands (Santa Cruz province, Argentina) as a case study. We highlight that soil protection is critical to sustain the capacity of rangeland ecosystems to supply provisioning (lamb and cattle meat, sheep wool) supporting (nutrient cycling, biodiversity, habitat) and regulating (carbon fixation, water flow regulation) ecosystem services for human well-being. We used a dendrogeomorphological method to determine soil erosion rates against datable exposed roots. Also, in each site soil samples were collected from nine randomly selected points in nondegraded patches to provide reference points from which to calculate loss of soil organic carbon and nutrients from erosion. The soil erosion rate in the degraded areas characterized by dwarf shrubs and shrubs with exposed roots was significantly different between sites and ranged from 1.6 to 4.1 mm year^{-1}. Soil mass loss rate ranged from 12.7 to 32.0 Mg ha^{-1} year^{-1} and soil carbon loss fluctuated from 85.3 to 250.1 kg C ha^{-1} year^{-1}. The main soil nutrient depleted during erosion processes was nitrogen (mean sites value of 17.9 kg N ha^{-1} year^{-1}) followed by potassium (mean of 9.2 kg K ha^{-1} year^{-1}) followed by phosphorus (mean of 0.6 kg P ha^{-1} year^{-1}). These results highlight the need for an early warning system by a soil erosion monitoring entity to prevent soil loss and prescribe sustainable management practices to maintain rangelands in an ecologically healthy state to conserve ecological functions and ecosystem service provision.

Introduction

Soil erosion is the most widespread form of land degradation in rangelands and the main driver of desertification in the world's drylands (Reynolds et al., 2007). This is a result of various factors, including climatic variations (e.g., severe drought events) and human activities (e.g., overgrazing of rangelands), that reduce potential land productivity (Hillel and Rosenzweig, 2002; Reynolds and Stafford Smith, 2002).

In Patagonia, the combination of extreme climatic conditions, low vegetation productivity, and overgrazing in dry steppe areas determine the highest values of desertification (Gaitán et al., 2019). In the region, there are more than 73.5 million ha that show different degrees of desertification (9.3% slight, 17.1% moderate, 35.4% moderate to severe, 23.3% severe, 8.5% very severe; del Valle et al., 1998), where annual pasture production in several areas does not exceed 40 kg DM ha^{-1}.

The vulnerability of dry steppes, where high desertification values generate soil removal, leads to biodiversity and ecosystem service supply losses (Del Valle et al., 1998; Peri et al., 2013; Rosas et al., 2018). Peri et al. (2018) reported that carbon stock in grasslands decreased under high stocking rates, mainly due to plant cover decrease and carbon loss from soil due to soil erosion by strong winds. Severe and frequent windstorms (wind speeds up to 100 km h^{-1}) that occur mainly during spring and summer often cause appreciable eolian sediment transport, especially on unprotected soils (Peri and Bloomberg, 2002). Wind erosion is the process whereby soil particles are lifted and carried away by the wind through creep, saltation, and suspension erosion processes. This process is influenced by the severity of climate and soil susceptibility (erodibility). When the wind removes soil particles together with organic matter and nutrients, land productivity is reduced (Zobeck and Fryrear, 1986).

In this context, because sustainable rangeland management depends mainly on soil conservation, measuring soil erosion becomes important to the management of land resources. However, there is a lack of information related to soil erosion rates in Southern Patagonia. For example, in Chubut province, Sterk et al. (2012) determined that storms with wind-speed peaks of 20 m s^{-1} caused a total soil loss of 248 Mg ha^{-1} in the control strip. In addition to the traditional methods for measuring soil erosion, dendrogeomorphological techniques determined soil erosion rates using datable exposed roots (Stoffel and Bollschweiler,

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2008). The vertical distance between the upper portion of the exposed stem-root interface of plants having annual wood ring structures and the actual soil surface can be used as an indicator of soil erosion since plant establishment (Chartier et al., 2009).

The objective of this chapter is to provide an overview of soil erosion as it relates to ecosystem services and to determine soil erosion rates from exposed roots of four shrub and dwarf-shrub species in nine sites of Southern Patagonia rangelands corresponding to severe and very severe desertification categories as a case study.

Soil erosion control as a regulating ecosystem services

The capacity of rangeland and agricultural ecosystems to provide ecosystem services (ES) for human well-being is strongly related to the condition of soils, their properties (physical, chemical, and biological characteristics), and their functions (Adhikari and Hartemink, 2016). While topsoil formation takes thousands of years, erosion can disintegrate all the organic matter and nutrients in the topsoil in a few years, dramatically reducing soil productivity. The extent of ecosystems' capacities to reduce erosion depends on many factors related to environmental conditions (amount of precipitation, wind velocity, soil properties, slope, and vegetation characteristics) and pressures (agricultural management practices and overgrazing) (Fig. 1). Many of those factors are taken into account in soil erosion models, such as the Universal Soil Loss Equation (USLE), to determine potential soil loss. Guerra et al. (2014) presented a modeling framework to assess actual soil erosion prevention that varies over time and space, mainly in regions at a high risk of desertification and soil degradation (Fig. 1).

Therefore soil protection is critical to sustain the capacity of rangeland ecosystems to supply provisioning ES (lamb and cattle meat, sheep wool), supporting ES (nutrient cycling, biodiversity, habitat), and regulating ES (carbon fixation, water flow regulation). Thus healthy soils, especially for agriculture and livestock production, contribute to a variety of ecosystem functions,

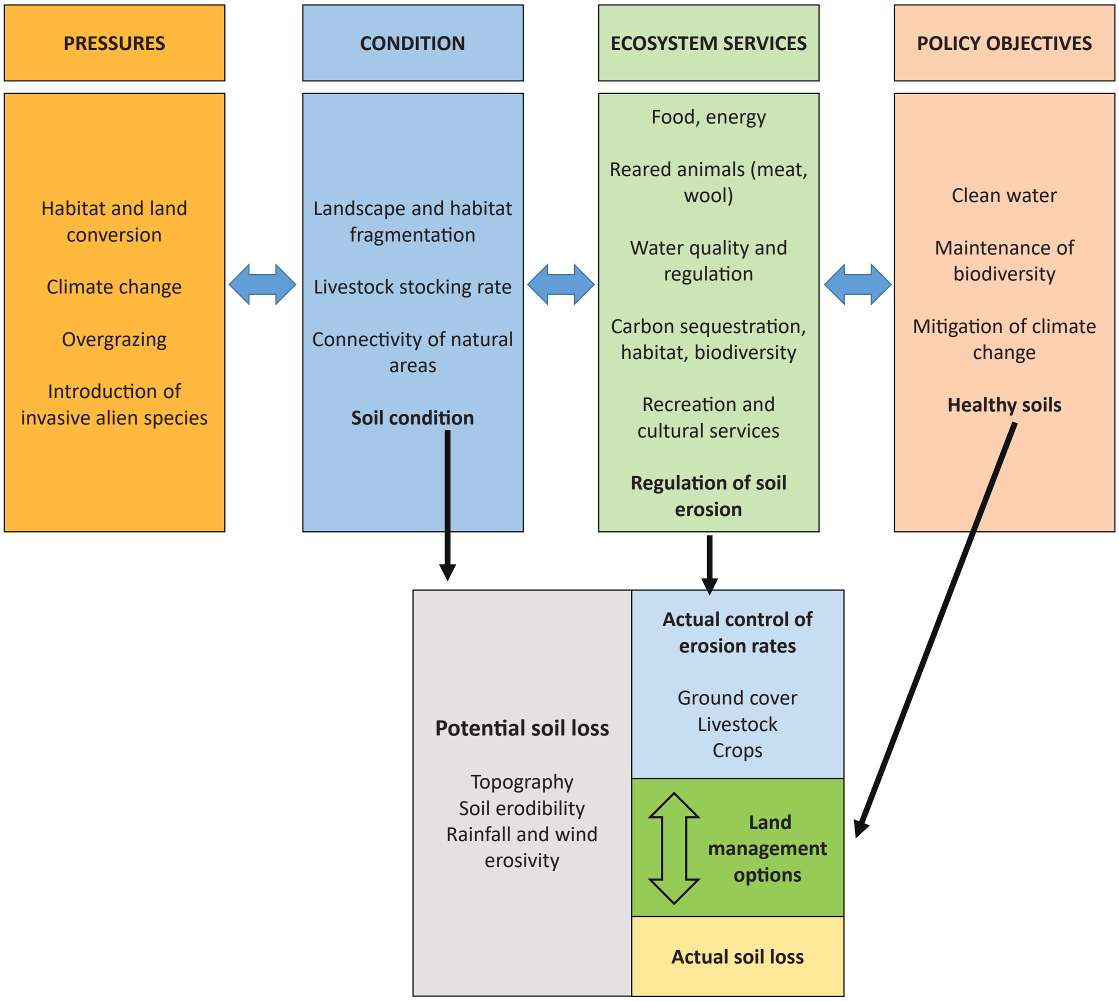

Fig. 1 Relationships between environmental pressures, condition, ecosystem services, and policy objectives in agroecosystems in the assessment of control of erosion rates. Adapted from Guerra C, Pinto-Correia T, and Metzger MJ (2014) Mapping soil erosion prevention using an ecosystem service modeling framework for integrated land management and policy. *Ecosystems* 17: 878–889.

including the ability to support biodiversity, land fertility, and the capacity to sustainably deliver multiple ES (Fig. 1). These services include food and fiber, climate and water regulation, water purification, carbon sequestration, nutrient cycling, and provision of habitat for biodiversity (Rinot et al., 2019). Water flow regulation and water filtration regulate ES related to soil, given that flow regulation and filtration hinge on the percolation rate and the absorption capacity of soils, which are determined by factors such as clay and organic material content and composition (Drobnik et al., 2018). These regulating ES related to soil-water storage capacity provide human benefits in the form of drought prevention, especially in arid and semiarid environments.

Declines in regulating provision of services like those related to soil can result in declines in ecosystem resilience, and affect the provision of other ES (Bennett et al., 2009). Healthy soils are important for biodiversity conservation and act as carbon storage pools. For example, Peri et al. (2019a) reported a strong trend between the number of threatened plant species and soil carbon in Southern Patagonia, and Rosas et al. (2018) showed that biodiversity among lizards in dry steppe habitats depends on the degree of desertification due to the human impacts (e.g., livestock). Soils are also the medium on which grassland plants are grown and their functionality is the basis for the biomass production that sustains animal provisioning ES. Thus degraded soils from heavy and unsustainable grazing conditions threaten the future of livestock productivity, and therefore the long-term well-being of the local economy. These impacts on farming development have driven sheep stocks to decline by half since the early 20th century, and left more than 500 farms abandoned (Coronato et al., 2015). In addition, soils preserve related cultural ES by storing geological and archeological heritage.

Soil erosion process as a threat to soils and mitigation actions

The main threats to soils that lead to soil degradation and negative impacts on the supply of soil-related ES include soil erosion, soil contamination, decline in soil organic matter, soil compaction, decrease in soil biodiversity, salinization, floods, and landslides. Soil erosion leads to soil degradation through loss of soil material and soil organic carbon and nutrients, resulting in a decrease in soil-related ES supply. Degraded soils with reduced biodiversity due to erosion processes drive a soil functions deterioration and consequent reduction of ES delivery. Overgrazing by livestock can affect soil-related ES by altering the structural condition of soils. For example, Peri (2011) reported that C stock in grasslands decreased from 130 Mg C ha^{-1} under low grazing intensity (0.10 ewe ha^{-1} year^{-1}) to 50 Mg C ha^{-1} at a heavy stocking rate (0.70 ewe ha^{-1} year^{-1}) mainly due to a decline in plant cover and C lost from soil (primarily from the organic layer in increasingly bare areas) as a consequence of soil erosion by strong winds.

Soil erosion has both local and regional impacts, which can often be mitigated by the adoption of suitable land management practices. Landscape management that reduces erosion risk and improves soil productivity directly translates into economic benefits generated by associated production systems (Alam et al., 2014). Thus soil conservation measures and best management practices could enhance yields and thereby ensure food security. For example, Lal (2004) estimated that improving soil quality with an increase of 1 t SOC ha^{-1} year^{-1} in the root zone can increase annual food production in developing countries by 24–32 million tons of food grains that could assist in achieving food security.

Land management decisions at the farm level such as those concerning stocking density of livestock, grazing pressure, and shrub control may mitigate soil erosion by maintaining adequate vegetation cover. Vegetation regulates soil erosion and therefore mitigates the impact that results from the combination of the erosive power of wind and/or precipitation and the biophysical conditions of a given area. Restoration of degraded drylands is urgently needed to mitigate climate change and reverse desertification. Restoration of degraded dryland ecosystems is frequently constrained by low and variable precipitation, low soil fertility, and a prevalence of invasive species. Passive restoration methods like reducing livestock grazing are often ineffective, as degraded dryland environments can exhibit stability and resilience in undesired states (Hoover et al., 2020). As an alternative, seeding is a widely used approach for dryland restoration, as it is a feasible strategy for reintroducing desired native species at large spatial scales (Kildisheva et al., 2016).

For policy makers, a specific understanding of the ES that soils provide can serve as a powerful tool to inform economically and environmentally beneficial management strategies. Healthy soils should be an important priority for national and regional policies, reflected in regulations that aim to sustainably secure and restore soil functions by protecting soils against harmful changes and remediating contaminated sites (Fig. 1). At the international level, recognizing the contribution of soils to ecosystem services, the Food and Agriculture Organization of the UN has established the Global Soil Partnership that advocates for and coordinates initiatives to ensure that soils are represented in global change dialogs and policy decisions.

1. Study case in Southern Patagonia: Quantifying the erosion rate

For this study, using the PEBANPA (Parcelas de Ecología y Biodiversidad de Ambientes Naturales en Patagonia Austral—Biodiversity and Ecological long-term plots in Southern Patagonia) network (Peri et al., 2016), we selected nine permanent plots across Santa Cruz province corresponding to severe and very severe desertification categories (Fig. 2). In the region, annual rainfall ranges from 800 to 1000 mm year^{-1} in the Andes Mountains (west) and decreases to 200 mm year^{-1} in the eastern steppe sector of Santa Cruz province. The mean annual precipitation to potential evapotranspiration ratio of the steppe fluctuates between 0.45 and 0.11, with marked soil water deficits in summer. The variations in local topographic and edaphic characteristics, combined with a significant precipitation gradient, substantially influence the grasslands' forage production. Mean annual temperatures range between 5.5 °C and 8.0 °C. The windiest season occurs between November and March, producing frequent and severe south-southwesterly windstorms sometimes reaching over 80 km h^{-1}. The main economic activities in the evaluated sites are related to extensive

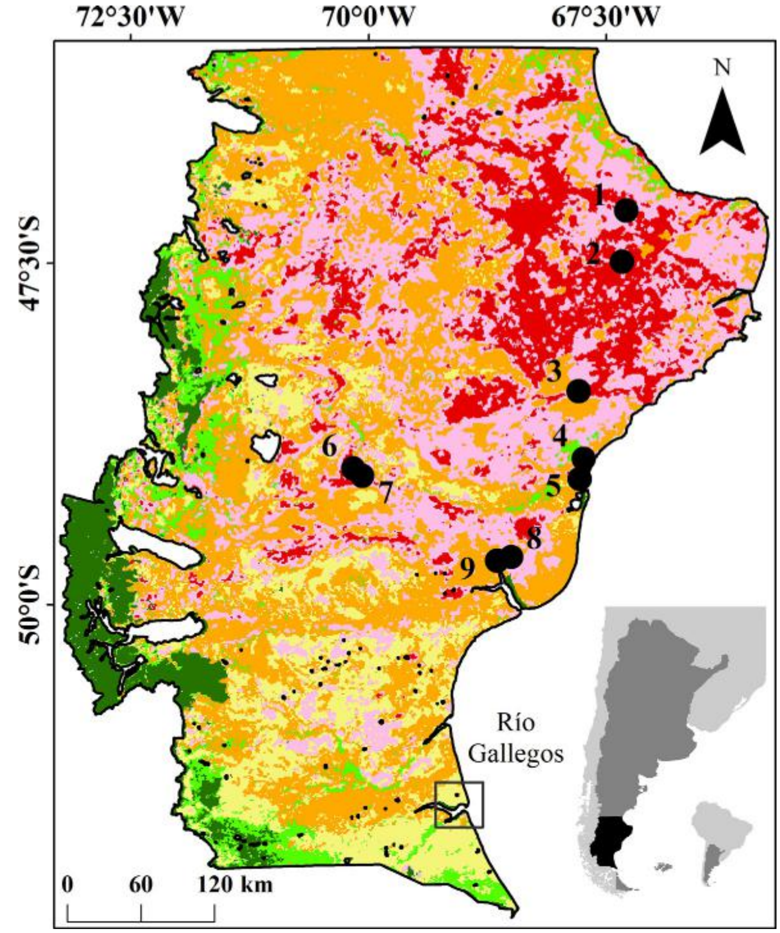

Fig. 2 Study area. Location (light gray = South America, dark gray = Argentina, black = Santa Cruz province). Sample sites are indicated by black dots. The desertification index is evidenced by: dark green = none; light green = slightly degraded; yellow = moderate desertification; orange = moderate to severe desertification; pink = severe desertification; red = very severe desertification. Modified from Del Valle HF, Elissalde NO, Gagliardini DA, and Milovich J (1998) Status of desertification in the Patagonian region: Assessment and mapping from satellite imagery. *Arid Land Research and Management* 12: 95–121.

sheep production, primarily of the Corriedale breed. The long-term intensity of grazing on each farm estimated from the mean sheep-stocking rates in the study area ranged from 0.12 ± 0.02 to 0.24 ± 0.10 ewe ha^{-1} $year^{-1}$.

All PEBANPA plots were permanently marked and assessed at least once during the flowering period (spring–summer) for accurate plant identification. At each sampling location, plant diversity was measured in a 20 m × 50 m quadrant (1000 m^2). This plot size enables regional comparisons of diversity-associated factors for the broad vegetation types (e.g., grasslands, shrublands, and forests). The species were classified as native, endemic, and exotic and by life-form (herb, graminoids, tussock grass, fern, shrub, dwarf shrub, tree); by life span (perennial, annual, biennial); and by location of the plant's growth point (meristem) based on the Raunkiær classification system (geophyte, chamaephyte, phanerophyte, hemicryptophyte, cryptophyte, therophyte). The bare eroded soil of the studied sites varied from 38% to 75%, and the vegetation of the grass-shrub steppe was dominated by grasses and sedges (*Bromus*, *Carex*, *Festuca gracillima*, *Hordeum*, *Jarava*, *Poa*, *Rytidosperma virescens*), dwarf shrubs and herbs (*Nardophyllum*, *Perezia*, *Azorella*, and *Nassauvia*), and shrubs (*Berberis*, *Colliguaja intergerrima*, *Chuquiraga*, *Mulguraea*, *Lycium*, *Schinus marchandii*) admixed. In the field, degraded areas within each site were identified by several signs of soil erosion: desert pavement, gravels in pedestals, and exposed roots of woody plants.

In each site, soil samples were collected from nine randomly selected points in nondegraded patches (reference areas without soil erosion) using a hand auger (30 cm depth). Samples from nondegraded patches were used to evaluate rates of soil organic carbon (SOC) and nutrient loss as a result of erosion. Coarse root debris >2 mm from soil samples was removed by sieving. To reduce the number of chemical analyses, we pooled individual soil samples into combined samples. From the nine samples collected within each quadrant we created three composite samples containing an equal proportion of soil from three auger holes ($n = 3$ for each site). The samples were finely ground to below 2 μm using a tungsten-carbide mill. Measurements of SOC concentration were derived from the dry combustion (induction furnace) method. Measurements of %N (nitrogen), %P (phosphorus), and %K (potassium) were made with a LECO auto-analyzer. Soil bulk density was estimated using the cylindrical core method (n = 3) (i.e., collecting a known volume of soil using a metal tube pressed into the soil (intact core), and determining the weight after drying).

The shrubs selected for growth ring and chronology measurements were *Schinus marchandii* Barkley, *Lycium ameghinoi* Speg, *Chuquiraga aurea* Skottsb., and *Mulguraea tridens* (Lag.) N.O'Leary & P.Peralta. Ten shrub plants with highly exposed roots were randomly selected in each site. For each selected plant, the distance between the upper root portion and the soil surface was measured with a ruler. Then the plants were harvested, put in plastic bags, and taken to the laboratory where they were transversely sectioned at the upper root portion (stem-root interface). The samples were processed according to conventional techniques used in dendrochronology (Stokes and Smiley, 1968). Growth rings were dated under a binocular magnifier, and the ring widths were measured to an accuracy of 0.001 mm. The ring widths of the selected shrub plants were cross-dated using the skeleton plot method to define pointer years (Schweingruber et al., 1990). The mean annual soil erosion rate was determined by dividing the heights of the exposed roots by the number of years each plant had lived.

We used the logistic model to describe the dynamic of the erosion process in degraded areas of the studied sites, as it was demonstrated by Chartier et al. (2009) to be the model that best reflects soil erosion rates in Patagonian rangelands.

Bulk soil densities, soil carbon, and nutrients in nondegraded patches are presented in Table 1. The mean length of all obtained chronologies ranged from 39 years for *Chuquiraga aurea* (1974–2013) to 61 years for *Schinus marchandii* (1952–2013) based on the mean annual growth ring index of dwarf shrub and shrub plants. The mean ring width ranged from a minimum of 0.07 (*Lycium ameghinoi*) to a maximum of 0.46 mm year^{-1} (*Mulguraea tridens*).

The soil erosion rate in the degraded areas (Fig. 3) characterized by dwarf shrubs and shrubs with exposed roots was significantly different between sites and ranged from 1.6 to 4.1 mm year^{-1} (Table 1). This is consistent with Chartier et al. (2009) who estimated soil erosion rates of 2.4 and 3.1 mm year^{-1} from exposed roots of the dwarf shrub *Margyricarpus pinnatus* in two ecological sites in the northeastern rangelands of Patagonia. Kliment'ev and Tikhonov (2001) reported that erosion under anthropic conditions might reach 2.0 mm year^{-1}, which significantly exceeds the rates of topsoil formation and natural erosion.

To estimate annual soil erosion mass and nutrient loss per hectare, we used the percentage of effective eroded soil in each site and the soil bulk densities; carbon and nutrient content measurements; and soil erosion rates from Table 1. The loss of soil mass, soil carbon, and nutrients from erosion varied across evaluated sites (Fig. 4). Soil mass loss rate ranged from 12.7 (site 7) to 32.0 Mg ha^{-1} year^{-1} (site 9).

In Chartier et al.'s study (2009), the erosion rate in degraded patches (10% of surface cover) was equivalent to 28.8 and 38.4 Mg ha^{-1} year^{-1} of sediment in the pediment-like plateau and the flank pediment, respectively. In an arid closed basin of northeastern Patagonia, Coronato and del Valle (1993) reported by using the universal soil loss equation a maximum value of 11.3 Mg ha^{-1} year^{-1} from the estimate of fluvial erosion. Sterk et al. (2012) by measuring eolian mass fluxes in the valley of Sarmiento (Chubut province, Argentina) reported a value of 248 Mg ha^{-1} of total soil loss in storms with wind speed that peaked at 20 m s^{-1} in the control strip, and heavily depleted the soil of its erodible fraction.

According to our findings, soil carbon loss fluctuated from 85.3 (site 1) to 250.1 kg C ha^{-1} year^{-1} (site 3). The soil nutrient most depleted by erosion was nitrogen (mean sites value of 17.9 kg N ha^{-1} year^{-1}) followed by potassium (mean of 9.2 kg K ha^{-1} year^{-1}) and phosphorus (mean of 0.6 kg P ha^{-1} year^{-1}) (Fig. 4). Soil nitrogen loss rate varied from 6.4 (site 7) to 35.2 kg N ha^{-1} year^{-1} (site 9); phosphorus loss from 0.3 (site 3) to 1.4 kg P ha^{-1} year^{-1} (site 6); and potassium loss from 3.3 (site 1) to 23.7 kg K ha^{-1} year^{-1} (site 9) (Fig. 4). In a regional study, Peri et al. (2018) determined that increased long-term animal stocking rates decreased soil organic content (SOC, 0–30 cm) values with the desertification gradient (from 10.6 without desertification presence to 4.4 kg C m^{-2} at sites where desertification was pronounced) due to erosion processes in sites with low vegetation cover (or high bare soil cover). Similarly, Peri et al. (2019b) reported that total soil nitrogen stocks (0–30 cm) decreased with desertification from 0.89 kg N m^{-2} at sites with little desertification to 0.32 kg N m^{-2} at sites where desertification was pronounced, probably reflecting erosion.

The differences in soil mass and nutrient losses between sites may be explained by potential differences in the susceptibility of soils to erosion, which depends on several interrelated factors such as climate, moisture availability, soil properties, topography, land cover, and management practices (Lal, 2001). The loss of soil mass, soil carbon, and nutrients described in the present study have probably been due to wind and water erosion processes, accelerated by heavy sheep grazing. Thus in Patagonia rangelands, as

Table 1 Range values of soil properties and plant parameters of nondegraded patches used to estimate the soil erosion rate at rangelands studied sites, Santa Cruz province, Southern Patagonia.

Site	*Soil bulk density (g cm^{-3})*	*Soil C (%)*	*Soil N (%)*	*Soil P (mg kg^{-1})*	*Soil K (cmol(+) kg^{-1})*	*Plant age (year)*	*Height of exposed root (mm)*	*Soil erosion rate (mm year^{-1})*
1	1.25	0.65	0.07	31	0.65	45–52	110–114	1.9–2.3
2	1.18	0.69	0.08	36	0.73	40–51	120–122	2.0–2.7
3	1.29	1.38	0.18	15	0.82	44–50	128–136	2.5–2.8
4	1.05	1.22	0.09	44	1.39	40–45	162–168	3.2–3.8
5	1.14	0.74	0.07	29	1.23	55–61	120–130	1.6–2.2
6	1.31	0.58	0.06	46	1.27	35–39	116–121	2.7–3.1
7	1.27	0.75	0.05	32	1.32	49–53	125–130	2.1–2.5
8	1.35	1.20	0.15	48	1.41	36–42	110–115	2.8–3.1
9	1.21	0.66	0.11	30	1.90	50–57	202–215	3.4–4.1

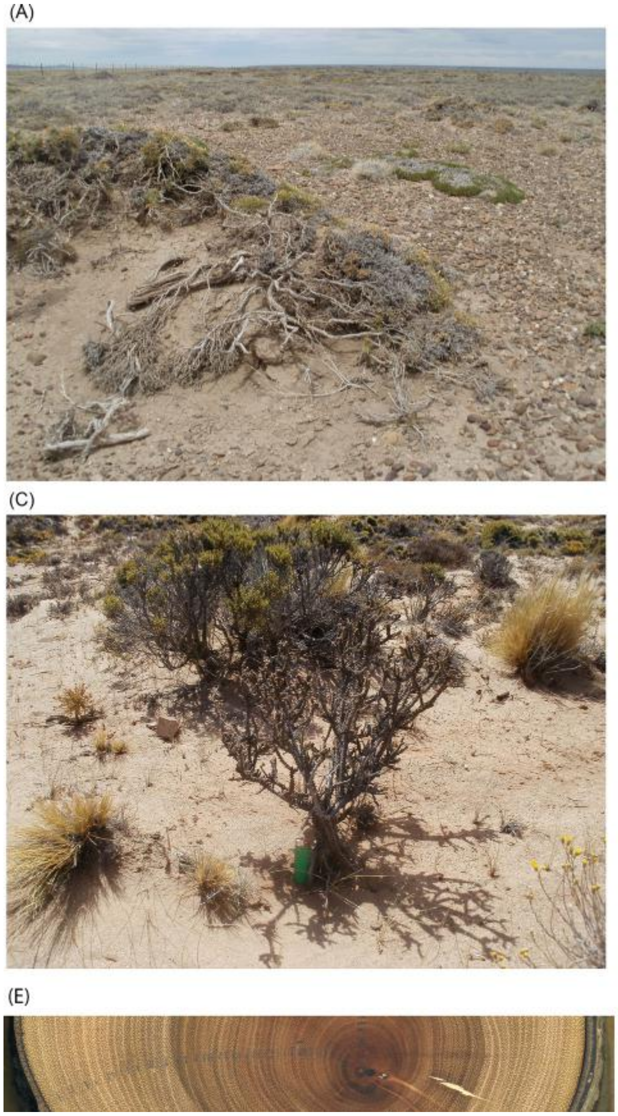

Fig. 3 Study areas (A) at site 7 and (B) site 6 near Gobernador Gregores city (see Fig. 2 for location), (C) site 3 north of San Julian city, (D) site 9 north of Piedra Buena city. (E) a wood transversal stem section of *Schinus marchandii* showing the annual rings with clearly discernible growth boundaries characteristic of this species. The vessels at the beginning of the growing season are larger and their cell walls thinner than those formed later (latewood) in the growing season.

in other arid and semiarid ecosystems, a decrease in the cover of perennial grasses due to overgrazing generally results in an acceleration of the soil erosion process (Chartier and Rostagno, 2006). Moreover, the most common form of land degradation in many dryland systems around the world is related to a relatively rapid change in the composition of the plant community, with a shift between grasses and woody plants (Van Auken, 2000). In our study areas, woody plants have been observed encroaching into degraded grasslands. Furthermore, the predicted increases in aridity and in the frequency of droughts in drylands globally, together with anthropogenic factors (e.g., overgrazing), indicate that there could be increasing land degradation through eolian soil erosion processes in future.

The soil loss rate estimated in our work was between 5.1 and 12.8 times greater than the soil loss tolerance value (*T*-value) of 2.5 Mg ha^{-1} year^{-1} based on the typical properties of root-limiting layers of subsurface soil at 0–25 cm depth (USDA-NRCS, 1999). Wischmeier and Smith (1978) defined *T*-value as "the maximum level of soil erosion that will permit a high level of crop productivity to be obtained economically and indefinitely." Because our results had a high *T*-value (severe erosion rates), the soil would not maintain productivity over time.

This situation brings a series of problems, such as decline of soil fertility and productivity, soil degradation, deterioration of grasslands, and reduction of ecosystem capacities to supply ES. Furthermore, soil erosion causes soil and nutrients loss and reduces depth of topsoil, which in turn reduces soil water storage capacity and soil fertility, and therefore grasslands yields. For example, Pimentel and Kounang (1998) reported that crop productivity is reduced by 20%–40% when water utilization efficiency of an agroecosystem decreases by 10%–25% due to soil erosion. At the global level, mean agricultural production loss reached 0.3% per year following soil erosion rate, but yields declined faster in more eroded regions (Biggelaar et al., 2003).

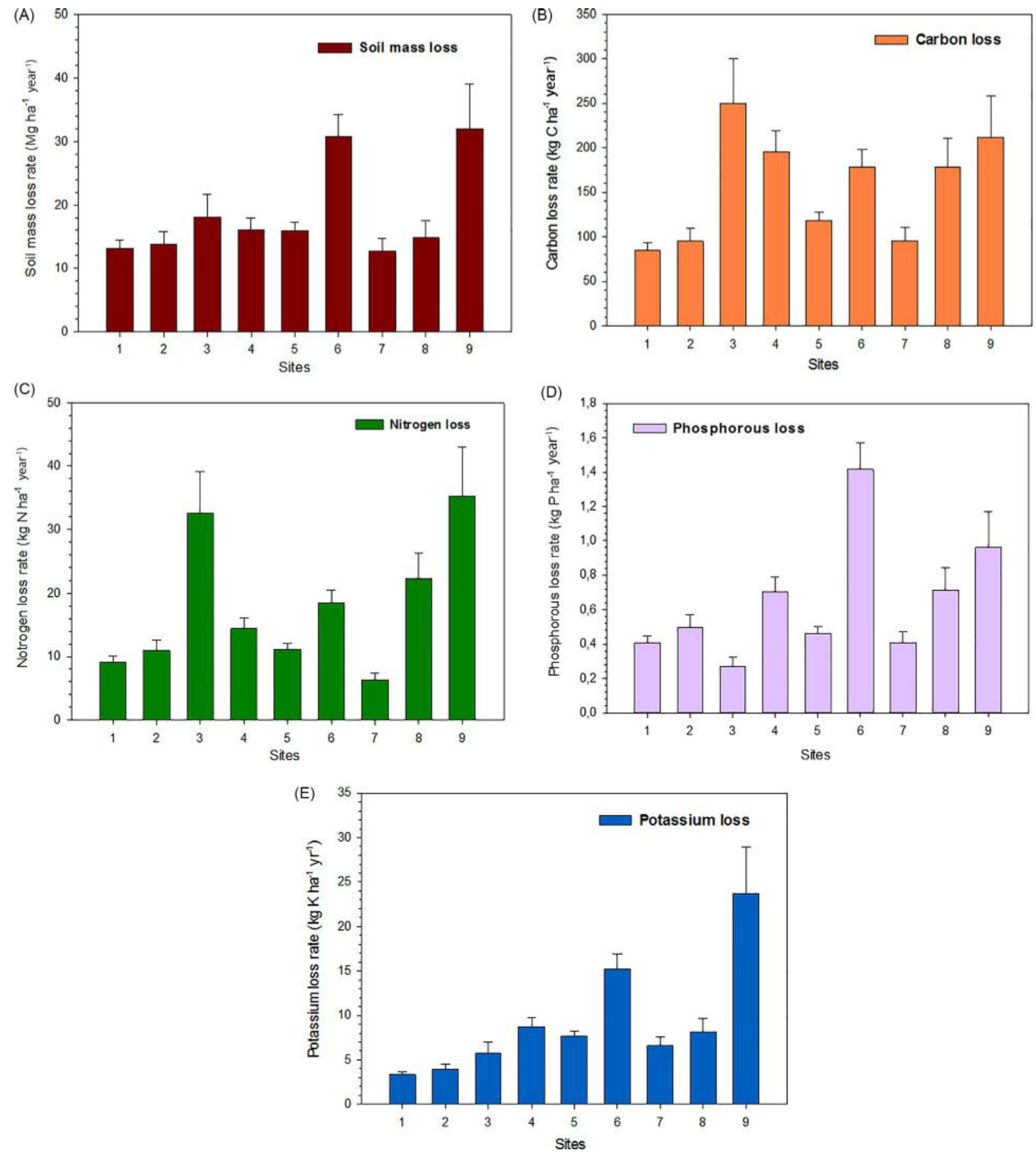

Fig. 4 Annual soil erosion mass, carbon, and nutrient loss rates per hectare in the evaluated sites (see Fig. 2 for locations) of Santa Cruz province, Patagonia, Argentina.

Given these dire circumstances, decision-makers and experts need to implement centralized soil-erosion monitoring and early-warning systems to prevent soil loss, prescribe sustainable management practices, and maintain rangelands in an ecologically healthy state. It is crucial that, in semiarid rangelands, management practices keep a high perennial grass and litter cover to avoid soil erosion and reduction of ecosystem productivity.

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