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The effect of natural infrastructure on water erosion mitigation in the Andes

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Abstract. Soil erosion by water is affecting natural and anthropogenic environments through its impacts on water quality and availability, loss of soil nutrients, flood risk, sedimentation in rivers and streams, and damage to civil infrastructure. Sustainable management aims to avoid, reduce and reverse soil erosion and can provide multiple benefits for the environment, population and livelihoods. We conducted a systematic review of 121 case-studies from the Andes to answer the following questions: (1) Which erosion indicators allow us to assess the effectiveness of natural infrastructure? (2) What is the overall impact of working with natural infrastructure on on-site and off-site erosion mitigation? and (3) Which locations and types of studies are needed to fill critical gaps in knowledge and research?

Three major categories of natural infrastructure were considered: protective vegetation, soil and water conservation measures, and adaptation measures that regulate flow and transport of water. From the suite of physical, chemical and biological indicators commonly used in soil erosion research, two indicators were particularly relevant: soil organic carbon (SOC) of topsoil, and soil loss rates at plot scale. In areas with protective vegetation and/or soil and water conservation measures, the SOC of topsoil is –on average– 1.3 to 2.8 times higher than in areas under traditional agriculture. Soil loss rates in areas with natural infrastructure were reported to be 38 % to 54 % lower than rates measured in untreated croplands. Further research is needed to evaluate whether the reported effectiveness holds during extreme events related to, for example, El Niño–Southern Oscillation.

1 Introduction

The Andes Mountains stretch about 8900 km and crosses tropical, subtropical, temperate and arid latitudes. Very few, if any, of the diverse physiographic, climatic and biogeographic regions in the Andes have been preserved from human impact. The area has been inhabited by humans for more than 15000 years (Jantz and Behling, 2012; Keating, 2007). By the mid-20[th] century, all Andean nations with exception of Argentina experienced an exponential population growth that caused substantial

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migration both within and between national borders (Little, 1981). More than 85 million people lived in the Andean region by 2020, with the Northern Andes being one of the most densely populated mountain regions in the world (Devenish and Gianella, 2012). The demographic growth and a stagnating agricultural productivity per hectare led to an expansion of the total agricultural land area, either upward to steep hillsides at high altitudes covered by native grassland-wetlands ecosystems, or downward to lands east and west of the Andes covered by tropical and subtropical forests (Wunder, 1996). Land abandonment is widespread where smallholders faced unfavorable economic conditions due to restricted land base, limited availability of farm credit and low productivity in fragile agro-ecological environments (Zimmerer, 1993).

The strong latitudinal gradients in climate and vegetation are reflected in the pronounced north-south gradient in natural erosion processes and rates (Latrubesse and Restrepo, 2014; Montgomery et al., 2001). Natural erosion rates are lowest (< 25 t km^{-2} yr^{-1}) in the hyper-arid and arid regions, but show high temporal variability as a result of extreme events, in particular during warm El Niño–Southern Oscillation (ENSO) conditions or earthquakes (Carretier et al., 2018; Morera et al., 2017). Erosion rates are usually higher (with rates of > 250 t km^{-2} yr^{-1}) in the humid regions where the catchments are deeply dissected by bedrock river channels, and where landslides are common (Blodgett and Isacks, 2007; Vanacker et al., 2020). Land use and management have significantly altered the magnitude and frequency of erosion events (Restrepo et al., 2015; Tolorza et al., 2014; Vanacker et al., 2007). Deforestation and agricultural practices (such as soil tillage and cattle grazing) increase erosion rates (Molina et al., 2007; Podwojewski et al., 2002), river sediment loads (Restrepo et al., 2015) and landslide occurrences (Guns and Vanacker, 2014). Changes in smallholders' livelihoods leading to the abandonment of agricultural land have a non-linear impact on soil erosion rates, as they are often associated with an initial increase in soil erosion, followed by a steady decrease in erosion rates on the long term (Harden, 2001).

To tackle soil erosion and mitigate its on-site and off-site effects, governmental and nongovernmental organizations in the Andean countries launched rural development and soil conservation programs in the 1970s and 1980s: for example, the programs by PRONAREG-MAG-ORSTOM and USAID in Ecuador (De Noni et al., 2001), IIDE and USAID in Bolivia (Zimmerer, 1993) and PRONAMACHCS in Peru (Torero Zegarra et al., 2010). The implementation of large-scale soil conservation and management programs and policies required considerable investments in labor and capital (Bilsborrow, 1992; Zimmerer, 1993). While their direct and indirect environmental benefits have been demonstrated on case-by-case basis (Farley and Bremer, 2017; Romero-Díaz et al., 2019), comprehensive evaluations of environmental programs rarely reach beyond case-by-case assessments (Bonnesoeur et al., 2019). For example, the PRONAMACHCS program of the Ministry of Agriculture of Peru promoted the implementation of a specific type of intervention, the infiltration trenches. They consist of dozens of earthen ditches dug over mountain slopes following contour lines with the objective of increasing water infiltration in the soils. They have been implemented in several catchments throughout the country for over three decades, before the impact of these practices was systematically assessed at the regional scale (Vásquez and Tapia, 2011). In a global systematic review, Locatelli et al. (2020) found that case-studies provide evidence that infiltration trenches are effective in reducing

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surface runoff and laminar erosion at plot scale, but they also highlight that their impacts on water infiltration are uncertain as well as their effects at catchment scale or on other erosion forms. There is an urgent need to identify which soil conservation and management practices are most effective to combat soil erosion and to mitigate its on-site and off-site effects in the Andean region.

Soil conservation measures are receiving renewed interest in the context of catchment management based on natural infrastructure. Natural infrastructure is defined by the IUCN as "services that nature provides such peatlands sequestering carbon, lakes storing large water supplies, and floodplains absorbing excess water runoff". These services perform as an infrastructure-like function, and working with natural infrastructure helps to protect, sustainably manage or restore ecosystems while simultaneously providing human well-being and biodiversity benefits (Cohen-Shacham et al., 2016). In the Andean context, three large groups of water-related interventions can be identified: (1) interventions based on land use and protective land cover including restoration and protection of native ecosystems such as montane forests, grasslands and forestation; (2) soil and water conservation measures including crop management, conservation tillage, and slow forming terraces, and the implementation of linear elements such as vegetation strips and check dams; and (3) interventions including elements of hydraulic regulation such as lakes, manmade reservoirs, and runoff diversion. Several studies have shown that working with natural infrastructure can help mitigate soil erosion and reduce risks to natural hazards (Cohen-Shacham et al., 2016).

To expand the knowledge base on natural infrastructure for erosion mitigation in the Andes, moving beyond case-by-case empirical studies to comprehensive assessments is needed (Bonnesoeur et al., 2019). This study reviews systematically the state of evidence on the effectiveness of interventions to mitigate soil erosion by water, and is based on Andean case studies published in grey and peer-reviewed literature. This study addresses the following research questions: (1) Which soil erosion indicators are useful to assess the overall effectiveness of natural infrastructure interventions from empirical studies in the Andes?; (2) What is the overall impact of implementing natural infrastructure on on-site and off-site erosion mitigation?; and, (3) Which locations and types of studies are needed to fill critical gaps in knowledge and research?

2 Materials and methods

The systematic review focuses on natural infrastructure interventions that are expected to influence erosion mitigation. We adapted the typology to the Andean region and defined three large groups of interventions: (i) the restoration and protection of native ecosystems and forestation; (2) the implementation of soil and water conservation measures, and (iii) the implementation of adaptation measures that regulate the flow and transport of water. We quantified their effect on the mitigation of water erosion by investigating measurable indicators of soil erosion. Besides common indicators of soil erosion such as soil loss rate, sediment yield, water turbidity and runoff coefficient, we also considered measures of soil quality such as soil organic carbon, soil nutrient content and bulk density. The definition of terms, and search criteria are provided in the

Supplement A and B, the database structure in Supplement C, and the studies that were included in the systematic review in Supplement D.

First, we systematically reviewed published case-studies from the Andean region and summarized the current state of knowledge, explored general patterns, and identified research gaps. We applied the reporting guidelines established in PRISMA, the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (Gurevitch et al., 2018; Moher et al., 2015). Then, we performed analyses of variance to explore systematic differences in soil erosion indicators in relation to interventions in natural infrastructure. Last, we estimated the overall effect of the interventions on soil quality, and on on-site and off-site erosion mitigation.

2.1 Literature search

The peer-reviewed literature search was conducted using the Scopus bibliographic database, and targeting studies published between 1980 and 2020. We used the keywords:

$$\begin{bmatrix} {*erosion}\ \text{OR}\ flood{*}\ \text{OR}\ landslid{*}\ \text{OR}\ mass\ movement\ \text{OR}\ alluv{*}\ \text{OR}\ runoff\ \text{OR}\ infiltration \\ \text{OR}\ gully\ \ \text{OR}\ sediment\ \text{OR}\ deposition\ \text{OR}\ soil \end{bmatrix}$$

$$\text{AND}$$

$$[Andes\ \text{OR}\ Colombia\ \text{OR}\ Venezuela\ \text{OR}\ Ecuador\ \text{OR}\ Peru\ \text{OR}\ Bolivia\ \text{OR}\ Chile\ \text{OR}\ Argentina]$$

$$\text{AND}$$

$$\begin{bmatrix} {*forest*}\ \text{OR}\ grazing\ \text{OR}\ grass{*}\ \text{OR}\ pasture\ \text{OR}\ agriculture\ \text{OR}\ crop\ {*}\ \text{OR}\ land\ use\ \text{OR} \\ puna\ \text{OR}\ paramo\ \text{OR}\ bofedal{*}\ \text{OR}\ dam\ \text{OR}\ reservoir\ \text{OR}\ conservation\ \text{OR}\ management \\ \text{OR}\ till{*}\ \text{OR}\ terraces\ \text{OR}\ irrigation\ \text{OR}\ lake{*}\ \text{OR}\ hydraulic{*}\ \text{OR} \\ ancient\ knowledge\ \text{OR}\ archaeology\ \text{OR}\ human\ \text{OR}\ people\ \text{OR}\ anthropogenic \end{bmatrix}$$

For the grey literature, we searched in 35 different databases from specialist organizations, public institutions, and local repositories of private and public universities in the Andean region. The above-mentioned search criteria were adapted for the grey literature given the limited search capabilities of some of the databases. Full details on the literature search are provided in the Supplement B, including the complete search terms, the number of records generated for specific searches, and the name, location, search dates in Scopus and the national and regional databases of research institutions, universities and specialist organizations. For international peer-reviewed literature, we used a test library of 20 references (Supplement E) that confirmed that the search strings captured relevant literature.

2.2 Inclusion and exclusion criteria

The number of studies that were identified, screened, selected and included in the analysis is shown in a PRISMA flow diagram (Fig. 1). Between 10 January and 27 February 2020, we identified 1798 potentially relevant studies: 91 % corresponding to peer-reviewed articles and 9 % to grey literature. After removing duplicate studies, the dataset was reduced to 813 studies.

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These records were screened, and articles that fulfilled the following criteria were included in the database: (1) they present quantitative data on soil erosion or soil quality comparing sites with different land use and protective land cover, soil and water conservation measures, or elements of hydraulic regulation, (2) are based on field-based studies or validated with field experiments or measurements, and (3) study the Andean region. During the screening stage, we excluded 623 studies because of absence of quantitative on-site or off-site soil erosion or soil quality measurements.

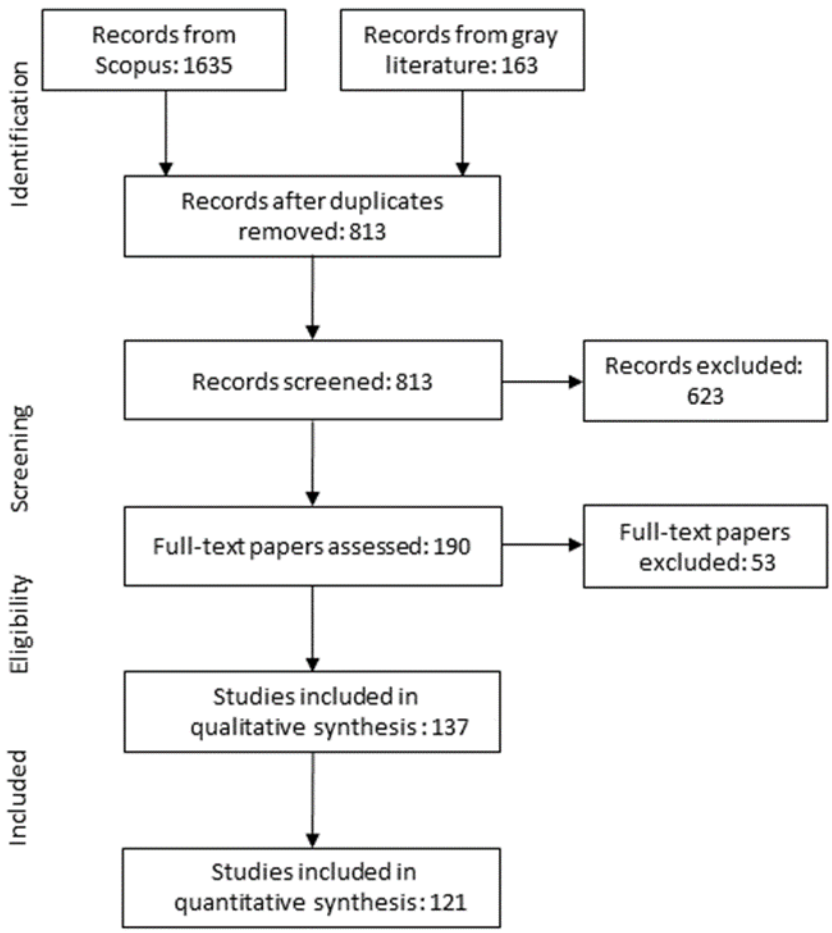

Figure 1. Flowchart resuming the results of literature search based on the PRISMA approach

We assessed 190 studies in full-text, and further excluded 53 studies based on the criteria described above. At this stage, this mainly concerned studies on landslides and landslide-related erosion events that did not allow us quantifying erosion rates on sites with different land use and protective land cover, soil and water conservation measures, or elements of hydraulic regulation.

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2.3 Database development

A total of 137 studies were included in the systematic review (Supplement D). Where a study encompassed several independent case studies, the case studies were included in the final database as separate entries. Each case study was coded by a unique study identifier and recorded in the georeferenced database (Supplement C). We recorded the following ancillary geographic data: (1) country, (2) site name, (3) coordinates (latitude and longitude in decimal degrees), altitude (meters above sea level, m a.s.l.), and information on (4) bioclimate, (5) surface lithology, (6) ecosystem and (7) landform. The latter four variables were derived from the 2005 Nature Conservancy datasets via the USGS dataviewer for South America (https://rmgsc.cr.usgs.gov/). We included additional information on the type of study: (8) the experimental design following the classification scheme of Nichols et al. (2011), (9) the modelling approach based on a classification in statistical, process-based and mixed models, (10) the existence of field-data, and (11) the spatial scale of study based on a classification in plot (< 0.01 km^2), small catchment (between 0.01 and 1000 km^2), large catchment (> 1000 km^2) and landscape scale (regional) analyses.

In the analyses, we quantified the effect of restoration and protection of natural vegetation such as forest or native grasslands (PRO), forestation with native or exotic species (FOR), and implementation of soil and water conservation measures (SWC) for soil erosion and mitigation (Fig. 2). Soil and water conservation measures (SWC) include crop management, conservation tillage, and slow forming terraces, and the implementation of linear elements such as vegetation strips and check dams. We compared the three natural infrastructure interventions (PRO, FOR and SWC) with untreated areas under traditional agriculture, either cropland (CROP) or rangeland (RANGE), and bare land (BARE). Bare land corresponds to abandoned cropland or degraded land with very low (< 10 %) vegetation cover.

The erosion indicators included in this study were (Fig. 2): Sloss = soil loss rate (determined as soil loss in t km^{-2} yr^{-1}), PRC = plot runoff coefficient (determined as event-based runoff coefficient from rainfall simulation experiments, in %), SSY = specific sediment yield (determined as the catchment-wide sediment yield per surface area, in t km^{-2} yr^{-1}), and RCC = catchment-wide runoff ratio (determined as the annual total runoff ratio of the catchment, in %). While Sloss and SSY are direct measures of soil erosion at the plot and catchment scale, the plot and catchment-wide runoff coefficients (RC and RCC) are indirect indicators of soil erosion by water: the rainfall regime plays a role as raindrop impact and runoff water are involved in the detachment of soil particles and transport of sediment in surface water flow. Empirical studies compiled by, for example, (Bonnesoeur et al., 2019; Valentin et al., 2008) have shown the strong association between runoff coefficients and soil erosion rates.

In addition to the four erosion indicators, two soil quality indicators were included: SOC (total soil organic carbon of the uppermost soil horizon, between 5 and 30 cm, in %), and BD (dry bulk density of the topsoil horizon, between 5 and 30 cm,

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in g.cm^{-3}). Soil organic carbon is the main indicator of soil quality (Franzluebbers, 2002) and directly linked to key soil functions (Wiesmeier et al., 2019) including soil water retention, erosion prevention and resilience to droughts and floods (Paustian et al., 2016). Bulk density is a commonly reported soil physical property that is related to soil aeration, water and air permeability, and soil microporosity (Horn et al., 1995). Increased bulk density can be indicative of soil compaction, and affect the water retention capacity and accelerate soil erosion (Molina et al., 2007; Patiño et al., 2021). Other erosion indicators were recorded in the database, but not included in the statistical analyses because of a lack of statistical representation. These include plot-based indicators like the stock in SOC over the entire soil depth or the saturated hydraulic conductivity of the topsoil, or catchment-wide indicators like the presence/relative occurrence of erosion signs or the suspended sediment concentration in the river channels. Mean, sample size and deviation metric were extracted from figures, supplementary materials and body text of the studies when available.

Of the 137 studies included in the systematic review, 121 studies contained sufficient information on the soil erosion and soil quality indicators to be statistically analyzed. Besides the above-mentioned information, the georeferenced database includes bibliographic details and a URL link to the individual case studies.

2.4 Statistical analyses

First, we tested whether sites with natural infrastructure interventions (PRO, FOR and SWC) are different in on-site (Sloss, RC) and off-site (SSY, RCC) soil erosion and soil quality (SOC, BD) compared to untreated areas under traditional agriculture (CROP, RANGE) or bare land (BARE) as illustrated in Figure 2. The comparison of the 4 erosion and 2 soil quality indicators between the treatments was performed using one-way analysis of variance (ANOVA). In this analysis, we pooled all observations from the 121 case-studies. Because of the limited number of quantitative case studies for the Andes, the number of observations is not the same for each group. Given the low number of observations per group, the Kruskal-Wallis ANOVA on the ranks was applied, with the Dunn's posthoc test. We rejected the null hypotheses (i.e. that there are no differences between the means of the groups) at the 0.05 significance level. We used R (R Core Team, 2017) with the "PMCMR" package (Pohlert, 2018) in R to perform the non-parametric comparisons.

Next, we analyzed the overall effect of natural infrastructure interventions on soil erosion and soil quality indicators. In this analysis, we only included case studies with a control–treatment design, where quantitative measures of soil erosion and quality were available to establish the control–treatment contrast. The response ratio (RR) was then used to determine the effect sizes. In this study, the response ratio was calculated for each natural infrastructure intervention (PRO, FOR, SWC) and soil erosion and quality indicator (Sloss, RC, SSY, RCC, SOC, BD) relative to untreated agricultural land (CROP and RANGE). We performed pairwise analysis for the pairs displayed in Figure 2 (e.g. Sloss for $RR_{PRO/CROP}$, BD for $RR_{FOR/RANGE, ...}$). For each pairwise comparison, we plotted the effect size of the individual studies in forest plots and explored the heterogeneity in the response among the case studies. The plots were used to identify the magnitude and sources of variation among the studies,

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and to identify eventual outliers. We then extracted the central tendency (mean effect) and confidence limits (standard error) for each indicator and pair-wise comparison. The mean effect and its standard error were then plotted in summary forest plots (per pair-wise comparison) to assess the overall effectiveness of a specific intervention on soil erosion and quality indicators. The graphs were produced using the R-package "metafor" (Viechtbauer, 2010).

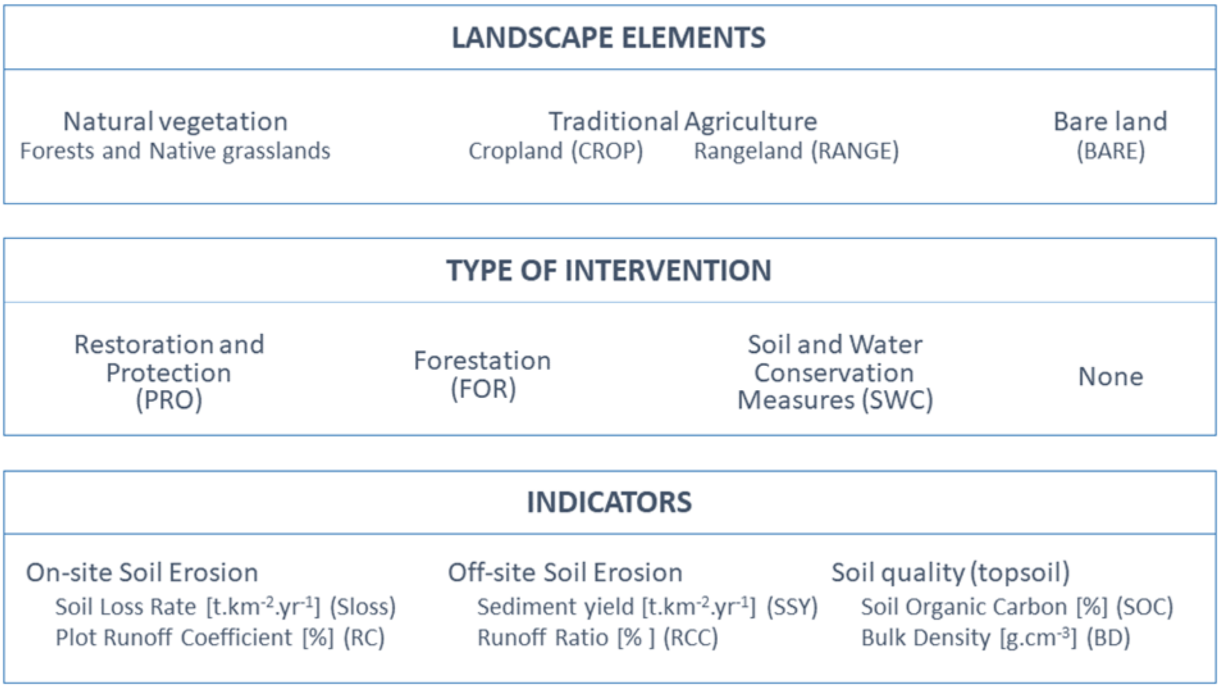

Figure 2. Schematic overview of study design

3 Results and discussion

3.1 Overall descriptive statistics

Of the 121 studies evaluating the effect of natural infrastructure interventions on soil erosion and quality indicators, 52 studies contained data on soil and water conservation practices (SWC), 65 studies on protective vegetation (PRO, FOR or both) and 14 studies on both (SWC, PRO and FOR). The majority of studies were journal articles (77 %), followed by chapters from books (6 %) and grey literature (17 %). The studies cover a 6500-km long stretch across the Andes, with 4 % of the studies in Venezuela (n = 5), 5 % in Colombia (n = 7), 38 % in Ecuador (n = 50), 36 % in Peru (n = 47), 6 % in Bolivia (n = 8), 6 % in Chile (n = 8) and 5 % in Argentina (n = 7). Ecuador and Peru have the highest concentration of case studies (Fig. 3). 89 % of the studies have investigated soil erosion in tropical climates, with 55 % of the studies performed in pluvial seasonal, 22 % in

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pluvial and 12 % in desertic or xeric climates. The remaining studies were performed in temperate or Mediterranean climate regimes. Field studies mostly involved erosion measurements at the plot scale (43 %, n = 57), small catchment scale (24 %, n = 32), and landscape scale (24 %, n = 32). Only 8 % of the studies included erosion assessment at the scale of large catchments (> 1000 km^2).

Table 1. Summary of the mean indicator values per treatment, with indication of the number of case studies between brackets. The values are reported for three interventions in natural infrastructure : PRO = restoration and protection of natural vegetation like forest or native grasslands, FOR = forestation with native and/or exotic species, and SWC = implementation of soil and water conservation measures, and for three untreated areas : CROP = cropland, RANGE = rangeland under traditional agricultural management, and BARE = bare land corresponding to abandoned cropland or degraded land with very low (< 10 %) vegetation cover. When the one-way analysis of variance was significant ($p < 0.05$), the results of the posthoc comparison test are shown. Treatments with different letter differ significantly.

	SOC	**BD**	**Sloss**	**RC**	**SSY**	**RCC**
	[%]	[g cm^{-3}]	[t km^{-2} yr^{-1}]	[%]	[t km^{-2} yr^{-1}]	[%]
PRO	8.67^{a} (16)	0.82^{ab} (12)	265 ^{a} (11)	14.0 ^{ab} (2)	1095 (10)	35.7 (6)
FOR	3.17^{ab} (8)	1.05 ^{ab} (4)	1858 ^{ab} (5)	15.7 ^{ab} (1)	1405 (7)	23.8 (4)
SWC	2.96^{b} (15)	1.37 ^{b} (6)	1662 ^{ab} (39)	6.4 ^{b} (9)	2427 (3)	30.1 (1)
CROP	2.49^{b} (19)	1.02 ^{ab} (6)	3193 ^{b} (40)	6.8 ^{ab} (10)	2889 (6)	39.6 (3)
RANGE	6.21^{ab} (17)	1.10 ^{ab} (12)	2371 ^{ab} (14)	24.5 ^{ab} (2)	464 (3)	41.0 (2)
BARE	1.88^{b} (10)	1.23 ^{ab} (6)	5137 ^{b} (16)	20.0 ^{a} (13)	6170 (8)	53.1 (2)
ALL						
$\bar{x} \pm 1SE$	4.47 ± 0.62	1.07 ± 0.05	2561 ± 292	12.9 ± 2.23	2599 ± 570	35.9 ± 4.54
n (#)	85	46	125	37	37	18

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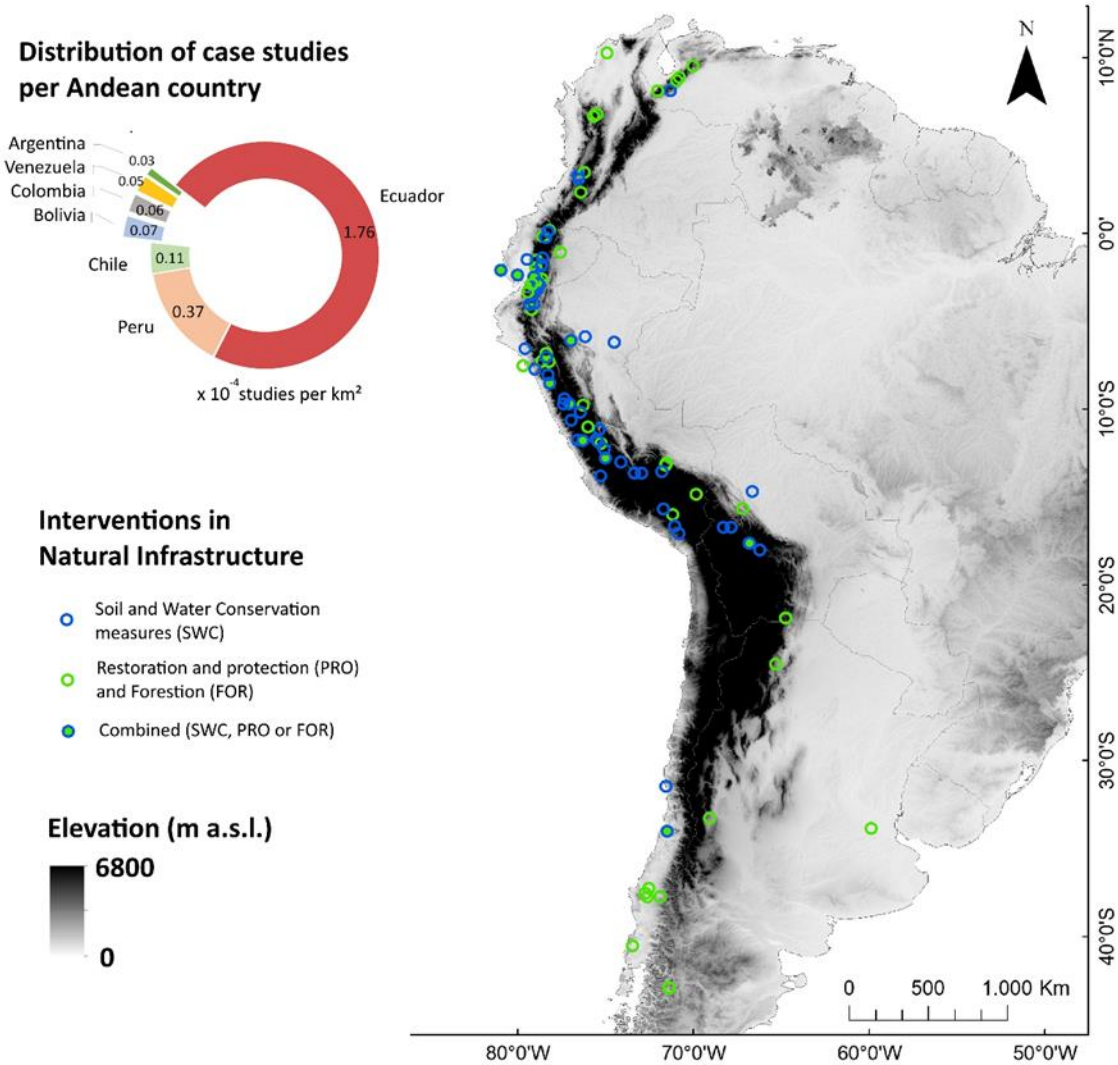

Figure 3. Spatial distribution of case studies in the Andean region, classified per type of natural infrastructure intervention. The background map corresponds to the 30 arc-second DEM of South America.

3.2 Erosion mitigation assessed from different soil erosion indicators

The one-way analysis of variance showed that the six categories of natural infrastructure (PRO, FOR, SWC, RANGE, CROP, BARE) were significantly different in soil quality and on-site soil erosion with notable differences in soil organic carbon ($p < 0.01$, $n = 85$), bulk density ($p = 0.02$, $n = 46$), soil loss ($p < 0.01$, $n = 125$) and plot runoff coefficient ($p = 0.03$, $n = 37$) (Table 1; Fig. 4). Notably, none of the erosion indicators that were measured at the catchment scale were significant at the 0.05 level,

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as we observe only small differences between categories for specific sediment yield ($p = 0.13$, $n = 37$) and no differences for catchment-wide runoff coefficient ($p = 0.54$, $n = 18$). The latter might be due to the limited number of observations documenting the effect of natural infrastructure interventions on SSY ($n = 37$) or RCC ($n = 18$) and the inherent variability in catchment response at the catchment scale (Romans et al., 2016). Below, we only present tendencies that are statistically significant at the 0.05 level.

Soil organic carbon concentration of topsoil is the indicator with the highest significance level, showing strong differences in soil quality between the six natural infrastructure categories (Fig. 4; Table 1). Based on 85 observations, we observe soil organic carbon values of the topsoil between 0.47 % and 34.06 %, with mean values of 4.47 % ± 0.62 %. Based on the results of the posthoc Dunn's test, two distinct groups can be identified: (1) areas covered by natural vegetation such as forests and grasslands with a mean SOC value of 8.67 %, and (2) areas covered by agricultural crops, with soil and water conservation measures and bare land having a mean SOC value of, respectively, 2.49 %, 2.96 % and 1.88 %. Areas with forestation and rangelands have intermediate SOC values of 3.17 % and 6.21 %, and are not statistically different from the two above-mentioned groups. These results are consistent with earlier observations by Bonnesoeur et al. (2019) who reported lower levels of topsoil organic matter in areas with forestation compared to native forests and grasslands. In this study, where we controlled for land use practices, we observed significant differences in SOC between pastures (grazed grasslands) and native grasslands.

Soil bulk density of the topsoil is reported in 15 % of the reported case studies for different natural infrastructure interventions. Soil bulk density ranges between 0.36 and 1.67 g cm^{-3} with a mean value of 1.07 ± 0.05 (Fig. 4; Table 1). The lowest mean BD values, i.e. 0.82 g cm^{-3}, are observed in soils covered by natural vegetation. Although the mean BD values are notably higher in areas with cropland, forestation, rangeland, and particularly bare land, the wide range in reported BD values per category does not allow to establish significant differences between categories at the 0.05 significance level. Remarkably, areas under soil and water conservation treatments have significantly higher bulk density values (mean value of 1.37 g cm^{-3}), which might reflect the advanced state of physical soil degradation due to compaction before SWC intervention. It also highlights that it may take several years to decades for impacts to be reversed, and that high levels of subsurface compaction may be irreversible without soil restoration.

The rate of soil loss measured at the plot scale (t km^{-2} yr^{-1}) is one of the most common indicators of soil erosion, as it is reported in 43 % of the case studies. The 125 quantitative measurements of Sloss reveal that soil loss rates vary widely with mean value of 2561 t km^{-2} yr^{-1} ± 292 and minimum and maximum values of resp. 0.001 and 14761 t km^{-2} yr^{-1}. Significant differences in Sloss are observed between areas covered by natural vegetation (mean of 265 t km^{-2} yr^{-1}) and areas covered by cropland or bare land (mean of resp. 3193 and 5137 t km^{-2} yr^{-1}). Areas with forestation, soil and water conservation and rangeland have intermediate values of Sloss (resp. 1858, 1662, 2371 t km^{-2} yr^{-1}) that show a wide spread. There are no significant differences between this group (FOR, SWC, and RANGE) and the other two groups (PRO, and CROP-BARE).

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The runoff coefficient (RC) is measured as surface runoff at the plot scale, and is here reported as the percentage of the rainfall that becomes runoff. The number of case studies that systematically report plot RC for different categories of natural infrastructure is low (12 %). Figure 4 illustrates the wide range of RC values (min: 0 %, max: 47 %) that are observed in the Andes, with mean values of 12.9 ± 2.23. The large variation might be the result of inherent spatial heterogeneity in rainfall-runoff response at the plot scale. However, methodological bias cannot be excluded as multiple field methods to estimate plot runoff coefficients were used: portable rainfall simulators covering a few cm^2 (e.g. Harden, 2001), runoff plots covering 1 m^2 (e.g. Molina et al., 2007), and experimental sites covering > 10 m^2 (Molina et al. 2009; Suescún et al. 2017). Also, the amount and intensity of the (simulated) rainfall often vary between case-studies. Notwithstanding, significant differences are observed in RC between areas with soil and water conservation measures and bare land, with RC values being, on average, 3.5 times lower in SWC compared to BARE.

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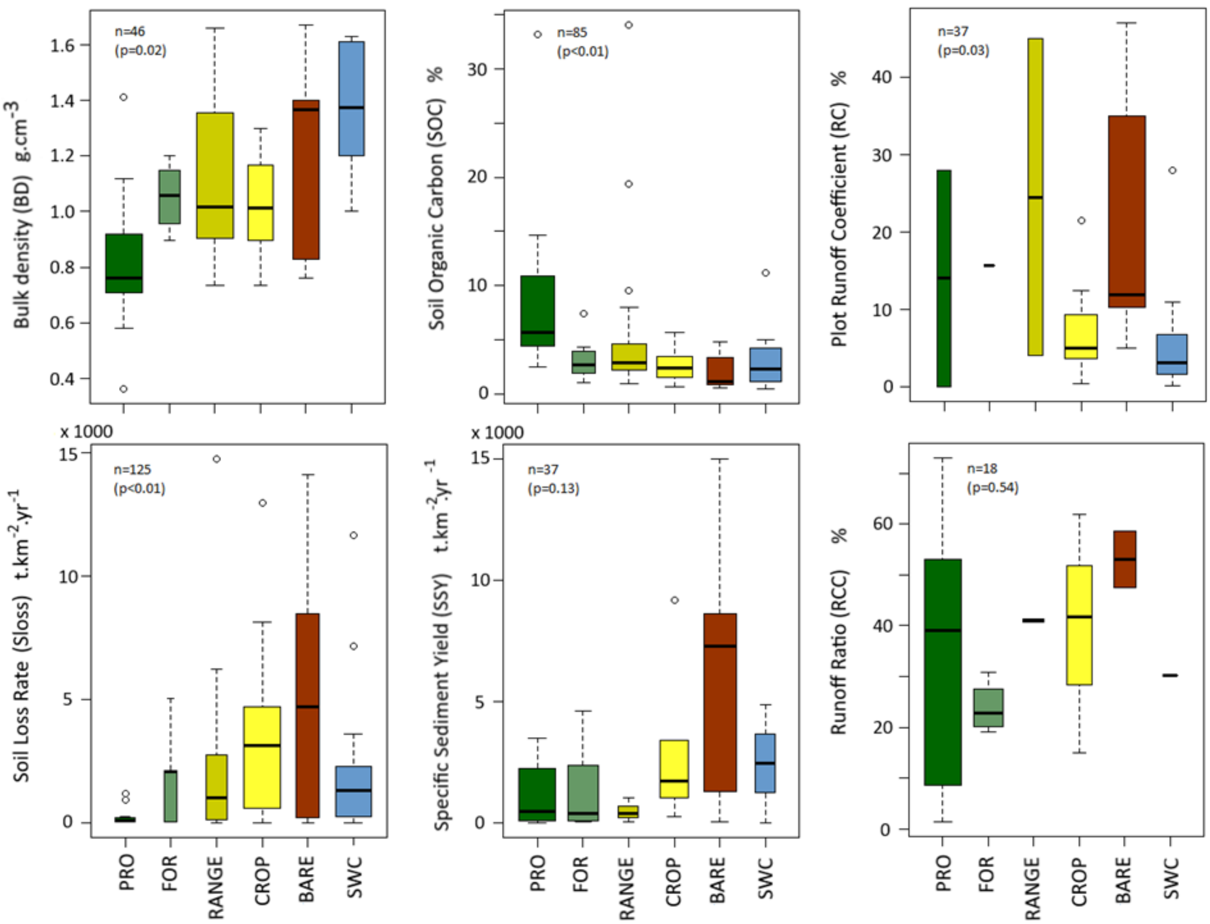

Figure 4. Variation in bulk density, soil organic carbon content, runoff coefficient, soil loss rate, specific sediment yield and catchment runoff coefficient between the three categories of natural infrastructure intervention (PRO, FOR, SWC), and the untreated agricultural land (RANGE, CROP) and bare land (BARE). Bold lines represent the median values, boxes extend to first and third quantiles and whiskers to 1.5 times the interquartile range from the box. The number of case studies that were included in the one-way analysis of variance test is shown, as well as the p-value.

3.3 Reported effectiveness of natural infrastructure interventions on soil erosion and soil quality

When limiting the quantitative analysis to matched pairs of control and single or multiple treatment(s), the number of independent empirical studies is reduced from 121 to 89 studies. As a result, the number of empirical case studies that counts

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with quantitative data on bulk density, catchment-wide runoff ratio and specific sediment yield is below 5 for all pairwise combinations. Below, we only discuss results that are based on a minimum of 3 independent case studies, and plotted the average (and standard error) of the response ratios in Figure 5.

The pairwise analyses showed that the effects of the protection and conservation of natural vegetation (PRO) were most notable on the SOC, Sloss, and SSY whereas there were no effects on BD. The areas maintained under natural vegetation had, respectively, 2.78 and 1.66 times higher SOC concentrations in the topsoil compared to areas under traditional agriculture (cropland and rangeland). Soil loss rates in conserved areas were on average 40 % lower than the rates measured in areas under traditional agriculture (cropland and rangeland). The data on specific sediment yield show a similar pattern with SSY in catchments with natural vegetation being 39 % of the SSY measured in cropland.

We find little evidence from control-treatment studies on the effect of forestation with native and/or exotic species (FOR) on soil quality, and on on-site and off-site soil erosion. The database counts less than 7 empirical studies per pairwise combination and erosion indicator. Compared to areas under traditional agriculture, forestation with native or exotic species (FOR) had no measurable effect on soil quality (SOC, BD; Fig. 5). Similarly, our analysis did not show evidence of a net effect on soil erosion (Sloss): the response ratio shows large scatter with RR values ranging between 0.37 for a study by Henry et al. (2013) and 7.75 for case study by Pesantez and Seminario (2010). Notwithstanding the high variability in response ratios for soil quality and on-site erosion, a positive and significant effect was observed for SSY, with specific sediment yields in catchments draining forested areas being –on average– 58 % of the yields measured in croplands. Similar observations were made by Bonnesoeur et al. (2019) who attributed the scatter in the empirical studies to the type of forestation (native vs. exotic species) and forestation age. In addition to this, the prior state of the environment (soil quality and erosion) has a major impact on erosion mitigation as Balthazar et al. (2015) showed for a case in the Ecuadorian Andes.

Areas with soil and water conservation measures (SWC) had 1.28 and 1.29 times higher SOC concentration in the topsoil compared to traditional agriculture (Fig. 5). The effect of SWC implementation on soil loss rates was dependent on the control group: Implementing SWC on croplands results in erosion rates that are on average 54 % lower than the rates measured on cropland, while SWC implementation on grasslands was reported to increase soil loss rates by 1.54 times.

The results for bare land are representative for the cases where no natural infrastructure was implemented on poorly vegetated land. These sites often correspond to abandoned croplands where the presence of erosion features (e.g. gully, rills, dunes) or soil degradation (e.g. salinization) limits agricultural production. Compared to sites with traditional agriculture, there was no clear effect of bare land on the soil quality (SOC, BD) of topsoil. Notwithstanding, there is strong effect on soil loss and specific sediment yield. Compared to cropland, the rates were – on average - respectively 20.2 and 10.7 times higher in bare land. Compared to grassland, we observe similar trends in catchment-wide erosion with effect sizes of 137 for SSY.

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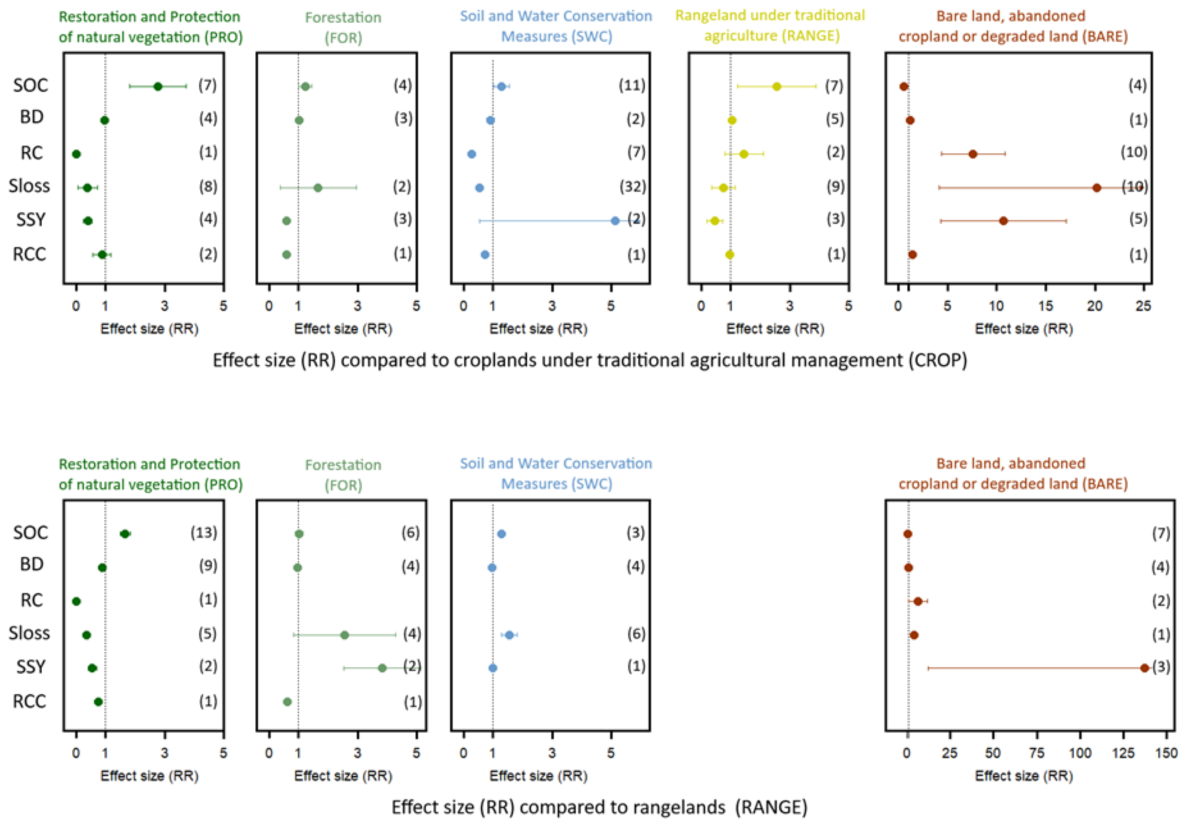

Figure 5. Response ratio of natural infrastructure interventions (PRO, FOR, SWC) and untreated degraded land (RANGE, BARE) relative to cropland and rangeland. The plots show the mean response ratio (points) and its standard error (solid lines) for soil organic carbon (SOC) and bulk density (BD) in the topsoil, runoff coefficient (RC) and soil loss rate (Sloss), specific sediment yield (SSY) and catchment-wide runoff ratio (RCC). The number of individual treatment-control studies are shown between brackets.

3.4 Knowledge gaps and prospects for future research

3.4.1 Representation of natural variability in environmental conditions within the Andean region

The literature reviewed in this study showed an unequal distribution of the empirical studies for the Andean countries, with an under-representation or documentation of studies from Argentina, Venezuela, Colombia and Bolivia. This is also reflected in the under-representation of tropical desertic, xeric, temperate, and Mediterranean climates (Fig. 3). There is a particular lack of knowledge on soil erosion processes before, after or during extreme rainfall or seismic events. Of the 121 quantitative studies, only 20 studies or 17 % explicitly referred to flooding or erosion processes during extreme (i.e. high-magnitude but rare and episodic) events. Reliable, quantitative information about the return period of extreme erosion and flooding events, and their influence on soil quality, long-term erosion rates and sediment discharge is scarce (Aguilar et al., 2020; Carretier et al., 2018). The severe scarcity of studies on the impact of extreme events has major implications for informing land use management practices (Coppus and Imeson, 2002), as the effectiveness of policy-based interventions on natural infrastructure could not be methodically evaluated for extreme events. A number of model applications by e.g. Bathurst et al. (2011, 2020) conveyed the limitations of forestation as an intervention for reducing peak discharges of floods derived from extreme but infrequent rainfall events. There is a clear need to thoroughly evaluate whether our results on the effectiveness of natural infrastructure interventions during frequent erosion events can be extended to extreme events related to e.g. El Niño–Southern Oscillation (ENSO).

3.4.2 Gap between plot-scale and catchment-scale erosion assessments

There is a clear gap between the number of case studies on water erosion at the plot-scale and the catchment-scale (Fig. 3) with about 43 % of all articles on small-scale erosion phenomenon and only 32 % on erosion at small and large catchment scale. The remaining 25 % of the studies are conducted at landscape scale, with observations made at different topographic positions within a larger geographical region. The strong research focus on mitigation of soil erosion on farmers' land is in line with past and ongoing efforts on sustainable and resilient agriculture. Only a handfull of studies in our database evaluated water erosion both at the plot and catchment scales. Transferring knowledge on erosion mitigation from the plot-scale to the catchment-scale remains a challenge. First, local-scale erosion phenomena might not be representative for the dominant erosion processes at the catchment scale (For example, terraced landscapes on the slopes with landslides along river channels).

Second, when the sediment that is generated by water erosion on the hillslopes is transferred downslope to the river network, sediment storage, erosion and remobilisation can occur across the river system (Romans et al., 2016). Therefore, human-induced changes in erosion rates on the hillslopes are not directly leading to changes in sediment yields at the outlet of the catchment (Verstraeten et al., 2017). Empirical studies from the Andean region are needed to decipher how environmental signals, such as changes in erosion rates after natural infrastructure interventions, are transferred through hillslopes, floodplains and river channels.

4 Conclusion

Our systematic review of the literature on natural infrastructure interventions and erosion mitigation in the Andean region resulted in 1798 potentially relevant case studies in the Andes. After screening the records, 121 empirical studies from peer-reviewed and grey literature (representing 6.7 % of the studies) were eligible and included in the quantitative analysis on soil quality and soil erosion. The studies covered a 6500-km stretch across the Andes with 89 % of the studies in tropical climates and the remaining 11 % in temperate and Mediterranean climates. More than 40 % of the empirical work was realized at the plot scale, followed by small catchments and landscape scale analyses. In our database, only 8 % of the studies were done at

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the scale of large catchments. As we restricted the database to empirical studies that isolated the effect of single or multiple natural infrastructure interventions, the number of observations at the large catchment scale was limited: with the increasing size of the catchments, the chance that multiple natural infrastructure interventions co-occur at the scale of large catchments is reduced.

Six indicators were commonly reported in the empirical studies: soil organic carbon and bulk density of the topsoil, soil loss rate and runoff coefficient at the plot scale, and specific sediment yield and catchment-wide runoff coefficient at the catchment scale. Soil organic carbon and soil loss rates were the two most frequently reported erosion indicators. The datasets contained, respectively, 85 and 125 records. The one-way analysis of variance showed significant differences in SOC, BD, Sloss and RC between areas with different natural infrastructure interventions (PRO, FOR, SWC). Areas under natural vegetation had significantly higher SOC and lower Sloss rates compared to cropland and bare land. In areas where soil and water conservation measures were implemented, soil loss and bulk density are not statistically different from natural vegetation, forestation, bare land and traditional agriculture. However, differences are observed for SOC and RC, with SOC values that are significantly lower than in natural vegetation, and runoff coefficients that are lower than in bare land.

When restricting the database to the 89 control-treatment studies, the overall effectiveness of the natural infrastructure interventions can be isolated. The response ratio then allowed us to quantify the mean outcome after implementing natural infrastructure interventions (PRO, FOR, and SWC) to that in the control group consisting of traditional agriculture. The protection and conservation of natural vegetation is the intervention with the highest effectiveness on SOC and Sloss. When avoiding conversion of natural vegetation to agricultural land, there was a strong and positive effect on the SOC of the topsoil (1.66 to 2.78 higher than in agricultural land) and a positive effect on soil loss rate (0.35 and 0.39 times the rates measured in agricultural land). The implementation of soil and water conservation measures in areas under traditional agriculture had positive effects on SOC (1.28 to 1.29 times higher SOC than in agricultural land). Soil loss rates were reduced to 54 % when implementing SWC on cropland. When implementing SWC in rangeland, the data indicated an increase in soil loss rate by 1.54 times. Untreated degraded land is reported to have significantly higher soil loss and specific sediment yield compared to cropland.

The systematic review of the existing literature not only allowed us to assess the overall effectiveness of commonly used nature-based solutions, but also to identify critical gaps in knowledge and research. The outcomes can be used to develop and guide implementation of natural infrastructure in the Andes.

Supplements

Supplement A. Definitions of terms used in the systematic review (in English and Spanish)
Supplement B. List of databases that were searched, with indication of search terms
Supplement C. Structure of the database
Supplement D. List of the literature studies that were included in the systematic review
Supplement E. Test library of 20 references compiled by experts in the fields

Data availability

Supplementary data are available for this paper. When accepted, the database will be uploaded to a data repository.

Author contribution

VV and AM conceived the study and conducted the statistical analyses, with backstopping of VB, FRD and BOT. AM and MR compiled and accurated the database from peer-reviewed and grey literature. All authors contributed to shaping the research and analyses, as well as writing the paper.

Competing interests

The authors declare that they have no conflict of interest.

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