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# Soil aggregation, erodibility, and erosion rates in mountain soils (NW Alps, Italy)

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**Abstract.** Erosion is a relevant soil degradation factor in mountain agrosilvopastoral ecosystems that can be enhanced by the abandonment of agricultural land and pastures left to natural evolution. The on-site and off-site consequences of soil erosion at the catchment and landscape scale are particularly relevant and may affect settlements at the interface with mountain ecosystems. RUSLE (Revised Universal Soil Loss Equation) estimates of soil erosion consider, among others, the soil erodibility factor ($K$), which depends on properties involved in structure and aggregation. A relationship between soil erodibility and aggregation should therefore be expected. However, erosion may limit the development of soil structure; hence aggregates should not only be related to erodibility but also partially mirror soil erosion rates. The aim of the research was to evaluate the agreement between aggregate stability and erosion-related variables and to discuss the possible reasons for discrepancies in the two kinds of land use considered (forest and pasture).

Topsoil horizons were sampled in a mountain catchment under two vegetation covers (pasture vs. forest) and analyzed for total organic carbon, total extractable carbon, pH, and texture. Soil erodibility was computed, RUSLE erosion rate was estimated, and aggregate stability was determined by wet sieving. Aggregation and RUSLE-related parameters for the two vegetation covers were investigated through statistical tests such as ANOVA, correlation, and regression.

Soil erodibility was in agreement with the aggregate stability parameters; i.e., the most erodible soils in terms of $K$ values also displayed weaker aggregation. Despite this general observation, when estimating $K$ from aggregate losses the ANOVA conducted on the regression residuals showed land-use-dependent trends (negative average residuals for forest soils, positive for pastures). Therefore, soil aggregation seemed to mirror the actual topsoil conditions better than soil erodibility. Several hypotheses for this behavior were discussed. A relevant effect of the physical protection of the organic matter by the aggregates that cannot be considered in $K$ computation was finally hypothesized in the case of pastures, while in forests soil erodibility seemed to keep trace of past erosion and depletion of finer particles. A good relationship between RUSLE soil erosion rates and aggregate stability occurred in pastures, while no relationship was visible in forests. Therefore, soil aggregation seemed to capture aspects of actual vulnerability that are not visible through the erodibility estimate. Considering the relevance and extension of agrosilvopastoral ecosystems partly left to natural colonization, further studies on litter and humus protective action might improve the understanding of the relationship among erosion, erodibility, and structure.

## 1 Introduction

Soil erosion is a key issue in mountain regions worldwide (Leh et al., 2013; Mandal and Sharda, 2013; Haregeweyn et al., 2013; Wang and Shao, 2013). Mountain soils develop in very sensitive environments subject to natural and anthropic disturbances (e.g., Cerdà and Lasanta, 2005; Vanwalleghem et al., 2011; Van der Waal et al., 2012; García Orenes et al., 2012), and they are often located at the interface with densely settled areas which may be considerably affected by sediment release from upstream erosion (Ziadat and Taimeh, 2013; Cao et al., 2014; Lieskovský and Kenderessy, 2014).

Considering that mountain soils are generally shallow and their fertility is often concentrated in the uppermost layers, soil erosion represents a crucial problem affecting the landscape at different scales and is a serious challenge for land management and soil conservation (García-Ruiz and Lana-Renault, 2011; Angassa et al., 2014; Bravo Espinosa et al., 2014).

Soil erosion can be assessed through a wide set of methods with different approaches as reviewed by Konz et al. (2012). RUSLE (Revised Universal Soil Loss Equation), derived from USLE (Wischmeier and Smith, 1978; Renard et al., 1997), is one of the most widely accepted empirical methods and, despite originally being applied at plot scale, is now being applied on catchments in a wide set of environments, including semi-natural ecosystems. Examples of mountain applications are widespread and reported by Meusburger et al. (2010) for the Swiss Alps, Haile and Fetene (2012) for Ethiopia, Ligonja and Shrestha (2013) in Tanzania, and Taguas et al. (2013) in Spain.

RUSLE gives an estimation of soil water erosion rates ($A$) in $\mathrm{Mg\,ha^{-1}\,yr^{-1}}$ obtained from the combination of five factors (rainfall erosivity, soil erodibility ($K$), topography, soil cover, protection practices). Among RUSLE factors, soil erodibility ($\mathrm{Mg\,ha\,h\,MJ^{-1}\,ha^{-1}\,mm^{-1}}$) expresses the intrinsic susceptibility of soil particles to be detached and consequently transported by surface runoff (Fernandez et al., 2003). Multiplying the rainfall erosivity factor $R$ by the soil erodibility, we get a measure of the potential erosion of a given soil that is then influenced by the topographic conditions and may be mitigated by vegetation cover and anthropic protection practices. RUSLE therefore combines intrinsic (soil erodibility) and exogenous (rainfall erosivity) factors to estimate an erosion rate which, in a second step, is linked to site conditions (topography and mitigation factors) to approach more closely the estimate of actual soil erosion.

The $K$ factor in its original formulation (Wischmeier and Smith, 1978) considers some physical and chemical variables, such as soil particle-size distribution and organic matter content, that are involved in the formation of soil structure. A good development of the structure of topsoil mineral horizons in terms of size and grade (i.e., well-developed and resistant aggregates) is therefore seen as fundamental in limiting erodibility, i.e., the combination of intrinsic properties affecting soil erosion.

Soil structure refers to the distribution and arrangement of soil voids and particles (Bronick and Lal, 2005); it cannot be measured directly and thus is commonly inferred by measuring the properties of the aggregates. Soil structure is thus often evaluated through aggregate stability that is promoted by organic and inorganic binding agents such as soil organic matter (SOM), clay, carbonates, and iron oxides (Tisdall and Oades, 1982). Soil aggregate stability can be assessed in a laboratory with a large set of methods (Cerdà, 1996; Pulido Moncada et al., 2013) and defines the resistance of soil aggregates to external stresses (e.g., dry or wet sieving, crushing). The existence of good relationships between soil aggregate stability and soil erodibility has been already investigated by several authors. For example, Barthès et al. (1999) observed that soil susceptibility to erosion is closely related to the topsoil aggregate stability. Tejada and Gonzalez (2006), in a study on amended soils, suggested adopting both erodibility and structural stability as soil vulnerability measures. However, these approaches do not take into account the complexity of the relationship: aggregation is indeed expected to mirror soil erodibility, but it can be considered in addition a proxy for soil erosion, as remarked by Cerdà (2000) who defined soil aggregate stability as a good indicator of soil erosion. Erosion is in fact expected to impede the development of soil structure (Poch and Antunez, 2010) as aggregates can build up only when losses of finer particles and cementing agents are limited (Shi et al., 2010) and, consequently, when erosion is not too intense.

We studied the relationships between soil aggregate stability (wet sieving test) and both erodibility (RUSLE $K$ factor) and erosion rates (RUSLE estimate) in a mountain catchment with two different vegetation covers (pasture and forest). The aim was to evaluate the agreement between aggregate stability and erosion-related variables and to discuss the possible reasons for discrepancies in the two kinds of land use.

## 2 Materials and methods

### 2.1 Study area

The study area (Fig. 1) is a mountain catchment (Perilieu river) in the Piedmont Alps (Susa Valley, NW Italy, 45°4′53″ E 6°42′1″ N.), very close to the town of Bardonecchia, the main ski resort in the valley. The altitude ranges from about 1200 to 2777 m a.s.l. (Mt. Jafferau ridge) with an extension of 219 ha. The predominant aspect is south and southwest. The climate is continental with around 720 mm rain and average temperature 10 °C (30-year time series). The precipitation peaks occur in May and October.

Large parts of the catchment were planted with tree species between the 1950s and the 1970s, while the rest of the forest cover was characterized by natural colonization of pioneer trees. In all cases, the canopy cover ranges from 50 to 75 % with a litter cover ranging from 75 to 80 %. The dominating species, depending on altitude, are larch (*Larix decidua* Mill.), Juniper (*Juniperus communis* L.), Scots pine (*Pinus sylvestris* L.), rhododendron (*Rhododendron ferrugineum* L.), and blackberry (*Vaccinium myrtillus* L.). The tree line is at around 2200 m, and the upper part of slopes is occupied by pastures, generally abandoned and with no relevant evidence of degradation. Geology is largely dominated by calcareous schists at higher elevation, while detritus and

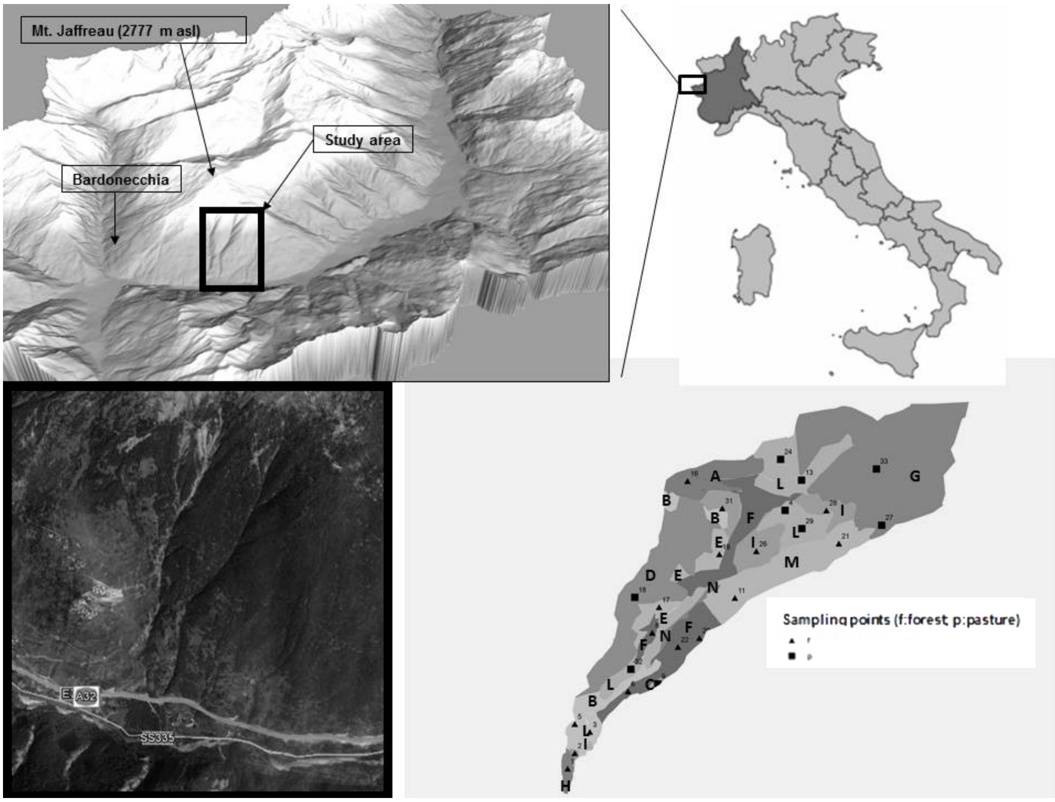


**Figure 1.** Digital elevation model of the study area (top left); catchment location (top right); Google Earth picture of the area (bottom left); LUT map (bottom right).

alluvial and colluvial materials dominate downslope. In particular, at the slope base an alluvial fan developed for river transport. The catchment is characterized by relevant slopes with a sharp reduction above 1900 m a.s.l., where pastures are present. Erosion evidences are visible in a large part of the study area, particularly where the vegetation cover is not complete. A large part of the area, mainly the southwest and southeast facing slopes, is interested by sheet erosion. Cattle trails and rill erosion phenomena are more common at high altitudes, while rill and interrill erosion, which are considered in RUSLE estimate, dominate at lower elevations. Rock outcrops are present at higher altitudes for a total area of ca. 20 ha (Mt. Jafferau summit). The south-facing slope (58.60 ha) is rather homogeneous and characterized by forest on detritus depositions with moderate slope, representing the largest land unit type in the catchment. The opposite slope is instead occupied forests on moderate slopes.

## 2.2 Soil sampling and analyses

Base maps and vector cartography were obtained from Regione Piemonte cartographic services, while the geology was digitized from the 1 : 50 000 geological map.

The catchment area was subdivided into 12 land unit types (LUTs), including non-soil units (e.g., rock outcrops), characterized by homogeneous vegetation cover, slope, and geology, obtained through an intersection procedure (Fig. 1) using the ArcGIS 9.3 software (ESRI Inc.). Out of the total area, around 199 ha were represented by soils while the rest was covered by rock outcrops. Considering a medium to high detail according to Deckers et al. (2002), we hypothesized a minimum sampling density of ca. 1 profile/10 ha, then distributed the sampling frequency according to the abundance and accessibility of LUTs. Twenty-five topsoils (i.e., always within A horizons, discarding the organic layers) were sampled at 0–10 cm ($n = 25$, of which 9 were represented by pasture, 16 by forest). The number of samples per LUT class was proportional to the LUT type abundance and considered the internal homogeneity of the LUT types. Sampling sites ranged from 1500 to ca. 2500 m a.s.l. and slope ranged from 0 to 80 %.

Soils were sampled in summer 2012, oven dried, and sieved to 2 mm. Soil structure grade, type, and size, as well as the skeleton content, were assessed in the field (Soil Survey Division Staff, 1993). Soil samples were characterized chemically and physically. All chemical and physical analyses were made in double and then averaged. Soil pH was determined potentiometrically (Soil Survey Staff, 2004), and total organic carbon (TOC) was determined by dry combustion with an elemental analyzer (NA2100 Carlo Erba Elemental Analyzer). The TOC content was calculated as the difference between C measured by dry combustion and carbonate C (Soil Survey Staff, 2004). The extractable carbon fraction (TEC, total extractable carbon) was obtained using a Na-hydroxide and Na-pyrophosphate 0.1 M solution (Sequi and De Nobili, 2000) to estimate the most transformed (i.e., humic) pool of organic matter. Carbonate content was measured by volumetric analysis of the carbon dioxide liberated by a 6 M HCl solution. Soil texture was determined by the pipette method with Na-hexametaphosphate without and with SOM oxidation with $H_2O_2$ (Gee and Bauder, 1986). The sand aggregation index ($\mathrm{Csand}_{H_2O_2}/\mathrm{Csand}_{Na}$), already applied in similar environments (Stanchi et al., 2102), was calculated and used as a measure of aggregation in the dimensional range of coarse sand. A pronounced aggregation is indicated by low ratios, while ratios close to 1 indicate almost negligible aggregation in the range of coarse sand.

Soil aggregates of 1–2 mm were separated from the 2 mm samples by dry sieving, The aggregate stability was determined by wet sieving. Soil samples (10 g, 1–2 mm fraction) were submerged on a rotating 0.2 mm sieve (60 cycles $\mathrm{min}^{-1}$) for fixed time intervals of 5, 10, 15, 20, 40, and 60 min. The aggregate loss at the different sieving times was computed as

$$\mathrm{loss\%} = 100\left(100 - \frac{\text{weight retained} - \text{weight of coarse sand}}{\text{total sample weight} - \text{weight of coarse sand}}\right). \quad (1)$$

Aggregate loss was then fitted to an exponential model described by the function (Zanini et al., 1998)

$$Y = a + b(1 - e^{-t/c}), \quad (2)$$

where $Y$ is aggregate loss (%), $t$ is the time of wet sieving (min), $a$ is the initial aggregate loss (%) upon water saturation, $b$ is the maximum aggregate loss for abrasion (%) and $c$ is the time parameter (min) related to the maximum aggregate loss (for $t = 3c$ the disaggregation curve approaches the asymptote). The curve parameters ($a$, $b$, and $c$) were estimated by non-linear regression, and goodness of fit was evaluated.

### 2.3 RUSLE application

RUSLE was developed from the original USLE equation (Wischmeier and Smith, 1978). The RUSLE model is formulated as follows:

$$A = RKLSCP, \quad (3)$$

where $A$ is the predicted average annual soil loss (Mg $\mathrm{ha}^{-1}\,\mathrm{yr}^{-1}$) and $R$ is rainfall-runoff-erosivity factor (MJ mm $\mathrm{ha}^{-1}\,\mathrm{h}^{-1}\,\mathrm{yr}^{-1}$) quantifying the eroding power of the rainfall. $R$ depends on rainfall amount and intensity. $K$ is the soil erodibility factor (Mg ha h $\mathrm{MJ}^{-1}\,\mathrm{ha}^{-1}\,\mathrm{mm}^{-1}$) that reflects the ease with which the soil is detached by impact of a splash or surface flow; $LS$ is the topographic factor (dimensionless), which considers the combined effect of slope length ($L$) and slope gradient ($S$) on soil erosion; $C$ is the cover factor (dimensionless) which represents the effects of land cover and management variables; $P$ (dimensionless) is the support practice factor, i.e., practices (mainly agricultural) for erosion control.

$R$ was calculated through six regression equations reported by Bazzoffi (2007) using meteorological data from the study area (Bardonecchia weather station, 30 years time series, monthly data) and then averaged. We therefore adopted in this study a unique average $R$ value of 1680 MJ mm $\mathrm{ha}^{-1}\,\mathrm{h}^{-1}\,\mathrm{yr}^{-1}$ (SD 576) for the study area despite the relatively wide altitude range, because for alpine continental areas such as Susa Valley the amount of precipitation does not show a clear gradient with elevation, as remarked by Ozenda (1985).

The $K$ factor (Mg ha h $\mathrm{MJ}^{-1}$ $\mathrm{ha}^{-1}$ $\mathrm{mm}^{-1}$) was calculated according to Wischmeier and Smith (1978) using the following equation adopted also by Bazzoffi (2007) for Italy:

$$K = 0.0013175((2.1\,M^{1.14}(10^{-4})(12 - a) + 3.25\,(s - 2) + 2.5(p - 3)), \quad (4)$$

where $M = (\text{silt } (\%) + \text{very fine sand } (\%)) \times (100 - \text{clay } (\%))$, and $a$ is organic matter (%) obtained as organic carbon content multiplied by the conversion factor 1.72. The coefficient $s$ is the structure code, varying from 1 to 4, based on aggregate shape and size assessed in the field during soil survey:

1. very fine or particulate < 1 mm;
2. fine granular and fine crumb, 1–2 mm;
3. granular and medium crumb, 2–5 mm, and coarse granular (5–10 mm);
4. very coarse granular or prismatic, columnar, blocky, platy, or massive, > 10 mm.

The coefficient $p$ is the profile permeability code, varying from 1 to 6 as follows:

1. rapid, i.e., > 130 mm $\mathrm{h}^{-1}$;
2. moderate to rapid, i.e., 60–130 mm $\mathrm{h}^{-1}$;
3. moderate, i.e., 20–60 mm $\mathrm{h}^{-1}$;
4. moderate to slow, i.e., 5–20 mm $\mathrm{h}^{-1}$;
5. slow (1–5 mm $\mathrm{h}^{-1}$);
6. very slow (<1 mm $\mathrm{h}^{-1}$).

The permeability code for the computation of $K$ factor was obtained after applying a pedotransfer function (PTF) for the estimation of $K_s$ (saturated hydraulic conductivity) and then classified according to RUSLE intervals. We adopted the PTF function proposed by Saxton et al. (1986):

$$K_s = 10\exp\left(12.012 - 0.0775\text{sand} + \frac{-3.895 + 0.03671\text{sand} - 0.1103\text{clay} + 0.00087546\text{clay}^2}{0.332 - 0.0007251\text{sand} + 0.1276\log_{10}\text{clay}}\right). \tag{5}$$

Estimated hydraulic conductivities ranged from 43 to $102\,\mathrm{mm\,h^{-1}}$, and therefore we attributed two discrete values to permeability codes (2 or 3).

Then the $K$ factor values calculated from sampling points were assigned to each LUT polygon.

The $LS$ factor was calculated from the digital elevation model (DEM) of the study area according to the procedure described in Desmet and Govers (1996) and Mitasova and Brown (2002). A flow accumulation raster was derived from a 10 m DEM and then the flow accumulation factor was computed using the ArcGIS (ESRI Inc.) Hydrologic extension. The equation adopted was

$$LS = (1+m)\left(F\,\frac{X}{22.13}\right)^m \left(\frac{\sin S}{0.0896}\right)^n, \tag{6}$$

where $F$ is the flow accumulation (Mitasova and Brown, 2002), $X$ is the grid size (10 m), $S$ is the slope angle, and 22.12 ($m$) and 0.09 are, respectively, the length and slope of the USLE experimental plot. $M$ and $n$ are coefficients related to the prevalent runoff type. Here we adopted $m = 0.4$ and $n = 1.3$.

As no specific survey was done, the $C$ factor was derived from tabular data proposed by Bazzoffi (2007) for forest and pasture vegetation cover, i.e., 0.003 for the forests of the study area (i.e., mixed forest with canopy cover ranging from 45 to 70 % and litter cover ranging from 75 to 85 %) and 0.02 for pasture (non-degraded pasture). The attribution to the classes was made on the basis of observations made during survey.

The $P$ factor was not applicable in the area and was therefore considered equal to 1. RUSLE was run using the input data of the 25 sampled slope sections.

### 2.4 Statistical analyses

A one-way ANOVA, using land use as the factor variable, was carried out for all soil properties. The homogeneity of variance was checked by the Levene test and all the variables showed homoscedasticity; therefore no variable transformation was needed. The correlation between variables was evaluated using the Pearson coefficient (two-tailed) after visual inspection of the data to verify that the dependence relationship was linear. Linear regression was also performed and the residuals saved for further data treatment.

All statistical analyses were performed using SPSS 19.

## 3 Results

Soil pH ranged from slightly acid to basic (Table 1) with an average of 7.3. The sand content always exceeded 50 %, while the clay content was scarce, always less than 11 %. The TOC content ranged from 16 to $53\,\mathrm{g\,kg^{-1}}$ and the TEC from 10 to $37\,\mathrm{g\,kg^{-1}}$; thus, on the average, 51 % of organic matter was extractable. The $a$ parameter, describing initial aggregate loss (Table 1), varied from 4.9 to 16.5 %; $b$, indicating the aggregate loss for abrasion, ranged from 30.8 to 52.5 %, while the $c$ parameter varied from 10.2 to 31.6 min. The sand aggregation index (Table 1) varied from 0.34 to 0.99. No significant differences in chemical, physical, and aggregation properties were observed between pasture and forest vegetation covers (Table 1).

The TOC content showed a good statistical correlation with the parameters of the aggregate breakdown fitting model (aggregate losses, time needed for aggregate disruption) and sand aggregation index (Fig. 2). With higher carbon contents, aggregates were globally more stable (Fig. 2a, c), needed a longer time for breakdown (Fig. 2b), and showed higher contents of sand-sized aggregates (Fig. 2d). A higher global stability corresponded to greater resistance to abrasion (Fig. 2c), as no significant relationships were found between TOC and initial losses upon water saturation ($r = 0.143$, $p = 0.25$). The correlation found between aggregate stability and organic matter also held when TEC was considered ($a{+}b$, $r = -0.690$, $p = 0.001$; $b$, $r = -0.656$, $p = 0.002$; $c$: $r = 0.755$, $p<0.01$, see Supplement S1). With regard to the RUSLE factors, soil erodibility (Table 2) ranged from 0.016 to $0.037\,\mathrm{Mg\,ha\,h\,MJ^{-1}\,ha^{-1}\,mm^{-1}}$ (average 0.025). In agreement with the lack of significant differences in soil chemical and physical properties, erodibility also did not differ significantly between pastures and forests. $K$ factors and soil aggregate stability were significantly correlated. In particular, a positive relationship was observed between $K$ values and aggregate loss (Fig. 3a), a negative relationship with the time parameter $c$ (Fig. 3b), and a positive relationship with the sand aggregation index (Fig. 3c). As expected, a negative correlation was observed with TOC (Fig. 3d). No correlation was visible when comparing $K$ and TEC ($r =$ 0.303, ns).

As a global marker for aggregate stability, total aggregate loss (Fig. 3a) explained about half of the $K$ variance ($R^2 =$ 0.453, $p<0.01$); most of the pasture samples fell above the fitting line, as confirmed by the positive average of residuals (Table 3), while forest samples showed a negative average of residuals (Table 3). Residuals were well correlated with the coarse and fine sand content on the whole (Supplement S2). Therefore, negative residuals (i.e., $K$ overestimation, typical of forest soils) corresponded to higher coarse sand contents and lower fine sand contents.

In Table 2 the other RUSLE factors and results are listed. The topographic factor $LS$ showed high spatial variability, reflecting the complexity of the study area, and ranged from

**Table 1.** Selected soil properties at sampling points.

| ID | pH | Sand % | Silt % | Clay % | TOC ($g\,kg^{-1}$) | TEC ($g\,kg^{-1}$) | TEC/ TOC | $CaCo_3$ ($g\,kg^{-1}$) | $a$ (%) | $b$ (%) | $a+b$ (%) | $c$ (min) | $Csand_{H_2O_2}$/ $Csand_{Na}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 7.3 | 70.9 | 23.0 | 6.1 | 53.3 | 30.1 | 0.56 | 113.7 | 13.7 | 32.1 | 45.8 | 28.14 | 0.34 |
| 2 | 8.1 | 66.1 | 28.4 | 5.5 | 42.3 | 21.0 | 0.50 | 112.7 | 13.4 | 33.7 | 47.1 | 23.29 | 0.64 |
| 3 | 7.3 | 71.1 | 22.2 | 6.7 | 21.8 | 14.0 | 0.64 | 4.80 | 14.9 | 39.5 | 54.4 | 16.41 | 0.83 |
| 4 | 7.4 | 66.6 | 28.5 | 4.9 | 46.2 | 22.0 | 0.48 | 108.0 | 13.5 | 33.4 | 46.9 | 28.19 | 0.60 |
| 5 | 7.2 | 72.1 | 22.2 | 5.7 | 16.3 | Nd | Nd | 66.5 | 4.9 | 52.5 | 57.4 | 10.25 | 0.99 |
| 6 | 7.2 | 70.7 | 22.9 | 6.4 | 41.4 | 20.0 | 0.48 | 92.0 | 11.4 | 36.1 | 47.5 | 16.08 | 0.69 |
| 7 | 8.2 | 65.1 | 28.2 | 6.7 | 23.5 | 13.0 | 0.55 | 47.6 | 15.1 | 38.3 | 53.4 | 17.01 | 0.80 |
| 8 | 6.3 | 59.9 | 30.6 | 9.5 | 52.0 | 37.0 | 0.71 | – | 12.7 | 32.8 | 45.5 | 31.64 | 0.55 |
| 9 | 7.6 | 60.6 | 31.5 | 7.9 | 33.5 | 24.0 | 0.72 | 103.20 | 16.5 | 35.1 | 51.6 | 23.57 | 0.69 |
| 10 | 7.3 | 80.1 | 16.3 | 3.6 | 21.2 | 10.0 | 0.47 | 85.4 | 13.8 | 40.3 | 54.1 | 15.29 | 0.85 |
| 11 | 7.5 | 69.5 | 23.1 | 7.4 | 47.0 | 19.0 | 0.40 | 116.00 | 13.2 | 32.4 | 45.6 | 26.77 | 0.63 |
| 12 | 7.2 | 72.7 | 22.1 | 5.2 | 39.1 | Nd | Nd | 98.0 | 13.7 | 35.8 | 49.5 | 22.10 | 0.72 |
| 13 | 7.7 | 65.6 | 26.7 | 7.7 | 36.5 | Nd | Nd | 111.3 | 15.1 | 35.4 | 50.5 | 20.10 | 0.73 |
| 14 | 7.1 | 66.7 | 26.7 | 6.6 | 33.1 | 17.0 | 0.51 | 90.6 | 13.7 | 35.9 | 49.6 | 16.60 | 0.70 |
| 15 | 7.0 | 65.2 | 28.0 | 6.8 | 23.8 | 16.0 | 0.67 | 79.6 | 15.4 | 38.6 | 54.0 | 17.96 | 0.78 |
| 16 | 8.0 | 71.1 | 21.2 | 7.7 | 42.4 | 22.0 | 0.52 | 107.6 | 10.7 | 35.9 | 46.6 | 15.36 | 0.69 |
| 17 | 7.2 | 59.8 | 33.1 | 7.1 | 29.5 | 21.0 | 0.71 | 81.9 | 10.0 | 42.0 | 52.0 | 14.02 | 0.72 |
| 18 | 7.5 | 61.9 | 26.9 | 11.2 | 33.6 | 17.0 | 0.51 | 4.6 | 10.4 | 39.9 | 50.3 | 13.51 | 0.71 |
| 19 | 7.6 | 63.9 | 27.3 | 8.8 | 47.6 | 21.0 | 0.44 | 19.0 | 12.6 | 32.3 | 44.9 | 19.35 | 0.59 |
| 20 | 8.0 | 62.4 | 27.6 | 10.0 | 37.9 | Nd | Nd | 104.0 | 10.2 | 38.7 | 48.9 | 14.93 | 0.72 |
| 21 | 7.7 | 58.7 | 30.9 | 10.4 | 27.8 | 10.0 | 0.36 | 4.60 | 10.8 | 41.5 | 52.3 | 12.77 | 0.83 |
| 22 | 6.1 | 62.3 | 29.2 | 8.5 | 27.9 | Nd | Nd | – | 15.7 | 38.4 | 54.1 | 19.27 | 0.78 |
| 23 | 6.5 | 66.5 | 27.0 | 6.5 | 42.2 | Nd | Nd | – | 14.9 | 34.3 | 49.2 | 26.67 | 0.66 |
| 24 | 7.0 | 65.7 | 29.8 | 4.5 | 44.1 | 22.0 | 0.50 | 104.60 | 15.0 | 30.8 | 45.8 | 23.72 | 0.65 |
| 25 | 7.5 | 65.5 | 24.8 | 9.7 | 40.5 | Nd | Nd | 96.0 | 12.2 | 36.0 | 48.2 | 16.71 | 0.74 |
| Av. forest | 7.4 | 67.0 | 26.1 | 6.97 | 35.3 | 18.99 | 0.50 | 71.1 | 12.8 | 37.2 | 50.1 | 19.1 | 0.68 |
| SD forest | (0.4) | (4.5) | (3.8) | (1.3) | (10.7) | (5.20) | (0.10) | (42.28) | (2.8) | (5.09) | (3.7) | (5.36) | (0.41) |
| Av. pasture | 7.2 | 65.7 | 26.7 | 7.7 | 37.8 | 21.40 | 0.50 | 68.2 | 13.2 | 36.2 | 49.4 | 20.4 | 0.70 |
| SD pasture | (0.6) | (5.9) | (4.3) | (2.7) | (9.43) | (9.70) | (0.1) | (50.58) | (2.0) | (3.4) | (3.2) | (6.3) | (0.20) |

The column $a$ represents initial soil loss after water saturation, $b$ the loss for abrasion, $a+b$ the total aggregates loss, and $c$ the time parameter related to maximum aggregate loss.

0 to ca. 25 RUSLE map is presented in Fig. 4. The erosion loss estimate $A$ ($Mg\,ha^{-1}\,yr^{-1}$) ranged from 0 (flat areas, with null $LS$ value) to ca. 26 $Mg\,ha^{-1}\,yr^{-1}$ (average 5.51, SD 7.69 $Mg\,ha^{-1}\,yr^{-1}$), thus showing high spatial heterogeneity. Around 50 % of the area was interested by moderate to severe soil erosion (i.e., 5–100 $Mg\,ha^{-1}\,yr^{-1}$) according to the scale proposed by Zachar (1982) and used in Fig. 4. Higher soil losses were concentrated in the channeled part of the catchment and a significant relationship, though not very strong, was found between RUSLE $A$ and the $LS$ factor (Supplement S3) in the whole data set. However, the correlation coefficients were much higher where forests and pastures were evaluated separately, as visible in the Supplement S3. The $LS$ factor (Table 2) did not show significant differences between vegetation covers, but the resulting erosion rate ($Mg\,ha^{-1}\,y^{-1}$) was much greater for pasture ($p<0.01$). Figure 5 shows the plot of RUSLE erosion rates against total aggregate loss. Considering forests and pasture points, two different trends were visible. In forests, aggregate stability did not explain the predicted soil erosion ($R^2 = 0.178$, n.s), while in pastures about 57 % of the RUSLE $A$ variance was explained by aggregate loss ($R^2 = 0.573$, $p<0.05$).

## 4 Discussion

In this work we wanted to assess the relationships between aggregate loss (wet sieving test) and both soil erodibility and erosion rates in a mountain agrosilvopastoral ecosystem characterized by two land cover types.

The relationships between RUSLE-related variables and aggregate loss (as a proxy of actual erosion in our initial hypothesis) showed a different behavior for the two land uses, i.e., soil erodibility was over- and underestimated from aggregate stability under forest and pasture cover, respectively (Fig. 3a). Moreover, the estimated erosion rate was not related at all with the total aggregate loss in the case of forest soils (Fig. 5).

Both aggregate stability (Fig. 2) and erodibility (Fig. 3) were deeply influenced by the soil organic matter content. Soil organic matter content did not differ between land covers, probably because of the concomitant presence of mor-

**Table 2.** Site characteristics and RUSLE input data at sampling points (slope sections used for RUSLE estimate). Vegetation, parent material, and average slope class refers to the land unit type. $LS$, $C$, and $K$ refer to sampling points.

| ID/ LUT code | Vegetation | Parent material | Altitude[a] | Slope class | $s$ | $p$ | Estimated $K_s$[b] | $K$[c] | $C$ | $LS$ | $A$[d] |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1/ | f | Mixed detritus | 1242 | Moderately steep | 3 | 2 | 74 | 0.017 | 0.003 | 2.90 | 0.25 |
| 2 | f | Mixed detritus | 1276 | Steep | 3 | 2 | 76 | 0.020 | 0.003 | 7.70 | 0.78 |
| 3 | f | Colluvium | 1336 | Very steep | 1 | 2 | 70 | 0.022 | 0.003 | 13.79 | 1.53 |
| 4 | p | Mixed detritus | 2161 | Steep | 3 | 2 | 80 | 0.025 | 0.020 | 20.37 | 17.11 |
| 5 | f | Colluvium | 1329 | Very steep | 1 | 2 | 78 | 0.037 | 0.003 | 8.75 | 1.63 |
| 6 | f | Carbonatic schists | 1476 | Very steep | 3 | 2 | 72 | 0.024 | 0.003 | 0.00 | 0.00 |
| 7 | f | Carbonatic schists | 1593 | Very steep | 2 | 2 | 69 | 0.023 | 0.003 | 24.66 | 2.86 |
| 8 | p | Colluvium | 1538 | Moderately steep | 3 | 3 | 53 | 0.016 | 0.020 | 5.63 | 3.03 |
| 9 | f | Colluvium/boulders | 1862 | Steep | 2 | 2 | 61 | 0.019 | 0.003 | 15.15 | 1.45 |
| 10 | p | Detritus | 2276 | Steep | 3 | 2 | 102 | 0.027 | 0.020 | 21.79 | 19.77 |
| 11 | f | Colluvium | 2104 | Steep | 3 | 2 | 64 | 0.013 | 0.003 | 14.21 | 0.93 |
| 12 | f | Carbonatic schists | 1704 | Steep | 3 | 2 | 82 | 0.022 | 0.003 | 26.16 | 2.90 |
| 13 | p | Colluvium | 1725 | Steep | 3 | 2 | 62 | 0.029 | 0.020 | 8.12 | 7.91 |
| 14 | f | Carbonatic schists | 1913 | Steep | 2 | 2 | 70 | 0.021 | 0.003 | 14.72 | 1.56 |
| 15 | f | Carbonatic schists | 1710 | Very steep | 2 | 2 | 68 | 0.033 | 0.003 | 7.07 | 1.18 |
| 16 | f | Colluvium | 2233 | Steep | 3 | 2 | 62 | 0.019 | 0.003 | 11.88 | 1.14 |
| 17 | f | Carbonatic schists | 1631 | Very steep | 2 | 2 | 66 | 0.031 | 0.003 | 16.66 | 2.60 |
| 18 | p | Detritus | 2334 | Steep | 3 | 3 | 43 | 0.032 | 0.020 | 20.83 | 22.40 |
| 19 | f | Detritus | 1978 | Steep | 3 | 3 | 56 | 0.024 | 0.003 | 16.74 | 2.02 |
| 20 | p | Colluvium/boulders | 2366 | Steep | 3 | 3 | 49 | 0.029 | 0.020 | 4.88 | 4.76 |
| 21 | f | Detritus | 2261 | Steep | 2 | 3 | 48 | 0.031 | 0.003 | 22.60 | 3.53 |
| 22 | p | Detritus | 2155 | Steep | 3 | 3 | 58 | 0.037 | 0.020 | 21.42 | 26.63 |
| 23 | f | Detritus | 2067 | Steep | 3 | 2 | 70 | 0.027 | 0.003 | 9.34 | 1.27 |
| 24 | p | Detritus | 1500 | Steep | 3 | 2 | 81 | 0.026 | 0.020 | 0.00 | 0.00 |
| 25 | p | Carbonatic schists | 2459 | Moderately steep | 3 | 3 | 50 | 0.024 | 0.020 | 11.47 | 9.25 |
| Av. Forest | – | – | | | – | – | 68 | 0.024 | 0.003 | 13.27 | 1.61 |
| SD forest | – | – | | | – | – | (8.3) | (0.006) | (–) | (7.32) | (0.98) |
| Av. Pasture | – | – | | | – | – | 64 | 0.027 | 0.02 | 12.72 | 12.43 |
| SD pasture | – | – | | | – | – | (19.3) | (0.006) | (–) | (8.50) | (9.57) |

[a] (m a.s.l.), [b] ($mm\,h^{-1}$), [c] ($Mg\,ha\,h\,MJ^{-1}\,mm^{-1}\,ha^{-1}$), [d] ($Mg\,ha\,yr^{-1}$).

**Table 3.** Residuals (unstandardized) of the relationship between erodibility ($K$) and total aggregates loss ($a+b$) for forest and pasture vegetation cover.

| Vegetation cover | Residuals (min) | Residuals (max) | Residuals (Average) | Residuals (SD) |
|---|---|---|---|---|
| Forest ($n=16$) | −0.00084 | 0.0052 | −0.00148 | 0.0045 |
| Pasture ($n=9$) | −0.00402 | 0.0068 | 0.00263 | 0.0040 |

phology and climate factors deeply affecting organic matter dynamics in mountain forest soils (Oueslati et al., 2013). Due to the lack of differences in SOM contents between pasture and forest soils, no differences in aggregate stability parameters or in the computed $K$ value (using texture, structure, and SOM as inputs) were found either. The importance of organic matter for topsoil structure conservation has been often reported in mountain soils with limited development in a variety of environments (e.g., Poch and Antunez, 2010; Stanchi et al., 2012). Relationships between aggregate stability and organic matter have often been observed in a wide range of climates, vegetation covers, and disturbance intensities (e.g., Cerdà, 1996, 2000; Amezketa, 1999; Gelaw et al., 2013).

Both the temporal stability of aggregates ($c$ parameter of the fitting equation) and the total aggregate loss ($a+b$) were related to soil erodibility. The soils displaying higher erodibility were therefore characterized by considerable and quick aggregate losses. Although the relationship was acceptable for both land uses (Fig. 3a), more than half of the variance of $K$ could not be accounted by aggregation. The systematic trend in the residuals indicated that predicting soil erodibility of pasture soils from aggregate loss generally led to an underestimation, i.e., pasture soils have higher $K$ (calculated with Eq. 4) than expected from aggregate stability (the measured $K$ values fall above the fitting line of Fig. 3a). The opposite occurred for forest soils (Table 3).

Several hypotheses can be formulated to assess the reasons of this systematic land-cover-dependent trend. First, to evaluate if this was linked to some systematic mathematical bias related to the use of discrete permeability classes, we recomputed $K$ by using a continuous distribution of permeability classes instead of the discrete values. The new erodibility values ($K_{cont}$, data not shown) were always positively related

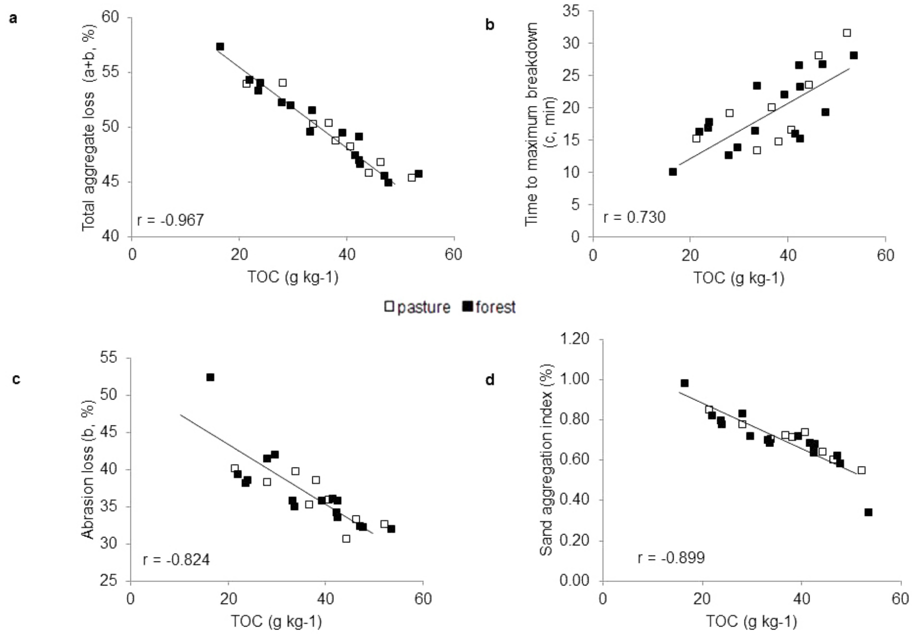


**Figure 2.** Correlations between organic carbon content (TOC) and aggregation stability parameters: **(a)** total loss of aggregates $a+b$; **(b)** time to maximum breakdown, $c$; **(c)** abrasion loss, $b$; **(d)** sand aggregation index. Black squares correspond to forest, open squares to pasture. All correlations had $p<0.01$.

with $K$ ($r = 0.94$, $p<0.01$) but always showed higher values, although not significantly different (paired $t$ test, $p<0.01$). Also, $K_{cont}$ showed significant relationships with aggregate stability characteristics; therefore any bias related to the use of discrete permeability classes could be excluded. Another possibility is that other cementing agents may influence soil aggregate formation and stability, such as pedogenic carbonates and iron oxides (Dimoyiannis, 2012; Campo et al., 2014), while only texture and organic matter content are used for $K$ computation. Although the role of these cementing agents may be important in later stages of pedogenesis, in poorly developed mountain soils the contribution of binding agents other than organic carbon is considered marginal. In fact, as reported by Tisdall and Oades (1982), in coarse sand-sized aggregates, organic matter acts as a relevant binding agent for aggregates. Moreover, $CaCO_3$ in the studied environment is of primary origin and not pedogenic, and thus is not expected to act as a cementing agent because of scarce reactivity and large grain dimensions (Le Bissonais, 1996). In addition, iron oxides are not relevant cementing agents for poorly developed soils (Bronik and Lal, 2005). In our dataset, the determination coefficients of the regressions between organic matter and aggregate stability (Fig. 2) suggested a relevant role of organic matter in aggregation (up to 93 % of variance explained), as found also by Zanini et al. (1998) during wet sieving experiments in similarly poorly developed soils. Considering that the effect organic matter has on aggregation is highly dependent on the degree of transformation of organic compounds (i.e., degree of alteration and/or incorporation in soil), differences in organic matter quality might account for the differences in residuals between pastures and forests. For example, Falsone et al. (2012) pointed out that not only organic matter quantity but also its quality affects soil structure development in surface horizons of poorly developed soils. In order to check this additional hypothesis, we introduced a qualitative variable describing SOM, i.e., the TEC content (besides the quantitative information given by TOC). In fact, an evaluation of the degree of SOM transformation can be provided by the ratio between TEC and TOC (Table 1). As the standard deviation was relatively high (0.16, i.e., more than 30 %), some variations in the degree of transformation of organic matter among sampling points can be hypothesized in the study area.

When trying to investigate the degree of transformation of organic matter in relation to erodibility (in terms of TEC and $K$) we did not observe any relevant correlation, which seems to suggest that the total amount of organic matter is more helpful for the purpose of erodibility studies. In fact, variations in SOM contents do not correspond to linear variations in $K$ values, as clearly visible from the original Wischmeier's nomograph (Wischmeier and Smith, 1978); thus the relationship disappearance may be caused by restricting the range of organic matter values.

To explain the underestimated $K$ values obtained for pasture soils, we therefore formulated a further hypothesis, i.e., a physical protection of organic matter due to its better incorporation in aggregates as a consequence of the annual turnover and the contribution of the root apparatus of herba-

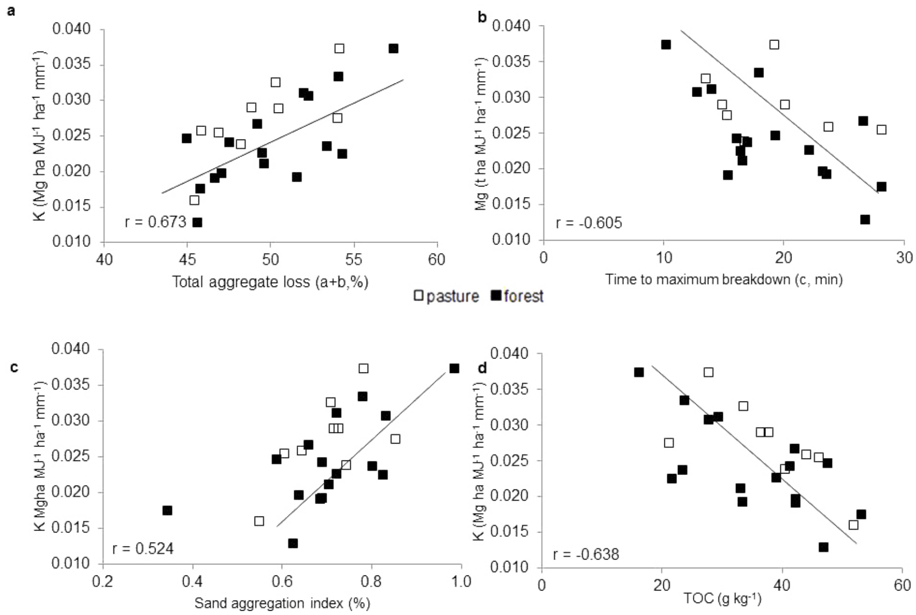


**Figure 3.** Correlations between $K$ and aggregation stability parameters: **(a)** total loss of aggregates $a+b$; **(b)** time to maximum breakdown, $c$; **(c)** sand aggregation index; **(d)** organic carbon content (TOC). Black squares correspond to forest, open squares to pasture. All correlations had $p<0.01$.

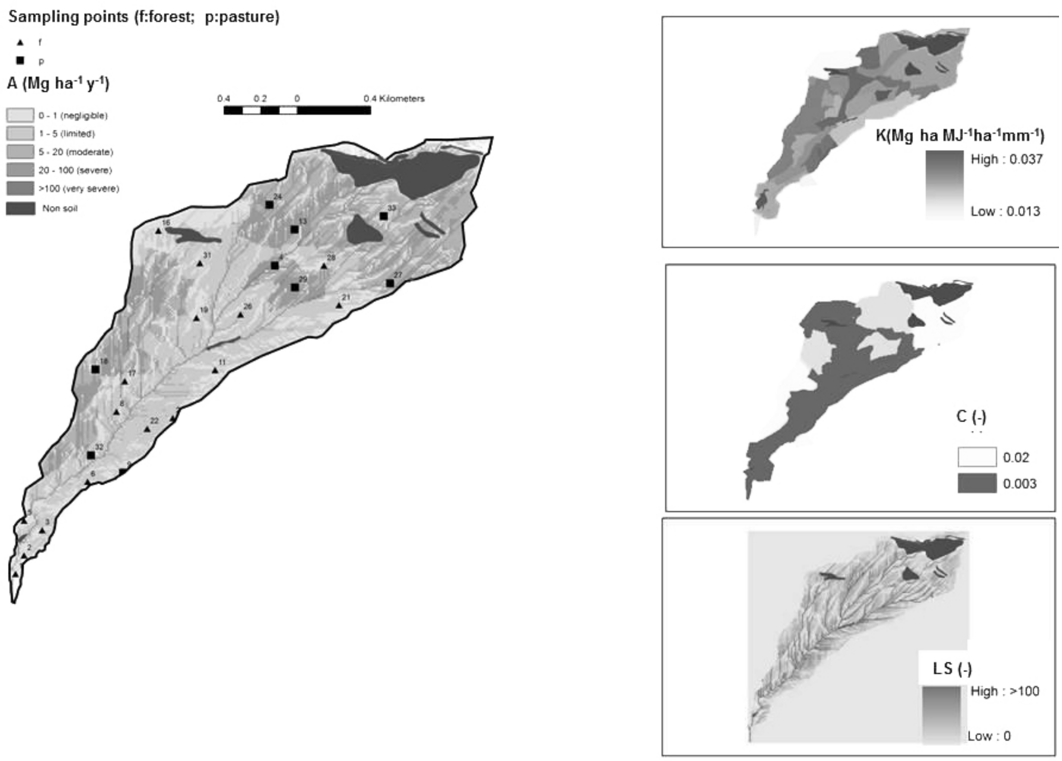


**Figure 4.** Map of RUSLE input factors and results.

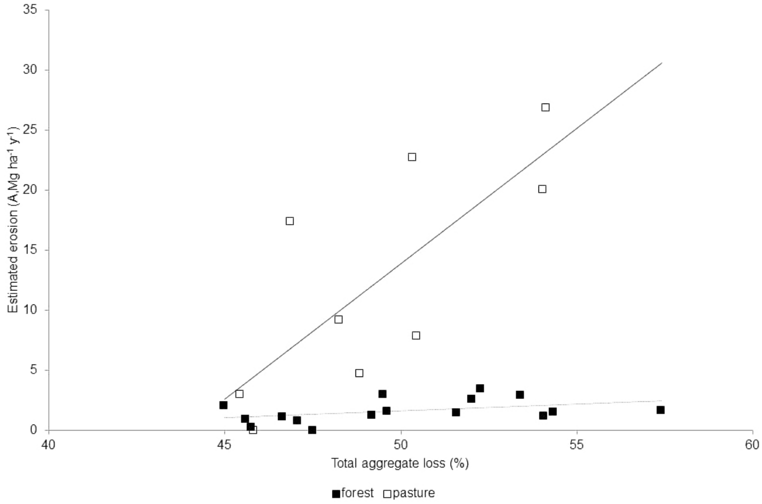


**Figure 5.** Regression lines between estimated erosion ($A$) and aggregate breakdown for the two vegetation covers.

ceous vegetation (Kalinina et al., 2011). The incorporation of organic matter into aggregates favors their stability, as stated by Puget et al. (2008), and increases their resistance to breakdown determining qualitative differences in SOM between grassland and forest topsoils (Wiesmeier et al., 2014). However, the formulation of the RUSLE $K$ factor cannot take these qualitative aspects into account. Conversely, this did not occur for forest soils or it was less marked.

The $K$ values calculated for forest soils might at present be lower than expected from aggregate stability (Table 3) if erosion has already been acting for a long time, leaving coarser particles that are by definition less erodible (Renard et al., 1997). The negative relationship (Supplement S2) observed between coarse sand content and residuals of $K$ estimate supported the hypothesis of past erosion effects of forest soils (lower residuals), which resulted in a depletion of fine particles and a relative enrichment of coarser, less erodible fraction (i.e., coarse sand). The forest stands in the study area are in fact relatively young, and thus the surfaces were previously exposed to erosion with the same intensity as pastures. Aggregate formation is, however, a fast and continuous process (Denef et al., 2002) and thus aggregates better represent the current land use.

The differences between land covers are maintained in the effect vegetation has on erosion rate, as expected due to the choice of the RUSLE $C$ factor; however, the relationships between the RUSLE $A$ parameter and aggregate loss were found only for pastures (Fig. 5). As the $LS$ was well correlated to A in both land uses, the lack of dependence of $A$ from aggregate loss observed in forests points could be due to a high heterogeneity in the actual effect of forest vegetation in mitigating erosion. In forests, the variability in litter quality and thickness is expected to be high, as indeed $C$ stocks in the humic episolum of northwestern Italian forest soils range from less than 3 to about 10 $\mathrm{kg\,m^{-2}}$ (Bonifacio et al., 2011), and could not be fully accounted by the range of $C$ factor provided for by the RUSLE tabular values, corresponding to rather wide vegetation densities ranges. As a consequence, aggregates may develop differently depending on the presence of organic layers, giving rise to a large variability in the erosion amounts.

## 5 Conclusions

The soil aggregate stability in a mountain catchment was assessed with a laboratory wet sieving test and the results were compared with the erodibility factor $K$ and the estimated erosion rate (RUSLE model). The $K$ factor was positively correlated with the aggregate loss (wet sieving test), i.e., the most erodible soils (higher $K$) also displayed higher aggregates losses and quicker breakdown. Land-use-dependent trends were, however, observed in the estimate of $K$ from aggregates loss. In fact, the residuals for forest soils were lower in absolute value and with average negative value, while the opposite behavior was found in pastures. Therefore, soil aggregate stability seemed to reflect better the actual vulnerability of topsoils to physical degradation. Several reasons for this behavior were discussed, and a relevant effect of the physical protection of organic matter by aggregates that cannot be considered in the traditional $K$ formulation was hypothesized for pastures. In forests, soil erodibility seemed to keep trace of past erosion and depletion of fine particles. Moreover, while the RUSLE erosion rate could be satisfactorily predicted from aggregates loss for pastures, this was not possible for forests. In forests, erosion estimates also seemed particularly problematic because of a high spatial variability of litter properties. The protecting role of the forest floor in terms of richness and diversity, and not only of cover, in the RUSLE $C$ factor definition would need further investigation in order to better understand the mechanisms that determine the relationship between soil erosion and structure for the different land uses.

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*Author contributions.* Silvia Stanchi carried out GIS modeling, result presentation and interpretation, and statistical analysis. Gloria Falsone was responsible for the aggregate stability analysis and SOM dynamics interpretation. Eleonora Bonifacio supervised the research and coordinated the manuscript writing and the discussion presentation.

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