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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, and can be enhanced by the abandonment of agricultural land and pastures, then 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 is therefore expected. Erosion is however expected to limit the development of soil structure, hence aggregates should not only be related to erodibility but also mirror soil erosion rates. We investigated the relationships between aggregate stability and the RUSLE erodibility and erosion rate in a mountain watershed at the interface with settlements, characterized by two different land use types (pasture and forest). 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. However, estimating $K$ from aggregate loss showed that forest soils always had negative residuals, while the opposite happened for pastures. A good relationship between RUSLE soil erosion rates and aggregate stability occurred in pastures, while no relationship was visible in forests. 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. In addition, in forests, the erosion rate estimate was particularly problematic likely because of a high spatial variability of litter properties. Considering the relevance and extension of agrosilvopastoral ecosystems partly left to natural colonization, further studies 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 affect 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 it was originally 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, by Haile and Fetene (2012) for Ethiopia, by Ligonja and Shrestha (2013) in Tanzania, and Taguas et al. (2013) in Spain.

RUSLE gives an estimation of soil water erosion rates in $\mathrm{Mg\,ha^{-1}\,y^{-1}}$ obtained from the combination of five factors (rainfall erosivity, soil erodibility, topography, soil cover, protection practices). Among RUSLE factors, soil erodibility ($K$, $\mathrm{Mg\,ha\,h\,MJ^{-1}\,ha^{-1}\,mm^{-1}}$) 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 $K$, 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 soil structure 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, thus it 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, clay, carbonates, and iron oxides (Tisdall and Oades, 1982). Soil aggregate stability can be assessed in 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 etc.). 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, which is quite easier to assess. 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.

The aim of this research was to verify the existence of the relationships between aggregate stability and RUSLE related variables in mountain areas, following the hypothesis that susceptibility of soil to erosion, erosion rate and aggregation should in principle agree. We thus studied the relationships between soil aggregate stability (wet sieving test), and both erodibility (RUSLE $K$ factor) and erosion rates ($A$, RUSLE estimate) in a mountain catchment with two different vegetation covers (pasture and forest).

## 2 Materials and methods

### 2.1 Study area

The study area is a mountain catchment (Perilieux river) in the Piedmont Alps (Susa Valley – Bardonecchia – 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. Jaffreau ridge) with an extension of 219 ha (Fig. 1). The predominant aspect is South and South-West. The climate is continental with around 720 mm rain and average temperature 10 °C (30 years time series). The precipitation peaks occur in May and October.

Large parts of the catchment were planted with tree species between the 50s and the 70s of the 20th century, while the rest of the forest cover was characterized by natural colonization by pioneer trees. In all cases, the canopy cover is discontinuous. The dominating species, depending on altitude, are larch, Juniper, Scots pine, rhododendron and blackberry. The tree line is at around 2200 m, and the upper part of slopes is occupied by pastures. Geology is largely dominated by calcareous schists at higher elevation, while detritus and 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. The riverbed is highly channeled, and erosion evidences are visible in a large part of the study area, and particularly where the vegetation cover is partial. A large part of the area, mainly the SW and SE facing slopes, is interested by sheet erosion. Cattle trails and rill erosion phenomena are very common at high altitudes, while rill and interrill erosion dominate at lower elevations. Rock outcrops are present at higher altitudes for a total area of ca 20 ha (Mt. Jaffreau 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 15 land unit types (LUTs), including non-soil units (e.g. rock outcrops), characterized by homogeneous vegetation cover, slope, geology, obtained through an overlay procedure using the ArcGIS 9.3 software (ESRI Inc.). Twenty-five topsoils (0–10 cm depth, A horizons) were sampled ($n = 25$, of which 9 were represented by pasture, 16 by forest) taking into account the relative % cover of each LUT. The site characteristics of the sampling points are summarized in Table 1. Sampling sites ranged from 1500 to ca. 2500 m a.s.l. and slope ranged from 0 to 80 %.

Soils were oven dried and sieved to 2 mm. Soil structure grade, shape and size were assessed in the field, as well as the skeleton content (Soil Survey Division Staff, 1993). Soil samples were characterized chemically and physically. All analyses were made in double and then averaged. Soil pH was determined potentiometrically (Soil Survey Staff, 2004), total organic C (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 C 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 soil organic matter (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:

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

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

$$y = a + b(1 - e^{-t/c}) \tag{2}$$

where $y$ is aggregate loss (%); $t$, time of wet sieving (min); $a$, initial aggregate loss (%) upon water saturation; $b$, maximum aggregate loss for abrasion (%); $c$, 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.

All statistical analyses were performed using SPSS 20.



### 2.3 RUSLE application

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

$$A = RK\,\mathrm{LS}CP \tag{3}$$

where:

$A$ = predicted average annual soil loss ($\mathrm{Mg\,ha^{-1}\,yr^{-1}}$);

$R$ = rainfall-runoff-erosivity factor ($\mathrm{MJ\,mm\,ha^{-1}\,h^{-1}\,y^{-1}}$) quantifying the eroding power of the rainfall. $R$ depends on rainfall amount and intensity;

$K$ = soil erodibility factor ($\mathrm{Mg\,ha\,h\,MJ^{-1}\,ha^{-1}\,mm^{-1}}$) that reflects the ease with which the soil is detached by impact of a splash or surface flow;

LS = topographic factor (dimensionless), it considers the combined effect of slope length ($L$) and slope gradient ($S$) on soil erosion;

$C$ = 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 with 6 regression equations reviewed by Bazzoffi (2007) using meteorological data from the study area (Bardonecchia weather station, 30 years time series) and then averaged. We adopted a unique value of 1680 $\mathrm{MJ\,mm\,ha^{-1}\,h^{-1}\,y^{-1}}$ 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 ($\mathrm{Mg\,ha\,h\,MJ^{-1}\,ha^{-1}\,mm^{-1}}$) was calculated according to Wischmeier and Smith (1978) using the following equation adopted also by Bazzoffi (2007) for Italy:

$$K = 0.013175(2.1M^{1.14}(10^{-4})(12 - a) + 3.25(s - 2) + 2.5(p - 3)) \tag{4}$$

Where $M$ = (silt (%) + very fine sand (%)) · (100 – clay (%)); $a$ = organic matter (%), obtained as organic C content multiplied by the conversion factor 1.72. The coefficient s is the structure code 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) and (4) very coarse granular or prismatic, columnar, blocky, platy or massive, > 10 mm. The coefficient p is the profile permeability code: (1) rapid, i.e. > 130 $mm\,h^{-1}$, (2) moderate to rapid, i.e. 60–130 $mm\,h^{-1}$, (3) moderate, i.e. 20–60 $mm\,h^{-1}$, (4) moderate to slow, i.e. 5–20 $mm\,h^{-1}$, (5) slow (1–5 $mm\,h^{-1}$) and (6) very slow (< 1 $mm\,h^{-1}$). 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 101 $mm\,h^{-1}$, and therefore we attributed two discrete values to permeability codes (2 or 3).The LS factor was calculated from the digital elevation model of the study area according to the procedure described in Desmet and Govers (1996) and Mitasova et al. (2002). A flow accumulation raster was derived from a 10 m digital elevation model (DEM) and then the flow accumulation factor was computed using the ArcGIS (ESRI Inc.) Hydrologic extension. The equation adopted was:

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

Where $F$ is the flow accumulation (Mitasova and Brown, 2002), $C$ is the grid size (10 m), $S$ is the slope angle, 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 $n = 1.3$.

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 and 0.02 for pasture. 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.

## 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 total organic C (TOC) content ranged from 16 to 53 $g\,kg^{-1}$ and the total extractable C (TEC) from 10 to 37 $g\,kg^{-1}$, 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 ($Csand_{H_2O_2}/Csand_{Na}$, 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).

As shown in Fig. 2, the organic C content showed a good relationship with the parameters of the aggregate breakdown fitting model (aggregate losses, time needed for aggregate disruption) and sand aggregation index. As the organic C content increased, aggregates were globally more stable (Fig. 2a), they 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 ($r = 0.143$, $p = 0.25$).

With regard to the RUSLE factors, soil erodibility ($K$, Table 2) ranged from 0.016 to 0.037 $Mg\,ha\,h\,MJ^{-1}\,ha^{-1}\,mm^{-1}$ (average 0.025). In agreement with the lack of significant differences in soil chemical and physical properties, also erodibility 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 losses ($b$: $r = 0.686$, $p < 0.01$; $a + b$: $r = 0.673$, $p < 0.01$), and a negative relationship with the time parameter $c$ ($r = -0.605$, $p < 0.01$). As expected, a negative correlation was observed with TOC ($r = -0.638$, $p < 0.01$) and a positive relationship with the sand aggregation index ($r = 0.524$, $p < 0.01$).

To better understand the relationships between soil erodibility and aggregate stability we plotted $K$ against total aggregates loss ($a + b$) in Fig. 3. The relationship 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 ($r = -0.758$, $p < 0.01$) and the fine sand content ($r = 0.601$, $p < 0.05$) for the whole dataset. Negative residuals (i.e. $K$ overestimation, typical of forest soils) corresponded therefore to higher coarse sand contents.

In Table 2 the other RUSLE factors and results are listed. The topographic factor LS (Table 2) showed high spatial variability, reflecting the complexity of the study area, and ranged from 0 to 25. RUSLE map is presented in Fig. 4. The erosion loss estimate $A$ ($\mathrm{Mg\,ha^{-1}\,y^{-1}}$) ranged from 0 (flat areas, with null LS value) to ca. 26 $\mathrm{Mg\,ha^{-1}\,y^{-1}}$ (average 5.51, SD 7.69 $\mathrm{Mg\,ha^{-1}\,y^{-1}}$) thus showing high spatial heterogeneity. Around 50 % of the area was interested by moderate to severe soil erosion (i.e. 5–100 $\mathrm{Mg\,ha^{-1}\,y^{-1}}$), 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 ($r = 0.412$, $p < 0.05$) in the whole dataset. However, the correlation coefficients were much higher where forests and pastures were evaluated separately ($r = 0.914$ and 0.963 for forests and pastures, respectively, $p < 0.001$). The LS factor (Table 2) did not show significant differences between vegetation covers, but the resulting erosion rate $A$ ($\mathrm{Mg\,ha^{-1}\,y^{-1}}$) was much greater for pasture ($p < 0.01$). If



the estimated erosion rates obtained from RUSLE application were plotted against aggregate stability, two different trends were visible (Fig. 5). In forests, aggregate stability did not explain the predicted soil erosion ($r = 0.18$, $p > 0.05$), while in pastures, about 57 % of the RUSLE $A$ variance was explained by aggregate losses ($r^2 = 0.573$, $p < 0.05$).

## 4 Discussion

In this work we wanted to assess the relationships between aggregate losses (wet sieving test) and both soil erodibility (RUSLE $K$ factor) and erosion rates ($A$, RUSLE estimate) 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 ($K$) was over/underestimated from aggregate stability under forest and pasture cover, respectively (Fig. 3). Moreover, the estimated erosion rate ($A$) 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 were deeply influenced by the soil organic matter content. Soil organic matter content was however not related with land cover, as visible from the lack of significant differences, probably because of the concomitant presence of morphology 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, nor 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; 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. 3), 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 losses 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. 4). 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 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 C is considered marginal. In our dataset, the determination coefficients of the regressions between organic matter and aggregate stability (Fig. 2) supported this hypothesis, as most of the variance of stability parameters (up to 93 %) was actually explained by SOM. 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 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 SD 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. The correlation found between aggregate stability and organic matter also held when the extractable carbon (TEC) was considered ($a+b$, $r = -0.690$, $p = 0.001$; $b$, $r = -0.656$, $p = 0.002$; $c$: $r = 0.755$, $p < 0.01$, data not shown). If the TEC content (instead of TOC) was considered as input parameter for $K$ calculation, the relationship between aggregate stability and erodibility disappeared. However, 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 herbaceous vegetation (Kalinina et al., 2011). The incorporation of organic matter into aggregates favors their stability 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 be at present 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 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, thus the surface 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 losses were found only for pastures (Fig. 5). As the LS was well correlated to $A$ in both land uses, the lack of dependence observed in forests points 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 by the RUSLE. 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 in agreement with the aggregate stability parameters derived from the wet sieving test, i.e. the most erodible soils in terms of $K$ value also displayed weaker structure and aggregation. The aggregate stability seems therefore a valuable indicator of the soil intrinsic susceptibility to erosion. However, land use dependent trends were observed in the estimate of $K$: forest soils always showed negative residuals and an opposite behavior was found in





pastures. 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. In addition, in forests, erosion estimate seemed particularly problematic also because of a high spatial variability of litter properties. Such aspects would need further investigation in order to better understand the mechanisms that determine the relationship between soil erodibility and structure for the different land uses.

*Author contributions.* Silvia Stanchi carried out GIS modelling, 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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**Table 1.** Selected soil properties at sampling points (slope sections used for RUSLE calculation).

| ID | Elevation (m a.s.l.) | Cover* | pH | Sand % | Silt % | Clay % | Total Organic C (g kg$^{-1}$) | TEC (g kg$^{-1}$) | $a$ (%) | $b$ (%) | $a+b$ (%) | $c$ (min) | Csand$_{H_2O_2}$/ Csand$_{Na}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 1242 | f | 7.3 | 70.9 | 23.0 | 6.1 | 53.3 | 30.1 | 13.7 | 32.1 | 45.8 | 28.14 | 0.34 |
| 2 | 1276 | f | 8.1 | 66.1 | 28.4 | 5.5 | 42.3 | 21.0 | 13.4 | 33.7 | 47.1 | 23.29 | 0.64 |
| 3 | 1336 | f | 7.3 | 71.1 | 22.2 | 6.7 | 21.8 | 14.0 | 14.9 | 39.5 | 54.4 | 16.41 | 0.83 |
| 4 | 2161 | p | 7.4 | 66.6 | 28.5 | 4.9 | 46.2 | 22.0 | 13.5 | 33.4 | 46.9 | 28.19 | 0.60 |
| 5 | 1329 | f | 7.2 | 72.1 | 22.2 | 5.7 | 16.3 | Nd | 4.9 | 52.5 | 57.4 | 10.25 | 0.99 |
| 6 | 1476 | f | 7.2 | 70.7 | 22.9 | 6.4 | 41.4 | 20.0 | 11.4 | 36.1 | 47.5 | 16.08 | 0.69 |
| 7 | 1593 | f | 8.2 | 65.1 | 28.2 | 6.7 | 23.5 | 13.0 | 15.1 | 38.3 | 53.4 | 17.01 | 0.80 |
| 8 | 1538 | p | 6.3 | 59.9 | 30.6 | 9.5 | 52.0 | 37.0 | 12.7 | 32.8 | 45.5 | 31.64 | 0.55 |
| 9 | 1862 | f | 7.6 | 60.6 | 31.5 | 7.9 | 33.5 | 24.0 | 16.5 | 35.1 | 51.6 | 23.57 | 0.69 |
| 10 | 2276 | p | 7.3 | 80.1 | 16.3 | 3.6 | 21.2 | 10.0 | 13.8 | 40.3 | 54.1 | 15.29 | 0.85 |
| 11 | 2104 | f | 7.5 | 69.5 | 23.1 | 7.4 | 47.0 | 19.0 | 13.2 | 32.4 | 45.6 | 26.77 | 0.63 |
| 12 | 1704 | f | 7.2 | 72.7 | 22.1 | 5.2 | 39.1 | Nd | 13.7 | 35.8 | 49.5 | 22.10 | 0.72 |
| 13 | 1725 | p | 7.7 | 65.6 | 26.7 | 7.7 | 36.5 | Nd | 15.1 | 35.4 | 50.5 | 20.10 | 0.73 |
| 14 | 1913 | f | 7.1 | 66.7 | 26.7 | 6.6 | 33.1 | 17.0 | 13.7 | 35.9 | 49.6 | 16.60 | 0.70 |
| 15 | 1710 | f | 7.0 | 65.2 | 28.0 | 6.8 | 23.8 | 16.0 | 15.4 | 38.6 | 54.0 | 17.96 | 0.78 |
| 16 | 2233 | f | 8.0 | 71.1 | 21.2 | 7.7 | 42.4 | 22.0 | 10.7 | 35.9 | 46.6 | 15.36 | 0.69 |
| 17 | 1631 | f | 7.2 | 59.8 | 33.1 | 7.1 | 29.5 | 21.0 | 10.0 | 42.0 | 52.0 | 14.02 | 0.72 |
| 18 | 2334 | p | 7.5 | 61.9 | 26.9 | 11.2 | 33.6 | 17.0 | 10.4 | 39.9 | 50.3 | 13.51 | 0.71 |
| 19 | 1978 | f | 7.6 | 63.9 | 27.3 | 8.8 | 47.6 | 21.0 | 12.6 | 32.3 | 44.9 | 19.35 | 0.59 |
| 20 | 2366 | p | 8.0 | 62.4 | 27.6 | 10.0 | 37.9 | Nd | 10.2 | 38.7 | 48.9 | 14.93 | 0.72 |
| 21 | 2261 | f | 7.7 | 58.7 | 30.9 | 10.4 | 27.8 | 10.0 | 10.8 | 41.5 | 52.3 | 12.77 | 0.83 |
| 22 | 2155 | p | 6.1 | 62.3 | 29.2 | 8.5 | 27.9 | Nd | 15.7 | 38.4 | 54.1 | 19.27 | 0.78 |
| 23 | 2067 | f | 6.5 | 66.5 | 27.0 | 6.5 | 42.2 | Nd | 14.9 | 34.3 | 49.2 | 26.67 | 0.66 |
| 24 | 1500 | p | 7.0 | 65.7 | 29.8 | 4.5 | 44.1 | 22.0 | 15.0 | 30.8 | 45.8 | 23.72 | 0.65 |
| 25 | 2459 | p | 7.5 | 65.5 | 24.8 | 9.7 | 40.5 | Nd | 12.2 | 36.0 | 48.2 | 16.71 | 0.74 |
| Average forest ($n$ = 16) | | | 7.4 | 67.0 | 26.1 | 6.97 | 35.3 | 18.99 | 12.8 | 37.2 | 50.1 | 19.1 | 0.68 |
| – | | | (0.4) | (4.5) | (3.8) | (1.3) | (10.7) | (5.20) | (2.8) | (5.09) | (3.7) | (5.36) | (0.41) |
| Average pasture ($n$ = 9) | | | 7.2 | 65.7 | 26.7 | 7.7 | 37.8 | 21.40 | 13.2 | 36.2 | 49.4 | 20.4 | 0.70 |
| – | | | (0.6) | (5.9) | (4.3) | (2.7) | (9.43) | (9.70) | (2.0) | (3.4) | (3.2) | (6.3) | (0.20) |

The column $a$ represent initial soil loss after water saturation, $b$ the loss for abrasion, $a+b$ the total aggregates loss. $c$ is the time parameter related to maximum aggregates loss.
Nd: not determined.
*f: forest, p: pasture.



**Table 2.** RUSLE factors at sampled points. $R$ and $P$ factors were constant for the area and therefore are not reported ($R$ = 1680 MJ mm ha$^{-1}$ h$^{-1}$ y$^{-1}$, $P$ = 1).

| ID | vegetation | $s$ (–) | $p$ (–) | $K$ (Mg ha h MJ$^{-1}$ mm$^{-1}$ ha$^{-1}$) | C (–) | LS (–) | $A$ (Mg ha y$^{-1}$) |
|---|---|---|---|---|---|---|---|
| 1 | f | 3 | 2 | 0.017 | 0.003 | 2.90 | 0.25 |
| 2 | f | 3 | 2 | 0.020 | 0.003 | 7.70 | 0.78 |
| 3 | f | 1 | 2 | 0.022 | 0.003 | 13.79 | 1.53 |
| 4 | p | 3 | 2 | 0.025 | 0.020 | 20.37 | 17.11 |
| 5 | f | 1 | 2 | 0.037 | 0.003 | 8.75 | 1.63 |
| 6 | f | 3 | 2 | 0.024 | 0.003 | 0.00 | 0.00 |
| 8 | f | 2 | 2 | 0.023 | 0.003 | 24.66 | 2.86 |
| 9 | p | 3 | 3 | 0.016 | 0.020 | 5.63 | 3.03 |
| 11 | f | 2 | 2 | 0.019 | 0.003 | 15.15 | 1.45 |
| 13 | p | 3 | 2 | 0.027 | 0.020 | 21.79 | 19.77 |
| 16 | f | 3 | 2 | 0.013 | 0.003 | 14.21 | 0.93 |
| 17 | f | 3 | 2 | 0.022 | 0.003 | 26.16 | 2.90 |
| 18 | p | 3 | 2 | 0.029 | 0.020 | 8.12 | 7.91 |
| 19 | f | 2 | 2 | 0.021 | 0.003 | 14.72 | 1.56 |
| 20 | f | 2 | 2 | 0.033 | 0.003 | 7.07 | 1.18 |
| 21 | f | 3 | 2 | 0.019 | 0.003 | 11.88 | 1.14 |
| 22 | f | 2 | 2 | 0.031 | 0.003 | 16.66 | 2.60 |
| 24 | p | 3 | 3 | 0.032 | 0.020 | 20.83 | 22.40 |
| 26 | f | 3 | 3 | 0.024 | 0.003 | 16.74 | 2.02 |
| 27 | p | 3 | 3 | 0.029 | 0.020 | 4.88 | 4.76 |
| 28 | f | 2 | 3 | 0.031 | 0.003 | 22.60 | 3.53 |
| 29 | p | 3 | 3 | 0.037 | 0.020 | 21.42 | 26.63 |
| 31 | f | 3 | 2 | 0.027 | 0.003 | 9.34 | 1.27 |
| 32 | p | 3 | 2 | 0.026 | 0.020 | 0.00 | 0.00 |
| 33 | p | 3 | 3 | 0.024 | 0.020 | 11.47 | 9.25 |
| Average forest ($n$ = 16) | | | | 0.024 (0.006) | 0.003 (–) | 13.27 (7.32) | 1.61 (0.98) |
| Average pasture ($n$ = 9) | | | | 0.027 (0.006) | 0.02 (–) | 12.72 (8.50) | 12.43 (9.57) |

**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 |





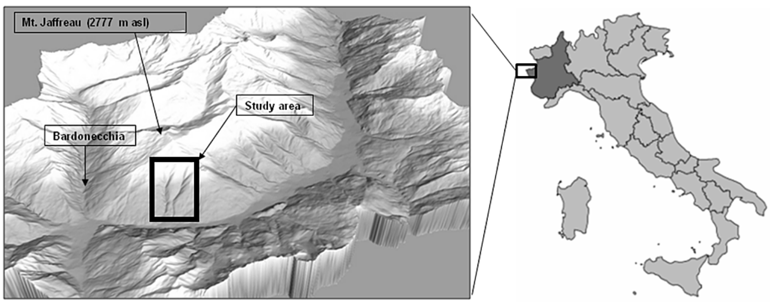


**Figure 1.** Digital elevation model of the study area (left) and catchment location.

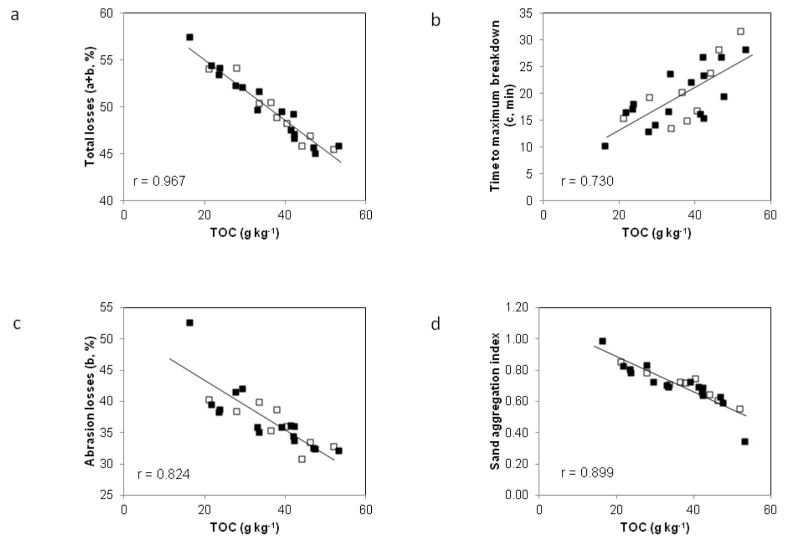


**Figure 2.** Relationships between organic C contents (TOC) and aggregation parameters. **(a)** Total losses of aggregates; **(b)** time to maximum breakdown; **(c)** abrasion losses; **(d)** sand aggregation index. Black squares correspond to forest, open squares to pasture.



Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper |

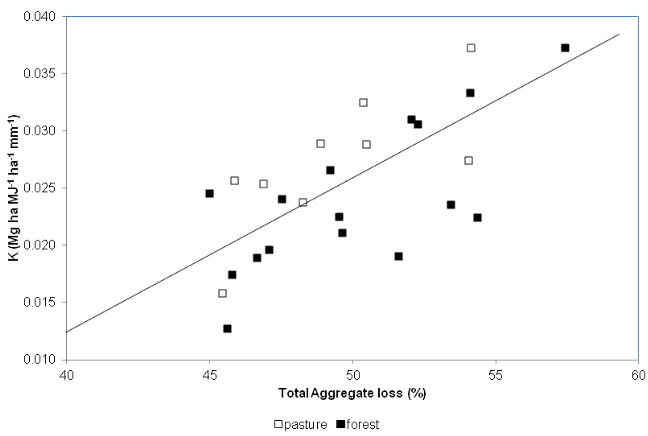


**Figure 3.** Plot of $K$ (Mg ha h MJ$^{-1}$ ha$^{-1}$ mm$^{-1}$) against total soil loss (%).

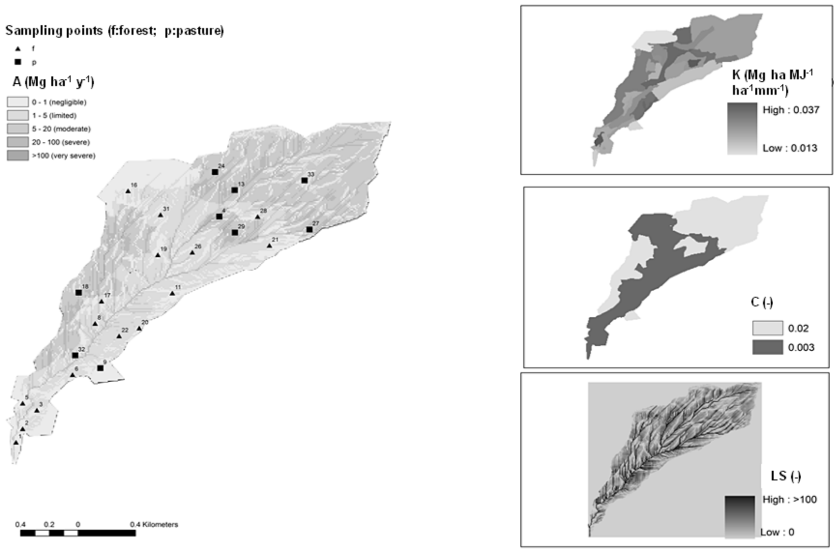


**Figure 4.** Map of RUSLE input factors and results.





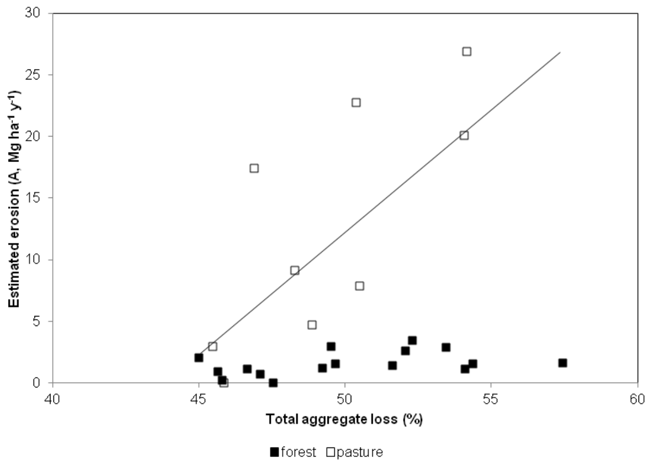


**Figure 5.** Relationship between estimated erosion (*A*) and aggregate breakdown.