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LAND DEGRADATION IN AGRICULTURAL LANDSCAPES – SOIL EROSION BY WATER

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CICLO XXIX

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Summary

Land degradation is a well-known problem throughout the world, due to its possible threat to land resources and strict connection with other global environmental issues such as biodiversity and climate. It is widely accepted that the major causes of land degradation include deforestation, soil erosion, overgrazing, inappropriate irrigation, abandonment and/or lack of maintenance of agricultural terraces, land use and cover change, especially because of urban sprawl and commercial development, soil pollution and quarrying. Among land degradation phenomena, soil erosion is one of the most significant issues that negatively influence the agricultural sector. In particular, soil erosion caused by water is one of the most important concern, especially in the Mediterranean area. Among the agricultural landscapes, vineyards deserve attention, because, not only they represent one of the most important crops in terms of income and employment, but they have also demonstrated to constitute, for the Mediterranean areas, the form of agricultural land use that has been causing the highest soil losses. Terraced vineyards deserve a particular mention too. In fact, they represent an important cultural heritage to preserve and if, if not properly maintained, can lead to local instabilities creating hazards for settlements and cultivations, and for the related economy. Although researchers have already dealt with the topic of soil erosion by water in agriculture, there are still some gaps in literature. The processes involved are complex and the analyses can be carried out at different spatial and temporal scales. Indeed, the lack of standardized procedures of collecting data and the variability of temporal and spatial conditions and measurement techniques for the analysis of soil water erosion processes require further research. To overcome these issues, this thesis aims to propose an integrated approach, by means of innovative remote-sensing technologies, field activities, and quantitative analyses to the investigation of soil erosion processes caused by water in agricultural landscapes. Furthermore, this thesis wants also to suggest a possible soil management technique, namely mulching, as an effective solution to mitigate soil and water losses in the before-mentioned environments. Among the remote-sensing technologies, light detection and ranging (LiDAR) and structure-from-motion (SfM) have been applied in this thesis. These have proven to be effective to obtain high-resolution Digital Elevation Models (DEMs). Experimental plots under simulated rainfalls have also been used to quantify and analyze the soil and water losses caused by water. Typical agricultural landscapes, especially Mediterranean vineyards, have been selected as study areas for this thesis.

Sommario

Titolo tesi: Degrado del suolo in ambienti agricoli – Erosione idrica del suolo

La degradazione del suolo è un problema conosciuto in tutto il mondo, a causa delle sue ripercussioni negative sulle risorse del territorio e della sua stretta connessione con altre questioni di tipo ambientale diffuse a livello globale quali la biodiversità ed il clima. E' ampiamente risaputo che le principali cause della degradazione del suolo sono la deforestazione, l'erosione del suolo, il pascolo intensivo, l'inadeguata irrigazione, l'abbandono e/o la scarsa manutenzione delle aree agricole terrazzate, il cambio di uso e copertura del suolo, soprattutto a causa dell'espansione urbana e dello sviluppo commerciale, l'inquinamento del suolo e le attività minerarie.

Tra i fenomeni di degradazione, l'erosione del suolo rappresenta uno degli aspetti che maggiormente influenzano negativamente il settore agricolo, ed in particolare, l'erosione idrica del suolo è uno di quelli che più colpisce il bacino del Mediterraneo. Tra i diversi tipi di colture, i vigneti sono quelli che meritano più attenzione. Infatti, non solo essi rappresentano una delle coltivazioni più redditizie, ma hanno anche dimostrato di essere caratterizzati, all'interno del bacino del Mediterraneo, dai più alti tassi di erosione del suolo. Anche i vigneti terrazzati meritano una menzione particolare. Essi costituiscono un importante patrimonio culturale da preservare e, se non vengono adeguatamente mantenuti, possono causare instabilità locali mettendo in pericolo la sicurezza dei centri abitati, e la produttività delle colture stesse, con conseguenze negative sulla relativa economia.

Nonostante l'erosione idrica del suolo in agricoltura sia già stato oggetto di studio da parte di molti ricercatori, permangono tuttavia delle lacune in letteratura. Infatti, i processi coinvolti sono complessi e le analisi che sono effettuate possono riguardare scale sia temporali che spaziali diverse. Per di più, la mancanza di procedure standardizzate per la raccolta dei dati e la variabilità delle condizioni temporali e spaziali, che influenzano i processi, e delle tecniche di misura utilizzate, fanno sì che ulteriori ricerche debbano essere portate avanti. Per ovviare a queste problematiche, questa tesi ha lo scopo di proporre un approccio integrato basato su tecnologie innovative e a basso costo per il telerilevamento, rilievi di campo, ed analisi quantitative al fine di studiare i processi di erosione idrica che caratterizzano gli ambienti agricoli. Inoltre, questa tesi si propone anche di suggerire una possibile tecnica di gestione del suolo, quale la pacciamatura, come soluzione efficace per attenuare le perdite di suolo ed acqua negli ambienti prima menzionati. Tra le tecnologie per il telerilevamento, il *LiDAR (light detection and ranging)* e la *SfM (structure-from-motion)* sono quelle che sono state impiegate in questa tesi. Queste tecnologie hanno dimostrato di essere efficaci al fine di ottenere modelli digitali dell'elevazione (*DEM – Digital Elevation Models*) ad alta risoluzione. Questa tesi ha visto l'impiego anche di parcelle sperimentali sottoposte a simulazioni di pioggia per quantificare ed analizzare le perdite di suolo ed acqua causate dalle precipitazioni. Le aree di studio per questa tesi sono state selezionate all'interno di tipici ambienti agricoli, specialmente vigneti situati nel bacino del Mediterraneo.

1 Introduction

1.1 State of the art

Land degradation is defined as ‘the reduction in the capacity of the land to provide ecosystem goods and services and to assure its functions over a period of time for the beneficiaries of these’ (Kellner et al., 2011). Land degradation has been originally associated with desertification and loss of biodiversity, particularly in arid, semi-arid and dry sub-humid areas, commonly called ‘drylands’ (Kellner et al., 2011; UNCCD, 1994). However, researches have gradually started to consider non-dryland areas too (Akhtar-Schuster et al., 2011; Kardjilov et al., 2006).

Recently, land degradation has increasingly been recognized to be a global issue because of i) its extent and proportion of the global population affected, ii) its inter-relation with other global environmental issues, and iii) its international and national environmental policy responses (Gisladottir and Stocking, 2005; Kellner et al., 2011). Land degradation is considered as a natural and socio-economic cause-effect phenomenon that is widespread over nearly 40 percent of the Earth’s surface. Its major causes are deforestation, poor farming practices leading to soil erosion, overgrazing, inappropriate irrigation, abandonment and/or lack of maintenance of agricultural terraces, land use and cover change, especially related to urban sprawl and commercial development, soil pollution, and quarrying (Abu Hammad and Tumeizi, 2012; Kellner et al., 2011; Tarolli et al., 2014). The driving forces and pressures are social, economic, ecological and physical but they act in an integrated way. In some cases, socio-economic factors are predominant in the occurrence of land degradation, for example, where urbanization and its related pollution take place (Prasad and Badarinh, 2004). In other cases, socio-economic factors have hampered the efforts to cope with land degradation (Akhtar-Schuster et al., 2011; Barrow, 1990; Bauer, 2006; Gebremedhin et al., 2003; Lubwama, 1999; UNCCD, 2009).

Soil erosion is recognized as one of the main driving factors that affect land degradation phenomena worldwide (Bai et al., 2013; Barton et al., 2004; Bhatt and Khera, 2006; Cerdà et al., 2009; Cerdan et al., 2010; García-Ruiz, 2010; Izzo et al., 2013; Ola et al., 2015; Sadeghi et al., 2015 a,b; Wang et al., 2013; Zhao et al., 2013; Zheng, 2006), and soil erosion by water is particularly one of the most significant types of erosion, especially in semi-arid and semi-humid areas of the world (Cerdà et al., 2009a; Cerdan et al., 2010; FAO, 2000; García-Ruiz, 2010; Mandal and Sharda, 2013; Martínez-Casasnovas and Ramos, 2006; Novara et al., 2016; Quinton and Catt, 2007; Ramos and Martínez-Casasnovas, 2004; Taguas et al., 2015; Verheijen et al., 2009; Verstraeten et al., 2003). Although soil erosion consists of natural processes, its severity mainly depends on anthropic activities such as i) overgrazing (Angassa, 2014; Mekuria and Aynekulu, 2013), ii) wildfires or controlled burnings (González-Pelayo et al., 2010; Robichaud, 2009), iii) mining (Martín-Moreno et al., 2016; Tarolli and Sofia, 2016), iv) quarries, waste disposal and construction sites (Albaladejo Montoro et al., 2000; Hayes et al., 2005), and v) unsustainable farming practices and land-use changes on large scales (Brevik, 2009; Cerdà et al., 2009a,b; Leh et al., 2013; Lieskovský and Kenderessy, 2014; Martínez-Casasnovas et al., 2016; Montgomery, 2007; Tarolli and

Sofia, 2016; Verheijen et al., 2009; Yuan et al., 2015). The effects of these anthropic activities have led to the definition of 'accelerated' soil erosion (Verheijen et al., 2009), which has received much concern due to both the increase of problems caused by the erosion itself, and the significant environmental and economic consequences (Cerdà et al., 2009a; Cerdan et al., 2010; Fernández-Calviño et al., 2012; Kellner et al., 2011; Ramos and Martínez-Casasnovas, 2004; Verheijen et al., 2009).

Soil erosion processes are recognized to cause both on-site and off-site effects. Among the on-site damages, are included: i) soil and nutrient losses (Corell et al., 1999; Douglas et al., 1998; Fernández-Calviño et al., 2012, 2013; Ramos and Martínez-Casasnovas, 2004; Steegen et al., 2001; Verstraeten and Poesen, 2002), ii) long-term productivity loss of degraded soils (Gunatilake and Vieth, 2000; Lal, 1995), iii) crop yield depression and iv) reduction in water storage capacity (Bakker et al., 2004; Boardman and Poesen, 2006; Pimentel et al., 1995). Off-site damages are linked to rills, ephemeral gullies and permanent gullies, because they all allow, one way or the other, the transfer of sediment particles with associated nutrients by runoff into the drainage networks and reservoirs (Avni, 2004, 2005; Daba, 2003; Eitel et al., 2002; Foster, 1986; Govers et al., 2004; Hansen et al., 2002; Nyssen et al., 2004; Verstraeten et al., 2003). Soil erosion by water may also have strong impact on other environmental and social issues such as pollution, inundations and muddy flows (Angás et al., 2006; Antoni et al., 2006; Auzet et al., 2006; Bechmann et al., 2009; Le Bissonais et al., 2002; OECD, 2001; Pieri et al., 2007; Schaffner et al., 2011). Moreover, if soil erosion is exacerbated to such an extent that it is able to modify the carbon dynamics, the environmental impact may be even broader (Bernoux et al., 2005; Lal, 2005; Robert, 2005).

Given the seriousness of this issue, the European Commission's Thematic Strategy for Soil Protection recommended an indicator-based approach for monitoring soil erosion. Indeed, defined baseline and threshold values have increasingly been considered essential for the evaluation of soil monitoring data. At this regard, Verheijen et al. (2009) proposed a modified definition of tolerable soil erosion and, for the European context, they estimated the upper limit of tolerable soil erosion, as equal to soil formation, to be ca. $1.4 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ while the lower limit to be ca. $0.3 \text{ Mg ha}^{-1} \text{ yr}^{-1}$. From their study, it emerged that actual soil erosion rates for tilled and cultivated European lands are, on average, 3 to 40 times greater than the upper limit of tolerable soil erosion. Montgomery (2007) confirmed the idea that erosion rates from conventionally plowed agricultural fields averaged 1-2 orders of magnitude greater than rates of soil production, erosion under native vegetation, and long-term geological erosion. His results proved that conventional plow-based agriculture increased erosion rates enough to be unsustainable. As a matter of fact, agricultural policies and practices have been playing a key role in soil erosion since the '60s. The intensification in agriculture resulted in increased farm and field sizes, greater mechanization, changed crop sequences and crop types, increased use of pesticides and herbicides, and extended use of conventional ploughing. All these factors have enhanced soil water erosion problems in agricultural lands (Arnáez et al., 2015; Bakker et al., 2005; Blavet et al., 2009; Cerdà et al., 2009a,b; Cerdan et al., 2010; Ciampalini et al., 2012; Cots-Folc et al., 2009; Johnsen et al., 2001; Laudicina et al., 2015; Prosdocimi et al., 2016a; Raclot et al.,

2009; Van-Camp et al., 2004). Figure 1 shows some typical examples of erosion processes caused by water and their consequences in agricultural lands.

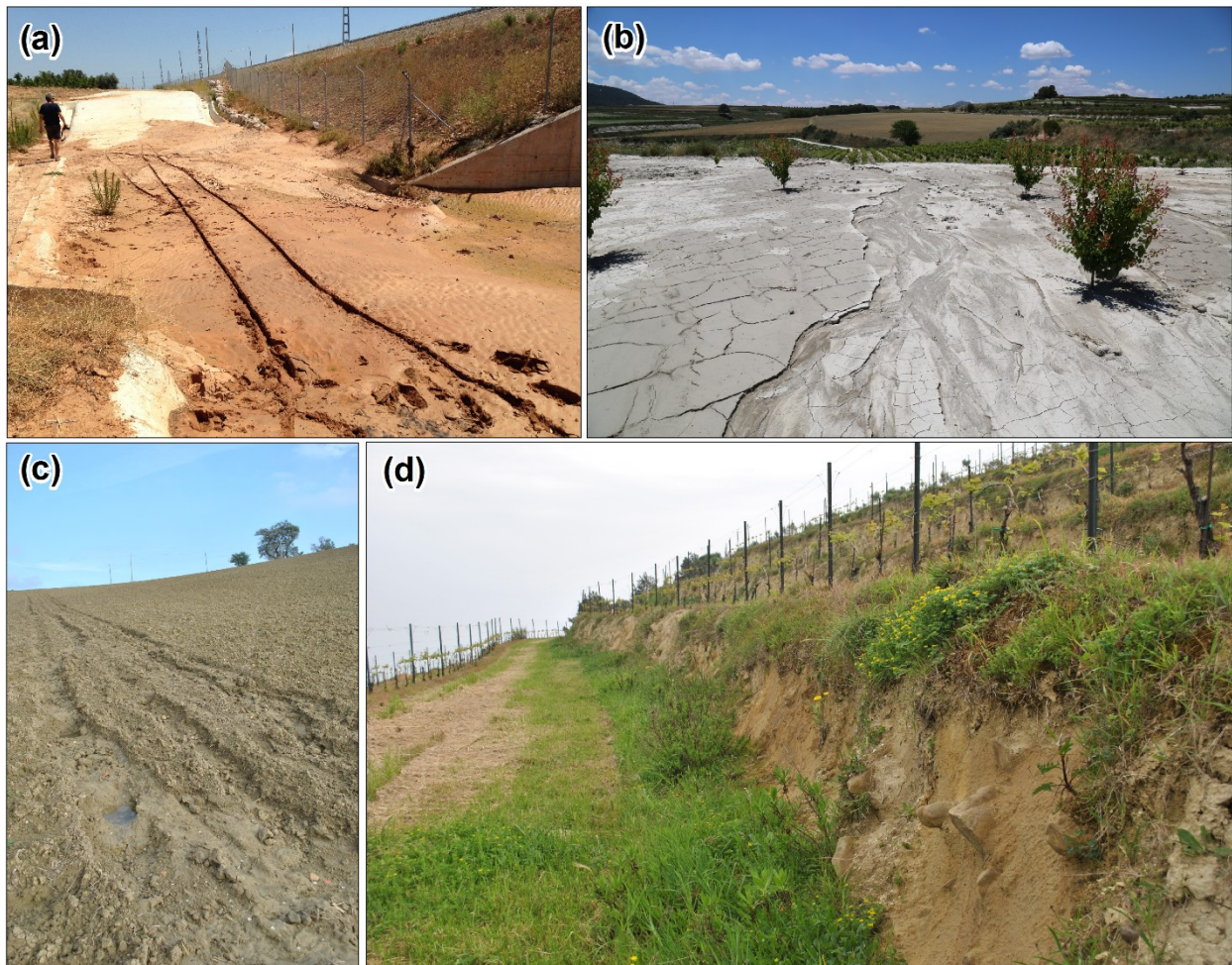


Figure 1 Evident examples of soil erosion processes caused by water and their consequences in agricultural lands in Eastern Spain (a,b) and Central Italy (c,d): significant deposition of soil in a road derived from an orange plantation (a), sheet erosion (b), rill erosion (c) and erosion affecting bench terraces (d) (photos by A. Cerdà, P. Tarolli and M. Prosdocimi).

Among the agricultural crops, vineyards are a matter of serious concern because they have proven to be the form of agricultural land that causes one of the highest soil loss in the Mediterranean areas (Cerdà and Doerr, 2007; Cerdan et al., 2002, 2010; García-Ruiz et al., 2010; Kosmas et al., 1997; Martínez-Casasnovas and Sánchez-Bosch, 2000; Martínez-Casasnovas and Ramos, 2006; Prosdocimi et al., 2016a; Raclot et al., 2009), and represent one of the most important crop in terms of income and employment (Anderson and Nelgen, 2011). Mediterranean vineyards are affected by high soil losses because of both natural and anthropogenic reasons (Prosdocimi et al., 2016a).

First, they are subject to intense rainfall events, especially in spring autumn and winter, and therefore, they are vulnerable to soil erosion processes (Beguiria and Vicente, 2006; Borga et al., 2011; Poesen and Hook, 1997; Ramos and Porta, 1994; Santos, 2000; Tarolli et al., 2012). Second, they are cultivated on steep hillsides (Arnáez et al.,

2007; Corti et al., 2011; Wichereck, 1993). Indeed, topography, intended as slope gradient and length, is one of the primary natural factors influencing soil erosion (Cerdan et al., 2010; Corti et al., 2011; Fox and Rorke, 1999; Hudson and Jackson, 1959; Koulouri and Giourga, 2007; McCool, 1982; Zachar, 1982; Morgan, 1986; Musgrave, 1947). As slope and the length of plot increase, the velocity of overland flow increases too and, hence, the erosive power and the capability of the flow to carry suspended sediments (Wischmeier and Smith, 1978). On large scale, topography is also recognized to strongly govern the spatial and temporal variability of soil water content within the top soil layer (Ramos and Martínez-Casasnovas, 2007).

In addition, soil properties also are recognized to enhance soil erosion on Mediterranean vineyards. Texture, organic matter content, stability of aggregates, rock fragments, soil chemistry, clay mineralogy, and soil moisture are all soil characteristics that influence erosion processes (Blavet et al., 2009; Cerdà, 1996; 2002; Corti et al., 2011; Lal, 1995; Novara et al., 2011; Ramos and Martínez-Casasnovas, 2007). The scarce presence of organic matter content in the Mediterranean soils result in a low stability of soil aggregates (Cerdà, 1996, 2002; Corti et al., 2011; Novara et al., 2011). Therefore, soil aggregates are used to collapse under the impact of raindrops, leading to the formation of soil sealing and crusts that reduce permeability and, in turns, favor runoff and the formation of rills and gullies (Robinson and Philips, 2001). This is in line with the general statement that erosion rate increases as the organic content decreases (Blavet et al., 2009; Prosdocimi et al., 2016a; Robinson and Philips, 2001). The scarce presence of organic matter content also influences the soil infiltration capacity and, thus, the water distribution within the soil profile (Ramos and Martínez-Casasnovas, 2007). Consequently, this results in a strong impact on runoff and soil erosion too.

Finally, the soil and water management techniques (SWCTs) (Maetens et al., 2012), the changes in land use and farming techniques and abandonment of agricultural land (Ballif, 1990; Blavet et al., 2009; Cerdà, 1994; Ferrero et al., 2005; Martínez-Casasnovas, 1998; Porta et al., 1994; Tarolli et al., 2014; Tropeano, 1984; Usón, 1998) are among the anthropogenic factors that further exacerbate soil erosion on Mediterranean vineyards. Starting from the SWCTs, they include: i) vegetation management (cover crops, mulching, grass buffer strips, strip cropping and enclosure), ii) soil management (no-tillage, reduced tillage, contour tillage, deep tillage, drainage and soil amendment), and iii) mechanical methods (terraces, contour bunds and geotextiles). In Mediterranean vineyards, tillage, where the weeds are usually removed mechanically, and no-tillage, where the weeds are usually removed chemically, are recognized to be the two most standard soil management techniques (Novara et al., 2011; Raclot et al., 2009). The common problem deriving from adopting these management techniques is usually a bare soil cover of the inter-rows during the whole year (Lasanta and Sobrón, 1998) and, consequently, a higher susceptibility of the soil to produce runoff and to be eroded by water. Alternative soil management techniques may consist of covering the inter-rows with straw mulching (Blavet et al., 2009; Carsouille et al., 1986; Gril et al., 1989; Louw and Bennie, 1991; Morvan et al., 2014; Prosdocimi et al., 2016c,d), rock fragments (Blavet et al., 2009; Nachtergaele et al., 1998), or in allowing a permanent or vegetation cover (Gril et al., 1989; Messer, 1980; Morvan et al.,

2014; Novara et al., 2011; Raclot et al., 2009; Schwing, 1978). The presence of a natural or permanent vegetation cover in the inter-rows always demonstrated to be recommended compared with bare soil. In fact, a vegetation cover facilitates the infiltration rate (Cerdà, 1998; Folorunso et al., 1992; Gulick et al., 1994; Ruiz-Colmenero et al., 2013), the stabilization of aggregates (Durán-Zuazo and Rodríguez-Plequezuelo, 2008; García-Orenes et al., 2005) and protects the soil surface from the impact of raindrops. Moreover, it enhances the organic matter formation and the microbiological function of soil (García-Orenes et al., 2009, 2010; Quinton and Catt, 2004). However, in dry areas especially during the summer, vegetation cover may be a problem because it competes with vines for water absorption. Therefore, it results to be a benefit only during the fall and period of winter frappe dormancy (Gristina et al., 2006).

Regarding the changes in land use and farming techniques, some authors stressed how the intensification of agriculture of the last four decades have had a strong and negative impact on soil erosion in Mediterranean vineyards (Cots-Folch et al., 2009; Lasanta and Sobrón, 1988; Martínez-Casasnovas and Sánchez-Bosch, 2000; Ramos and Martínez-Casasnovas, 2007). Larger fields with low to moderate slopes were needed to maximize wine production and therefore, land leveling and vine-rows along the slope replaced the traditional tillage techniques (Ramos and Porta, 1997) and terrace systems (Martínez-Casasnovas and Sánchez-Bosch, 2000; Tarolli et al., 2014, 2015). The benefits of terracing have been deeply discussed by Tarolli et al. (2014). In particular, terraces were built to retain more water and soil, to reduce the slope gradient and length and to intercept surface runoff, leading in this was to a reduction in soil erosion with positive effects on agriculture (Gachene et al., 1997; Li et al., 2014; Louwagie et al., 2011; Wakindiki and Ben-Hur, 2002; Tarolli et al. 2014). In Mediterranean areas, terraces are still considered one of the most important anthropic sign on the relief (Gallart et al., 1994; Trischitta, 2005) and they represent an important European heritage to preserve (Arnáez et al., 2011; Varotto, 2008). However, Tarolli et al. (2014) also highlighted the problem of terrace abandonment in relation to an increase of soil erosion risk. In fact, terrace abandonment or the lacks of maintenance cause an increase of erosion that can cause the terraces to collapse (Canuti et al., 2004; Crosta et al., 2003; Gallart et al., 1994; Lasanta et al., 2001). A more complete description of the effects of topography, soil properties, rainfall characteristics and SCT on soil erosion processes caused by water on Mediterranean vineyards can be found at Chapter 2.

Although the topic of soil erosion by water in agriculture has already been researched, it needs further investigation, because there are still some gaps due to the complexity of the processes involved, measurement techniques adopted, and the spatial and temporal variabilities of the analyses carried out (Prosdocimi et al., 2016a; García-Ruiz et al., 2015). Among the possible measurement methods, this thesis focuses on the use of experimental plots under simulated rainfalls (Figure 2a,b) and on innovative remote-sensing technologies which allow, through the construction of high-resolution Digital Elevation Models (DEMs), to detect the topographic changes over time (Figure 2c,d).

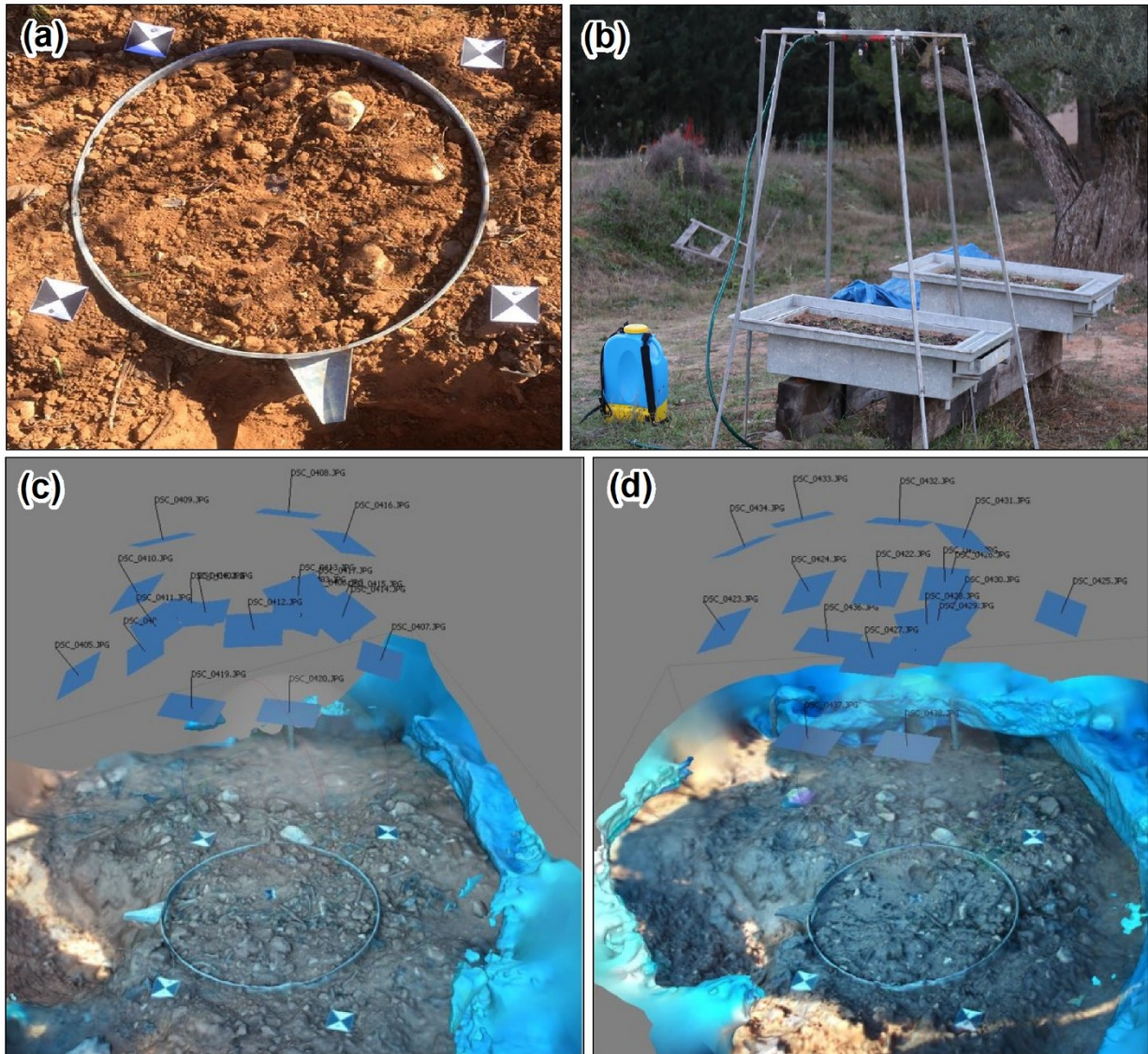


Figure 2 Examples of experimental plots installed: 0.25 m² circular plots (a), and 2 m² rectangular lifted up plots where the portable rainfall simulation, as described by Cerdà et al. (1997), is located (b). The use of structure-from-motion (SfM) technique to detect the surface elevation changes due to a simulated rainfall event: reconstruction of the pre- (c) and post-event (d) soil surfaces of the 0.25 m² plots. (photos by M. Prosdocimi).

The use of experimental plots under simulated rainfall has become a very effective method for quantifying soil erosion rates, particle detachment and overland flow in situ and in the laboratory (Arnáez et al., 2007; Blavet et al., 2009; Cerdà, 1999, 2001; Gril et al., 1989; Iserloh, et al., 2013; Jordán et al., 2010; Lasanta et al., 2000; Morvan et al., 2014; Tossel et al., 1987; Wainwright, 1996). Several types of rainfall simulators have been designed to meet the objectives of researchers (Iserloh et al., 2013; Lassu et al., 2015; Tossel et al., 1987). This latter approach has also been considered for this thesis where a portable rainfall simulator as described in Cerdà et al. (1997) was considered, because it revealed to be effective in rugged terrain conditions in semi-arid environments. Its detailed description can be found at Chapters 3 and 7.

Remote-sensing technologies have proven to be effective to achieve high-resolution DEMs (Tarolli, 2014; Tarolli et al., 2015). These can be used for quantifying the soil

erosion and deposition rates in agricultural environments (Martínez-Casasnovas and Sánchez-Bosch, 2000; Martínez-Casasnovas et al., 2002; Prosdocimi et al., 2015, 2017), and can also serve as basic topographic information to derive morphometric attributes, such as the roughness index (RI) (Cavalli et al., 2008) and the index of connectivity (IC) (Cavalli et al., 2013). Further information can be found at Chapters 3, 4 and 5. Among the most significant remote-sensing technologies, light detection and ranging (LiDAR) and structure-from-motion (SfM) have proven to be the ones that have been successfully applied in several environments to obtain high-resolution DEMs (Dandois and Ellis, 2013; Eltner et al., 2015; James and Robson, 2012; Jones et al., 2007; Masiero et al., 2015; Notebaert et al., 2009; Passalacqua et al., 2014; Piermattei et al., 2016; Prosdocimi et al., 2015, 2017; Sofia et al., 2014a,b, 2016; Tarolli, 2014; Tarolli et al., 2015; Westoby et al., 2012; Whitehead et al., 2013; Woodget et al., 2015). Lidar is an active remote sensing technology that consists in a laser scanner that emits intense, focused beam of light and measures the time it takes for the reflections to be detected by the sensor. The laser scanner unit consists of a pulse generator of laser with a wavelength in the range of 0.8 μm to 1.6 μm (near-infrared) and a receiver to get the signal of scattered and reflected pulses from targets (Mukai et al., 2006; Pfeifer and Brieese, 2007; Wehr and Lohr, 1999). The lidar survey can be distinguished based on the platform on which the sensor has been installed: Airborne Laser Scanning (ALS - Wehr and Lohr, 1999) refers to surveys, in which lidar sensor is mounted on airplanes or helicopters, while Terrestrial Laser Scanning (TLS) on ground-based, fixed positions. In the former, as the instrument is in motion, the aircraft trajectory and its absolute position are needed. To achieve this, the laser scanner unit is coupled with a Global Positioning System (GPS) receiver and an Inertial Measurement Unit (IMU) (Habib et al., 2005; Hollaus et al., 2005; Pfeifer and Brieese, 2007; Reutebuch et al., 2005; Webster and Dias, 2006) (Figure 3).

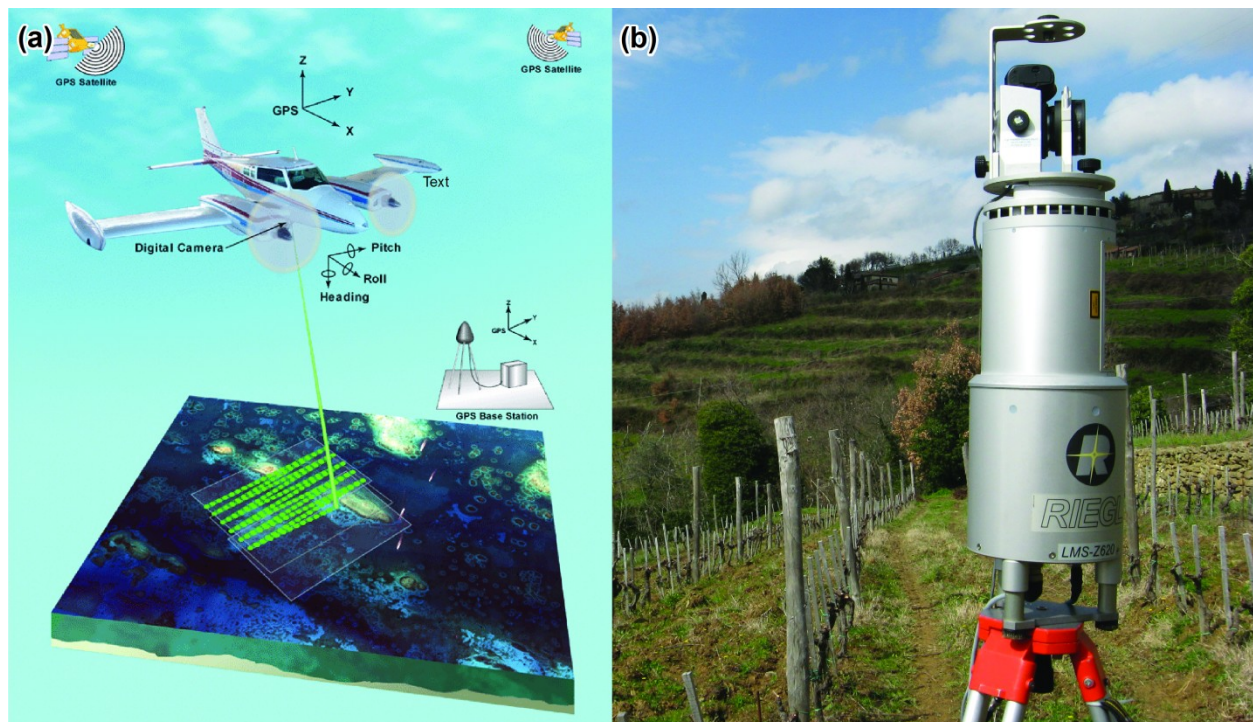


Figure 3 Schematic components of an ALS survey: scanning laser unit, scan patterns on the ground (green) and aircraft positioning and altitude measurement systems (DGPS for aircraft position and an inertial measurement unit -IMU for recording pitch, yaw and roll of the aircraft (Wehr and Lohr, 1999) (a) (picture taken from <http://www.geolidar.it/dettagliotecnologia.php?idtecnologia=11>). TLS survey carried out in a terraced vineyard located in Tuscany (Lamole case study - see Chapt. 5.3.2.2) (b) (photo by S. Calligaro).

This makes an ALS survey more complicated to carry out than a TLS one. However, the scanning of impervious areas through the use of a TLS may reveal to be challenging, especially for what concerns the transport of the equipment needed. Furthermore, due to the fixed position of the laser scanner, multiple scans from different positions (Scan Positions - SP) may be required (Buckley et al., 2008; Dassot et al., 2011). During the last decades, lidar data have been used in a wide of areas, such as building extraction and 3D urban modeling, hydrological modeling, glacier monitoring, landform or soil classification, river bank or coastal management, and forest management. However, terrain modeling has been the primary focus of most lidar surveys (Tarolli, 2014). The literature has flourished with works using high-resolution topography derived from lidar for both natural and engineered landscapes (Casas et al., 2012; Grove et al., 2013; Heritage and Hetherington, 2007; Passalacqua et al., 2012; Schaefer and Inkpen, 2010; Sofia et al., 2014a,b; Tarolli, 2014). However, lidar technology remains still expensive and unwieldy and are thus not suitable for every environment. Much interest has risen, therefore, in obtaining high-resolution DEMs using more flexible and affordable technologies. At this regard, SfM has proven to be a valid and cheaper alternative to lidar for topographic data acquisition (Carrivick et al., 2013; Eltner et al., 2015; Fonstad et al., 2013; Hugenholtz et al., 2013; Piermattei et al., 2015; Prosdocimi et al., 2015, 2017; Westoby et al., 2012; Woodget et al., 2015). It is a digital low-cost photogrammetric technique, which uses overlapping, unconstrained 2D images to

create 3D models (Koenderink and Van Doorn, 1991). Differently from stereo vision, SfM uses a single, moving camera. The movement between sequential images creates enough parallax to infer 3D data. The working principles of SfM are similar to those of stereoscopic photogrammetry, namely that the 3D model can be created from overlapping, offset images. However, in traditional photogrammetry either the position of the camera or the positions of some points are known prior to scene reconstruction (Fonstad et al., 2013; Verhoeven et al., 2012; Westoby et al., 2012). On the contrary, in the SfM, matches are made between points across many photographs without prior knowledge of the camera position (Lowe, 2004) (Figure 4). Further information about this technique can be found at Chapters 3 and 4.

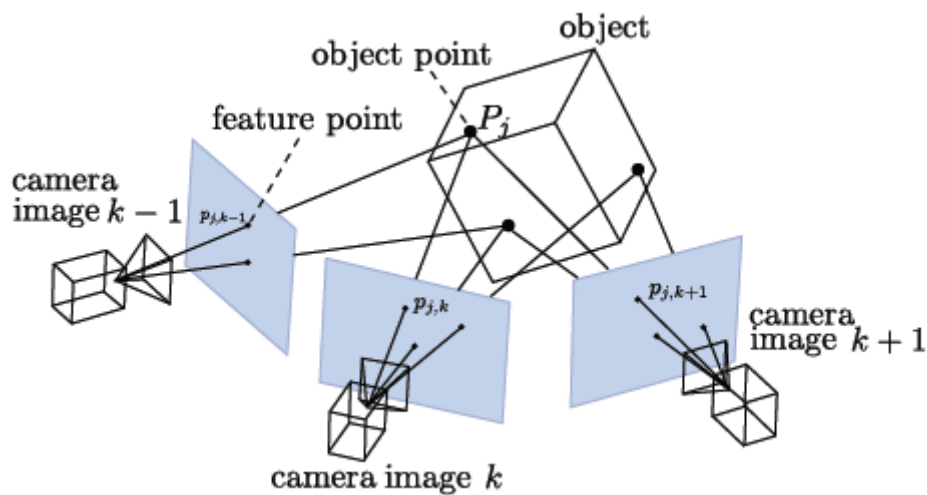


Figure 4 The SfM concept, and 3D model reconstruction from a sequence of overlapping images by using feature detection and matching algorithm (picture taken from <http://openmvg.readthedocs.io/en/latest/openMVG/sfm/sfm/>).

The interest for SfM technique is continuously increasing because can even rely on built-in cameras of smartphones (Micheletti et al., 2014; Prosdocimi et al., 2015, 2016b). Furthermore, the possibility of being used within different sectors (agricultural, geomorphological, environmental, and engineering), thanks to the recent development and diffusion of unmanned aerial vehicles (UAVs) (e.g. Colomina and Molina, 2014, Chen et al., 2015), make the SfM technique even more competitive and versatile.

Given the seriousness of soil water erosion in agricultural landscapes, adequate soil management practices are required to contribute to a more sustainable agriculture. The Food and Agriculture Organization of the United Nations (FAO), among others, has started to promote the concept of conservation agriculture (CA) (see FAO CA web site: <http://www.fao.org/ag/ca/1a.html>). CA is based on three main crop management principles: (i) minimal soil disturbance (no-till), (ii) permanent soil cover derived by crop residues (mulch) or cover crops, and (iii) crop rotation (FAO, 2011; Hobbs et al., 2008). Therefore, alternative and more conservation-minded soil management practices have been promoting. These include catch crops (Bonfante et al., 2015; García-Orenes et al., 2009), mulching (Costantini et al., 2015; Jordán et al., 2011; Prosdocimi et al., 2016b,c), hydromulching (Prats et al., 2013), geotextiles (Giménez-Morera et al., 2010), natural

grassing (Raclot et al., 2009) and rock fragments (Blavet et al., 2009). Among these, this thesis focuses on mulching, because it has proven to be particularly effective in reducing water and soil loss rates and improving soil condition (Cerdà et al., 2016; Cook et al., 2006; García-Orenes et al. 2009; Jordán et al., 2010; Mwango et al., 2016; Mulumba and Lal, 2008; Prosdocimi et al., 2016b,c; Sadeghi et al., 2015a; Tebrügge and Düring, 1999; Winteraeken and Spaan, 2010).

Several materials can be used for the mulching practice, such as vegetative residues, biological geotextiles, gravel and crushed stones (Blavet et al., 2009; Cerdà, 2001; Gilley et al., 1986b; Jiménez et al., 2016; Jordán et al., 2010; Keizer et al., 2015; Mandal and Sharda, 2013; Robichaud et al., 2013a,b; Smets et al., 2008; Xu et al., 2012; Zhao et al., 2015). Figure 5 shows examples of four types of vegetative residues that are used as mulch to reduce soil water erosion in agricultural environments: straw mulching (a), and mulching with prunings (b) applied to vine inter-rows, mulching with chopped prunings (c) in an apricot orchard, and mulching with maize residues in a soy field (d).

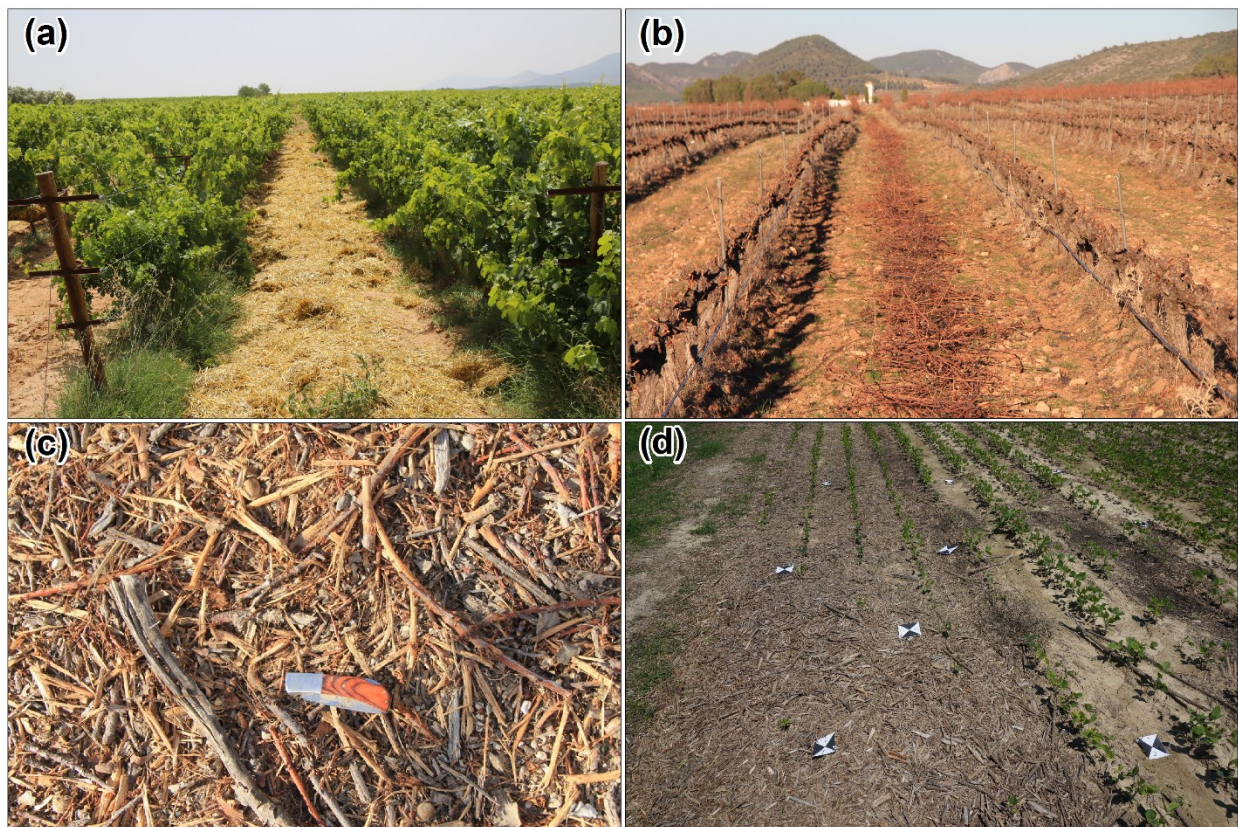


Figure 5 Straw mulching (a), and mulching with prunings (b) applied to vine inter-rows, mulching with chopped prunings (c) in an apricot orchard, and mulching with maize residues in a soy field (d). (photos by A. Cerdà and M. Prosdocimi).

The vegetative residues have particularly proven to be very effective at reducing the soil erosion rates and water losses in agricultural lands, rangelands, fire-affected areas and anthropic sites (Bautista et al., 1996; Cook et al., 2006; Fernández et al., 2012; Hayes et al., 2005; Prats et al., 2014; Prosdocimi et al., 2016b; Robichaud et al., 2013a,b;

Sadeghi et al., 2015a; Shi et al., 2013). A detailed description of the effects of mulching practice for reducing soil water erosion can be found at Chapters 6 and 7.

1.2 Research questions and objectives

This thesis proposes an integrated approach, by means of innovative remote-sensing technologies, especially those relying on low-cost and fast techniques, field activities, and quantitative analyses to the investigation of soil erosion processes caused by water in agricultural landscapes. Furthermore, this thesis wants also to suggest a possible soil management technique, namely mulching, as an effective solution to mitigate soil and water losses in the before-mentioned environments.

This thesis relies on the following research questions and specific objectives (Figure 6). From these, six papers have been obtained and included in this thesis, whose format is described in detail in Chapter 1.3.

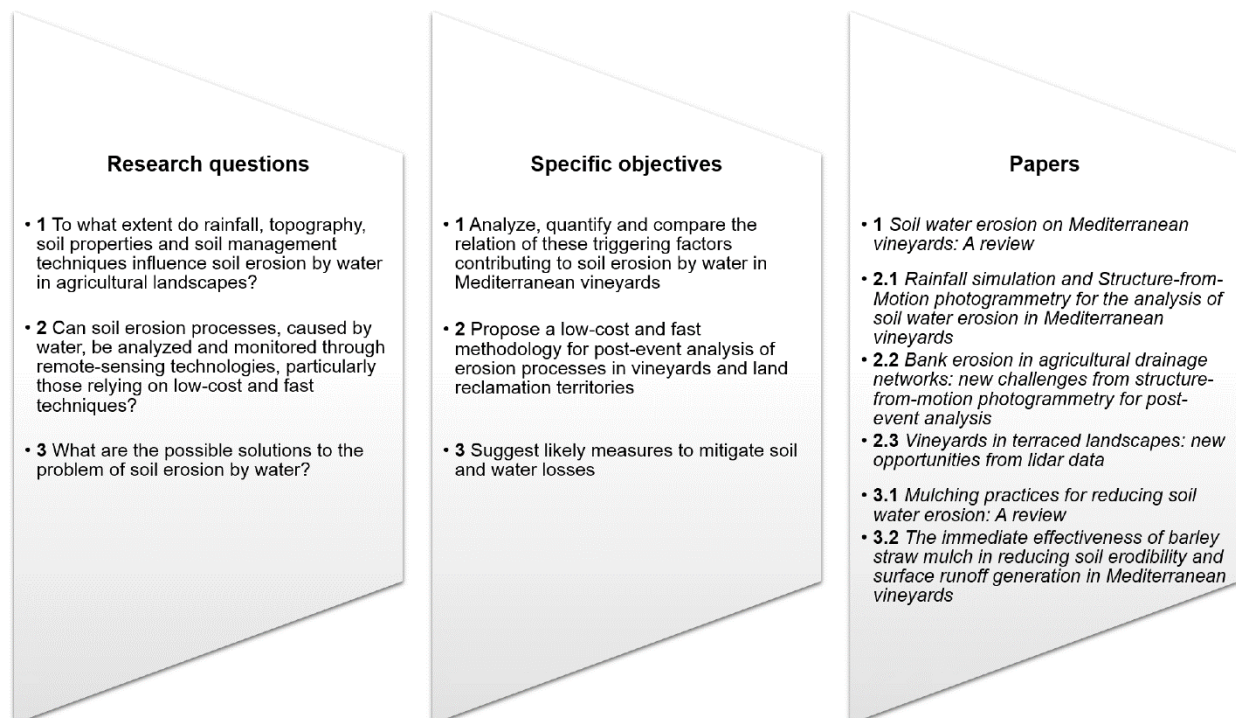


Figure 6 Research questions, specific objectives and corresponding papers of this thesis.

1.3 Explanation of thesis format

This thesis consists of six papers coming from my PhD activity (Chapters from 2 to 7). The first paper (Chapter 2) has been published in *Catena* journal (IF: 2.612) in 2016 and is a review article about the soil water erosion in Mediterranean vineyards. It provides with the state of the art of the problem by analyzing, quantifying and comparing the relation of the main triggering factors, which contribute to this issue. It also suggests, as future guidelines, the use of remote-sensing technologies for a better understanding of the processes involved.

The second paper (Chapter 3) has been accepted in *Science of the Total Environment* journal (IF: 3.976) in 2016 and published online as 2017. It proposes the combination of two methodologies for the analysis of soil water erosion processes in Mediterranean vineyards. These two methodologies include rainfall simulation and surface elevation change analyses based on high-resolution Digital Elevation Models (DEMs) derived from innovative remote-sensing technologies such as Structure-from-Motion (SfM).

The third paper (Chapter 4) has been published in *Earth Surface Processes and Landforms* journal (IF: 3.505) in 2015 and provides the application of remote-sensing technologies for post-event analysis of erosion processes in land reclamation territories.

The fourth paper (Chapter 5) has been published in *Land Degradation & Development* journal (IF: 8.145) in 2015 and proposes the use of remote-sensing technologies for a better understanding of possible terraces and roads' failures in Mediterranean vineyards.

The fifth paper (Chapter 6) has been published in *Earth-Science Reviews* journal (IF: 6.991) in 2016 and it is a review article about the effectiveness of mulching practice to mitigate the problem of soil erosion by water.

The sixth paper (Chapter 7) has been published in *Science of the Total Environment* journal (IF: 3.976) in 2016 and proposes a practical application of mulch for reducing soil erodibility and surface runoff in Mediterranean vineyards.

A *Summary* (the Italian version *Sommario* is included too) and an *Introduction* (Chapter 1) to the thesis precede the papers. The *Introduction* consists in three sub-chapters: (1.1) *State of the art*, (1.2) *Research questions and objectives* and (1.3) the current chapter *Explanation of thesis format*.

The thesis concludes with the *Conclusion* (Chapter 8) and *References* (Chapter 9).

2 Soil water erosion on Mediterranean vineyards: A review ¹

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2.1 Abstract

Soil water erosion on cultivated lands represents a severe threat to soil resources in the world, and especially in Mediterranean areas, due to their topographic, edaphic and climatic conditions. Among the cultivated lands, vineyards deserve a particular attention because, aside representing one of the most important crops in terms of income and employment, they also have proven to be the form of agricultural use that causes one of the highest soil losses. Although the topic of soil water erosion on vineyards has been studied, it still raises uncertainties, due to the high variability of procedures for data acquisition, which consists into different scales of analysis and measurement methods. There is still a great gap in knowledge about the effect of triggering factors on soil water erosion and, so far, an effort to quantify this effect on the Mediterranean viticulture has not been made yet. Therefore, this paper review aims to (i) develop a documented database on splash, sheet and rill erosion rates in Mediterranean vineyards, (ii) identify and, if possible, quantify the effect of triggering factors such as topography, soil properties, rainfall characteristics and soil conservation techniques on soil water erosion, and (iii) provide suggestions for a more sustainable viticulture. Although the large variability of data, some general trends between erosion rates and triggering factors could be found, as long as data were categorized according to the same measurement method. However, no general rule upon which to consider one triggering factor always predominant over the others came out. This paper review supports the importance of monitoring soil water erosion by field measurements to better understand the relationship between the factors. However, protocols should be established for standardizing the procedure of collecting data and reporting results to enable data comparison among different study areas.

2.2 Introduction

Throughout the world, the topic of soil water erosion on cultivated lands has received much concern, due to both the increase of problems caused by the erosion itself and the significant environmental and economic consequences (Cerdà et al., 2007, 2009; Fernández-Calviño et al., 2012; Galati et al., 2015; Gunatilake and Vieth, 2000; Leh et al., 2013; Lieskovský and Kenderessy, 2014; Mandal and Sharda, 2013; Martínez-Casasnovas and Ramos, 2006; Quinton and Catt, 2007; Ramos and Martínez-Casasnovas, 2004; Verstraeten et al., 2003; Zhao et al., 2013; Tarolli and Sofia, 2016). In Europe, 12% of the emerged lands are estimated to be subject to erosion by water (CEC, 2006), which is considered to be one of the most critical forms of soil degradation (FAO, 2000), capable of causing both on-site and off-site (Antoni et al., 2006; Corell et al., 1999; Douglas et al., 1998; Fernández-Calviño et al., 2012, 2013; Pieri et al., 2007; Pimentel et al., 1995; Ramos and Martínez-Casasnovas, 2004; Steegen et al., 2001; Verstraten and Poesen, 2002; Verstraeten et al., 2003). Fig. 1 shows some typical erosion processes caused by water affecting agricultural lands.

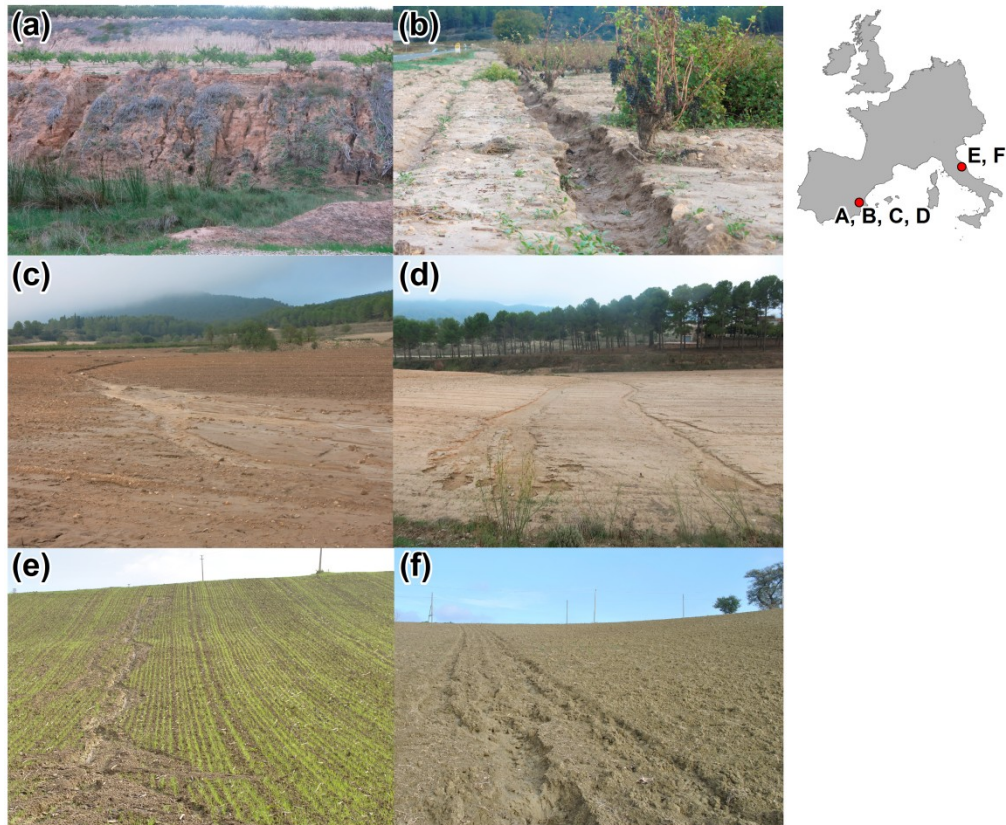


Fig. 1. Evident examples of soil erosion processes on agricultural lands in Eastern Spain and province of Ancona (Marche region, central Italy): (a) badlands, (b) gully erosion, (c), (d) and (e) sheet erosion and (f) rill erosion (photo by A. Cerdà and M. Prosdocimi).

In terms of economic consequences, soil erosion by water on cultivated lands may cause annual costs that have been estimated to be £205 million in England and Wales in 2009 (Verheijen et al., 2009) and \$44 billion in the U.S.A. in 1995 (Pimentel et al., 1995). The strong impact of soil erosion on society has raised the need for severe threshold values against which to assess the soil monitoring data. At this regard, Verheijen et al. (2009) proposed a modified definition of tolerable soil erosion as ‘any actual erosion rate at which a deterioration or loss of one or more soil functions does not occur’. For Europe, they estimated the upper limit of tolerable soil erosion, as equal to soil formation, to be ca. 1.4 Mg ha⁻¹ yr⁻¹ while the lower limit to be ca. 0.3 Mg ha⁻¹ yr⁻¹. Having said that, actual soil erosion rates for tilled, cultivated lands in Europe resulted to be, on average, 3 to 40 times greater than the upper limit of tolerable soil erosion (Verheijen et al., 2009). Montgomery (2007) adopted and updated the approach of Bennett and Lowdermilk (1938) against which to evaluate sustainable and tolerable soil erosion rates. His results quantitatively confirmed the contention that erosion rates from conventionally plowed agricultural fields averaged 1–2 orders of magnitude greater than rates of soil production, erosion under native vegetation and long-term geological erosion. This indicated that conventional plow-based agriculture increased erosion rates enough to prove unsustainable (Montgomery, 2007). Among the cultivated lands, vineyards merit a particular attention, because, aside from representing one of the most important crops in terms of income and employment (Anderson and Nelgen, 2011), they also constitute, for the Mediterranean areas, a form of agricultural land use that causes the highest soil losses (Cerdà and Doerr, 2007; Cerdan et al., 2002, 2010; García-Ruiz, 2010; García-Ruiz et al., 2010; Kosmas et al., 1997; Martínez-Casasnovas and Sánchez-Bosch, 2000; Martínez-Casasnovas and Ramos, 2006; Raclot et al., 2009; Tropeano, 1983). This can be explained by a combination of natural and anthropogenic factors. First, Mediterranean vineyards are usually located on steep slopes (Arnáez et al., 2007; Wichereck, 1993) and, therefore, they are more susceptible to soil water erosion (Corti et al., 2011). In fact, topography is recognized to be one of the most significant factors affecting soil water erosion (i.e. Cerdan et al., 2010; Koulouri and Giourga, 2007; Musgrave, 1947) and the transport of sediments. Then, Mediterranean vineyards are usually characterized by poor nutrient and organic matter contents (Cerdà, 1996; Ibáñez et al., 1996; Muñoz-Rojas et al., 2013; Novara et al., 2011). This is mainly due to the climatic conditions of the Mediterranean but also as a consequence of erodible soils and parent materials. At this regard, the Plio-Pleistocene marine sediments are recognized to be very susceptible to soil water erosion (Corti et al., 2011). Soils evolved from these fine-textured marine sediments are characterized by low stability of aggregates and low organic matter. Consequently, these weak soil aggregate collapse under the impact of raindrops, leading to the formation of soil sealing and crusts that reduce permeability and, in turns, favour runoff and the formation of rills (Robinson and Phillips, 2001). Furthermore, Mediterranean vineyards have to bear high intensity rainfall events, mainly concentrated in spring and autumn that are recognized to have an influence on soil water erosion and runoff processes (Borga et al., 2011; Tarolli et al., 2012). In addition, in Mediterranean vineyards the soil under the vines is usually artificially maintained without plant cover, leaving large areas

exposed to the rainfall (Arnáez et al., 2007; García-Ruiz, 2010; Novara et al., 2011; Tropeano, 1984). In fact, the two most common soil conservation techniques (SCTs) are considered to be tillage (mechanical weeding) and no-tillage (chemical weeding) (Raclot et al., 2009; Novara et al., 2011), and both of them generally turn out in bare soil management during the whole year (Lasanta and Sobrón, 1988). Fig. 2 shows two examples of Mediterranean vineyards affected by soil water erosion. Fig. 2a was taken in a terraced vineyard located in the Marche region, central Italy. Here, the wrong positioning of the drainage system facilitated the formation of surface wash that eroded the terrace risers, exposing in this way the roots of vines (black arrow). On the other hand, Fig. 2b was taken in an almost flat vineyard located in the province of Valencia, Spain. Here, a severe rainstorm caused sheet erosion processes, as pointed out by the black arrow, that were enhanced even by the absence of grass cover in the inter-rows and by the early stage of the vineyard implant.

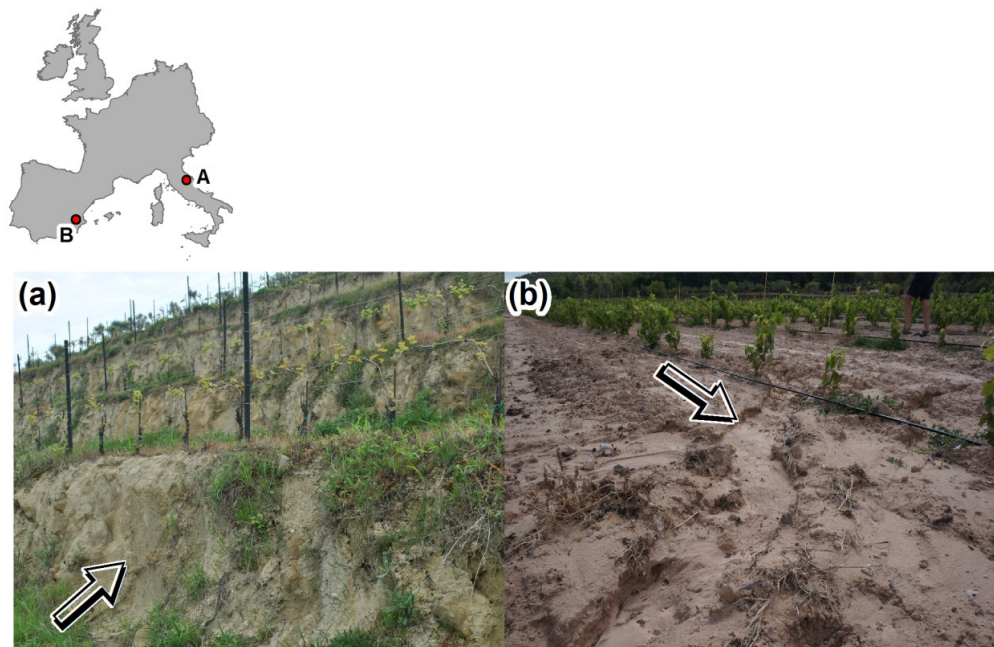


Fig. 2. Evident signs of erosion caused by water in two different vineyards, located in the province of Pesaro (Marche region, central Italy) (a) and in the province of Valencia (Spain) (b), respectively (photo by M. Prosdocimi and P. Tarolli).

Alternative systems to chemical weed control in the vine inter-rows are straw and prunings mulching (Blavet et al., 2009; Carsouille et al., 1986; Cerdà et al., in press. Gril et al., 1989; Keesstra et al., 2016; Louw and Bennie, 1991; Morvan et al., 2014; Prosdocimi et al., 2016; Sadeghi et al., 2015; Tejada and Benítez, 2014), natural or permanent grassing (Gril et al., 1989; Messer, 1980; Morvan et al., 2014; Novara et al., 2011; Raclot et al., 2009; Schwing, 1978) and covering the soil with rock fragments (Blavet et al., 2009; Nachtergaele et al., 1998). Fig. 3 shows an example of three alternative systems to chemical weed control and reduction of soil erosion in the vine inter-rows: chopped prunings mulching (3a), permanent grassing (catch crops) (3b) and straw mulch cover (3c).

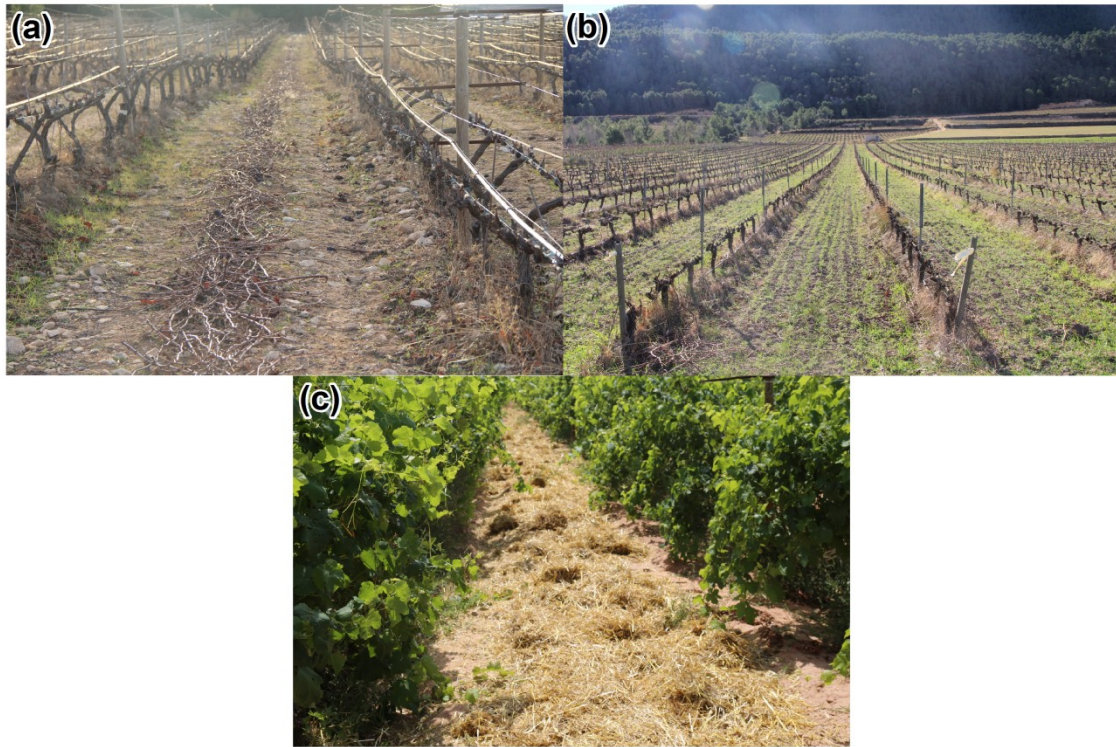


Fig. 3. Chopped pruned branches and straw mulch left in the inter-rows of a vineyard to reduce soil erosion at Celler del Roure in Les Alcusses de Moixent, (Province of Valencia, Spain) (a and c, respectively) and inter-rows covered with catch crops such as legumes at Casa Pago Gran in Les Alcusses de Moixent (Province of Valencia, Spain) (b) (photos by A. Cerdà).

Furthermore, the abandonment of land, land use changes and farming techniques have accelerated erosion processes in vineyards too (Ballif, 1990; Blavet et al., 2009; Cerdà, 1994; Dunjó et al., 2003; Ferrero et al., 2005; García-Ruiz, 2010; Martínez-Casasnovas, 1998; Novara et al., 2011; Porta et al., 1994; Tarolli et al., 2014; Tarolli et al., 2015; Usò, 1998). At this regard, García-Ruiz (2010), focusing on the effects of land uses on soil erosion in Spain, highlighted how vineyards expanded to steep slopes, sometimes on new unstable bench terraces, leading to increased soil erosion. Another problem related to soil water erosion in Mediterranean vineyards is the variability among the measurement methods applied to study it. The lack of a standardized procedure of collecting data and reporting results make data comparison among different study sites very difficult (García-Ruiz et al., 2015). Most of the soil erosion research studies are based on experimental plot stations that measure the soil erosion rates under simulated or natural rainfalls (Arnáez et al., 2007; Blavet et al., 2009; Cerdan et al., 2010; Rodrigo Comino et al., 2015, 2016; Corti et al., 2011). Fig. 4 shows an example of experimental plots used to measure surface wash and sediment yield under simulated (4a, b) and natural rainfall (4c, d).

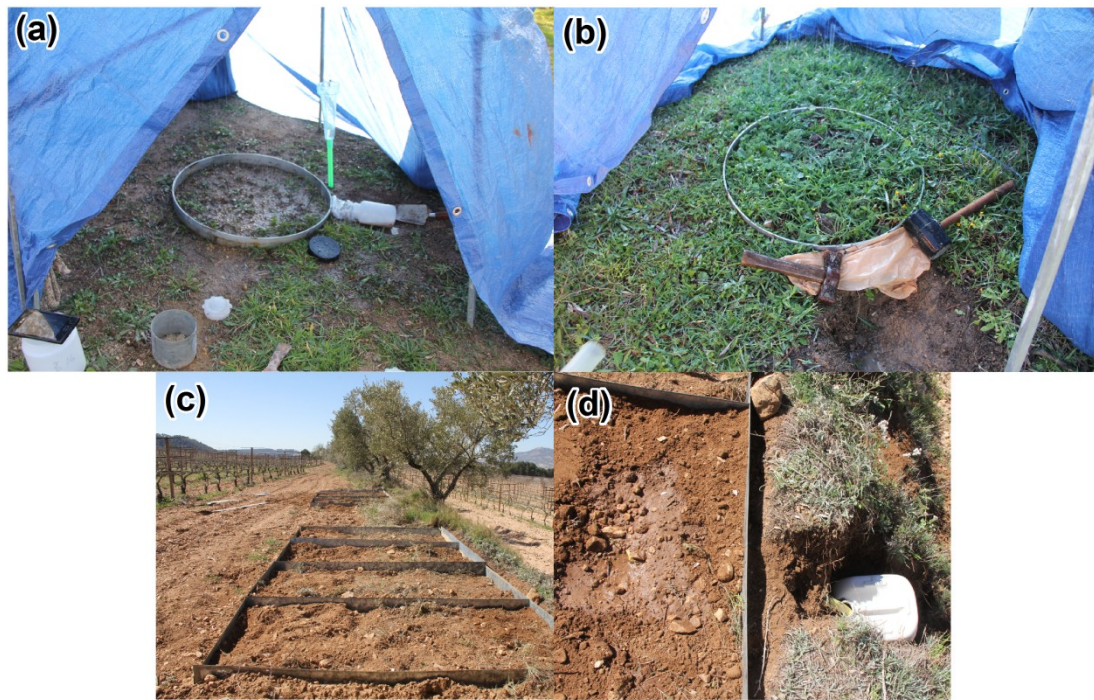


Fig. 4. Experimental plots installed for measuring surface wash and sediment yield under simulated (a, b) and natural rainfalls (c, d) at the soil erosion research station of El Teularet and at the Celler del Roure in Les Alcusses de Moixent, respectively (Province of Valencia, Spain). For rainfall simulation, 0.25-m² steel ring plots were placed on both bare (a) and grass covered soils (b). For natural rainfall, 2-m² bounded rectangular plots were installed on bare soil (photos by A. Cerdà and M. Prosdocimi).

Another way of assessing soil erosion in vineyards involves the use of erosion markers (Brenot et al., 2008; Casalí et al., 2009; Rodrigo Comino et al., 2015; Novara et al., 2011). Erosion markers allow carrying out analyses at larger temporal and spatial scales than those that are achieved through experimental plots, but their accuracy is lower. By applying Dendrogeomorphology, bio-markers such as tree ring characteristics have been used to estimate the rates of soil erosion from decennial to millennium time scales (Alestalo, 1971; Carrara and Carroll, 1979; Danzer, 1996; Dunne et al., 1978; Eardley and Viavant, 1967; Gärtner, 2007; LaMarche, 1968). In fact, environmental changes, such as erosion processes, are recognized to affect tree growth expressed in ring-width variations that can be dated. Research in Dendrogeomorphology has also been focused on exposed tree roots (Biot, 1990; Bodoque et al., 2005; Gärtner, 2007; Gärtner et al., 2001; Schnabel, 1994). Estimation of soil erosion rates is possible thanks to wood anatomical research that revealed the possibility of determining the first year of exposure by analyzing changes in the anatomical structure within annual rings of exposed roots (Gärtner, 2007). However, this method is affected by the uncertainty over the original depth of the root development that may lead to an erroneous estimation of erosion rates (Schnabel, 1994). Furthermore, as it is not always easy to identify the original land surface level, the vertical distance from an exposed root to the present ground surface may represent an underestimation of the total depth of the material (Casalí et al., 2009). Therefore, this method has been overtaken by more recent approaches that consider the graft point as botanical benchmarks (Brenot et al., 2008;

Casalí et al., 2009; Vanwalleghem et al., 2010). These approaches are based on the fact that, in grafted vines, the tissue between the rootstock and the scion expands horizontally most of the time (Baldini, 1992), in such a way that an identifiable callus develops around the fused stems. Because of this, the position of the callus within the stem remains unchanged and, given that grafting in vine is done directly in the field and almost at soil surface level (Brenot et al., 2008; Casalí et al., 2009), erosion and deposition rates can be quantified around a single plant using this botanical benchmark. The use of biomarkers is useful for long-term erosion quantification but it is subject to errors due to the natural variability of plants. To solve this issue, Novara et al. (2011) and Lieskovský and Kenderessy (2014) suggested using simple poles as erosion markers. In Novara et al. (2011) an espalier structure was built using poles 2.20 m in height. During the plantation of the vines, poles were planted using a machine for pile driving to a standard depth (0.60 m). Therefore, their novelty consisted in the fact that the exact original depth was known because of the machinery used. In this way, as time passed by, it was possible to measure the pole over-ground height without the uncertainty given by the vine stocks. This method was inexpensive, did not hamper soil management practices and, therefore, it could be applied for analyses at larger scales. Another common way of estimating soil erosion rates on a long-term scale is by using some of the current existing models. Various models exist for simulating erosion, sediment generation and sedimentation processes. These models differ from one another in terms of complexity, processes considered, data required for model calibration and applicability (Merriit et al., 2003). The well-known Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) and its improved version, the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997) represent the most commonly applied empirical models to estimate soil erosion (Bagarello and Ferro, 2004; Bewket and Teferi, 2009; Ferro and Porto, 2000; López-Vicente and Navas, 2010; Martínez-Casasnovas and Sánchez-Bosch, 2000; Mutua et al., 2006). With the advance of remote-sensing technologies and Geographic Information Systems, it is possible to compute spatially distributed USLE maps (Li et al., 2014). Fig. 5 shows an example of USLE equation spatially computed for a small basin located in Veneto region, northeast Italy. Erosion rates range from few (blue color) up to hundreds (red color) $\text{Mg ha}^{-1} \text{ yr}^{-1}$. In this area, climatic erosivity, topography and land use, management and conservation support practices play a predominant role in soil loss, rather than the soil erodibility. In fact, the highest erosion rates occur on vineyards located along steep slopes (Fig. 5).

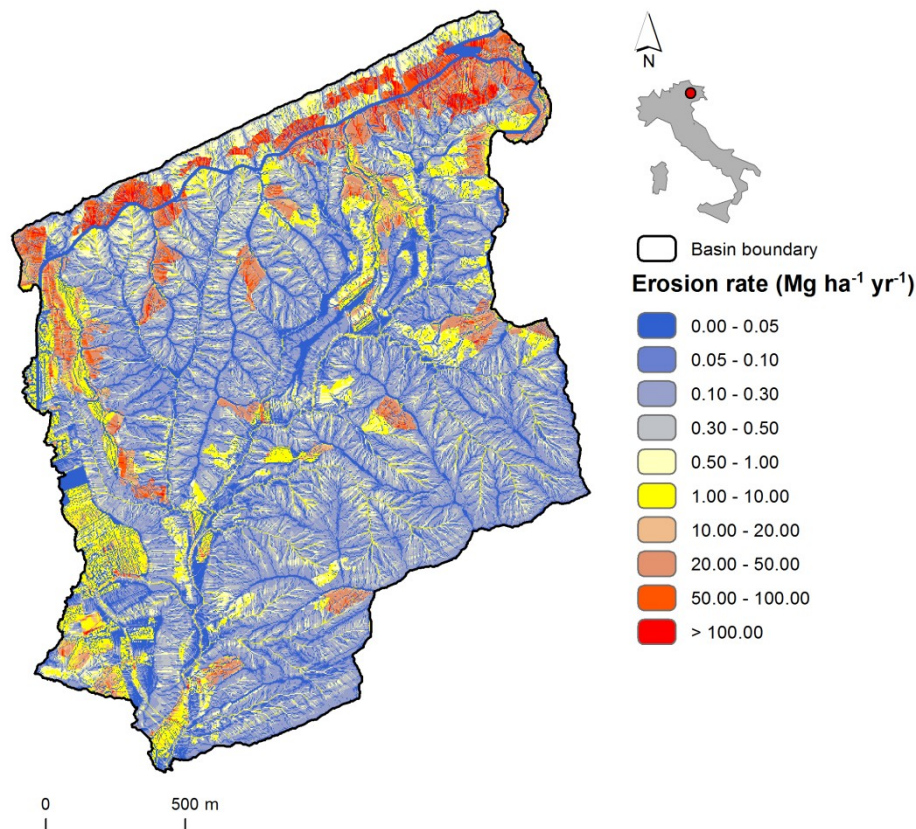


Fig. 5. Erosion rates estimated according to USLE equation, spatially computed for a small basin located in Veneto region (northeast Italy).

However, unless these empirical models are calibrated with local sediment yields, soil erosion estimation may be subject to large errors (De la Rosa et al., 2005; Fornes et al., 2005; Wang et al., 2001). A physics-based model such as WATEM/SEDEM (Van Oost et al., 2000; Van Rompaey et al., 2001) has been used by Lieskovský and Kenderessy (2014) to estimate the total soil loss from vineyards in Slovakia under different soil management conditions. The WATEM/SEDEM model has also been applied and validated in other vineyards by Alatorre et al. (2010), Bakker et al. (2008) and Jordan et al. (2005). An estimation of erosion rates may also be derived from the so-called “surface elevation change-based” methods, which measure the topographic evolution of a hillslope over time. As a result, Digital Elevation Models (DEMs) can be used as basic topographic information to derive morphometric attributes and quantify soil erosion and deposition rates too (Martínez-Casasnovas, 2003; Martínez-Casasnovas and Sánchez-Bosch, 2000; Martínez-Casasnovas et al., 2002). High resolution DEMs obtained from remote sensing technologies can also be used to model the topographic component of instability (Tarolli, 2014) in terraced vineyards (Tarolli et al., 2015) through the computation of a morphometric index: the Relative Path Impact Index (RPII) (Tarolli et al., 2013). Fig. 6 shows an example of RPII computed in three terraced vineyards located in Tuscany (6a), Veneto (6b) and Marche (6c) regions, respectively.

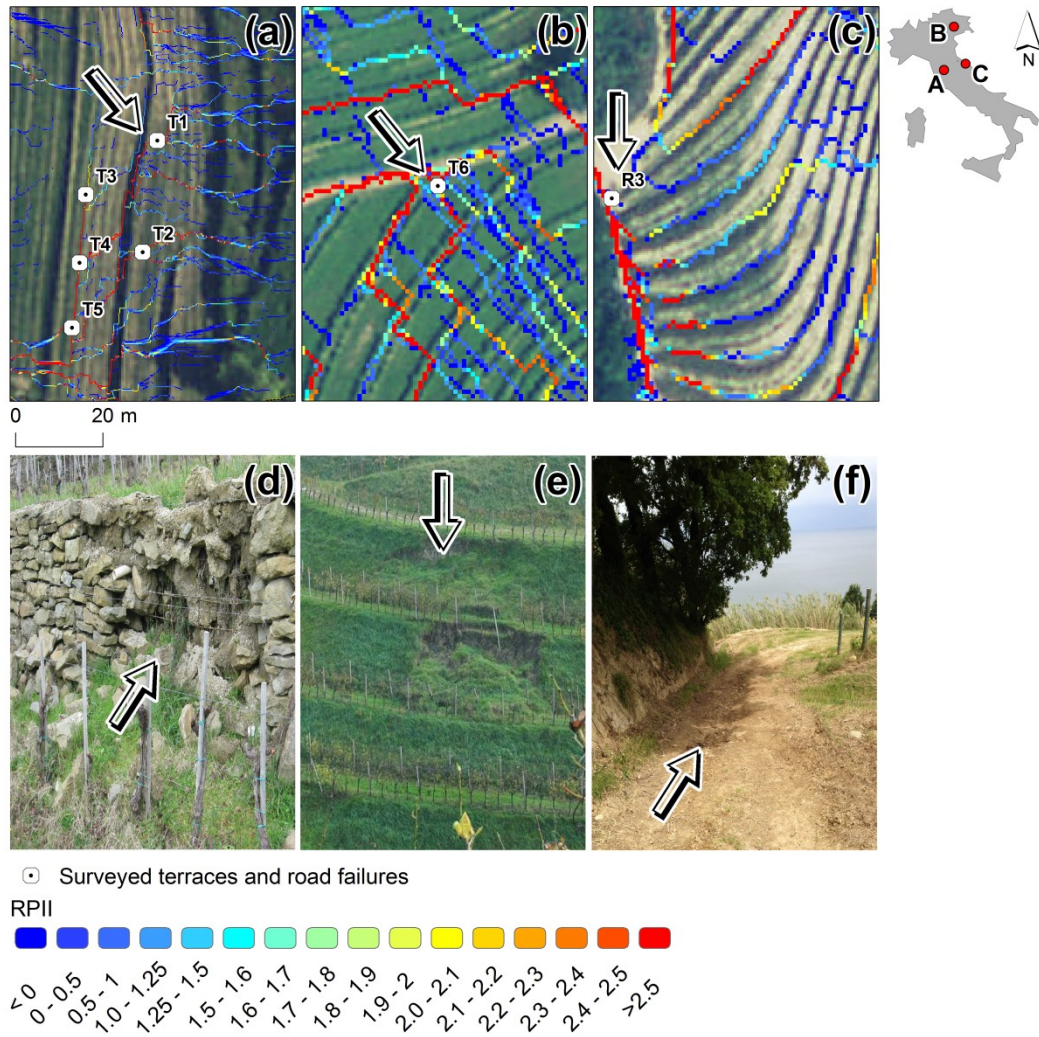


Fig. 6. RPII maps computed for three different vineyards located in Tuscany (central Italy) (a), Veneto (northeast Italy) (b) and Marche regions (central Italy) (c). Pictures highlight the flow modifications induced by the terraces ((a) and (b)), the agricultural road ((c)), and the related terraces (T1 in (d) and T6 in (e)) and road failures (R3 in (f)). In correspondence of these critical points, the index shows the highest values (red color). Fig. 6a, c, d and f are modified from Tarolli et al., 2015).

The index quantifies the effect of agricultural roads, trail networks and terraces on contributing area distributions, and therefore on flow paths. As it is expressed in a logarithmic form, it stresses and maps areas presenting an increased drainage area because of the presence of anthropogenic features (Tarolli et al., 2015). The higher the RPII and the higher the potential runoff-induced erosion is.

Over the last few decades, geochemical methods have also been used to quantify erosion rates at different temporal scales. Radionuclides such as caesium-137 (^{137}Cs) (Quine and Walling, 1991; Ritchie et al., 2005) and lead-201 (^{201}Pb) (Walling and He, 1999) have been successfully adopted for estimating erosion rates at decennial time scales (ca. 45 years). This has been possible because of their affinity for fine sediment soil particles, which make them good particle tracers. Loughran et al. (1992) and Loughran and Balog (2006) used the ^{137}Cs method to calculate net soil loss from vineyards in Australia for a period of several decades. However these methods require

a long measurement time and a high cost of laboratory analysis. Therefore, researchers have recently started to use carbon (C) stable isotopes to trace the movements of sediments (Alewell et al., 2009; Turnbull et al., 2008) and quantify soil losses too (Novara et al., 2015) still at decennial time scales. Given the complexity of this topic and the variability of the possible measurement methods, the effect of triggering factors on soil water erosion are not still completely clear, especially in Mediterranean vineyards. Therefore, this paper review aims to (i) develop a documented database on splash, sheet and rill erosion rates in Mediterranean vineyards, (ii) identify and, if possible, quantify the effect of triggering factors, such as topography, soil properties, rainfall characteristics and soil conservation techniques on soil water erosion, and (iii) provide suggestions for a more sustainable viticulture. Data published in literature are collected and the choice of focusing on the Mediterranean area is justified by its high soil losses estimated due to the topographic, edaphic and climatic characteristics. This is confirmed also by the great concern among the researches in literature.

2.3 Methods

2.3.1 Data collection

A dataset of erosion rates and soil loss measurements derived from Mediterranean vineyards was constructed from published literature. For each entry in the dataset, the following variables were collected, if available: (i) erosion rate and/or soil loss ($\text{g m}^{-2} \text{h}^{-1}$, $\text{Mg ha}^{-1} \text{yr}^{-1}$ or Mg ha^{-1}), (ii) spatial location (country), (iii) spatial scale, (iv) measurement method, (v) observation time (years), (vi) slope gradient (m m^{-1}), (vii) organic content (g kg^{-1}), (viii) mean rainfall intensity (mm h^{-1}), (ix) maximum rainfall intensity (mm h^{-1}), (x) soil conservation technique, (xi) runoff coefficient (–) and (xii) soil texture. Depending on the spatial and temporal scales of analyses, erosion rate can be expressed with different units of measurement (Boix-Fayos et al., 2006). In this work, to make data comparable with one other, soil erosion rates are expressed as (i) $\text{g m}^{-2} \text{h}^{-1}$ or (ii) $\text{Mg ha}^{-1} \text{yr}^{-1}$. The former refers to data collected at a very fine scale (microplots) in short periods (minutes or hours) (i.e. Arnáez et al., 2007; Blavet et al., 2009; Morvan et al., 2014). The latter is associated with data collected at fine, hillslope and field scales over longer periods (up to several years in some cases) (i.e. Brenot, 2007; Brenot et al., 2008; Cerdan et al., 2010; Kosmas et al., 1997; Marques et al., 2010; Tropeano, 1983). In case data of soil erosion are related to single natural events (i.e. Ballif, 1990; Litzler, 1988; Martínez-Casasnovas et al., 2002; Quiquerez et al., 2008; Raclot et al., 2009), the term soil loss, expressed as Mg ha^{-1} , is preferred and used in this work. Spatial scale, if the exact plot size was reported, is here classified as very fine (microplots) ($< 1 \text{ m}^2$), fine ($1\text{--}1000 \text{ m}^2$), hillslope ($1000 \text{ m}^2\text{--}1 \text{ ha}$) and field scale ($> 1 \text{ ha}$) (Boix-Fayos et al., 2006; Verheijen et al., 2009). As spatial scale is related to the measurement method, if the exact plot size was not reported, the spatial scale was defined according to the measurement method. This justifies the fact that, for some works, a spatial scale ranging from fine to hillslope (fine–hillslope) was assigned. The measurement methods considered are: (i) rainfall simulation (RS) (i.e. Arnáez et al., 2007; Morvan et al., 2014), (ii) runoff plot (RP) (i.e. Corti et al., 2011; Raclot et al., 2009), (iii) erosion markers (EM)

(i.e. Brenot, 2007; Novara et al., 2011) and (iv) DEMs (Martínez-Casasnovas et al., 2002; Quiquerez et al., 2008). RS method is associated to very fine scales and refers to splash and sheet erosion. RP method is associated to both fine and hillslope scales because runoff plots may have different sizes and usually refer to sheet and rill erosion. In the same way, EM and DEMs methods are associated to fine and hillslope scales. In addition, they may be applied to field scale too (Boix-Fayos et al., 2006; Verheijen et al., 2009). Gully erosion is not accounted for in this work because of the insufficient number of studies carried out in vineyards. Regarding the observation time, studies that applied RS method and those that applied DEMs method in relation to a single natural rainfall event are associated to an observation time less than 1 year. On the other hand, studies that applied RP method are usually associated to longer periods (up to some years). Finally, those studies that used EM method are always associated to longer periods, up to several years in some cases. Regarding the SCTs, their classification is based on the information provided for each study and the categorization made by Maetens et al. (2012a). SCTs used in this work include: (i) tillage (T), (ii) reduced tillage (RT), (iii) no-tillage (NT), (iv) grass cover (GC), (v) grass cover with mulching (GCM), (vi) mulching (M) and (vii) cover crops (CC). Respecting soil texture of topsoil, the retrieved information varied from a detailed description of a soil profile to simple qualitative information on soil texture. Therefore, we used the information to classify soils into five textural classes (Cerdan et al., 2010): (i) very fine (VF), (ii) fine (F), (iii) medium fine (MF), (iv) medium (M) and (v) coarse (C), following the scheme of the soil database and map of Europe (European Commission, 2004). While soil types are recognized to influence erosion processes (Cerdà, 2002; Corti et al., 2011), they were not systematically reported in the literature from which data were extracted. In addition, where soil types were specified, they were not categorized according to the same classification system. Therefore a quantitative analysis of the effect of the soil types on erosion rate and soil loss could not be made, as also pointed out by others authors who dealt with similar data (i.e. Cerdan et al., 2010; García-Ruiz et al., 2015; Maetens et al., 2012b). This process of data collection identified a total of 34 articles from which 492 records from 9 countries were obtained, although not every record included data for every variable.

2.3.2 Statistical analyses

Given the large variability of data collected, statistical analyses were applied to identify the main trends with respect to the effects of slope gradient, organic content, mean and maximum rainfall intensities and soil conservation techniques on erosion rates and/or soil loss, depending on the measurement method, using linear models (Cerdan et al., 2010). Erosion rates and soil loss data measured with the same method were separated from those obtained with a different one. This was done to reduce the uncertainty of comparing published data, derived from different erosion measurement methods, with each other (García-Ruiz et al., 2015). Analyses were applied to data derived from RS, RP and EM measurement methods. With respect to DEMs method, analyses were not carried out because of an insufficient number of available data. Ordinary least square linear regression analysis was used to examine the relative contribution of slope

gradient, organic content, mean and maximum rainfall intensities (explanatory variables) on erosion rates and/or soil loss (response variable) (Cerdan et al., 2010; García-Ruiz et al., 2015; Ziadat and Taimeh, 2013). In case of low R^2 values, the variability among the different works from which data were collected, could have had a more significant effect than the explanatory variables. Therefore, in these cases, to understand whether explanatory variables had a significant effect on the response variable or not, linear mixed models with the source work as random effect and explanatory variables as fixed effect were applied. These models were then compared to linear mixed models without the fixed effect through the likelihood ratio test (Bates et al., 2012) to assess whether explanatory variables had a significant effect or not. Multiple linear regression models were used to test the ability of the combination of explanatory variables to predict the response variable. At this regard, a backward manual deletion procedure starting from the most complex model was carried out. Linear, quadratic and cubic terms for the explanatory variables were considered in the most complex model. The manual deletion procedure consisted in removing each time the non-significant terms ($p > 0.05$), starting from the cubic, to the quadratic and, finally, to the linear terms. All the assumptions for the regression models were checked and, in case of non-linearity between explanatory and response variables, a logarithmic transformation was applied (García-Ruiz et al., 2015). In addition, one-way ANOVA was used to evaluate the presence of significant differences between the SCTs in regard to soil erosion rate and/or soil loss. In case the assumptions of ANOVA were seriously violated, the non-parametric Kruskal–Wallis test was applied. For all the analyses, a value of $p < 0.05$ was considered statistically significant and the open-source software R was used.

2.4 Results

2.4.1 Description of dataset

Erosion rates and soil loss data considered for the purposes of this work are summarized in Table 1. Eight out of the 34 articles reported are cited works. For each article, minimum, maximum and mean values (in brackets) of the above-mentioned variables are pointed out, if available. Articles are put in chronological order.

Table 1 Minimum, maximum and mean values (in brackets) of all the variables collected from published literature, if available: erosion rate (ER), soil loss (SL), spatial location (SI), spatial scale (Ss), measurement method (Mm), observation time (Ot), slope, organic content (OC), mean rainfall intensity (Rlmean), maximum rainfall intensity (Rlmax), soil conservation techniques (SCTs), runoff coefficient (RC) and soil texture (T). For the variable Mm, the following types of measurement methods are reported: rainfall simulation (RS), runoff plot (RP), erosion markers (EM) and digital elevation models (DEMs). For the variable of SCTs, the following types of soil conservation techniques are reported: tillage (T), reduced tillage (RT), no-tillage (NT), grass cover (GC), grass cover with mulching (GCM), mulching (M) and cover crops (CC). Articles are reported in chronological order.

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	Rlmean	Rlmax	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Tropeano (1983)*	-	32.55	>70.18	Italy	Fine-hillslope	RP	<1	0.35	-	-	-	-	-	-
Tropeano (1983)**	-	0.32-2.73	-	Italy	Fine-hillslope	RP	<1	0.35	-	-	-	-	-	-
Litzler (1988)*	-	32	-	France	-	-	>5	-	-	-	-	-	-	-

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	RI _{mean}	RI _{max}	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Litzler (1988)*	-	1.16	3.2-4.6	France	-	-	<1	-	-	-	-	-	-	-
Litzler (1988)*	-	15-25	-	Switzerland	-	-	<1	-	-	-	-	-	-	-
Litzler (1988)*	-	5.5	-	France	-	-	40	-	-	-	-	-	-	-
Gril et al. (1989)	21-388 (125.13)	-	-	France	Very fine	RS	<1	0.16- 0.29 (0.19)	14-28.9 (18.74)	60	-	NT	0.7	MF- M- C
Ballif (1990)*	-	0.6-1.5	-	France	Fine	RP	3	0.34	-	-	-	-	-	-

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	RI _{mean}	RI _{max}	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Ballif (1990)*	-	-	0.13-1.3	France	Fine	RP	<1	0.34	-	-	-	-	-	-
Wicherek (1991)*	-	35	-	France	-	RP	3	-	-	-	-	-	-	-
Figueiredo and Ferreira (1993)**	-	0.39-2.8	-	Portugal	-	-	10	-	-	-	-	-	-	-
Kosmas (1993)**	-	0.38-2.53	-	Greece	-	-	2	-	-	-	-	-	-	-
Viguier (1993)**	-	2.1-53.85	-	France	-	-	-	-	-	-	-	-	-	-

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	RI _{mean}	RI _{max}	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Wainwright (1996)	252	-	-	France	Very fine	RS	<1	0.13	-	100	-	NT	-	-
Kosmas et al. (1997)	-	1.43	-	France	Fine	RP	4	0.07	-	-	-	T	-	-
Usón (1998)	-	22	-	Spain	Fine-hillslope	RP	-	-	-	-	-	-	-	-
Bazzoffi and Chisci (1999)	-	0.02-41.5 (20.76)	-	Italy	Fine-hillslope	RP	-	-	-	-	-	-	-	-
Schaller and Emde (2000)*	-	-	0.2-52	Germany	-	-	<1	-	-	-	-	-	-	-

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	RI _{mean}	RI _{max}	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Lisa et al. (2001)	-	-	0.0001-0.02 (0.0052)	Italy	Hillslope	RP	7	-	-	1.7-26.3 (6.39)	4.5-31.6 (11.16)	T-GCM	0.0008-0.072 (0.01)	-
Martínez-Casasnovas et al. (2002)	-	-	207	Spain	Field	DEM	<1	0.09	-	-	-	NT	-	-
Blavet et al. (2004)	71.60-309 (144.73)	-	-	France	Very fine	RS	<1	0.12	12.4-17.3 (15.85)	60	-	NT-T-GC-M	0.12-0.78 (0.52)	M
Brenot et al. (2006)	-	7.61-14.94 (12.02)	-	France	Hillslope	EM	32-54	0.19	<10	-	-	T	-	M
Ramos and Martínez-Casasnovas (2006)	-	-	0.04-11.51 (1.36)	Spain	Fine-hillslope	RP	2	0.09	9	-	4.2-60.6 (23.24)	T	0.029-0.558 (0.183)	M

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	Rlmean	Rlmax	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Arnáez et al. (2007)	2.6-93.2 (39.59)	-	-	Spain	Very fine	RS	<1	0.07	8	30-117.5 (67.8)	-	T	0.02-0.37 (0.19)	M
Brenot (2007)	-	1.51-21.02 (8.34)	-	France	Fine-Hillslope-Field	EM	20-69	0.19	<10	-	-	T	-	M
Brenot et al. (2008)	-	23	-	France	Hillslope	EM	20	0.21	<10	-	-	T	-	M
Quiquerez et al. (2008)	-	-	24-48	France	Hillslope	DEM	<1	0.19	<10	20	-	T	-	M
Blavet et al. (2009)	11-242 (101.4)	-	-	France	Very fine	RS	<1	0.05-0.15 (0.10)	5-16 (12.6)	60	-	T-NT-GC-M	0.06-0.56 (0.34)	M

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	RI _{mean}	RI _{max}	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Raclot et al. (2009)	-	-	0.001-10.5 (1.5)	France	Hillslope	RP	4	0.09-0.12 (0.11)	9-10 (9.5)	3.34-31.35 (9.3)	7-107 (26.72)	T	0.02-0.8 (0.37)	M
Casalí et al. (2009)	-	30	-	Spain	Fine-hillslope	EM	22-71	0.08	4.1-14.9	-	-	T	-	M
Cerdan et al. (2010)	-	8.62	-	Mediterranean	Fine-hillslope	RP	-	-	-	-	-	-	-	-
Marques et al. (2010)	-	0.62-0.70 (0.66)	-	Spain	Fine	RP	1	0.14	10.45	-	-	GCM	0.16-0.30 (0.23)	-
Paroissien et al. (2010)	-	10.5	-	France	Fine-hillslope	EM	7-25	0.03	-	-	-	T	-	-

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	RI _{mean}	RI _{max}	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Corti et al. (2011)	-	-	0.00058-23.48 (1.80)	Italy	Hillslope	RP	10	0.15	-	0.5-24.5 (3.59)	1.2-76.2 (12.59)	T-RT-GCM	0.03-1.49 (0.16)	F-M
Novara et al. (2011)	-	-	0.001-18.62 (3.67)	Italy	Hillslope	RP	2	0.16	7.81	-	-	CC	-	M
Novara et al. (2011)	-	102.2	-	Italy	Hillslope	EM	9	0.16	7.81	-	-	-	-	M
Vrsic et al. (2011)	-	0.088-2.012 (0.99)	-	Slovenia	Fine	RP	3	0.34	-	-	-	T-GCM	-	-
Bernardoni (2012)	-	-	0.70-42.66 (16.30)	Italy	Fine	RP	3	0.17	15	-	8.4-101 (35.86)	GCM	0.001-0.313 (0.07)	M

Table 1 (*continued*)

References	ER		SL	SI	Ss	Mm	Ot	Slope	OC	RI _{mean}	RI _{max}	SCTs	RC	T
	(g m ⁻² h ⁻¹)	(Mg ha ⁻¹ yr ⁻¹)	(Mg ha ⁻¹)				(yr)	(m m ⁻¹)	(g kg ⁻¹)	(mm h ⁻¹)	(mm h ⁻¹)			
Maetens et al. (2012a)	-	1.8	-	Mediterranean	Fine-hillslope	RP	-	-	-	-	-	-	-	-
Lieskovský and Kenderessy (2014)	-	3.89-24.08 (14.6)	-	Slovakia	Fine-hillslope	EM	-	0.19	15	-	-	T	-	M
Morvan et al. (2014)	0.8-13.4 (4.01)	-	-	France	Very fine	RS	<1	0.05	22.8-47.8 (32.48)	20-48 (32.75)	-	T-GC-M	0.004-0.77 (0.29)	M

Spatial scale (Ss), if the exact plot size was reported, is here classified as very fine (microplots) (< 1 m²), fine (1 -1000 m²), hillslope (1000 m² – 1 ha) and field scale (> 1 ha). As spatial scale is related to the measurement method, if the exact plot size was not reported, the spatial scale was defined according to the measurement method. This justifies the fact that, for some works, a spatial scale ranging from fine to hillslope (fine – hillslope) was assigned.

* in Quiquerez et al., 2008

** in Hooke, 2006

First, from Table 1 emerges that not every work includes data for every variable. Based on the full dataset, observation time, measurement method and spatial scale were the most included variables with 98.0, 97.2 and 97.0% of data accounted for, respectively. Follow the SCTs, slope, soil texture, runoff coefficient, mean and maximum rainfall intensities and organic content with 93.9, 92.7, 88.4, 61.0, 54.9, 53.3 and 52.2% of data accounted for, respectively. Furthermore, from Table 1 emerges that Italy, France and Spain are the countries where most of the studies on soil water erosion in vineyards have been carried out. In fact, these countries together account for 96.1% of the full dataset. This may be justified by the fact that these countries are three of the world top ten grape-producing countries (Anderson and Nelgen, 2011). Respecting the spatial scale, hillslope scale is the most representative one with 77.4% of data accounted for of the full dataset. Follow very fine, fine, fine – hillslope and field scales, with 7.8, 7.1, 6.5 and 1.3%, respectively. Concerning the measurement methods, RP is the most frequent one with 81.2% of data considered of the full dataset. This is in line with the fact that hillslope scale is the most accounted for. Follow EM, RS and DEMs methods with 10.7, 7.7 and 0.4%, respectively. Regarding the SCTs, T is the most accounted for with 40.7% of data considered. Follow CC, GCM, RT, NT, GC and M, with 22.1, 18.8, 13.0, 3.0, 1.7 and 0.6%, respectively. With respect to soil texture, M and F are the most frequent ones with 98.9% of data accounted for of the full dataset. Follow C and MF, with 0.7 and 0.5%, respectively. VF is not accounted for at all in the dataset. In the end, from Table 1, also emerges that Mediterranean vineyards are characterized by very variable erosion rates and soil losses that may be even much greater than the upper limit of the European tolerable soil erosion rate (Verheijen et al., 2009). The large variability is mainly due to the different temporal and spatial scales of analyses carried out and measurement methods adopted. If data are not categorized, a comparison is worthless. Therefore, this justifies the choice of separating data derived from the same measurement method from those obtained with a different one and running divided analyses.

2.4.2 Effect of slope on soil erosion rate and soil loss

The effect of slope gradient on soil erosion rate and/or soil loss was evaluated for RS, RP and EM measurement methods through linear regression models. Starting from RS method, data were log-transformed to have linearity between explanatory and response variables. The linear regression analysis confirmed a significant and positive relationship between erosion rate and slope gradient. The model took into account for 40.7% of the total variance ($R^2 = 0.407$). Fig. 7 shows the corresponding scatterplot and the fitting regression line.

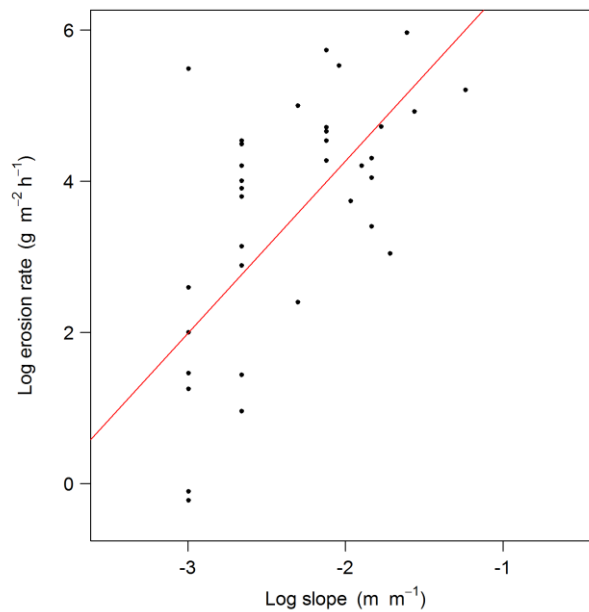


Fig. 7. Relationship between erosion rate and slope derived from RS method. Data were log-transformed to have linearity. The red line represents the fitting regression line ($R^2 = 0.407$).

Regarding RP method, it was not necessary to log-transform data to have linearity between explanatory and response variables. The linear regression analysis confirmed a significant and positive relationship between soil loss and slope gradient. However, in this case, the model took into account only for 4.6% of the total variance ($R^2 = 0.046$). Fig. 8 shows the corresponding scatterplot and the fitting regression line.

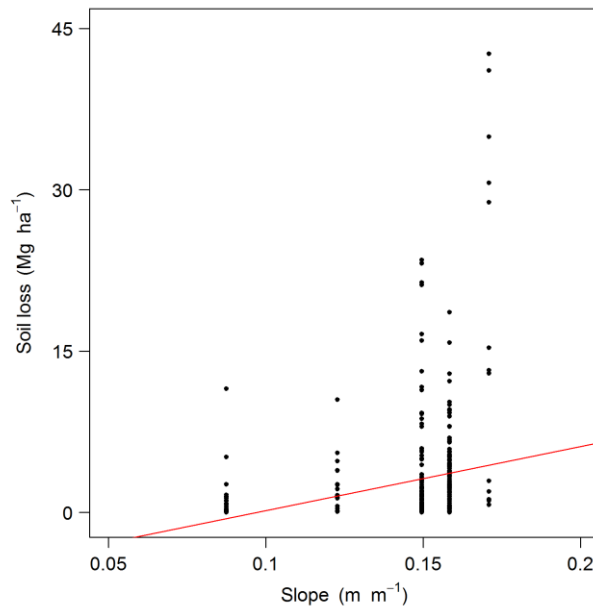


Fig. 8. Relationship between soil loss and slope derived from RP method. The red line represents the fitting regression line ($R^2 = 0.046$).

The low R^2 value may be due to the variability among the different works from which data were collected. Therefore, to understand whether slope had a significant effect on soil loss or not, a linear mixed model with the source work as random effect and slope as fixed effect was applied and was compared to a linear mixed model without the fixed effect. From the comparison of the two models through the likelihood ratio test, slope did not prove to affect significantly soil loss. With respect to EM method, also in this case it was not necessary to log-transform data to have linearity between explanatory and response variables. The linear regression analysis revealed a significant and positive relationship between erosion rate and slope gradient also for EM measurement method. The model took into account only for 5.7% of the total variance ($R^2 = 0.057$). Fig. 9 shows the corresponding scatterplot and the fitting regression line.

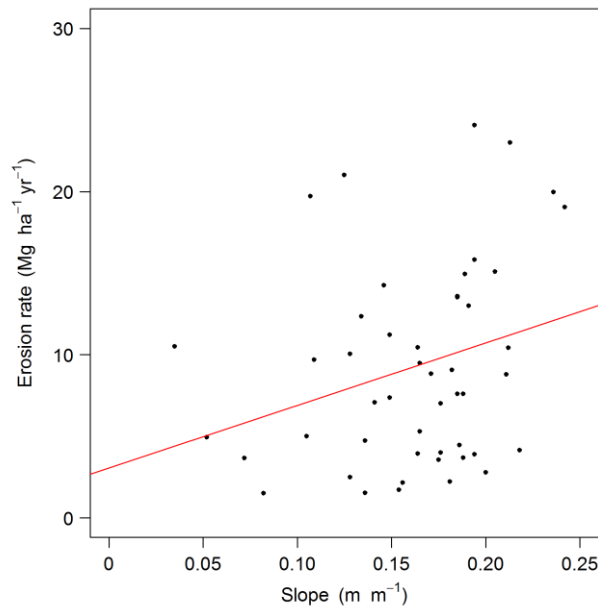


Fig. 9. Relationship between erosion rate and slope derived from EM method. The red line represents the fitting regression line ($R^2 = 0.057$)

As done for RP method, a linear mixed model with the source work as random effect and slope as fixed effect was applied and was compared to a linear mixed model without the fixed effect. After the comparisons of the two models still performed with the likelihood ratio test, the results highlighted that slope did not significantly affect soil loss also for RP method.

2.4.3 Effect of organic content on soil erosion rate and soil loss

The effect of organic content on soil erosion rate and/or soil loss was evaluated only for RS and RP measurement methods because of a lack of available data for EM. The relationship between the explanatory and response variables was still evaluated through linear regression models. Starting from RS method, data were log-transformed to have linearity between explanatory and response variables. The linear regression analysis confirmed a significant and negative relationship between erosion rate and organic content. The model took into account for 20.8% of the total variance ($R^2 = 0.208$). Fig. 10 shows the corresponding scatterplot and the fitting regression line.

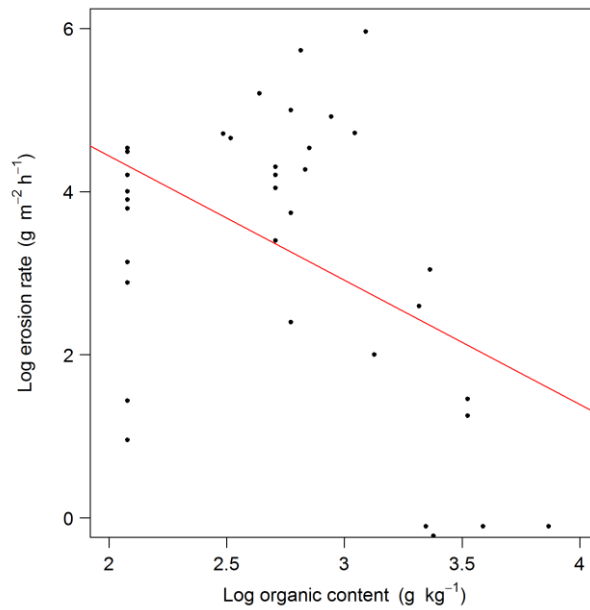


Fig. 10. Relationship between erosion rate and organic content derived from RS method. Data were log-transformed to have linearity. The red line represents the fitting regression line ($R^2 = 0.208$).

Regarding RP method, it was not necessary to log-transform data to have linearity between explanatory and response variables. The linear regression analysis confirmed a significant and negative relationship between soil loss and organic content. However, in this case, the model took into account only for 3.5% of the total variance ($R^2 = 0.035$). Fig. 11 shows the corresponding scatterplot and the fitting regression line.

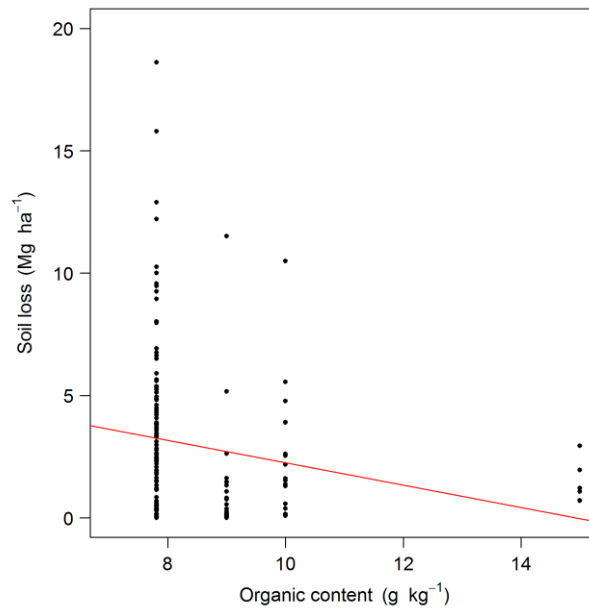


Fig. 11. Relationship between soil loss and organic content derived from RP method. The red line represents the fitting regression line ($R^2 = 0.035$).

Given the low R^2 value, a linear mixed model with the source work as random effect and organic content as fixed effect was applied and was compared to a linear mixed model without the fixed effect. From the comparison of the two models through the likelihood ratio test, organic content did not prove to affect significantly soil loss for RP method.

2.4.4 Effect of mean rainfall intensity on soil erosion rate and soil loss

The effect of mean rainfall intensity on soil erosion rate and/or soil loss was evaluated only for RS and RP measurement methods because of a lack of available data for EM. The relationship between the explanatory and response variables was still evaluated through linear regression models. Starting from RS method, data were log-transformed to have linearity between explanatory and response variables. The linear regression analysis confirmed a significant and positive relationship between erosion rate and mean rainfall intensity. The model took into account for 55.0% of the total variance ($R^2 = 0.550$). Fig. 12 shows the corresponding scatterplot and the fitting regression line.

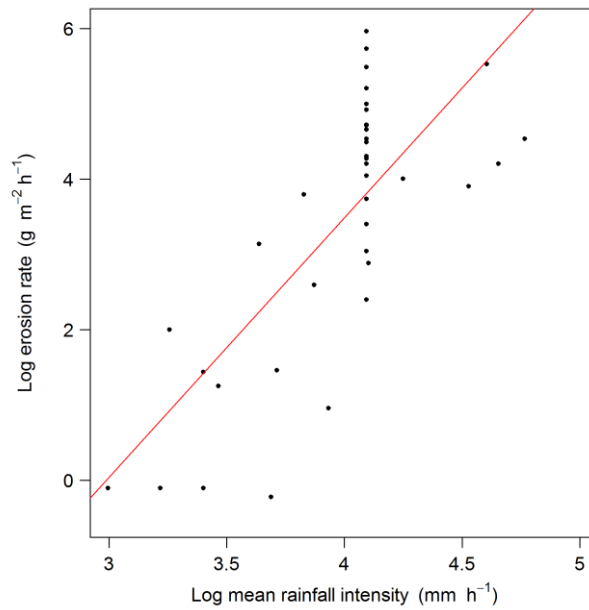


Fig. 12. Relationship between erosion rate and mean rainfall intensity derived from RS method. Data were log-transformed to have linearity. The red line represents the fitting regression line ($R^2 = 0.550$).

Regarding RP method, data were log-transformed to have linearity between explanatory and response variables too. The linear regression analysis confirmed a significant and positive relationship between soil loss and mean rainfall intensity. However, in this case, the model took into account only for 7.3% of the total variance ($R^2 = 0.073$). Fig. 13 shows the corresponding scatterplot and the fitting regression line.

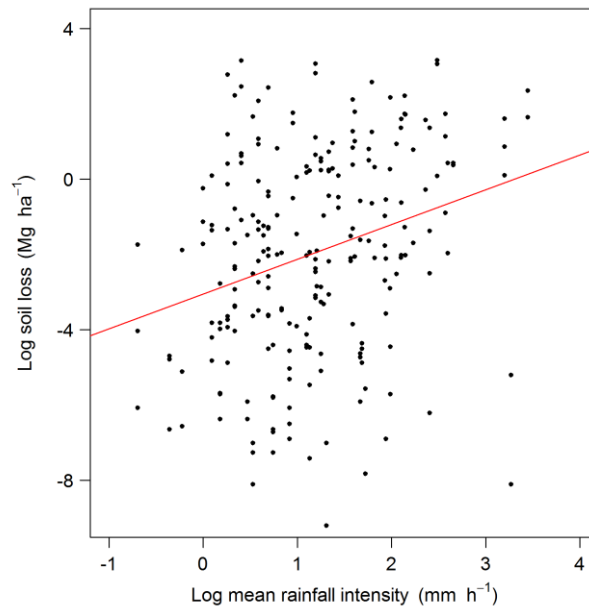


Fig. 13. Relationship between soil loss and mean rainfall intensity derived from RP method. Data were log-transformed to have linearity. The red line represents the fitting regression line ($R^2 = 0.073$).

Given the low R^2 value, a linear mixed model with the source work as random effect and mean rainfall intensity as fixed effect was applied and was compared to a linear mixed model without the fixed effect. The two models were compared each other through the likelihood ratio test and the results confirmed that mean rainfall intensity significantly affected soil loss.

2.4.5 Effect of maximum rainfall intensity on soil loss

The effect of maximum rainfall intensity on soil loss was evaluated only for RP measurement methods because of a lack of available data for RS and EM. The relationship between the explanatory and response variables was still evaluated through linear regression models. Data were log-transformed to have linearity between explanatory and response variables. The linear regression analysis confirmed a significant and positive relationship between soil loss and maximum rainfall intensity. The model took into account for 21.4% of the total variance ($R^2 = 0.214$). Fig. 14 shows the corresponding scatterplot and the fitting regression line.

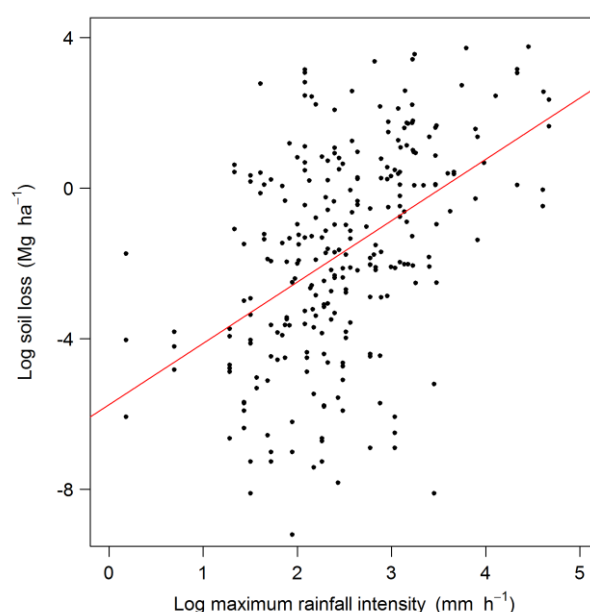


Fig. 14. Relationship between soil loss and maximum rainfall intensity derived from RP method. Data were log-transformed to have linearity. The red line represents the fitting regression line ($R^2 = 0.214$).

Table 2 reports an overview of the results obtained from the abovementioned linear regression analyses carried out for each explanatory variable and measurement method considered.

Table 2 Overview of the results obtained from the linear regression analyses carried out for each explanatory variable (slope, organic content, mean and maximum rainfall intensities) and measurement method (rainfall simulation (RS), runoff plot (RP) and erosion markers (EM)) considered. For each model are reported the R^2 and the p values (considered statistically significant if < 0.05).

	Measurement method	Erosion rate	Soil loss	p -Value
Slope	RS	0.407		0.00001
	RP		0.046	> 0.05
	EM	0.057		> 0.05
Organic content	RS	0.208		0.003
	RP		0.035	> 0.05
Mean rainfall intensity	RS	0.550		< 0.00001

Table 2 (*continued*)

	Measurement method	Erosion rate	Soil loss	<i>p</i> -Value
Mean rainfall intensity	RP		0.073	< 0.00001
Maximum rainfall intensity	RP		0.214	< 0.00001

From Table 2 emerges that, for the relationships between slope and soil loss/erosion rate (RP and EM methods), and between organic content and soil loss (RP method), the explanatory variables do not have a significant influence on erosion rate and soil loss. On the other hand, for the others relationships, the explanatory variables prove to significantly affect erosion rate and soil loss. Furthermore, considering the variance explained by each model (R^2), the one that links the mean rainfall intensity with the erosion rate (RS method) is the one that takes into account the highest percentage of the total variance (55.0%). Follow the models that link the slope with the erosion rate (RS method) and the mean rainfall intensity with soil loss (RP method), with 40.7% and 21.4% of the total variances taken into account, respectively.

2.4.6 Effect of slope, organic content, mean and maximum rainfall intensities on soil erosion rate and soil loss

The ability of the combination of explanatory variables to predict the response variable was tested through multiple linear regression models only for RS and RP measurement methods, because of a lack of available data for EM. Starting from RS method, all the explanatory variables were considered because the Pearson's correlation matrix did not point out any collinearity among them. Therefore, in the first and most complex model, linear, quadratic and cubic terms were considered for each variable. Erosion rate data were log-transformed to have linearity. The final model took into account for 67.0% of the total variance ($R^2 = 0.670$) and indicated a significant influence of the slope and organic content on soil erosion rate (Table 3).

Table 3 Multiple regression model testing slope, organic content and mean rainfall intensity on soil erosion rate for the rainfall simulation (RS) measurement method.

	Estimate	Std. Error	<i>t</i> -Value	<i>p</i> -Value
Intercept	3.203	0.538	5.952	< 0.05
Slope	17.897	3.268	5.475	< 0.05
Organic content	- 1.060	0.174	-6.085	< 0.05

From Table 3 emerges that slope had a positive linear relationship with erosion rate, while organic content had a negative one. Furthermore, the results revealed that slope

and organic content had much more influence on erosion rate than the mean rainfall intensity. Regarding RP method, in the first and most complex model, linear, quadratic and cubic terms were considered only for slope and maximum rainfall intensity as explanatory variables. In fact, maximum rainfall intensity was preferred to mean rainfall intensity because of its one greater order R^2 value (0.214 vs. 0.073). Furthermore, the Pearson's correlation matrix revealed collinearity between slope and organic content, and therefore, only slope was considered. Soil loss data were log-transformed to have linearity. The final model took into account for 62.3% of the total variance ($R^2 = 0.623$) and indicated a significant influence of all the variables considered on soil loss (Table 4).

Table 4 Multiple regression model testing slope, organic content and maximum rainfall intensity on soil loss for the runoff plot (RP) measurement method.

	Estimate	Std. Error	t-Value	p-Value
Intercept	- 6.995	0.717	- 9.755	< 0.05
Slope	37.397	5.311	7.041	< 0.05
Maximum rainfall intensity	0.118	0.025	4.716	< 0.05
Maximum rainfall intensity ²	0.000	0.000	- 3.162	< 0.05

From Table 4 emerges that slope had a positive linear relationship with soil loss while maximum rainfall intensity showed a non-linear relationship with it.

2.4.7 Effect of soil conservation techniques on soil erosion rate and soil loss

The effect of different SCTs on soil erosion rate was conducted with a one-way ANOVA test for data derived from RS measurement method. On the other hand, for data derived from RP method, the non-parametric Kruskal–Wallis test was applied because the assumptions of ANOVA were seriously violated. Starting from RS method, the SCTs considered were T, NT, GC and M. Erosion rate data were square-root transformed to have normality. The one-way ANOVA revealed significant differences between the four categories of SCTs in regard to soil erosion rate ($F(3,36) = 6.041$, $p < 0.05$). Post hoc pairwise comparisons using t tests and Bonferroni p value adjustment method indicated that only NT and GC were significant different one from the other. Regarding RP method, the SCTs considered were T, RT, GCM and CC. Kruskal–Wallis test revealed a statistically significant difference between the four categories of SCTs in regard to soil loss ($X^2(3) = 94.736$, $p < 0.05$). Post hoc pairwise comparisons using Tukey and Kramer (Nemenyi) test with Tukey-Dist approximation for independent variables revealed that all the four categories of SCTs were significant different one from the other, except for T and GCM.

2.5 Discussion

2.5.1 Influence of topography

Topography, intended as slope gradient and length, is one of the primary factors influencing soil water erosion (Musgrave, 1947; Hudson and Jackson, 1959; McCool, 1982; Zachar, 1982; Morgan, 1986; Fox and Rorke, 1999; Koulouri and Giourga, 2007; Cerdan et al., 2010). Our results confirmed a significant and positive relationship between erosion rates and slope gradient only for the RS method ($R^2 = 0.407$, p value = 0.00001) (Table 2). On the other hand, regarding the RP and EM methods, our results pointed out a positive, but non-significant, relationship between the two variables, due to the large variability of the studies from which data were collected (Table 2). Furthermore, with respect to multiple regression analyses, slope confirmed to have a significant and positive effect on erosion rate and soil loss, for RS and RP methods, respectively (Tables 3 and 4). This is in line with the general statement that erosion rate and soil loss increase as the slope becomes steeper (Fu et al., 2011; Mah et al., 1992; Quansah, 1981). In fact, as slope and the length of plot increase, the velocity of overland flow increases too and, hence, the erosive power and the capability of the flow to carry suspended sediments (Wischmeier and Smith, 1978). The fact that, in Mediterranean areas, vineyards are usually located on hillslopes and are subject to high intensity rainfall events, mainly concentrated in spring and autumn, exacerbates the frequency and magnitude of soil erosion processes (Corti et al., 2011). Cerdan et al. (2010) related slope gradient and length of plot to erosion rates for different land uses such as arable lands, bare lands and permanent cultures, within which vineyards were included. They found out weak but statistically positive relationship between erosion rates and slope length, especially for permanent cultures ($R^2 = 0.42$; $p = 0.05$). Corti et al. (2011) proved the importance of slope gradient in soil erosion, by comparing each other two vineyards characterized by the same soil management techniques (grass cover of the inter-rows). They found out that, at slopes of about 0.15, grass cover was able to reduce soil erosion, while at slopes of about 0.05, grass cover did not offer any advantage in reducing soil erosion. In addition, at large scale, topography is one of the main factors governing the spatial and temporal variability of soil water content within the top soil layer, as pointed out by Ramos and Martínez-Casasnovas (2007). Furthermore, topography not only influences soil erosion rates but also soil management techniques. In fact, in the past, to make cultivation easier, vineyards located on hillslopes were generally terraced and cultivated with vine-rows oriented perpendicularly to slope (Corti et al., 2011; Martínez-Casasnovas and Sánchez-Bosch, 2000; Tarolli et al., 2014; Tarolli et al., 2015). In particular, terraces were built to reduce the slope gradient and length and to intercept surface runoff, leading in this was to a reduction in soil erosion (Gachene et al., 1997; Li et al., 2012; Louwagie et al., 2011; Tarolli et al., 2014; Wakindiki and Ben-Hur, 2002). Fig. 15 reports pictures of four Italian terraced vineyards. Fig. 15a represents an example of vines cultivated on drystone wall terraces that represent a typical agricultural landscape of Tuscany region, central Italy. These terraces have a height of about 2 m and the terraces benches are about 8 to 10 m wide, and they are slightly outward sloped for drainage purposes. Fig. 15b is taken in

another terraced vineyard, still in Tuscany region. Differently from the previous one (Fig. 15a), here terraces are not made of dry-stone walls but compacted soil. Fig. 15c shows vines cultivated on bench terraces made of compacted soil in which terrace risers are covered with grass (Marche region, central Italy). Finally, Fig. 15d is taken in the Veneto region, northeast Italy, where the D.O.C.G. wine called “Prosecco of Conegliano Valdobbiadene” is produced. Here, vines are cultivated along bench terraces made of compacted soil and the inter-rows are covered with grass to prevent soil water erosion.

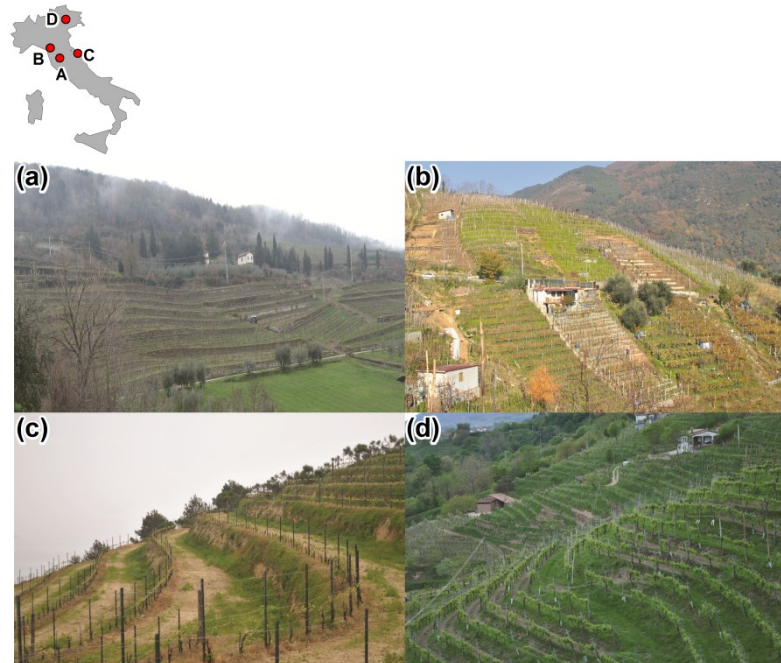


Fig. 15. Examples of terraced vineyards located in Tuscany (a) and (b) (central Italy), Marche (c) (central Italy) and Veneto (d) (northeast Italy) regions (photo by S. Calligaro and M. Prosdocimi).

2.5.2 Influence of soil properties

Particle size distribution (soil texture), organic matter content, stability of aggregates, soil chemistry, clay mineralogy, rock fragments, and soil moisture are all soil characteristics that influence erosion processes (Barthès and Roose, 2002; Blavet et al., 2009; Bryan, 2000; Cerdà, 2002; Cerdan et al., 2006, 2010; Corti et al., 2011; Lal, 1994; Poesen et al., 1994; Ramos and Martínez-Casasnovas, 2007; Sanchis et al., 2008; Torri et al., 1997). Our results confirmed a significant and negative relationship between erosion rate and organic content only for the RS measurement method, for which the linear regression model took into account for 20.8% of the total variance ($R^2 = 0.208$, p value < 0.00001). On the other hand, regarding the RP method, our results pointed out a negative, but non-significant, relationship between the two variables, due to the large variability of the studies from which data were collected (Table 2). With respect to multiple regression analyses, organic content confirmed to have a significant and negative effect on erosion rate for RS method (Table 3), while for RP method, organic content resulted to be highly correlated to slope and hence it was excluded from the analysis. The results derived from RS method are in line with the general statement that erosion rate increases as the organic content decreases (Blavet et al., 2009;

Robinson and Phillips, 2001). In fact, soils that have low organic content and, hence, stability of aggregates, are recognized to be very susceptible to soil water erosion. One such example is given by those soils, characterized by non-conservative soil management techniques, which develop from the Plio-Pleistocene silty clayey marine sediments. At this regard, Corti et al. (2011) compared the effects of two different soil management techniques (T and GCM) on water erosion of vineyards developed from these soils in the Mediterranean area (central of Italy). They found out that a greater amount of coarse sand in the grass-covered soils was related to a higher stability of soil aggregates. Aggregate stability was also enhanced by the presence of plant roots, as already pointed out by Cerdà (1999). Soils prone to erosion are also developed from marls (Cerdà, 2002). Furthermore, Ramos and Martínez-Casasnovas (2007) demonstrated how sealing susceptibility, favored by the low organic matter content, influenced the soil infiltration capacity and the water distribution within the soil profile. Consequently, the spatial distribution of soil moisture at different soil depths had an effect on runoff and soil erosion rates. Regarding the presence of rock fragments in the top soil layers, they constitute a characteristic feature for arid and semiarid environments (Cerdà, 2001; Poesen, 1990; Poesen et al., 1998). Rock fragment cover and size are demonstrated to have a great influence on various hydrological and soil degradation processes such as surface sealing, infiltration, evaporation, runoff generation, and erosion by water (Abrahams and Parsons, 1994; Blavet et al., 2009; Brakensiek and Rawls, 1994; Cerdà, 2001; Poesen and Lavee, 1994; Poesen et al., 1994, 1998; Valentin, 1994; vanWesemael et al., 1995, 1996). Cerdà (2001) studied the effects of rock fragments cover on soil infiltration, runoff and erosion under field conditions using simulated rainfall. His results revealed that surface rock fragments cover retarded ponding and surface runoff due to the increase of the surface roughness. Furthermore, he pointed out how the presence of a rock fragments cover increased steady-state infiltration rates. This was mainly due to the contact between the stones and the soil matrix that favored a faster and deeper flow (Poesen et al., 1990). Rock fragments diminished runoff discharge, sediment concentrations and erosion rates too by enriching in organic matter the soil beneath them and reducing the erosivity of the rain. This proved to be in line with Adams (1966); Blavet et al. (2009); Box (1981); Grant and Struchtemeyer (1959); Meyer et al. (1972), and Tromble (1976). Regarding soil chemistry, the presence of inorganic pollutants in vineyard soils may become a serious concern if water erosion transports these metals associated with sediment particles (Banas et al., 2010; Quinton and Catt, 2007; Xue et al., 2000). Zinc content of agricultural soils is usually higher than that of natural soils due to the use of fertilizers, pesticides and fungicides (Komárek et al., 2010; Senesi et al., 1999). Also regarding copper, its presence in agricultural soils may reach alarming levels because of the use of copper-based fungicides (Fernández-Calviño et al., 2009; Komárek et al., 2008; Paradelo et al., 2008). In these cases, zinc and copper fractions may have a strong impact on the environment, rather than being principal factors favouring soil erosion (Fernández-Calviño et al., 2012, 2013).

2.5.3 Influence of rainfall characteristics

Rainfall characteristics intended as rainfall amount, intensity and frequency are among the natural factors that most influence soil erosion and runoff (Cerdà, 2002; Greer, 1971; Serrano-Muela et al., 2013; Ziadat and Taimeh, 2013). Above all, the impact of raindrops plays a key role in detaching soil particles. Rainfall erosivity is even enhanced in Mediterranean vineyards because i) they have to bear high intensity and short duration rainfall events (Raclot et al., 2009; Ramos and Martínez-Casasnovas, 2007; Tropeano, 1984), ii) they are characterized by low organic content (Cerdà, 1996; Novara et al., 2011), and by bare soil cover in the inter-rows for most of the year (Arnáez et al., 2007; Novara et al., 2011; Tropeano, 1984). Our results confirmed a significant and positive relationship between erosion rate and mean rainfall intensity for both RS and RP measurement methods considered. In particular, the linear regression model computed with data obtained from RS method was the one that explained the highest total variance ($R^2 = 0.550$, p value < 0.00001) (Table 2). However, still regarding the RS method, in multiple regression analysis mean rainfall intensity did not prove to have a significant effect on erosion rate if compared to slope and organic content (Table 3). With respect to maximum rainfall intensity, it proved to affect significantly soil loss ($R^2 = 0.214$, p value < 0.00001) (Table 2). This behavior was confirmed by multiple regression analysis too, in which maximum rainfall intensity showed a non-linear and significant relationship with soil loss (Table 4). Regarding the effect of rainfall characteristics on erosion rate and soil loss, Arnáez et al. (2007) found out that the variability in soil erosion rate was not exclusively related to rainfall intensity, rather to the combination of others factors such as storm kinetic energy, runoff, resistance of the soil material, slope gradient and gravel cover. Similarly, Ramos and Martínez-Casasnovas (2007) obtained significant correlations at a 95% confidence level when considering the maximum intensity in a 30-minute period too.

2.5.4 Influence of soil conservation techniques

In literature, various works have dealt with the role of SCTs on soil erosion and runoff in vineyards (Arnáez et al., 2007; Ballif and Herre, 1990; Blavet et al., 2009; Biddoccu et al., 2016; Corti et al., 2011; Marques et al., 2010; Martínez-Casasnovas and Sánchez-Bosch, 2000; Maetens et al., 2012a; Morvan et al., 2014; Novara et al., 2011; Prosdocimi et al., 2016; Raclot et al., 2009; Ramos and Martínez-Casasnovas, 2007). By comparing the frequency distributions of annual soil erosion rates with and without the application of SCTs, Maetens et al. (2012a) found out that the exceedance probability of tolerable soil erosion rates was about 20% lower when SCTs were applied. Based on the one-way ANOVA test applied to data derived from RS method, among the SCTs considered, only NT and GC revealed to be significant different one from the other with respect to soil erosion rate. On the other hand, according to the Kruskal–Wallis test applied to data derived from RP method, all the four categories of SCTs were significant different one from the other, except for T and GCM. The presence of a natural or permanent vegetation cover in the inter-rows of vineyards resulted to be recommended for any slope and rainfall intensity if compared with bare

soil and tillage techniques proved to be always advocated in respect to no-tillage techniques characterized by chemical weeding (Ballif and Herre, 1990; Battany and Grismer, 2000; Blavet et al., 2009; Martínez et al., 2006; Gómez et al., 2003; Gril, 1986; Raclot et al., 2009). In fact, the effects of vegetation cover for reducing soil erosion proved to (i) increase the infiltration rate (Cerdà, 1998; Folorunso et al., 1992; Gulick et al., 1994; Ruiz-Colmenero et al., 2013), (ii) protect the soil surface from the impact of raindrops, and (iii) facilitate the formation and stabilization of soil aggregates (Durán-Zuazo and Rodríguez-Plequezuelo, 2008; García-Orenes et al., 2005). Furthermore, vegetation management practices proved to reduce soil erosion by enhancing the organic matter and microbiological function of soil (García-Orenes et al., 2009, 2010; Quinton and Catt, 2007). However, in xeric moisture regimes, such as the ones characterizing the south of Italy, grass cover resulted to be a benefit only during the fall and period of winter grape dormancy (Gristina et al., 2006; Lavezzi et al., 1999). Vineyards mulched with straw confirmed to guarantee the best protection against soil erosion and runoff (Blavet et al., 2009; Corti et al., 2011; Maetens et al., 2012a). Novara et al. (2011) found out that the most effective cover crop was constituted of *Trifolium subterraneum*, *Festuca rubra* and *Lolium perenne* that decreased the mean erosion rate by 76% compared to tillage technique. This result was in line with Marques et al. (2010). Similarly, Morvan et al. (2014) measured low erosion rates for the grass covered plots and pointed out the importance of grass cover density in the wheel tracks of agricultural machinery to prevent runoff coefficient and erosion rate. In fact, tractor traffic in the inter-rows of vineyards may represent a remarkable issue because it causes soil compaction (Coulouma et al., 2006), leading to an increase of bulk density (Voorhees et al., 1979), a smaller infiltration rate (Boiffin and Monnier, 1994) and consequent higher runoff and soil erosion (Leonard and Andrieux, 1998; Quiquerez et al., 2008). Regarding SCTs, some authors pointed out the effects of the intensification of agriculture during the last four decades on soil erosion in Mediterranean vineyards (Lasanta and Sobrón, 1988; Martínez-Casasnovas and Sánchez-Bosch, 2000; Ramos and Martínez-Casasnovas, 2007; Cots-Folch et al., 2009). To maximize wine production, larger fields with low to moderate slopes were required. Consequently, land levelling and vine-rows along the slope replaced the traditional tillage techniques (Ramos and Porta, 1997) and terrace systems (Martínez-Casasnovas and Sánchez-Bosch, 2000; Tarolli et al., 2014; Tarolli et al., 2015). Lasanta and Sobrón (1988) found out that soil erosion rates on plots with modern tillage techniques (parallel to the maximum slope gradient) were 18 times higher than on those with traditional ones (perpendicular to the maximum slope gradient). Ramos and Martínez-Casasnovas (2007) proved that plots, with a high disturbance of the soil profile (HD), in which up to 3 m of soil was cut from the upper part of the plot and filled at the bottom, tended to seal quickly and show higher runoff volumes compared to the ones with a low disturbance of the soil profile (LD). The benefits of terracing have been deeply discussed by Tarolli et al. (2014), who also stressed the problem of terrace abandonment in relation to soil erosion risk. Terrace abandonment or the lack of maintenance determines an increase of erosion that can cause the terraces to collapse (Canuti et al., 2004; Crosta et al., 2003; Gallart et al., 1994; Lasanta et al., 2001) (Fig. 16).



Fig. 16. Examples of terrace abandonment (a) and lack of maintenance of terraces with consequent failure (b), of a vineyard located in the Tuscany region (central Italy) (photo by S. Calligaro).

Furthermore, changes in terracing systems may lead to higher erosion rates too, as pointed out by Cots-Folch et al. (2009). In fact, they demonstrated how the transition from stone-wall terraces to bench terraces, built by bulldozers to facilitate mechanized work, resulted in a more unstable system.

2.6 Future challenges

2.6.1 New guidelines

This literature review confirmed the concerning issue of soil water erosion in Mediterranean vineyards and the large variability of conditions that worked as a barrier to the interpretation of the environmental factors affecting this phenomenon. This is in line with García-Ruiz et al. (2015) who pointed out that no general rule had been applied to the study of erosion processes across the world. Furthermore, given the high erosion rates that can be reached, we believe that much more attention should be paid to the viticulture sector to reduce its impact on soil erosion issue. More researches should be carried out in several others Mediterranean areas to enrich the knowledge about the relationship between the triggering factors and links between the different spatial scales of analysis. This would help in filling the gap in knowledge about the negative effects of soil water erosion on vineyards and setting possible sustainable management practices to handle it. In fact, if no proper measures were taken to protect the soil, intensive agriculture would accelerate soil erosion processes to meet the increasing demand on food (Ziadat and Taimeh, 2013). At this purpose, this review suggests that more resources should be addressed to keep monitoring soil erosion by field measurements over longer periods. At the European level, a set of soil erosion indicators have already been developed and validated to evaluate the impact of agricultural and other land use policies (Gobin et al., 2002; Gobin et al., 2004).

Furthermore, these indicators have been included in the design of a European soil monitoring system by the ENVASSO project — Environmental Assessment of Soil for Monitoring — funded under the European Commission's 6th Framework Program (Morvan et al., 2008). However, as pointed out by García-Ruiz et al. (2015), currently, no protocols exist for standardizing the measurement methods and reporting the results of soil erosion studies. Therefore, this review stresses the fact that standardized procedures should be established for measuring soil erosion and reporting the results to enable data to be compared among different study areas. We think that the criteria we adopted in this work to organize the collected published data may serve as a guidance for future research. In our opinion, the variables we considered to fill the dataset represent the absolute minimum that every future scientific article, dealing with soil erosion by water in vineyards, should report. Depending on the methodology used and the temporal and spatial scale of analyses, we propose, to make data comparable with one other, to express soil erosion rates as $\text{kg m}^{-2} \text{s}^{-1}$, and soil loss data as kg m^{-2} . Regarding the spatial scale, measurement method, soil conservation techniques and soil texture, we recommend to use the corresponding classifications adopted in our work. In case of the rainfall simulation method, we suggest to make the experiments last until a constant infiltration rate over time is reached, at a determined reasonable rainfall intensity, whose value must be justified by an analysis of the rainfall characteristics of the study area. For this reason, we recommend to associate the rainfall intensity to its return period. Furthermore, attention must also be paid to the rain water quality used during the simulations. As the erosional and hydrological response of dispersive soils is strongly influenced by the chemical composition of the rain, the use of distilled water, unlike tap water, is highly recommended (Agassi et al., 1981, 1994; Borselli et al., 2001). Associated with the soil conservation techniques, we also suggest to report the percentages of bare and vegetated soil, and eventually, soil covered by rock fragments. In case of mulching application, we recommend to specify the type of mulch, its application rate, its cover at the moment of the analysis, and its thickness too. Furthermore, more researches should be carried out to understand better the role of alternative soil conservation techniques to chemical weed control in preventing the issue of soil water erosion. The use of grass cover, straw mulch and rock fragments gave promising results in reducing erosion rates and surface runoff but little is yet known. In addition, in order that these alternative soil conservation techniques can be really effective, they must be feasible and of real help to the farmers whose vineyards are affected by soil water erosion. Therefore, researches should also be at farmers' disposal to find out the best and possible sustainable solutions. Furthermore, concerning terraced vineyards, this literature review supports the advice from Tarolli et al. (2014) that terraced landscapes should need to be maintained, well managed and protected for both (i) reducing soil water erosion risk and (ii) offering economic benefits by improving tourism and the commerce of products related to viticulture.

2.6.2 Remote sensing

Given the great importance of measuring soil erosion processes over time, this may also benefit from remote-sensing technologies that allow to create high-resolution DEMs. The low-cost and flexible photogrammetric technique called 'Structure-from-Motion' (SfM) may represent a valid and cheaper alternative to the established airborne and terrestrial lidar (Light Detection and Ranging) technology for measuring soil surface changes (Westoby et al., 2012; Tarolli, 2014; Prosdocimi et al., 2015). Currently, through the SfM technique, it is possible to produce high-resolution DEMs from multi-stereo images without expert knowledge in photogrammetry, thanks to the advances of digital photogrammetry and computer vision. The interest for SfM technique is even enhanced by the fact that it can rely on built-in cameras of smartphones (Micheletti et al., 2014; Prosdocimi et al., 2015) and on the diffusion of unmanned aerial vehicles (UAVs). In fact, UAVs can be accoutered with ordinary cameras, thermal cameras and even laser scanners (Colomina and Molina, 2014) and allow to detect large areas without disturbing the investigated plot (Eltner et al., 2014). High-resolution topographic data obtained from SfM technique may provide new insights into the knowledge of erosion processes that affect vineyards at field scales. The results can help to schedule appropriate soil and water management techniques for a sustainable viticulture.

2.7 Conclusions

This work proposed a review of published studies that dealt with the problem of soil water erosion in Mediterranean vineyards. We believe that the dataset used in this work is one of the most complete and up to-date databases for studying the effect of triggering factors on soil water erosion. The high complexity of processes involved and variability of the conditions under which studies were carried out made the comparison between erosion rates difficult, unless data were separated from one to another according to the measurement method, and consequently, the temporal and spatial scales of analysis. Furthermore, the fact that very few studies provided complete information about all the considered variables concerned restricted our analyses. However, we believe that our analyses identified and confirmed significant general trends that can help to fill the gap in knowledge of soil water erosion in Mediterranean vineyards. The general statement that erosion rate increases as slope becomes steeper was proved to be statistically significant by the linear regression analyses only for the rainfall simulation method. Furthermore, linear regression analysis confirmed a significant and negative relationship between erosion rate and organic content only for the runoff simulation method, by proving that erosion rates usually increases as the organic content decreases. In addition to this, linear regression analysis revealed significant and positive relationships between erosion rates and mean rainfall intensity for both runoff simulation and runoff plot methods, and between soil loss and maximum rainfall intensity for runoff plot method. Multiple regression analysis confirmed the important role played by slope in affecting erosion rates and soil loss for rainfall simulation and runoff pot methods, respectively. Organic content confirmed to have a significant and negative effect on erosion rate only for runoff simulation method and

maximum rainfall intensity showed a non-linear and significant relationship with soil loss for runoff plot method. One-way ANOVA test, only applied to data derived from runoff simulation method, highlighted a significant difference between no-tillage and grass covered plots with respect to soil erosion rate. On the other hand, Kruskal–Wallis test, applied to data derived from runoff plot method, revealed a significant difference among all the four categories of soil conservation techniques considered in relation to soil loss with the exception of tillage and grass covered plots with mulching. Although these general trends emerged, we believe that more research should be carried out in other Mediterranean vineyards. This would lead to collect more spatially distributed data to improve the knowledge about the effects of the environmental factors on soil water erosion processes and to find out the most feasible and sustainable soil conservation techniques, which are of real help to the farmers. To achieve this task, a continuous and prolonged monitoring of soil erosion processes is necessary, but we strongly believe that it should rely on standardized procedures to allow the comparison of data derived from different study areas.

2.8 References

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3 Rainfall simulation and Structure-from-Motion photogrammetry for the analysis of soil water erosion in Mediterranean vineyards ²

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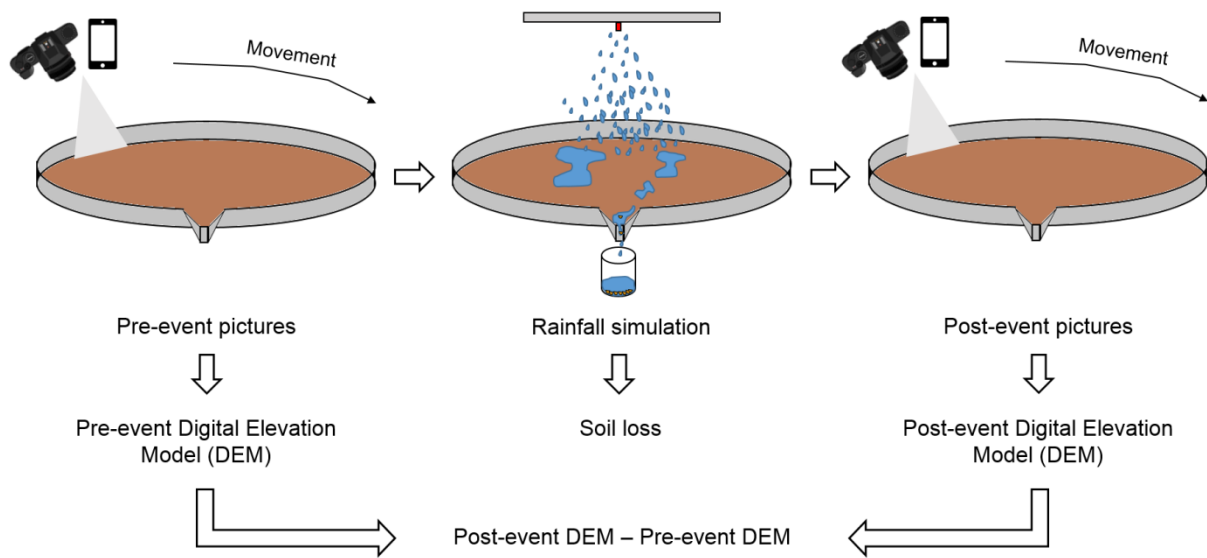
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3.1 Graphical abstract



3.2 Abstract

Soilwater erosion is a serious problem, especially in agricultural lands. Among these, vineyards deserve attention, because they constitute for the Mediterranean areas a type of land use affected by high soil losses. A significant problem related to the study of soil water erosion in these areas consists in the lack of a standardized procedure of collecting data and reporting results, mainly due to a variability among the measurement methods applied. Given this issue and the seriousness of soil water erosion in Mediterranean vineyards, this work aims to quantify the soil losses caused by simulated rainstorms, and compare them with each other depending on two different methodologies: (i) rainfall simulation and (ii) surface elevation change-based, relying on high-resolution Digital Elevation Models (DEMs) derived from a photogrammetric technique (Structure-from-Motion or SfM). The experiments were carried out in a typical Mediterranean vineyard, located in eastern Spain, at very fine scales. SfM data were obtained from one reflex camera and a smartphone built-in camera. An index of sediment connectivity was also applied to evaluate the potential effect of connectivity within the plots. DEMs derived from the smartphone and the reflex camera were comparable with each other in terms of accuracy and capability of estimating soil loss. Furthermore, soil loss estimated with the surface elevation change-based method resulted to be of the same order of magnitude of that one obtained with rainfall simulation, as long as the sediment connectivity within the plot was considered. High-resolution topography derived from SfM revealed to be essential in the sediment connectivity analysis and, therefore, in the estimation of eroded materials, when comparing them to those derived from the rainfall simulation methodology. The fact that smartphones built-in cameras could produce as much satisfying results as those derived from reflex cameras is a high value added for using SfM.

3.3 Introduction

Throughout the world, soil erosion by water is a serious problem, especially in semi-arid and semi-humid areas (Cerdà et al., 2009, 2015; Cerdan et al., 2010; García-Ruiz, 2010; Ligonja and Shrestha, 2015; Novara et al., 2016; Taguas et al., 2015; Rodrigo Comino et al., 2016a). Although soil erosion by water consists of physical processes that vary significantly in severity and frequency according to when and where they occur, they are also strongly influenced by anthropic factors such as land-use changes on large scales and unsustainable farming practices (Cerdà, 2000; León et al., 2015; López-Vicente et al., 2015; Ochoa-Cueva et al., 2015; Montgomery, 2007; Mwango et al., 2016; Nanko et al., 2015; Tarolli et al., 2014). This has led to the definition of 'accelerated' soil erosion as being the result of human impact on the landscape (Tarolli and Sofia, 2016) and this is found in all the continents (Borrelli et al., 2015; Cao et al., 2015; Gessesse et al., 2015; Rodrigo Comino et al., 2016b). The impact of soil erosion on modern society has required to set threshold values against which to assess the monitoring of soil data, especially in agriculture (Montgomery, 2007). Among the cultivated lands, vineyards merit a particular attention, because, aside from representing one of the most important crops in terms of income and employment, they also constitute, for the Mediterranean areas, a form of agricultural land use that causes the highest soil losses (Cerdà and Doerr, 2007; Cerdan et al., 2010; Martínez-Casasnovas and Sánchez-Bosch, 2000; Prosdocimi et al., 2016a; Raclot et al., 2009; Rodrigo Comino et al., 2015; Rodrigo Comino et al., 2016c). One of the main reasons for this is the bare soil under the vines that is exposed to high intensity rainfall events, mainly concentrated in spring, autumn and winter, which characterize the Mediterranean climate (Arnáez et al., 2007; Borga et al., 2011; García-Ruiz, 2010; Prosdocimi et al., 2016a). For this cultivation, the two most common soil management techniques are considered to be tillage, where the weeds are usually removed mechanically, and no-tillage, where the weeds are usually removed chemically (Novara et al., 2011; Raclot et al., 2009), and both of them generally turn out in bare soil management during the whole year. Extreme rainfall events that occur in the Mediterranean area are able to cause significant soil water erosion processes, especially when no protective material covers the soil (Fig. 1) (Bisantino et al., 2015; Keesstra et al., 2016; Novara et al., 2016; Prosdocimi et al., 2016c).

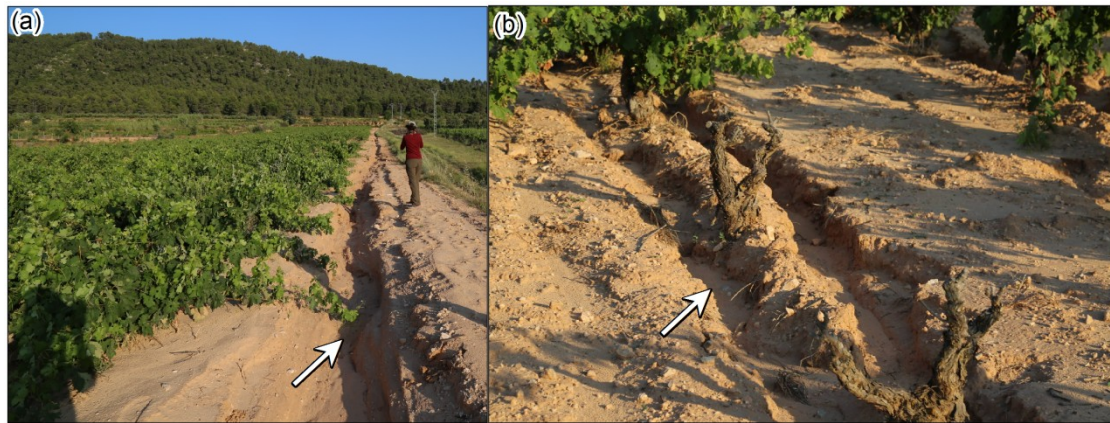


Fig. 1. Examples of soil water erosion processes caused by a 40 mm in 30 min thunderstorm occurred in mid-June 2015 in the study area. The white arrows point out a gully (a) and a rill (b).

However, to reduce the high soil erosion rates, more conservation-minded soil management practices have also been used such as mulching (Cerdà et al., 2015; Costantini et al., 2015; Jordán et al., 2011; Prosdocimi et al., 2016b, 2016c), cover crops (Novara et al., 2011), rock fragments (Blavet et al., 2009), natural grassing (Grimaldi et al., 2015; Mekonnen et al., 2015a; Mekuria et al., 2016; Raclot et al., 2009) and geotextiles (Giménez Morera et al., 2010; Mekonnen et al., 2015b; Mengistu et al., 2016). Furthermore, new approaches to evaluate incentives for the adoption of agri-environment measures in degraded and eroded vineyards have been implemented (Galati et al., 2015) and mulching is one of those successful strategies (Prosdocimi et al., 2016c). Another issue related to soil water erosion in Mediterranean vineyards is the lack of a standardized procedure of collecting data and reporting results, mainly due to a great variability among the measurement methods applied to quantify it (Prosdocimi et al., 2016a; García-Ruiz et al., 2015). This induces difficulties in comparing data coming from different studies and obtained with different methodologies. Based on the paper review of Prosdocimi et al. (2016a), six different methodologies to assess soil water erosion in vineyards have been identified: (i) experimental plot stations under simulated or natural rainfalls, (ii) erosion markers, (iii) models, (iv) the surface elevation change-based methods, (v) geochemical methods, and (vi) carbon stable isotopes. This work focuses on the use of plot stations under simulated rainfall and on the surface elevation change-based method. Rainfall simulation has become a very effective technique for assessing soil erosion, particle detachment and overland flow at very fine scales (Arnáez et al., 2007; Cerdà et al., 1997; Iserloh et al., 2013; Rodrigo Comino et al., 2016b). Several types and designs of rainfall simulators have been realized to meet the objectives of researchers (Iserloh et al., 2013; Lassu et al., 2015). In particular, the advantages of using a portable rainfall simulator are: i) its versatility, ii) low cost and easy operation, and iii) capability of obtaining data under controlled conditions and over relatively short periods of time. The surface elevation change-based method is able to detect the topographic changes over time. It relies on Digital Elevation Models (DEMs) that can be used as basic topographic information to derive morphometric attributes and quantify soil erosion and deposition rates (Martínez-Casasnovas and Sánchez-Bosch,

2000; Martínez-Casasnovas et al., 2002; Prosdocimi et al., 2015). Remote-sensing technologies have proven to facilitate significantly the creation of high-resolution DEMs (Aucelli et al., 2016; Tarolli, 2014; Tarolli et al., 2015), and the availability of DEMs at multiple scales in terms of resolution but also temporal coverage is becoming essential to the understanding of global issues, such sediment production and anthropogenic changes to the Earth system, among others (Sofia et al., 2016). The recent development of the photogrammetric technique 'Structure-from-Motion' (SfM) has confirmed to represent a valid and cheaper alternative to the established airborne and terrestrial lidar (Light Detection and Ranging) technology for measuring soil surface changes in different environments (Dandois and Ellis, 2013; Eltner et al., 2015; James and Robson, 2012; Masiero et al., 2015; Piermattei et al., 2016; Westoby et al., 2012; Whitehead et al., 2013; Woodget et al., 2015). All this information can shed light into the connectivity within the soil and water losses (López-Vicente et al., 2016; Marchamalo et al., 2016; Masselink et al., 2016). The growing interest for SfM has been enhanced by the fact that it is a user-friendly technique, and that it can also rely on smartphone built-in cameras (Masiero and Vettore, 2016; Micheletti et al., 2014; Prosdocimi et al., 2015) and on the diffusion of unmanned aerial vehicles (UAVs) (Chen et al., 2015; Colomina and Molina, 2014). Given the seriousness of soil water erosion in Mediterranean agricultural lands and the issue of putting data obtained with different methodologies in relation to each other, this work intends to quantify the soil losses caused by simulated rainstorms, and compare them with each other depending on two different methodologies used: (i) rainfall simulation and (ii) surface elevation change-based, relying on high-resolution DEMs derived from SfM. Furthermore, this work aims to compare the results obtained from SfM with each other, depending on the type of camera used. The objectives are pursued by carrying out the experiments in a typical Mediterranean vineyard, under tillage conditions, located within the province of Valencia (Spain), at very fine scales (0.25 m^2).

3.4 Materials and methods

3.4.1 Study area

The study area consists in a 25-year-old vineyard, located at El Celler del Roure in Les Alcusses de Moixent, within the Canyoles river watershed in the province of Valencia (La Costera District, eastern Spain) ($38^{\circ} 48' 33.12'' \text{ N}$, $0^{\circ} 49' 3.27'' \text{ O}$). Vines are located parallel to the contour lines and the inter-rows, which are about 2.5 m wide, are artificially maintained bare during the whole year through tillage operations carried out with a Landini Rex 95 tractor which adopts a tooth arrow as farm implement. The portion affected by the tractor wheel tracks results to be about 36% of the total inter-row area (Fig. 2).



Fig. 2. Visual perspective of the tilled inter-rows where the tractor wheel tracks are well visible (black arrows) (a). The white arrows stress the soil sediments that were transported following the 40 mm in 30 min thunderstorm occurred in mid-June 2015.

Climate is typically Mediterranean with 3–5 months of summer drought (June–September). Mean annual rainfall is about 350 mm yr^{-1} . Rainfall is distributed among autumn, winter and spring, with maximum peak rainfall intensities during the autumn season, where values higher than 200 mm day^{-1} were recorded during the last 50 years. Mean annual temperature is about $13.8 \text{ }^{\circ}\text{C}$ while the hottest month (August) has average temperatures of about $23 \text{ }^{\circ}\text{C}$. The parent materials in this area belong to Cretaceous limestones and Tertiary Marly deposits that develop Typic Xerothent soils (Soil Survey Staff, 1998). The soils are characterized by low levels of soil organic matter ($< 1\%$) due to the millennia of agricultural use and soil disturbance (ploughing), basic pH (8) (Prosdociimi et al., 2016b), sandy loam soil textures (clay 19.3%, silt 13.4% and sand 67.3%), and low bulk density (1.109 g cm^{-3}). To better characterize the climate of our study site, Walter-Lieth climate diagram (Walter and Lieth, 1960) has been obtained by using data derived from Ontinyent climate station as it is the one with the longest records (29 years) closest to the study site (about 17 km) (Fig. 3). The diagram displays monthly averages for temperature and precipitation over a year. When the precipitation curve undercuts the temperature curve, the area in between them indicates dry season. When the precipitation curve supersedes the temperature curve, the area in between them indicates moist season. For further information, readers may refer to <http://www.globalbioclimatics.org/>.

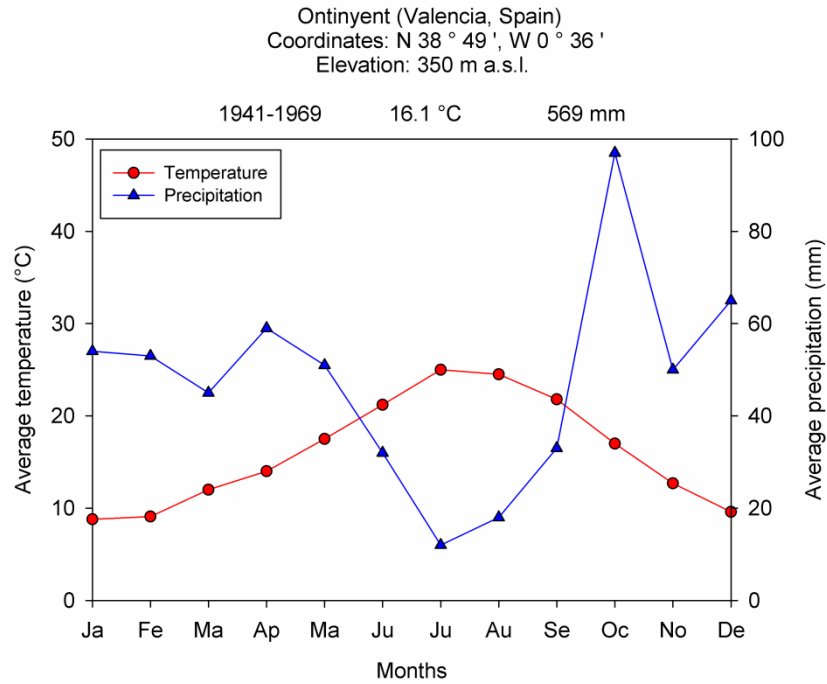


Fig. 3. Walter-Lieth climate diagram (Walter and Lieth, 1960) computed for the Ontinyent climate station as it is the one with the longest records (29 years) closest to our study site (about 17 km). The information above the panel corresponds to station location, the period of years recorded, the mean annual temperature and the mean annual precipitation.

3.4.2 Experimental plot design

Four circular steel plots (0.25 m²) were located in the bare inter-rows of the vines managed with conventional tillage, and are referred to in the text as 1, 2, 3 and 4. Each plot was placed in a different inter-row and had an outlet, which allowed to converge and collect the surface runoff samples during the runoff simulation experiments. For each plot, five targets (SfM-targets), made of black and white polythene squares, were used: four (5.5 cm × 5.5 cm) were placed outside the circular plots and one (2.5 cm × 2.5 cm) inside the plot (Fig. 4). SfM-targets centroids were surveyed using a Topcon GRS-1 rover receiver running in real time kinematic (RTK) mode. In addition, other thirteen ground-control points (GCPs) were surveyed in the immediate neighborhood of each plot.

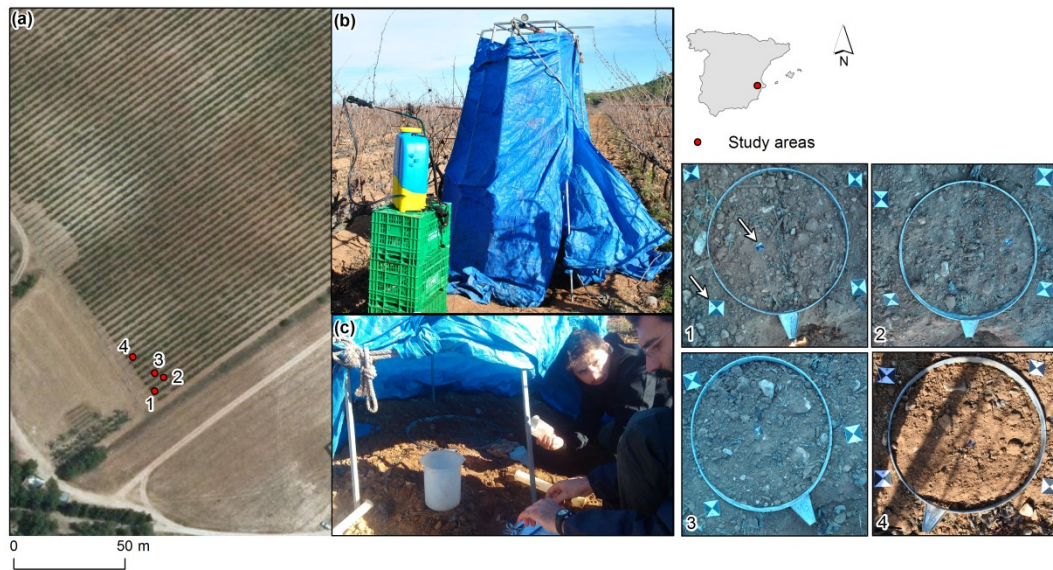


Fig. 4. Localization of the study areas (a), that correspond to the four circular plots (1, 2, 3 and 4) where the rainfall simulation and photogrammetric surveys were carried out. Views of the rainfall simulator (b) and of the rainfall simulation experiment in action (c) are also shown.

3.4.3 Rainfall simulation

A one-nozzle (Hardi-1553-12) rainfall simulator was used to reproduce seven rainstorms at 55 mm h^{-1} rainfall intensity for one hour on the 4 circular plots of 0.25 m^2 . For plots 1, 2 and 3, a single rainfall experiment was carried out, while for plot 4, four rainfall experiments were carried out during four consecutive days, and are referred to in the text as 4A, 4B, 4C and 4D. Storms similar to the ones simulated have a return period of 10 years in the study area (Cerdà, 1996; Prosdocimi et al., 2016b). The rainfall simulator used was the one described by Cerdà et al. (1997) because it revealed to be effective in rugged terrain conditions proving to give good results in semi-arid environments. Its basic components are a nozzle, a structure that holds the nozzle, the connection with the water supply, the pumping system and a tarpaulin to protect the rainfall simulation from wind. As the nozzle was kept at about 2 m height over a plane surface, the 0.25 m^2 plots were established at the centre of the 1 m^2 sprinkling area, to avoid border interference. Readers are referred to Cerdà et al. (1997) and Iserloh et al. (2013) for a further description of the rainfall simulator used and Cerdà (1996, 1997) for more information about the distribution of rainfall parameters. Surface runoff from the plots were collected and measured at 1-min intervals during each simulated rainfall event. Every tenth 1-min runoff sample was collected for laboratory analysis in order to determine sediment concentration, that was obtained after the desiccation of the samples in the laboratory. Then, runoff rates and sediment concentration were used to calculate the soil loss, runoff, runoff coefficient, and erosion rates.

3.4.4 Surface elevation changes through Structure-from-Motion

Photographs of each plot were taken using two different types of camera: (i) a standalone digital reflex camera (Nikon D3000 at 10.2 MP resolution, set at a focal length of 35 mm) and (ii) a smartphone, precisely a BQ Aquaris E5, built-in camera (13 MP resolution) with both automatic focusing and exposure enabled. The choice of using two cameras was due to test the effectiveness of SfM, also when it relies on an image dataset derived from a smartphone. Twenty photographs were taken before and after the rainfall simulation using each camera. A 1 m high support having two boxes, that were 0.3 m far from each other and capable of holding the cameras, was used to take the pictures (Fig. 5). Photographs were taken inside the rainfall simulator covered by the tarpaulin to have a homogeneous light over the plots.

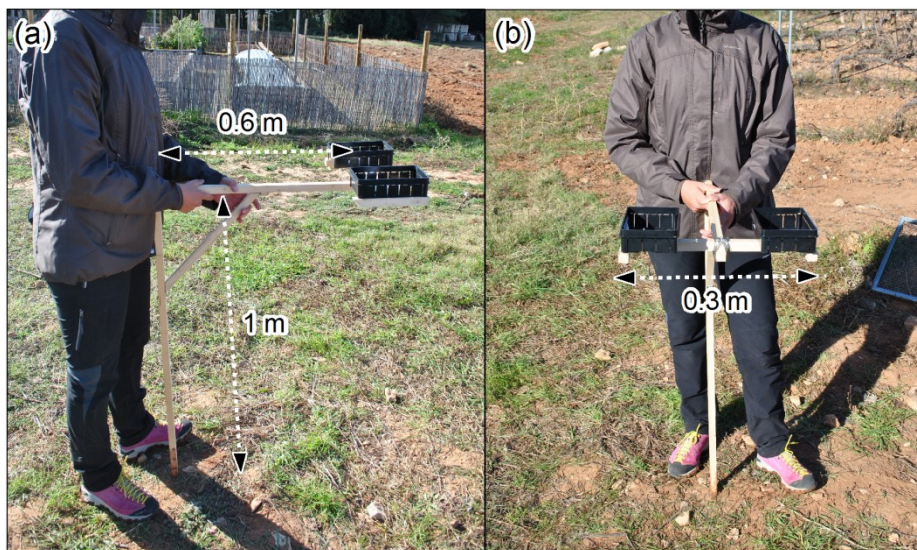


Fig. 5. Two visual perspectives of the support used to take the pictures. The support consists in a main pole, 1 m high, with two boxes that stick out the main pole for 0.6 m (a) and are 0.3 m far from each other (b). The boxes were designed to hold the cameras with the lens downwards facing.

The SfM technique was then used to obtain three-dimensional (3D) georeferenced point clouds and to generate 0.01 m resolution DEMs for each plot. The thirteen points collected in the immediate neighborhood of each plot (see the previous chapter Experimental plot design) were used as GCPs to assess the accuracy and precision of the DEMs through the computation of the root-mean-square-error (RMSE), mean error, and standard deviation of error (SDE). The working principles of SfM are similar to those of stereoscopic photogrammetry, namely that the 3D model can be created from overlapping, offset images. However, unlike traditional photogrammetry, in which either the position of the camera or the positions of some points are known prior to scene reconstruction (Fonstad et al., 2013; Verhoeven et al., 2012; Westoby et al., 2012), in the SfM, matches are made between points across many photographs without prior knowledge of the camera position (Lowe, 2004). The images acquired were processed using the commercial software Agisoft PhotoScan®, as already successfully considered

in different analyses (Doneus et al., 2011; Javernick et al., 2014; Piermattei et al., 2016; Prosdocimi et al., 2015; Verhoeven et al., 2012; Woodget et al., 2015). A custom algorithm similar to the Lowe's (2004) Scale Invariant Feature Transform (SIFT) object recognition system was used by the software to determine the 3D location of matching features in multiple images. Then, camera position was calculated by estimating the camera's intrinsic (focal length, principal point, and lens distortion) and extrinsic (projection centre location and the six exterior orientation parameters that define the image) orientation parameters. This was done by using a bundle-adjustment algorithm (Javernick et al., 2014; Robertson and Cipolla, 2009; Verhoeven et al., 2012). Afterwards, the software created a dense surface, usually referred to as mesh, by using these parameters and a dense multi-view stereo reconstruction (DMVR) (Agisoft, 2016). The mesh was generated in a relative 'image-space' coordinate system (Westoby et al., 2012), and therefore, it required to undergo a linear similarity transformation using seven parameters (three translation, three rotation, and one scaling), based on known GCPs, to be transformed to an absolute coordinate system. The GCPs corresponded to the SfM-targets centroids, whose x, y and z coordinates were previously recorded with Topcon GRS-1. As the linear similarity transformation could not remove non-linear model misalignments (Woodget et al., 2015), an optimization transformation method was applied to minimize geometric distortions within the mesh (Agisoft, 2016). Thereafter the mesh was rebuilt and the 3D georeferenced point could be exported. The georeferenced point clouds are referred to in the text as $GEOPre_{NKN}$ and $GEOPost_{NKN}$, for those derived from the Nikon camera before and after the rainfall simulation, respectively, and $GEOPre_{PHO}$ and $GEOPost_{PHO}$ for those derived from the smartphone camera before and after the rainfall simulation, respectively. Furthermore, the number of the plot is also included (1, 2, 3, 4A, 4B, 4C and 4D). Then, the SfM final point clouds were further manipulated using the open source program CloudCompare® (Girardeau-Montaut, 2015) to remove additional noise that typically affects these data (Javernick et al., 2014; Prosdocimi et al., 2015). In this case, given the small size of the plots, the noise removal was accomplished manually. Finally, the elevation points were interpolated by the natural neighbor method (Sibson, 1981) to generate 0.01 m resolution DEMs. The DEMs are referred to in the text as $DEMPre_{NKN}$ and $DEMPost_{NKN}$, for those derived from the Nikon camera before and after the rainfall simulation, respectively, and $DEMPre_{PHO}$ and $DEMPost_{PHO}$ for those derived from the smartphone camera before and after the rainfall simulation, respectively. Furthermore, the number of the plot is also included (1, 2, 3, 4A, 4B, 4C and 4D). The $DEMPre_{NKN}$ obtained for each plot are shown in Fig. 6.

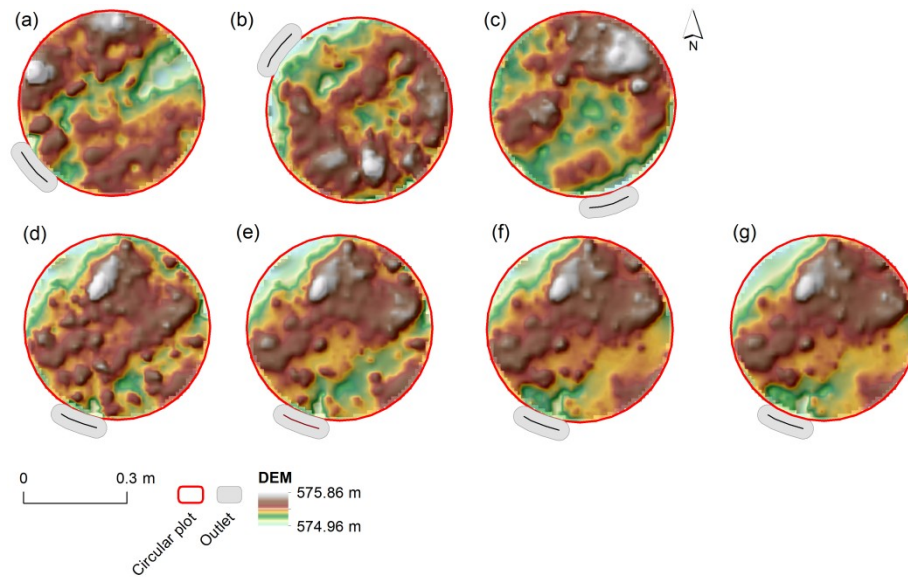


Fig. 6. DEMsPre_{NKN} (0.01 m resolution) obtained for each plot: (a) DEMs1Pre_{NKN}, (b) DEMs2Pre_{NKN}, (c) DEMs3Pre_{NKN}, (d) DEMs4APre_{NKN}, (e) DEMs4BPre_{NKN}, (f) DEMs4CPre_{NKN}, and (g) DEMs4DPre_{NKN}.

For the objectives of this work, all the analysis was based on the final DEMs, as done by Bangen et al. (2014), Calligaro et al. (2013), Javernick et al. (2014), Prosdocimi et al. (2015), Tarolli et al. (2015), and Wechsler (2007). The DEMs derived from the smartphone were then directly compared to the DEMs derived from the camera, by assuming a normal distribution and using robust statistical methods (Höhle and Höhle, 2009; Prosdocimi et al., 2015). This entailed the computation of the mean error, SDE, RSME, median, and normalized median absolute deviation (NMAD).

3.4.5 Computation of soil loss

Soil loss was computed for both rainfall simulation and surface elevation change-based methodologies. For rainfall simulation methodology, the runoff samples were used to determine the sediment concentration and, then, the runoff rates and sediment concentration were used to calculate the total soil loss (g). For the surface elevation change-based methodology, SfM was applied to obtain high-resolution DEMs before (DEMsPre) and after (DEMsPost) the rainfall simulation. Then, the so-called morphological method (Ashmore and Church, 1998) was used to estimate the soil loss. The morphological method consists in carrying out repeated topographic surveys from which DEMs can be obtained and differenced to produce DEMs of difference (DoDs). The volumes of eroded materials (cm³) were computed by considering the DEMsPre and DEMsPost for each plot and for each camera by using the Geomorphic Change Detection (GCD) 6.1.14 toolbar embedded in an ESRI® add-in for ArcGIS 10.X that is freely downloadable from <http://gcd.joewheaton.org/downloads>. Then, the volumes of eroded materials were turned into soil loss expressed in grams, by knowing the bulk density. The GCD allows to compute the volumes of deposited materials too, but, for this work, only eroded materials have been considered, to make a comparison with the

soil loss derived from the rainfall simulation methodology. The DoDs are referred to in the text as DoDs_{NKN} and DoDs_{PHO} for those derived from the Nikon and smartphone cameras, respectively. DEMs' uncertainty in DoDs has also been considered (Brasington et al., 2000; Lane et al., 1994; Lane, 1998; Lane et al., 2003; Prosdocimi et al., 2015; Wheaton, 2008; Wheaton et al., 2010). In this case, DEMs' uncertainties were evaluated according to a probabilistic thresholding that can be carried out with a user-defined confidence interval (Brasington et al., 2003; Lane et al., 2003; Taylor, 1997):

$$U_{crit} = t \left(\sqrt{SDE_{new}^2 + SDE_{old}^2} \right) \quad (1)$$

where U_{crit} is the critical threshold error propagated in the DoD and SDE_{new} and SDE_{old} are the individual standard deviation errors in DEM_{new} (post-event) and DEM_{old} (pre-event), respectively. U_{crit} is based on a critical student's t -value at a chosen confidence interval where:

$$t = \frac{|z_{DEM_{new}} - z_{DEM_{old}}|}{\delta u_{DoD}} \quad (2)$$

where $|z_{DEM_{new}} - z_{DEM_{old}}|$ is simply the absolute value of the DoD. The probability of a DoD predicted elevation change occurring due the uncertainty can then be calculated by relating the t -statistic to its cumulative distribution function. In this work, we used the 95% confidence interval as a threshold, as also suggested by Wheaton et al. (2010).

3.4.6 Sediment connectivity

Sediment connectivity is defined as the connected transfer of sediment from a source to a sink in a system through processes of sediment detachment and transport (Bracken et al., 2015). The concept of connectivity has increasingly been used in quantitative process-based sediment dynamics research, especially at catchment scales (Ali et al., 2014; Baartman et al., 2013; Bracken and Croke, 2007; Bracken et al., 2015; Brierley et al., 2006; Cavalli et al., 2013; Fryirs et al., 2007; Heckmann and Schwanghart, 2013; Lexartza-Artza and Wainwright, 2011; López-Vicente et al., 2013; Wainwright et al., 2011). Geomorphology has been considered as a major driver on determining sediment connectivity (Heckmann and Schwanghart, 2013; Theler et al., 2010), and geomorphometric indices have increasingly been developed to assess it (Borselli et al., 2008; Cavalli et al., 2013; López-Vicente et al., 2013; Reid et al., 2007; Sougnez et al., 2011). In this study we applied the index of connectivity (IC) as proposed by Cavalli et al. (2013) based on the work of Borselli et al. (2008), to evaluate the potential effect of sediment connectivity within the plots. The reasons for this choice relied on the facts that the IC (i) is a distributed geomorphometric index that can be easily derived from a DEM, (ii) can be computed with reference to specific target features, and (iii) has been adapted for high-resolution DEMs. The IC has been developed as a ToolBox for ArcGis 10.1 or as stand-alone application based on Python scripting with bindings for processing geographical datasets. It uses functionalities and algorithms available in TauDEM 5.2 tool (Tarboton, 2013) and it is freely downloadable from

<http://www.sedalp.eu/download/tools.shtml>. This index mainly focuses on the influence of topography on sediment connectivity, and takes into account the characteristics of the drainage area (upslope component, D_{up}) and the flow path length that a particle has to travel to arrive at the nearest sink (downslope component, D_{dn}). The IC is computed as follows:

$$IC = \log_{10} \left(\frac{D_{up}}{D_{dn}} \right) = \log_{10} \left(\frac{\overline{W} \overline{S} \sqrt{A}}{\sum_i \frac{d_i}{W_i S_i}} \right) \quad (3)$$

where $\overline{W}$ is the average weighting factor of the upslope contributing area (dimensionless), $\overline{S}$ is the average slope gradient of the upslope contributing area (m/m), A is the upslope contributing area (m^2), d_i is the length of the flow path along the i^{th} cell according to the steepest downslope direction (m), W_i and S_i are the weighting factor and the slope gradient of the i^{th} cell, respectively. IC can assume values ranging from $-\infty$ to $+\infty$, with connectivity increasing for larger IC values.

3.5 Results and discussion

3.5.1 Nikon and smartphone built-in cameras comparisons

Regarding the comparisons between the Nikon and smartphone built-in cameras, the georeferentiation errors (RMSE) calculated by the Agisoft PhotoScan® software along the x, y and z-axes for each SfM point cloud are reported (Table 1).

Table 1 Georeferentiation errors (RMSE) calculated by Agisoft PhotoScan® along the x, y and z-axes for each point cloud derived from SfM technique. GEOPre_{NKN} and GEOPost_{NKN} refer to the point clouds derived from the Nikon camera before and after the rainfall simulation, respectively, and GEOPre_{PHO} and GEOPost_{PHO} for those derived from the smartphone camera before and after the rainfall simulation, respectively. The number of the plot is also included (1, 2, 3, 4A, 4B, 4C and 4D).

	X Error (± m)	Y Error (± m)	Z Error (± m)
GEO1Pre _{NKN}	0.0119	0.0030	0.0038
GEO1Pre _{PHO}	0.0119	0.0030	0.0041
GEO1Post _{NKN}	0.0113	0.0029	0.0045
GEO1Post _{PHO}	0.0113	0.0029	0.0046
GEO2Pre _{NKN}	0.0123	0.0024	0.0043
GEO2Pre _{PHO}	0.0125	0.0026	0.0071
GEO2Post _{NKN}	0.0126	0.0028	0.0034

Table 1 (*continued*)

	X Error ($\pm$ m)	Y Error ($\pm$ m)	Z Error ($\pm$ m)
GEO2Post _{PHO}	0.0138	0.0017	0.0060
GEO3Pre _{NKN}	0.0085	0.0033	0.0105
GEO3Pre _{PHO}	0.0074	0.0044	0.0094
GEO3Post _{NKN}	0.0093	0.0042	0.0120
GEO3Post _{PHO}	0.0091	0.0042	0.0118
GEO4APre _{NKN}	0.0125	0.0062	0.0041
GEO4APre _{PHO}	0.0131	0.0059	0.0044
GEO4APost _{NKN}	0.0133	0.0079	0.0008
GEO4APost _{PHO}	0.0142	0.0065	0.0010
GEO4BPre _{NKN}	0.0126	0.0083	0.0008
GEO4BPre _{PHO}	0.0127	0.0083	0.0009
GEO4BPost _{NKN}	0.0129	0.0082	0.0006
GEO4BPost _{PHO}	0.0130	0.0083	0.0006
GEO4CPre _{NKN}	0.0127	0.0083	0.0016
GEO4CPre _{PHO}	0.0126	0.0083	0.0017
GEO4CPost _{NKN}	0.0128	0.0084	0.0011
GEO4CPost _{PHO}	0.0127	0.0084	0.0011
GEO4DPre _{NKN}	0.0128	0.0084	0.0011
GEO4DPre _{PHO}	0.0132	0.0085	0.0009
GEO4DPost _{NKN}	0.0132	0.0083	0.0011
GEO4DPost _{PHO}	0.0131	0.0085	0.0011

The SfM point clouds show an average error of the order of about 0.01 m along the x-axis, and an even lower order error along the y and z-axes. These good results support the choice of setting the DEMs resolution equal to 0.01 m and can be explained by the fact that: (i) the plots were very small, (ii) the 5 SfM-targets were well distributed over each plot, and (iii) the pictures were taken in a correct way, thanks to the support used, the expedient of shooting photographs inside the tarpaulin, and the short distance between the position of the cameras and the plots (about 1 m). Furthermore, differences between the DEMs_{PHO} and DEMs_{NKN} for the unthresholded DEMs (where no uncertainty analysis was carried out) were also evaluated with accuracy measures assuming a normal distribution and more robust parameters too (Table 2).

Table 2 Accuracy measures of DEMs_{PHO} checked by DEMs_{NKN} with the assumption of normal distribution and more robust parameters too. DEMPre_{NKN} and DEMPost_{NKN} refer to DEMs derived from the Nikon camera before and after the rainfall simulation, respectively, and DEMPre_{PHO} and DEMPost_{PHO} for those derived from the smartphone camera before and after the rainfall simulation, respectively. The number of the plot is also included (1, 2, 3, 4A, 4B, 4C and 4D).

	Minimum (m)	Maximum (m)	Mean (m)	SDE (m)	Kurtosis	Skeweness	NMAD (m)	Median (m)
DEM1Pre _{PHO} - DEM1Pre _{NKN}	-0.0160	0.0210	0.0003	0.0022	12.5108	0.2772	0.0015	0.0003
DEM1Post _{PHO} - DEM1Post _{NKN}	-0.0344	0.0336	-0.0002	0.0026	88.9927	-1.3843	0.0010	-0.0002
DEM2Pre _{PHO} - DEM2Pre _{NKN}	-0.0135	0.0142	0.0015	0.0031	4.1464	-0.2322	0.0024	0.0017
DEM2Post _{PHO} - DEM2Post _{NKN}	-0.0063	0.0173	0.0049	0.0029	3.9343	-0.0287	0.0022	0.0049
DEM3Pre _{PHO} - DEM3Pre _{NKN}	-0.0062	0.0054	-0.0002	0.0019	2.5106	0.1547	0.0016	-0.0003
DEM3Post _{PHO} - DEM3Post _{NKN}	-0.0056	0.0059	-0.0003	0.0010	6.3428	0.1691	0.0007	-0.0003
DEM4APre _{PHO} - DEM4APre _{NKN}	-0.0139	0.0168	-0.0009	0.0026	8.5218	0.6003	0.0018	-0.0009
DEM4APost _{PHO} - DEM4APost _{NKN}	-0.0201	0.0242	-0.0012	0.0043	5.6034	0.3439	0.0031	-0.0015
DEM4BPre _{PHO} - DEM4BPre _{NKN}	-0.0193	0.0239	0.0003	0.0046	4.9291	0.0854	0.0034	0.0002
DEM4BPost _{PHO} - DEM4BPost _{NKN}	-0.0067	0.0078	-0.0001	0.0014	6.2354	0.0027	0.0010	-0.0002
DEM4CPre _{PHO} - DEM4CPre _{NKN}	-0.0057	0.0061	0.0001	0.0012	5.3686	-0.1376	0.0009	0.0002
DEM4CPost _{PHO} - DEM4CPost _{NKN}	-0.0117	0.0128	0.0002	0.0028	5.6941	0.2353	0.0020	0.0002
DEM4DPre _{PHO} - DEM4DPre _{NKN}	-0.0068	0.0092	-0.0001	0.0017	5.7170	0.5328	0.0012	-0.0002
DEM4DPost _{PHO} - DEM4DPost _{NKN}	-0.0104	0.0115	0.0000	0.0023	5.8356	0.2322	0.0016	-0.0001

From Table 2, emerges that all the DEMs_{PHO} are comparable to DEMs_{NKN}. Mean values are of the order of about 0.0001 m and SDE values of the order of about 0.001 m. Skewness and kurtosis confirm the fact that the elevation differences do not follow normal distributions (Höhle and Höhle, 2009; Sofia et al., 2013), and this supports the choice of considering more robust parameters too such as NMAD and median. However, also when considering these more robust approaches, DEMs_{PHO} confirm to be comparable to DEMs_{NKN}, showing NMAD and median values of the order of about 0.001 and 0.001 m, respectively.

3.5.2 Soil loss

Fig. 7 shows the DoDs derived from SfM, by considering the DEMs_{PreNKN} and DEMs_{PostNKN} for each plot, thresholded according to the probabilistic thresholding with a 95% confidence interval. The fact that, the thresholding of DoDs entails a loss of information, is expected and occurs at the expense of a better geomorphic plausibility (Wheaton et al., 2010). Elevation differences range from negative values (red colour), to which correspond net eroded sediments, to positive values (blue colour), to which correspond net deposited sediments.

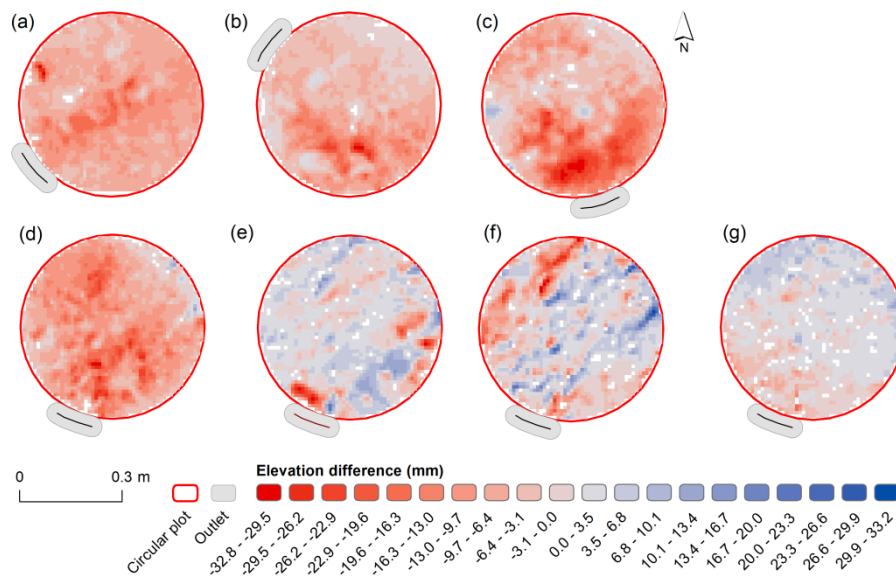


Fig. 7. DoDs derived from the Nikon dataset, thresholded according to the probabilistic thresholding with a 95% confidence interval and obtained for each plot: (a) Plot 1, (b) Plot 2, (c) Plot 3, (d) Plot 4A, (e) Plot 4B, (f) Plot 4C, and (g) Plot 4D.

From Fig. 7 emerges that plots 1, 2, 3 and 4A mainly show negative elevation differences. This means that the single simulated rainfall event caused more erosion than deposition, and this can be explained by the fact that the plots, at the beginning, have more material which is prone to be washed away. In contrast, plots 4B, 4C and 4D show greater elevation differences. This suggests that, as rainfall events follow one another, the soil particles, that are susceptible to be eroded, diminish, and therefore, the soil shows elevation differences which are closer to zero values, where zero

corresponds exactly to no difference at all between before and after the rainstorm. Fig. 8 shows the soil loss data, expressed in grams, derived from both the methodologies applied. For the surface elevation change-based method, the data coming from the DoDs obtained with both the Nikon and smartphone cameras are reported.

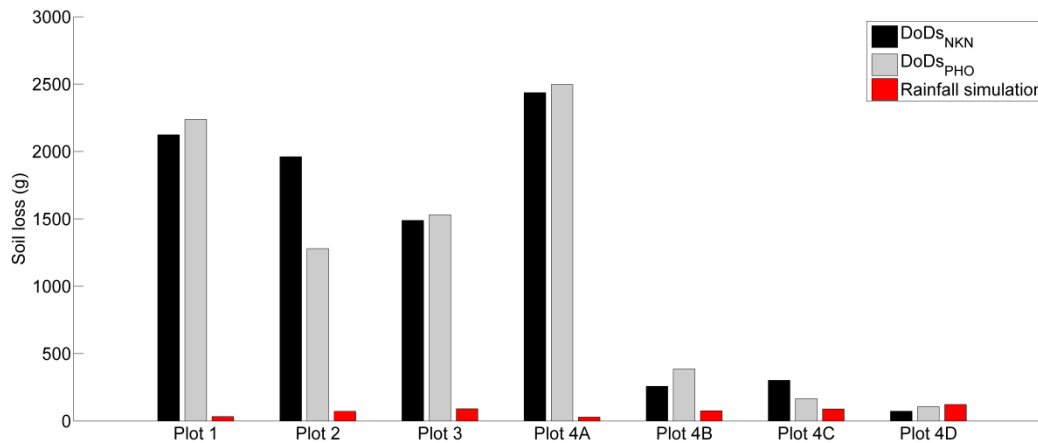


Fig. 8. Soil loss data, expressed in grams, derived for each plot from both the methodologies applied: rainfall simulation and surface elevation change-based relying on DoDs. DoDs_{NKN} and DoDs_{PHO} refer to soil loss estimated from Nikon and smartphone cameras, respectively.

From Fig. 8 emerges how the soil loss data estimated with the two methodologies are not comparable with each other, especially for the plots 1, 2, 3 and 4A, where only a single rainstorm was artificially reproduced. On the contrary, soil loss data derived from the same methodology, namely surface elevation change-based, are comparable with each other, independently from the type of camera used. Soil loss derived from the surface elevation change-based method result to be of two orders of magnitude greater than the one obtained with rainfall simulation. However, this discrepancy is in line with the processes that are involved and analysed with the two different methodologies. Rainfall simulation accounts for splash and initial inter-rill erosion processes and allows to study the impact of rain drops on sediment detachment, transport and runoff initiation. However, when it rains the water is able to disintegrate some of the soil aggregates, leading to the collapse of micro-pores and to the surface seal formation. Furthermore, the water that infiltrates makes also the soil heavier, causing a lowering of the soil surface, which is the process that DoDs are able to detect. To overcome this discrepancy between the two methodologies, sediment connectivity within the plots has been taken into consideration too.

3.5.3 Sediment connectivity analysis

Other than rainfall intensity and kinetic energy, also micro-topography plays a key role in the collection of eroded materials, especially when the experiments are carried out at very fine scales, as in our case. To prove this, Fig. 9 shows the maps of the connectivity index calculated with regard to the plots outlets, by considering, as inputs, the DEMsPre_{NKN}. As no reference theory exists for the partitioning of the connectivity index

into classes, we relied on the same classification provided by Tarolli and Sofia (2016), in which they proposed to adopt a relative classification into four classes (High, Medium-High, Medium-Low and Low) by considering break points that best grouped similar values and maximized the differences between classes (natural breaks).

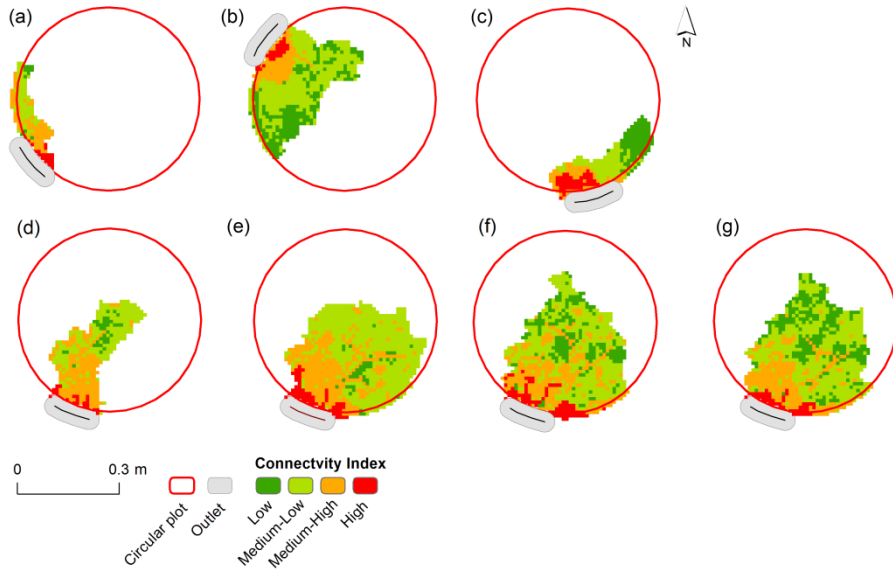


Fig. 9. Connectivity index maps calculated with regard to the plots outlets, by considering, as inputs, the DEMsPre_{NKN}, for each plot: (a) Plot 1, (b) Plot 2, (c) Plot 3, (d) Plot 4A, (e) Plot 4B, (f) Plot 4C, and (g) Plot 4D.

From Fig. 9 emerges how (i) each plot has different patterns of sediment connectivity, which vary whether or not consecutive rainstorms occur (Fig. 9d–g), and (ii) not all the soil within the plots is connected to the outlet. This proves the fact that the placement of the plots in the field is extremely important because micro-reliefs with their roughness can facilitate sediment dis-connectivity. The portions of soil that are more connected to the outlet are those that are closer to it. Therefore, these portions, which correspond to the Medium-High and High classes of the connectivity index maps, are reasonably those that will be more prone to erosion, once the rainstorm occurs. As a consequence, by masking the elevation differences maps (Fig. 7) with the Medium-High and High classes of the connectivity index maps (Fig. 9), we recomputed the soil loss derived from the surface elevation change-based method, considering both the Nikon (DoDs_{NKN} IC) and smartphone (DoDs_{PHO} IC) DoDs (Fig. 10).

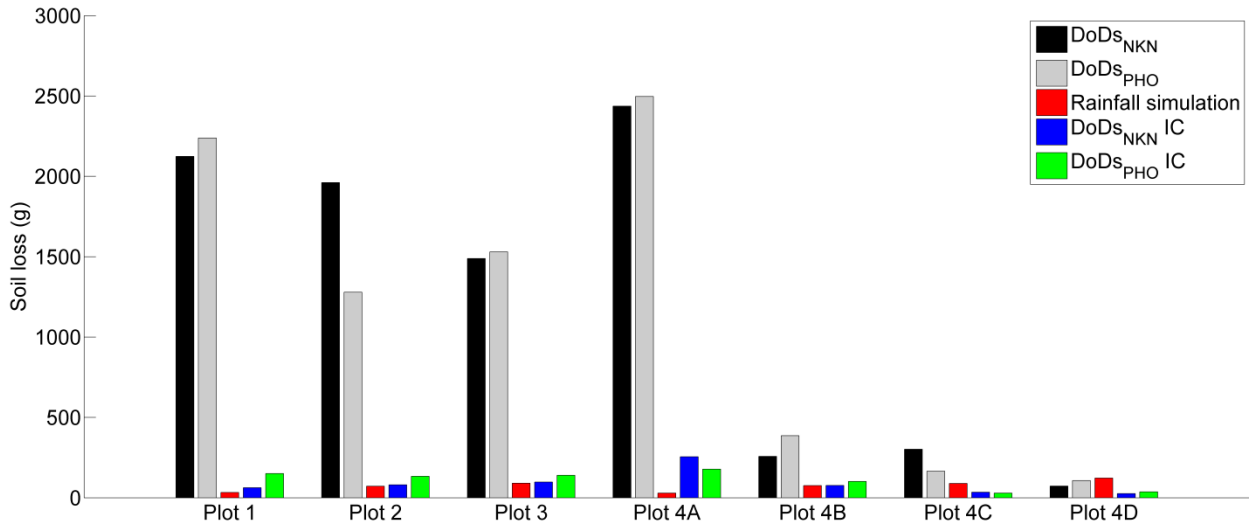


Fig. 10. Soil loss data, expressed in grams, derived for each plot from both the methodologies applied: rainfall simulation and surface elevation change-based relying on DoDs. DoDs_{NKN} and DoDs_{PHO} refer to soil loss estimated from Nikon and smartphone cameras, respectively. DoDs_{NKN} IC and DoDs_{PHO} IC refer to soil loss estimated from Nikon and smartphone cameras, respectively, by considering the connectivity index computed according to the DEMsPre.

Differently from what emerged from Fig. 8, Fig. 10 illustrates that the soil loss data, estimated with the two methodologies, are of the same order of magnitude, as long as the sediment connectivity within the plot is taken into consideration. These results confirm the importance of micro-topography in the sediment connectivity and, consequently, in the estimation of eroded materials.

3.6 Conclusions

In this work, we quantified the soil losses caused by water and compared them with each other, depending on two different methodologies applied: rainfall simulation and surface elevation change-based, relying on high-resolution DEMs derived from SfM. The experiments were carried out in a typical Mediterranean vineyard, under tillage conditions, at very fine scales. SfM data were derived from one standalone digital reflex camera and a smartphone built-in camera. We also applied an index of connectivity (IC) to evaluate the potential effect of sediment connectivity within the plots. Compared to the DEMs_{NKN}, we evaluated the DEMs_{PHO} in terms of (i) accuracy, and (ii) capability to estimate soil loss with regard to the results derived from the rainfall simulation methodology. In terms of accuracy, the DEMs_{PHO} revealed to be comparable with the DEMs_{NKN}, by assuming a normal distribution of errors and with more robust parameters too. Also regarding the estimation of soil losses, caused by the rainstorms artificially reproduced, through the surface elevation change-based methodology, the results between the two different types of cameras used were comparable with each other. What they differed from was the soil losses data estimated with the rainfall simulation. However, this discrepancy was overcome when the sediment connectivity within the plot was taken into consideration by computing the IC index. In conclusion, high-resolution

topography derived from SfM revealed to be essential in the sediment connectivity analysis and, therefore, this, proved to play a key role in the estimation of eroded materials, if compared them to those derived from another methodology such as the rainfall simulation. SfM confirmed to be a useful approach to quantify topographic changes in agricultural lands, also at very fine scales, and revealed to be capable of detecting the more random changes, less easily traceable, induced by the rainstorms. In addition, the fact that smartphones built-in cameras can produce as much satisfying results as those derived from standalone digital reflex cameras is undoubtedly a high value added. Nowadays, smartphones are commonly available for anyone, from farmers to researchers, and will become increasingly important for fast and cheap post-event analyses, as long as they are provided with a high-resolution camera. The increasing development of computer vision technologies and digital camera sensors makes the process of taking good pictures quite easy. A farmer would require few hours of training to learn how to take good pictures of a specific case study, i.e. a rill process, located in its own land. Afterwards, he would be completely independent during the whole field survey, and then he could send the pictures taken to a researcher for further analyses. In this way, the farmer could easily keep monitoring some of the erosion processes that occur in his land and the researcher could provide him quantitative information about net erosion and deposition rates. However, it also should be said that the spatial scale plays a fundamental role in the feasibility of using smartphones for post-event analyses. For erosion processes that occur at field or catchment scales, the use of aerial photogrammetry, supported by the increasing diffusion of UAVs, is more recommended.

3.7 References

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4 Bank erosion in agricultural drainage networks: new challenges from structure-from-motion photogrammetry for post-event analysis³

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4.1 Abstract

Drainage channels are an integral part of agricultural landscapes, and their impact on catchment hydrology is strongly recognized. In cultivated and urbanized floodplains, channels have always played a key role in flood protection, land reclamation, and irrigation. Bank erosion is a critical issue in channels. Neglecting this process, especially during flood events, can result in underestimation of the risk in flood-prone areas. The main aim of this work is to consider a low-cost methodology for the analysis of bank erosion in agricultural drainage networks, and in particular for the estimation of the volumes of eroded and deposited material. A case study located in the Veneto floodplain was selected. The research is based on high-resolution topographic data obtained by an emerging low-cost photogrammetric method (structure-from-motion or SfM), and results are compared to terrestrial laser scanning (TLS) data. For the SfM analysis, extensive photosets were obtained using two standalone reflex digital cameras and an iPhone5® built-in camera. Three digital elevation models (DEMs) were extracted at the resolution of 0.1 m using SfM and were compared with the ones derived by TLS. Using the different DEMs, the eroded areas were then identified using a feature extraction technique based on the topographic parameter Roughness Index (RI). DEMs derived from SfM were effective for both detecting erosion areas and estimating quantitatively the deposition and erosion volumes. Our results underlined how smartphones with high-resolution built-in cameras can be competitive instruments for obtaining suitable data for topography analysis and Earth surface monitoring. This methodology could be potentially very useful for farmers and/or technicians for post-event field surveys to support flood risk management.

4.2 Introduction

Currently, agricultural ditches and channels are an integral part of cultivated landscapes, especially in urbanized European floodplains such as those located in northern Italy, the Netherlands, and the UK, and they have a clear influence on the hydrology, land use planning, and water resources management of cultivated catchments (Gallart et al., 1994; Longfield and Macklin, 1999; Robinson et al., 2000; Moussa et al., 2002; Pfister et al., 2004; Duke et al., 2006; Dagès et al., 2009; Levavasseur et al., 2012). During the past half century, numerous countries in Europe, such as England (Nicholson, 1943; Trafford, 1970; Robinson, 1990; O'Connell et al., 2007; Wheeler and Evans, 2009; Macklin et al., 2010), France (e.g. Gallart et al., 1994), Germany (e.g. Krause et al., 2007), and Italy (e.g. Camorani et al., 2005; Sofia et al., 2014c), as well as the United States (e.g. Changnon and Demissie, 1996), witnessed changes in the spatial configuration of the drainage network due to urbanization, demographic pressure (Fumagalli, 1976; Millar and Hatcher, 1978), and changes in both agricultural techniques and technological innovation (Magnusson, 2001; van Dam, 2001). Among the causes of network changes, urbanization has been most studied (e.g. Leopold, 1968; Hollis, 1975; Pizzuto et al., 2000), but it is known that agriculture itself has transformed the hydrological processes and sediment dynamics of fluvial systems and irrigation/reclamation channels (Wischmeier and Smith, 1960; Pimentel et al., 1995; Trimble, 1999; Knox, 2001; Montgomery, 2007a, 2007b). Various authors have pointed out how changes in the spatial configuration of the drainage network may have (i) increased the risk of flooding (Bronstert et al., 2001; Wheeler, 2006; Evrard et al., 2007; Palmer and Smith, 2013), (ii) changed flood lag times (i.e. Robinson et al., 1985; Robinson, 1990; Wright et al., 2012; Smith et al., 2013), and (iii) changed the discharge regime in agricultural lands due to dikes and embankment constructions (Ebel and Engel, 1994; Pfister et al., 2004). Among the other natural disturbances that affect fluvial systems and agricultural channels, bank erosion deserves particular attention. Bank erosion is a natural geomorphic process and it cannot be eliminated. Therefore, the occurrence of erosion is not the main concern, but the location and rate at which erosion occurs are critical. Bank erosion may represent a severe threat to the function played by agricultural channels (Thorne, 1982, 1999; Simon et al., 1991; Rosgen, 1996; Simon and Darby, 1999; Simon et al., 2000; De Rose and Basher, 2011; Sass and Keane, 2012). Bank erosion, other than representing a remarkable source of sediments (Bull, 1997; Simon and Darby, 1999; Sekely et al., 2002; Evans et al., 2006; Fox et al., 2007; Grove et al., 2013), may even cause bank failure through a combination of hydraulic and gravitation forces (Carson and Kirkby, 1972; Thorne, 1982; Fischenich, 1989; Simon et al., 1991), and this, especially during intense rainfall events, may rapidly cause hundreds of square kilometres of land to be flooded. The last major flood affecting northern Italy, in 2010, brought numerous clear examples of these effects (Structure of the Extraordinary Commission for Recovering from the Flooding, 2011). Figure 1 shows a bank failure caused by flooding of the Alpone stream (province of Verona, Italy) (Figure 1a) and its effects on the adjacent agricultural lands (Figure 1b) during the major flood of November 2010.

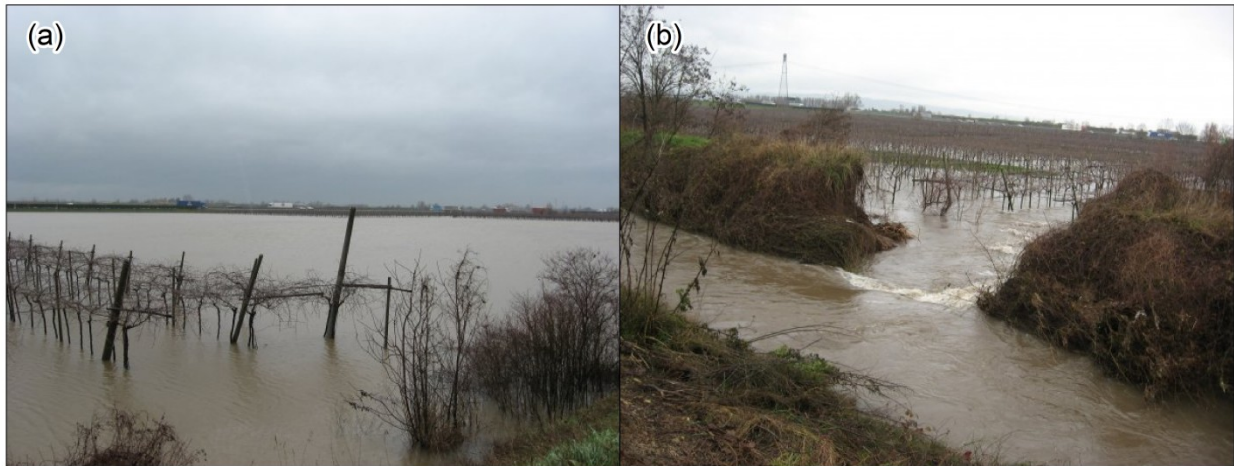


Figure 1. Effects of bank failures in flood management. Bank failure of the Alpone stream (province of Verona) (a) and consequent damage on the adjacent agricultural plot (b) during the major flood of November 2010 (photograph courtesy of Eng. Silvia Tizian).

Land loss due to bank erosion may also have a strong impact on floodplain inhabitants, agricultural lands, bridge crossings, and other hydraulic constructions (Nasermoaddeli and Pasche, 2008). Gravitational forces on in situ bank material combined with hydraulic forces on the bank bottom determine rates of bank erosion. Failure usually occurs when erosion at the bank toe and the channel bed adjacent to the bank have increased the height and angle of the bank to such a point that gravitational forces exceed the shear strength of the bank material (Simon et al., 1991). A stable bank can be transformed into an unstable bank because of prolonged rainfall through: (i) increase in soil unit weight, (ii) loss of matric suction and, therefore, apparent cohesion, (iii) generation of positive pre-water pressures and, therefore, reduction of frictional strength, and (iv) loss of confining pressure during recession of stormflow hydrographs (Simon et al., 2000). Clearly, the analysis of bank erosion in agricultural ditches and channels is important in hazard assessment and evaluation of restoration options (Grove et al., 2013). Numerous approaches have been developed to study bank erosion at different spatial and temporal scales (Lawler, 1993), but advances in remote sensing and digital terrain analysis allow for a more sophisticated study of bank erosion processes nowadays (Fonstad and Marcus, 2010). The literature has flourished with works using high-resolution topography for the identification and study of artificial features in floodplains (i.e. Casas et al., 2012; Passalacqua et al., 2012; Grove et al., 2013; Sofia et al., 2014a). Papers have demonstrated the value of high-resolution topography for both natural and engineered landscapes (see Tarolli, 2014, for a full review). Aside from airborne laser scanning (ALS), among the high-resolution survey techniques, terrestrial laser scanning (TLS) has demonstrated to be capable of generating very high-resolution digital terrain models (DEMs) (i.e. Heritage and Hetherington, 2007; Schaefer and Inkpen, 2010). However, these instruments remain expensive and unwieldy and are thus not suitable for every environment. Much interest has risen, therefore, in obtaining high-resolution DEMs using more flexible and affordable technologies. Recently, an emerging time-of-flight technology, called range

imaging, has been tested for geomorphic applications in complex environments (Nitsche et al., 2013). Such methods are capable of obtaining similar degrees of topographic detail showed by the DEM derived from TLS. Structure-from-motion (SfM) has proven to be a new frontier for topographic data acquisition (Westoby et al., 2012; Carrivick et al., 2013; Fonstad et al., 2013; Hugenholtz et al., 2013). SfM has already been used for quantifying coastal erosion rates (James and Robson, 2012) and submerged fluvial topography (Woodget et al., 2014), for mapping vegetation spectral dynamics (Dandois and Ellis, 2013), and for measuring rates of glacier variation (Whitehead et al., 2013). Micheletti et al. (2014) investigated the geomorphological potential of freely available and accessible SfM photogrammetric software with smartphones focusing on a river bank and an Alpine alluvial fan. In addition, SfM flourished in different sectors (agricultural, geomorphological, environmental, and engineering) thanks to the recent development and diffusion of unmanned aerial vehicles (UAVs) (e.g. Colomina and Molina, 2014; Chen et al., 2015). Considering this review, this work intends to test SfM as a user-friendly, low-cost, and flexible technique to derive high-resolution topography for the analysis of bank erosion in agricultural drainage networks in Veneto. Among the urbanized floodplains, the Veneto (northeast of Italy) is often taken as representative (Ranzato, 2011; Fabian, 2012), being characterized by different levels of urban sprawl and settlements (Fregolent, 2005) and by a dense network of channels and ditches (Cazorzi et al., 2013). The landscape of the northeast of Italy has also undergone significant changes due to a prolonged period of control and governing of water (Viganò et al., 2009; Fabian, 2012), witnessing abrupt changes in the drainage network system (Rusconi, 1991; Regione Veneto, 2007). For the work, the photogrammetric survey was conducted using two standalone reflex cameras, and one smartphone built-in digital camera. In addition, we planned a simultaneous TLS survey for comparison of results. The main goal of this work is to show the effectiveness of a user-friendly, low-cost, and flexible technique that may even rely on smartphones, for the post-event analysis in support of flood risk management in agricultural floodplains.

4.3 Study Area

The study area is a small section of an agricultural drainage channel called Scolo Orsaro, about 30 m long, located within the Bacchiglione Land Reclamation Consortium (BLRC) in the Veneto region (Italy) (Figure 2).



Figure 2. Localization of the study area along the agricultural channel Scolo Orsaro, within the Bacchiglione Land Reclamation Consortium (BLRC) in the Veneto region, Italy.

Overall, the BLRC reclamation territory includes about 916 km of hydrographic network, which plays a key role in both irrigation and drainage. About 25% of the land of the whole territory lies below mean sea level, and therefore most of the drainage network is controlled by pumping stations (BLRC, 2015). The study area was involved in the main flood event of 2010, which affected most of the southern part of the province of Padua (about 130 municipalities), affected about 500 000 people, and cost more than 400 million euros (Structure of the Extraordinary Commission for Recovering from the Flooding, 2011). The drainage channel considered in the analysis was chosen because it was affected by bank failure (Figure 3) in February 2014, after an intense two-week rainy period in which 210 mm of precipitation fell in the whole province of Padua.

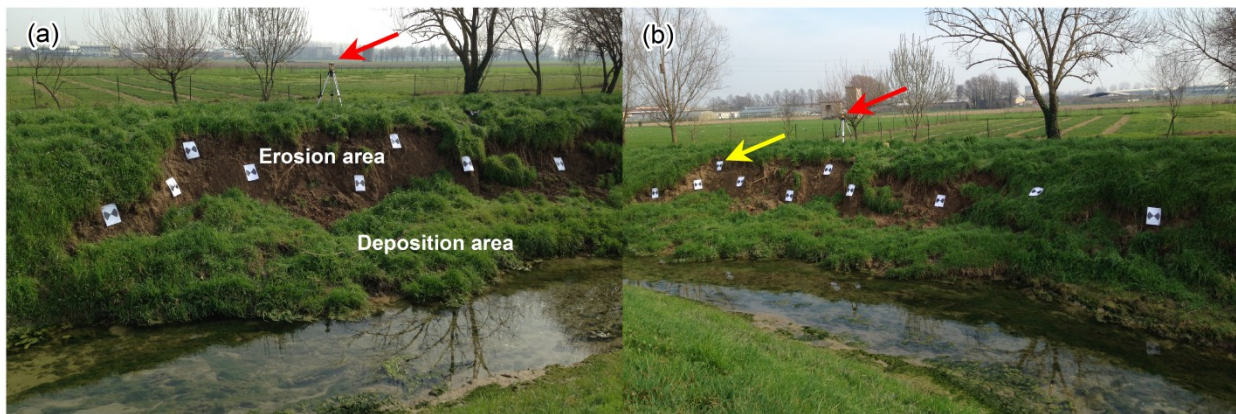


Figure 3. Two visual perspectives of the channel bank affected by erosion (a, b). Eroded areas, characterized by bare soil, are easily distinguishable from the deposited ones, which are covered by grass. In both pictures a tripod supporting a global positioning system (GPS) receiver (base station), used for the terrestrial laser scanning (TLS) survey (red arrows), and the black and white polythene rectangular targets (yellow arrow) required to georeference the structure-from-motion (SfM) point clouds are visible.

The bank failure was caused by a combination of hydraulic and geotechnical factors. The soils that can be found in this study area are characterized by a moderately high runoff potential (C according to the USDA Hydrological Group definition) and by a groundwater level between 0 and 2 m below the surface (Piani Territoriali di Coordinamento Provinciale, 2009). Using a Topcon HiPer Pro® rover receiver running in real time kinematic (RTK) mode, areas that had been eroded and deposited by the bank failure were mapped precisely in the field, collecting a total of 222 points. The differential receiver guarantees a horizontal precision of 10 mm + 1.0 ppm and a vertical precision of 15 mm + 1.0 ppm when used in RTK mode.

4.4 Material and Methods

4.4.1 Considered datasets

4.4.1.1 Reference dataset: terrestrial laser scanning (TLS)

In February 2014, using a Riegl® LMS-Z620 ‘time-of-fly’ TLS system, we performed a detailed survey of the section of the channel bank affected by erosion. The Riegl® LMS-Z620 terrestrial laser scanner operates in the near-infrared wavelength region and offers a maximum measurement range of up to 2 km, with an accuracy of 10 mm and a speed of acquisition of up to 11000 pts/s. For each measured point, the system records the range, horizontal and vertical alignment angles, and the backscattered signal amplitude. The TLS survey was carried out from three scan positions to give full coverage of the area of interest. From these scan positions, 983 277 elevation points were collected with a point spacing of 0.02 m. Each scan position was georeferenced with its own reflective target (TLS-target), whose location was precisely measured through two Topcon HiPer Pro® dual-frequency and dual-constellation

(GPS+GLONASS) receivers used in differential mode. Afterwards, the three TLS point clouds were co-registered within the RiSCAN PRO® software (RiSCAN, 2009), using the Iterative Closest Point (ICP, Chen and Medioni, 1991; Besl and McKay, 1992) algorithm. Vegetation filtering was not necessary because the area mainly features bare soil and a thick coverage of grass. The georeferenced elevation points were then interpolated by the natural neighbours method (Sibson, 1981) to generate a 0.1 m resolution DEM (DEM_{TLS}) (Figure 4).

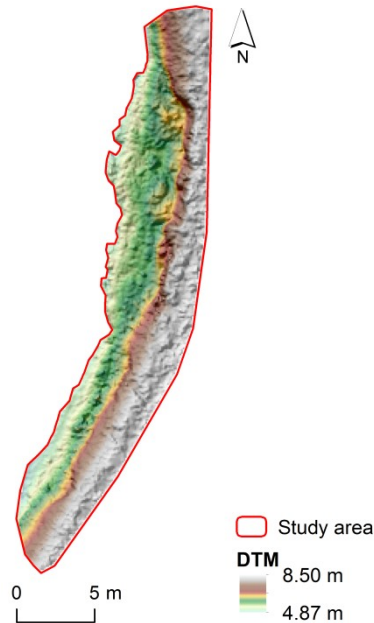


Figure 4. Digital elevation model (DEM) derived from TLS (0.1 m resolution).

The points collected for the bank erosion (see the previous chapter Study Area) were used as ground control points (GCPs) to assess the accuracy and precision of the DEM_{TLS} through the computation of the root-mean-square-error (RMSE), mean error, and standard deviation of error (SDE). The SDE of the DEM_{TLS} was estimated to be within ± 0.1 m. For the purposes of this work, we based all our analyses on the final DEMs (as in Wechsler, 2007; Calligaro et al., 2013; Bangen et al., 2014; Javernick et al., 2014; Ouédraogo et al., 2014; Tarolli et al., 2015), and we focused on only the error related to the DEM itself, rather than that in the point cloud. Considering the DEM quality and the fact that high-resolution topography derived by LiDAR (light detection and ranging) technology has been successfully applied for geomorphometric analyses in different contexts (i.e. Jones et al., 2007; Notebaert et al., 2009; Passalacqua et al., 2014; Sofia et al., 2014a, 2014b; Tarolli, 2014; Tarolli et al., 2015), the DEM_{TLS} was considered as a benchmark to compare DEMs obtained using the SfM methodology (as in Micheletti et al., 2014).

4.4.1.2 Low-cost, high-resolution datasets: structure-from-motion (SfM)

Simultaneously with the TLS survey, photographs of the channel bank were taken using two standalone digital reflex cameras at different sensor resolutions (a Canon EOS 5D Mark III at 22.3 MP resolution, set at a focal length of 35 mm, and a Nikon D3000 at

10.2 MP resolution, set at a focal length of 35 mm), and an iPhone5® built-in camera (8 MP resolution), with both automatic focusing and exposure enabled. Prior to taking photographs, 12 targets (SfM-targets) were distributed throughout the channel bank to georeference the photograph-derived topography. Each SfM-target was made of a black and white polythene rectangle of approximately 29.5 cm × 42 cm. SfM-target centroids were surveyed using the terrestrial laser scanner. For homogeneity, all the cameras were handheld carefully, and 60 photographs were taken using each camera. The pictures were taken from different positions in a row along two different altitude levels. The pictures were taken standing first at the foot (first altitude level) and then at the top (second altitude level) of the bank facing the eroded areas. From each position, a picture was taken with all three cameras, keeping the camera angle unvaried. The SfM technique was then used to obtain three-dimensional (3D) georeferenced point clouds and to generate three 0.1 m resolution DEMs (referred to as DEMs_{sfm}). For this purpose, the images acquired were processed using the commercial software Agisoft PhotoScan®, as already successfully considered in different analyses (Doneus et al., 2011; Verhoeven et al., 2012; Javernick et al., 2014; Woodget et al., 2014). The workflow consisted in the following steps, repeated for each dataset (Canon EOS, Nikon, and iPhone5®): image import, image alignment, geometry building, georeferencing, optimization of image alignment, rebuilding of scene geometry, and export of the 3D georeferenced point cloud. Once the photographs have been uploaded to the system, Agisoft PhotoScan®, using a custom algorithm similar to the Lowe's (2004) Scale Invariant Feature Transform (SIFT) object recognition system, examines them to find matching points across the images. As a further step, the software estimates the camera's intrinsic (focal length, principal point, and lens distortion) and extrinsic (projection centre location and the six exterior orientation parameters that define the image) orientation parameters by initially using a greedy algorithm to calculate camera positions and later improving their positions with a bundle-adjustment algorithm (Robertson and Cipolla, 2009; Verhoeven et al., 2012; Javernick et al., 2014). Unlike traditional photogrammetry, in which either the position of the camera or the positions of some points are known prior to scene reconstruction (Verhoeven et al., 2012; Westoby et al., 2012; Fonstad et al., 2013), in the SfM, matches are made between points across many photographs without prior knowledge of the camera position (Lowe, 2004). By using the previously determined intrinsic and extrinsic camera-orientation parameters and a dense multi-view stereo reconstruction (DMVR), Agisoft PhotoScan® allows the generation of a dense surface (mesh or model geometry – Agisoft, 2015). The XYZ positions of the 12 SfM-target centroids were imported into Agisoft PhotoScan®; the coordinates of the target centroids were then extracted from the TLS point cloud and used for georeferencing purposes (Micheletti et al., 2014). At this point, a linear similarity transformation using seven parameters (three translation, three rotation, and one scaling) is automatically performed by Agisoft PhotoScan® to register the model to real-world coordinates. The georeferencing process provides a linear, affine transformation of the model but cannot remove non-linear model misalignments (Woodget et al., 2014). Therefore, an optimization transformation method is applied to minimize geometric distortions within the 3D model (Agisoft, 2015).

As a result, re-projection and reference coordinate misalignment errors are reduced in the final output model geometry (Woodget et al., 2014; Javernick et al., 2014; Agisoft, 2015). Thereafter the model geometry is rebuilt and the 3D georeferenced point cloud is exported. Although Agisoft PhotoScan® does not require great photogrammetry experience to produce 3D georeferenced point clouds, various parameter settings are performed throughout the SfM stages (Agisoft, 2015). To achieve the best possible performance, after various tests we decided to apply the same parameters as were proposed by Javernick et al. (2014). The three georeferenced point clouds will be referenced in the manuscript as GEO_{EOS} (Canon camera), GEO_{NKN} (Nikon camera), and GEO_{IPH} (iPhone5®). The SfM final point clouds were sampled at 0.02 m point spacing to make them computationally manageable and comparable with the TLS point cloud. After trimming, they were further manipulated using the open source program CloudCompare® (<http://www.danielgm.net/cc/>) to remove additional noise that typically affects these data (Javernick et al., 2014). In this case, given the small study area and the trimming procedure already applied, the noise removal was accomplished manually. Finally, the elevation points were interpolated by the natural neighbours method (Sibson, 1981) to generate 0.1 m resolution DEMs. The three DEMs_{sfm} georeferenced according to the TLS survey will be referred to in the text as DEM_{EOS} , DEM_{NKN} , and DEM_{IPH} (Figure 5).

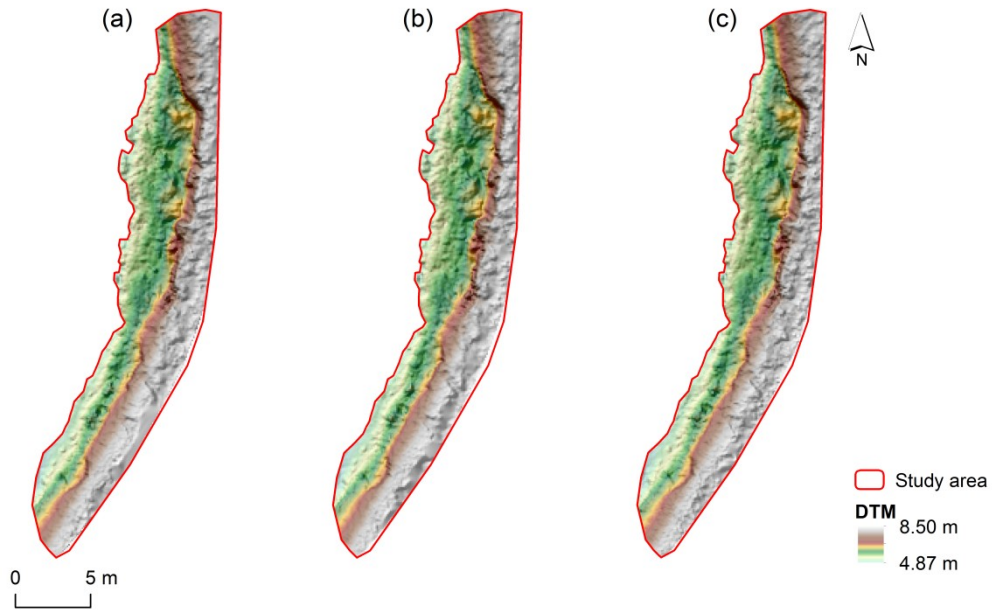


Figure 5. DEMs derived from SfM methodology (0.1 m resolution): (a) DEM_{EOS} , (b) DEM_{NKN} , and (c) DEM_{IPH} .

The three DEMs_{sfm} were then directly compared to the DEM_{TLS} , considered as the reference model, by assuming a normal distribution and using robust statistical methods (Höhle and Höhle, 2009). This entailed the computation of the mean error, SDE, RSME, median, normalized median absolute deviation (NMAD), and the quantiles (68.3 and 95%) of the distribution of absolute elevation differences.

4.4.2 Roughness index (RI)

Areas characterized by severe erosion (landslides and/or slumps) tend to have a rougher terrain than those not affected by erosion at all (Tarolli, 2014); topographic indexes of surface roughness can therefore automatically map rough morphologies (e.g. landslides – McKean and Roering, 2004; Glenn et al., 2006, channel bed morphology – Cavalli et al., 2008, or alluvial fan morphology – Frankel and Dolan, 2007). Although several methods have been proposed to measure the surface roughness by LiDAR data (i.e. McKean and Roering, 2004; Haneberg et al., 2005; Glenn et al., 2006; Frankel and Dolan, 2007; Turner et al., 2014), for the purposes of this work, we considered the topographic parameter Roughness Index (RI) (Cavalli et al., 2008) (Equation (1)). The reasons for this choice relied on the facts that (i) the RI can be computed easily because it only requires a DEM and (ii) it takes into consideration the average width of the feature located in the study area. The RI is computed as:

$$\sigma = \sqrt{\frac{\sum_{i=1}^n (x_i - x_m)^2}{n}} \quad (1)$$

where σ is the RI or the standard deviation of residual topography, n is the number of processed cells within the moving window, x_i is the elevation value of one specific cell within the moving window, and x_m is the mean of the n -cell values (Cavalli et al., 2008). The higher the index, the rougher the surface and the more likely the occurrence of erosion. The moving window size must be selected based on the average width of the feature within the study area. Pirotti and Tarolli (2010) and Tarolli et al. (2012) suggested that the best ratio between information and noise and thus the best results for geomorphic features recognition are found when the window size is about two to three times the maximum size of the investigated feature. In our case study, most of the clods were about 0.4 m wide. Therefore, also considering computational constraints that require the moving window to be an integer odd number, we considered a 1.1 m moving window (11 x 11 cells). Given the RI, the residual topography can be considered as a DoD (DEM of difference). Therefore, the DEMs' uncertainty on the RI has been evaluated through the methodology developed by Wheaton et al. (2010). Their novelty was to develop a new technique for estimating the magnitude of DEM uncertainty in a spatially variable way using fuzzy set theory. The Fuzzy Inference System (FIS) (Klir and Yuan, 1995) requires the definition of some rules upon which the input rasters used as a proxy for the uncertainty [i.e. point density, slope, global positioning system (GPS) point quality, etc.] (Wheaton, 2008; Bater and Coops, 2009; Milan et al., 2011; Wheaton et al., 2010) are analysed. Following this methodology, the elevation uncertainty (δz) is spatially variable and is evaluated on a cell-by-cell basis. Readers are referred to recent papers by Wheaton (2008) and Wheaton et al. (2010) for further details on the FIS. Here, a two-input rule FIS system based on slope and point density is used. This type of FIS system has been chosen because it proved to be applicable to any topographic survey (Wheaton, 2008). Matlab's Fuzzy Logic Toolbox, developed by Jang and Gulley (2009), was used to implement this FIS. The output variable, elevation uncertainty (δz),

was finally organized into four categories: low, average, high, and extreme. Class ranges in the output variable were operationally established as follows:

- Low: The upper limit of this class was chosen by considering the best accuracy that DEMs derived from SfM can achieve (approximately ± 0.02 m) (Woodget et al., 2014).
- Average: The upper limit of this class was set by considering the resolution of DEMs (0.1 m).
- High: The upper limit of this class was set by considering an acceptable value of accuracy for many LiDAR analyses in the field of geomorphology (Frankel and Dolan, 2007; Glenn et al., 2006; McKean and Roering, 2004) (0.3 m).
- Extreme: The upper limit was set by considering any value greater than 0.3 m that was in line with the literature (Wheaton, 2008; Wheaton et al., 2010) (0.4 m).

In addition to this, recent literature underlined the effectiveness of statistical analysis of variability of the topographic attributes (i.e. landform curvature) derived from high-resolution topography for the analysis of geomorphic (Lin et al., 2013; Sofia et al., 2013) or anthropogenic (Cazorzi et al., 2013; Sofia et al., 2014a) processes. In this work, we considered m -times the standard deviation of RI (σ_{RI}) as an objective method for the recognition of erosion areas. To assess how well the automatic extraction represents the ground truth, we converted the erosion area, mapped in the field through a differential global positioning system (DGPS), into a raster. This map was compared with each extraction carried out considering multiples of the σ_{RI} as a threshold. The extracted pixels that match the actual erosion area are considered as true positives (TP), the non-matching extracted pixels are defined as false positives (FP), and the pixels within the eroded area that have not been extracted are considered as false negatives (FN). Finally, the goodness of each extraction was evaluated through a quality index (Heipke et al., 1997) (Equation (2)), which considers the percentage of the reference data explained by the extracted areas as well as the percentage of correctly extracted features.

$$QUALITY = \frac{TP}{(TP + FP + FN)} \quad (2)$$

The index varies from zero, for extraction with no overlap between extracted and observed features, and one, for extraction where these coincide perfectly. The same quality index was adopted in Tarolli et al. (2012) and Lin et al. (2013).

4.4.3 Estimation of erosion and deposition volumes

In the field, the boundaries of erosion and deposition areas were surveyed using DGPS. Then, we estimated the volumes of eroded and deposited materials (in m^3). Generally, the estimation of erosion and deposition volumes relies on the so-called morphological method (Ashmore and Church, 1998), an affordable approach for monitoring geomorphic changes and estimating sediment budgets. The simplest way to apply the morphological method consists in carrying out repeated topographic surveys from which

DEMs can be obtained and differenced to produce DoDs. Therefore, an available DEM that showed the original bank not affected by erosion would have been of great assistance. In the absence of such a DEM, and given that banks in agricultural channels and ditches in the Veneto floodplain are standard man-made structures characterized by a regular and well known homogeneous geometry (Cazorzi et al., 2013), we reconstructed a DEM that followed the geometry of the undisturbed channel (original bank surface prior to erosion issues). With the terrestrial laser scanner, we also surveyed a portion of channel bank not affected by erosion and used it as reference data for the reconstruction of the original bank. We refer to this as the pre-event-DEM. As a further step, we computed the volumes of eroded and deposited materials considering the pre-event-DEM and the post-event DEM_{TLS} and DEMs_{Sfm}. We did this by using the Geomorphic Change Detection 6 (GCD 6) toolbar embedded in an ESRI® add-in for ArcGIS 10.X that is freely downloadable from <http://gcd.joewheaton.org/downloads>. Erosion and deposition volumes have been estimated only for the areas mapped in the field. DEMs' uncertainty in DoDs has also been evaluated for estimating sediment budgets (Lane et al., 1994; Milne and Sear, 1997; Lane, 1998; Brasington et al., 2000; Lane et al., 2003; Wheaton, 2008; Wheaton et al., 2010). In this case, DEMs' uncertainties were evaluated both (i) by assuming that they are spatially uniform (Brasington et al., 2000) and (ii) by estimating them in a spatially variable way using fuzzy set theory (Wheaton et al., 2010), as already done for RI. Following the first way of evaluating DEMs' uncertainties in DoDs, a spatially uniform minimum level of detection threshold ($\delta_{\min LoD}$) is used to distinguish actual surface changes from noise (Brasington et al., 2000; Fuller et al., 2003). Elevation changes that are beneath this detection limit are typically discarded, while those that are above it are treated as real. In this regard, Lane et al. (2003) and Brasington et al. (2003) show that the individual errors in the DEMs could be propagated into the DoD as:

$$\delta u_{DoD} = \sqrt{(\delta z_{new})^2 + (\delta z_{old})^2} \quad (3)$$

where δu_{DoD} is the propagated error in the DoD, while δz_{new} and δz_{old} are the individual errors in the DEM_{new} and DEM_{old}, respectively. In this regard, different values of $\delta_{\min LoD}$ were set according to Equation (3). The values used to threshold the DoDs are shown later. Regarding the second way of evaluating DEMs' uncertainties in the DoDs, readers are referred to the previous section about the RI, because the same two-input rule FIS system and class ranges in the output variable (elevation uncertainty) were used.

4.5 Results

4.5.1 TLS and SfM comparisons

Regarding the comparison between TLS and SfM, first, Table I reports the georeferentiation errors (RMSE) calculated by the Agisoft PhotoScan® software along the X, Y, and Z-axes for each SfM point cloud (Table I).

Table I. Georeferentiation errors (RMSE) calculated by PhotoScan® along X, Y, and Z axes for each three-dimensional point cloud obtained by the SfM methodology (GEO_{EOS}, GEO_{NKN}, and GEO_{IPH}). The point clouds were georeferenced by using target coordinates recorded by the terrestrial laser scanner.

	X Error (± m)	Y Error (± m)	Z Error (± m)
GEO _{EOS}	0.0150	0.0081	0.0050
GEO _{NKN}	0.0119	0.0081	0.0049
GEO _{IPH}	0.0141	0.0112	0.0048

All of the SfM point clouds show an error of the order of about 0.01 m along the X and Y axes, and an even lower order error along the Z-axis. These good results can be explained by the fact that: (i) the area of interest was relatively small, (ii) the 12 SfM-targets were well distributed over the study area, and (iii) the distance between the position of the cameras and the study area was short (about 7 m). Differences between the DEMs_{sfm} and DEM_{TLS} for the unthresholded DEMs (where no uncertainty analysis was carried out) were evaluated with the accuracy measures assuming a normal distribution, for both the whole study area and the erosion and deposition mapped areas (Table II).

Table II. Accuracy measures of DEMs_{sfm} (DEM_{EOS}, DEM_{NKN}, and DEM_{IPH}) and checked by DTM_{TLS} with the assumption of normal distribution, for both (i) the whole study area and (ii) the erosion and deposition mapped areas. Outliers have been classified using the boxplot approach (Tukey, 1977; Sofia et al., 2014a).

	DEM _{EOS} - DEM _{TLS}	DEM _{NKN} - DEM _{TLS}	DEM _{IPH} - DEM _{TLS}
<i>(i) Whole study area</i>			
Mean (m)	0.037	0.034	0.019
SDE (m)	0.054	0.045	0.040
RSME (m)	0.078	0.068	0.057
Minimum (m)	-0.524	-0.313	-0.429
Maximum (m)	0.501	0.435	0.492
Number of cells DoD	18711	18711	18711
Number of outliers	1073	920	1038
Percentage of outliers (%)	5.73	4.92	5.55
Mean (after removal of outliers) (m)	0.032	0.030	0.017

Table II. (continued)

	$DEM_{EOS} - DEM_{TLS}$	$DEM_{NKN} - DEM_{TLS}$	$DEM_{IPH} - DEM_{TLS}$
SDE (after removal of outliers) (m)	0.038	0.033	0.028
<i>(ii) Erosion and deposition mapped areas</i>			
Mean (m)	0.035	0.038	0.023
SDE (m)	0.052	0.045	0.042
RSME (m)	0.062	0.059	0.048
Minimum (m)	-0.524	-0.313	-0.374
Maximum (m)	0.501	0.435	0.492
Number of cells DoD	10116	10116	10116
Number of outliers	632	594	643
Percentage of Outliers (%)	6.25	5.87	6.36
Mean (after removal of outliers) (m)	0.031	0.035	0.021
SDE (after removal of outliers) (m)	0.034	0.031	0.027

The accuracy of DEM_{sfm} , checked by DEM_{TLS} (benchmark) is displayed instead in Figure 6.

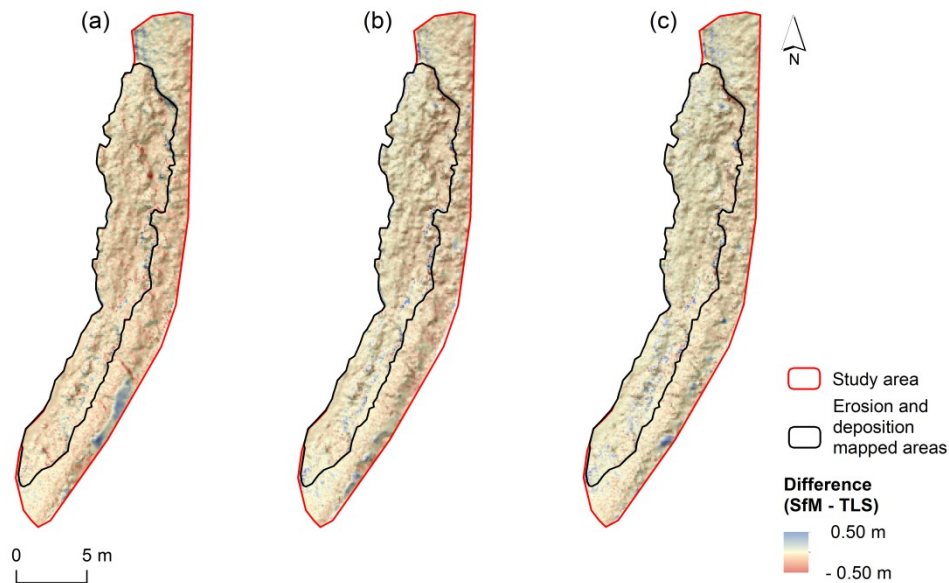


Figure 6. Distribution of elevation differences for each DoD: (a) $DEM_{EOS} - DEM_{TLS}$, (b) $DEM_{NKN} - DEM_{TLS}$, and (c) $DEM_{IPH} - DEM_{TLS}$.

Considering both the whole study area and the erosion and deposition mapped areas, all the DEMs_{sfm} are comparable to the reference DEM_{TLS} model. In particular, the DEM_{IPH} is the one that shows the lowest SDE and RSME values for both the case studies (0.040 and 0.057 m for the whole study area and 0.042 and 0.048 m for the erosion and deposition mapped areas, respectively). The DEM_{NKN} and DEM_{EOS} show slightly greater values instead. After the removal of the outliers (points outside the boxplot graphs, as in Tukey, 1977), mean and SDE values are reduced. Again, the DEM_{IPH} remains the one with the lowest SDE value (0.028 and 0.027 m for the whole study area and the erosion and deposition mapped areas, respectively). Histograms of each DoD, for both the case studies, are shown in Figure 7.

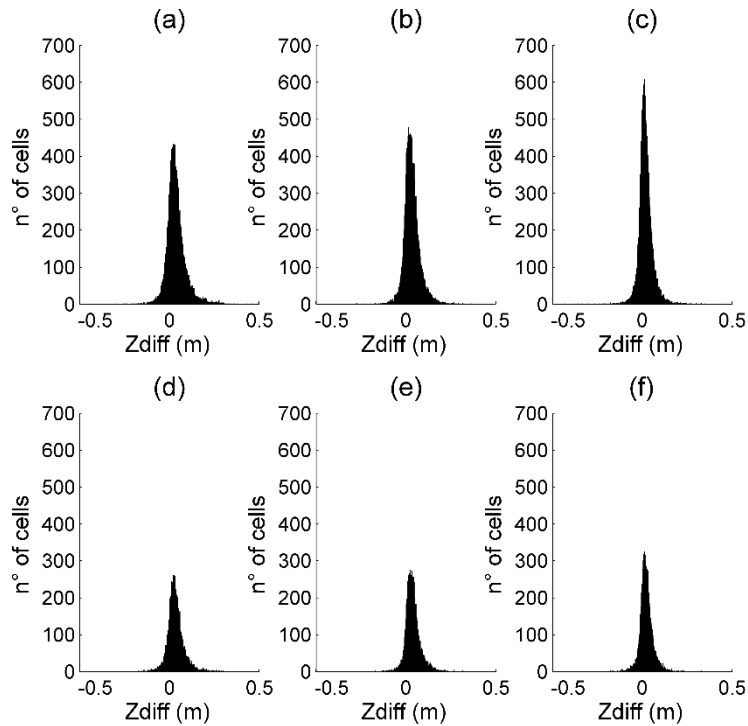


Figure 7. Histograms of DoDs: DEM_{EOS} – DEM_{TLS} [(a) and (d), for the whole study area and for the erosion and deposition mapped areas, respectively], DEM_{NKN} – DEM_{TLS} [(b) and (e), for the whole study area and for the erosion and deposition mapped areas, respectively], and DEM_{IPH} – DEM_{TLS} [(c) and (f), for the whole study area and for the erosion and deposition mapped areas, respectively].

As highlighted by Höhle and Höhle (2009), histograms give a first impression of the normality of the error distribution. From Figure 7 and Table II, it emerges that other approaches should be taken in order to obtain reliable accuracy measures for the DoDs. In fact, the histograms reported in Figure 7 show skewness and distributions that are more peaked around their means (kurtosis), and these factors are confirmed by the values reported in Table III. Among the DoDs, the one with DEM_{IPH} shows the highest values of kurtosis but, at the same time, the lowest values of skewness. The non-normality of errors in the DEMs was also shown in Sofia et al. (2013).

Table III. Skewness and kurtosis computed for each DoD between DEMs from TLS and those from SfM (DEM_{EOS}, DEM_{NKN}, and DEM_{IPH}) for both (i) the whole study area and (ii) the erosion and deposition mapped areas.

	DEM _{EOS} - DEM _{TLS}	DEM _{NKN} - DEM _{TLS}	DEM _{IPH} - DEM _{TLS}
<i>(i) Whole study area</i>			
Skewness	1.083	0.804	0.518
Kurtosis	9.191	8.762	14.213
<i>(ii) Erosion and deposition mapped areas</i>			
Skewness	0.961	0.835	0.523
Kurtosis	12.429	10.695	15.931

Table IV reports the accuracy measures of DEMs_{sfm} using more robust parameters (Höhle and Höhle, 2009).

Table IV. Accuracy measures of DEMs_{sfm} (DEM_{EOS}, DEM_{NKN}, and DEM_{IPH}) as checked by DEM_{TLS} through robust statistical methods, for both (i) the whole study area and (ii) the erosion and deposition mapped areas.

	DEM _{EOS} - DEM _{TLS}	DEM _{NKN} - DEM _{TLS}	DEM _{IPH} - DEM _{TLS}
<i>(i) Whole study area</i>			
Median (m)	0.029	0.028	0.015
NMAD (m)	0.038	0.032	0.027
Q _{δz} (0.683) (m)	0.052	0.048	0.034
Q _{δz} (0.95) (m)	0.134	0.113	0.088
<i>(ii) Erosion and deposition mapped areas</i>			
Median (m)	0.029	0.033	0.019
NMAD (m)	0.035	0.031	0.028
Q _{δz} (0.683) (m)	0.050	0.051	0.037
Q _{δz} (0.95) (m)	0.124	0.119	0.093

The DEM_{IPH} is the one that shows the lowest values of the median and NMAD for both the whole study area (0.015 and 0.027 m, respectively) and the erosion and deposition mapped areas (0.019 and 0.028 m, respectively). This is in line with the results displayed in Table II. The DEM_{NKN} and DEM_{EOS} follow, with slightly higher values. The

quantiles of the distribution of absolute elevation differences for DEM_{IPH} are 0.034 m (68.3%) and 0.088 m (95%) for the whole study area and 0.037 m (68.3%) and 0.093 m (95%) for the erosion and deposition mapped areas. These values, the lowest among the DEM_{Sfm} , are not influenced by outliers or non-normality of the error distribution (Höhle and Höhle, 2009). The good results provided by DEM_{IPH} , in terms of accuracy, might suggest findings similar to those found when dealing with the scale of topographic parameters (i.e. Pirotti and Tarolli, 2010; Tarolli et al., 2012; Sofia et al., 2013; Sofia et al., 2014a). Given a sufficiently high resolution dataset, the smallest scale of analysis is not always the most appropriate for feature extraction: the rougher the maps, the higher the noise bias and the lower the capability to provide a reliable measurement of morphological indices. In the case of photogrammetry, this could mean that, as long as the sensor has a sufficient pixel resolution, using an average megapixel value (8 MP on the iPhone versus 22.3 MP on the Canon EOS) results in smoothing of the surface and of the errors. Furthermore, as also pointed out by Westoby et al. (2012), a wide variety of imaging sensors can be used for SfM, from video stills through to low-grade compact digital cameras. The primary requirement is well-exposed photographs of the features of interest, rather than the highest resolution possible.

4.5.2 Roughness index (RI) analysis

Figure 8 shows the RI maps computed for each DEM using the 11-cells moving window and considering DEMs' uncertainties. The colour follows the m-times of σ_{RI} . Looking at these maps, the results are encouraging, showing how high values of the RI are actually related to the erosion section of the bank.

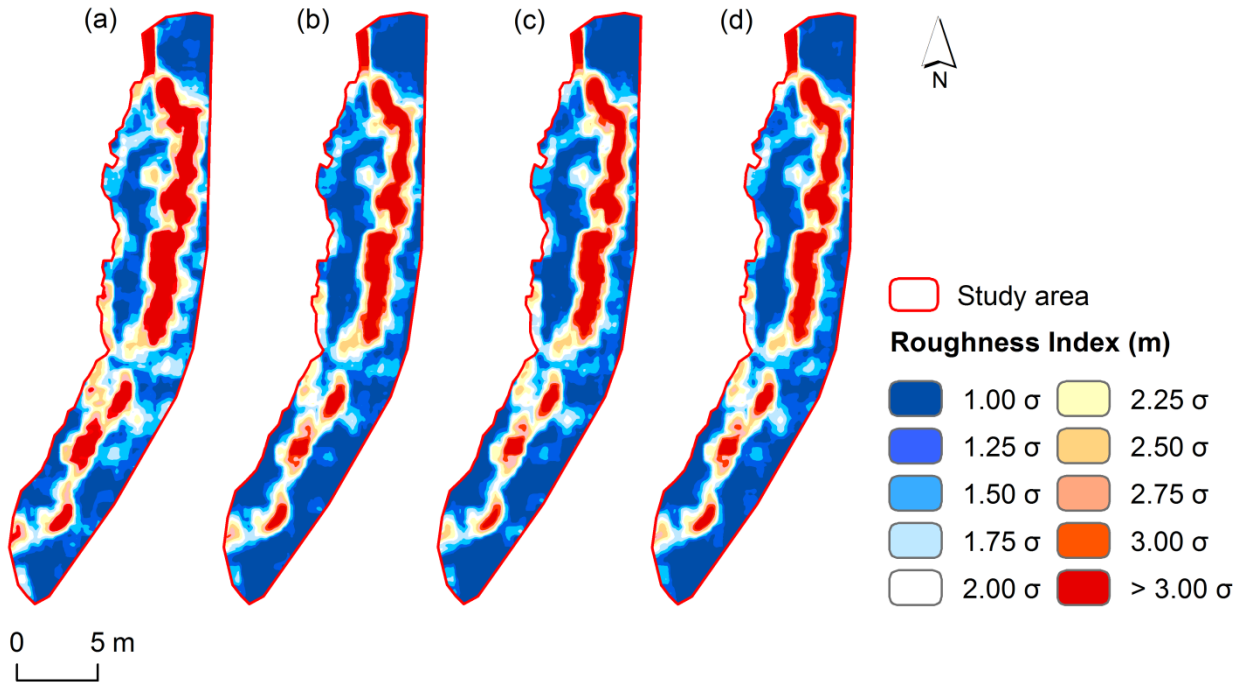


Figure 8. Roughness index (RI) maps computed for each DEM: (a) DEM_{TLS} (considered as a reference), (b) DEM_{EOS} , (c) DEM_{NKN} , and (d) DEM_{IPH} . The colour follows the considered m-times of σ_{RI} used for erosion extraction.

The erosion area extracted using a progressive increase of m-times (1.00, 1.25, 1.50, 1.75, 2.00, 2.25, 2.50, 2.75, 3.00) the σ_{RI} are shown in Figure 9.

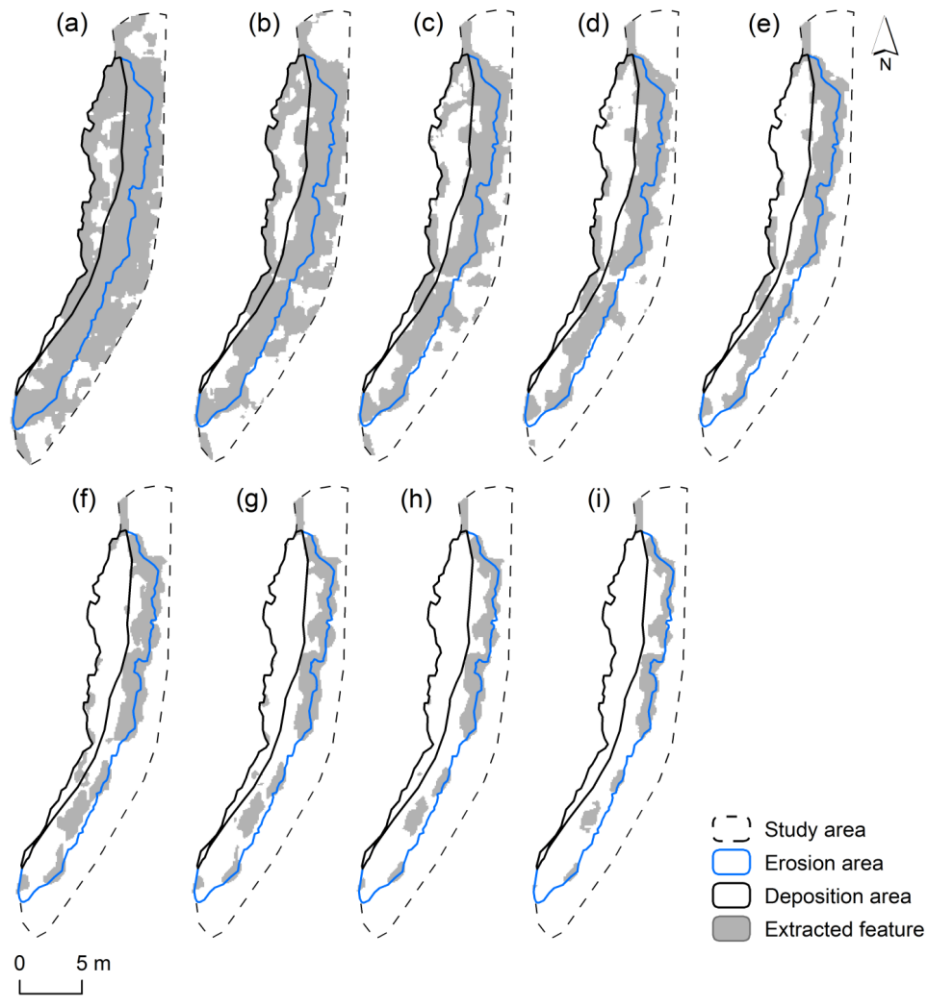


Figure 9. Geomorphic features extracted using m-times the defined threshold (σ_{RI}) for DEM_{TLS} : 1.00 (a), 1.25 (b), 1.50 (c), 1.75 (d), 2.00 (e), 2.25 (f), 2.50 (g), 2.75 (h), and 3.00 (i).

Clearly, a progressive increase of the m value is related to a better identification of the eroded area. However, at the same time, large values of m cause a progressive loss of information on the extracted features. Quality indexes (Equation (2)) are computed for each multiple of σ_{RI} and for each DEM available (Figure 10).

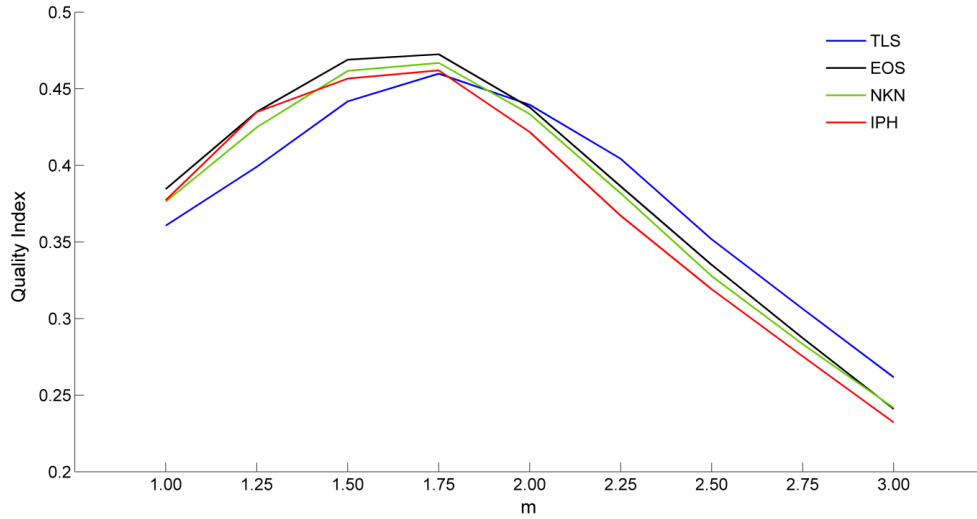


Figure 10. Plot of quality index values computed according to m-times the defined threshold (σ_{RI}) obtained for each DEM available.

From Figure 10, it emerges that: (i) the highest values of the quality index are reached at $1.75 \sigma_{RI}$ for each DEM, (ii) DEMs_{sfm} extraction tends to present similar quality indexes, and (iii) DEMs_{sfm} extraction using the reflex camera presents a slightly higher (0.473 and 0.467) quality index than the one achieved with the DEM_{TLS} (0.460). Among the DEMs_{sfm}, DEM_{EOS} is the one whose quality index is the highest (0.473). Figure 11 shows the best extraction of erosion areas (highest quality index) for each available DEM.

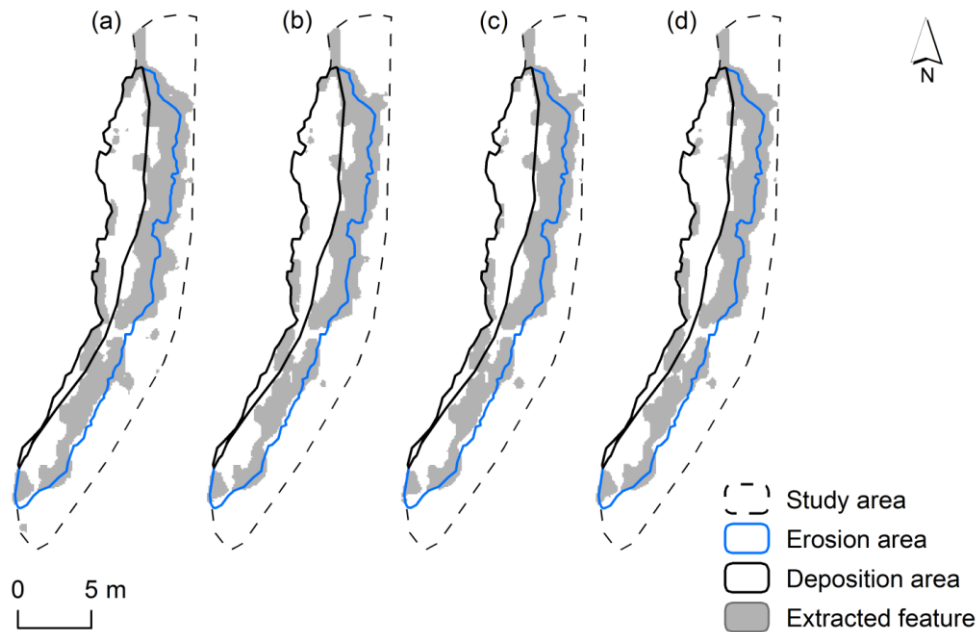


Figure 11. Erosion area extracted for: (a) DEM_{TLS} with $1.75 \sigma_{RI}$, (b) DEM_{EOS} with $1.75 \sigma_{RI}$, (c) DEM_{NKN} with $1.75 \sigma_{RI}$, and (d) DEM_{IPH} with $1.75 \sigma_{RI}$.

Figure 11 shows that this procedure also automatically extracts pixels that are out of the mapped erosion area. However, the results obtained are in line with the recent literature (Tarolli et al., 2012; Lin et al., 2013; Sofia et al., 2013; Sofia et al., 2014a), and prove the effectiveness of RI for identifying eroded areas. The next step consists in computing the erosion volume by considering these automatically extracted eroded areas based on the best σ_{RI} performance.

4.5.3 Erosion and deposition

Table V reports the volumes of eroded and deposited materials computed for each DoD, between the post-event DEMs (DEM_{TLS} and DEMs_{sfm}) and the pre-event-DEM. The values reported in Table V refer to DoDs not subject to uncertainty analyses.

Table V. Volumes of eroded and deposited materials computed for each DoD between the post-event DEM_{TLS} and DEMs_{sfm} (DEM_{EOS}, DEM_{NKN}, and DEM_{IPH}) and the pre-event-DEM, respectively. The column Δ erosion reports the difference between the erosion volumes of each DEM_{sfm} and the DEM_{TLS} (reference). The same differences are computed for deposition volumes (Δ deposition). Volumes of eroded and deposited materials are computed using the GCD 6 toolbar.

	Erosion (m ³)	Δ Erosion ($\pm$ m ³)	Deposition (m ³)	Δ Deposition ($\pm$ m ³)
DEM _{TLS} – pre-event-DEM	36.77	-	18.60	-
DEM _{EOS} – pre-event-DEM	34.74	-2.03	20.09	1.49
DEM _{NKN} – pre-event-DEM	34.71	-2.06	20.42	1.82
DEM _{IPH} – pre-event-DEM	35.59	-1.18	19.71	1.11

From Table V, it emerges that all the DEMs_{sfm} make it possible to compute deposition and erosion volumes that are comparable with those considered as a reference (TLS). The DEMs_{sfm} tend to slightly overestimate the deposition volumes (1.49, 1.82, and 1.11 m³ for DEM_{EOS}, DEM_{NKN}, and DEM_{IPH}, respectively) and, at the same time, underestimate the erosion volumes (–2.03, –2.06, and –1.18 m³ for DEM_{EOS}, DEM_{NKN}, and DEM_{IPH}, respectively). Table VI shows the different $\min LoD$ values that were set according to Equation (3) in order to propagate uncertainty analysis in a spatially uniform way.

Table VI. Individual DEMs' errors are reported under δz , while the spatially uniform propagated errors in the DoDs, which represent the $\min\text{LoD}$, are reported under δu_{DoD} . As the pre-event-DEM was built based on a portion of DEM_{TLS} not affected by erosion, we assumed that its standard deviation error (SDE) was the same as that of DEM_{TLS} (0.1 m). Regarding the SDE of DEMs_{sfm} , as we considered the DEM_{TLS} as a benchmark, we obtained them by adding 0.1 m to the SDE of each DEM_{sfm} reported in table II.

	δz ($\pm$ m)	δu_{DoD} ($\pm$ m)
pre-event-DEM	0.10	
DEM_{TLS}	0.10	
DEM_{EOS}	0.15	
DEM_{NKN}	0.15	
DEM_{IPH}	0.14	
$\text{DEM}_{\text{TLS}} - \text{pre-event-DEM}$		0.14
$\text{DEM}_{\text{EOS}} - \text{pre-event-DEM}$		0.18
$\text{DEM}_{\text{NKN}} - \text{pre-event-DEM}$		0.18
$\text{DEM}_{\text{IPH}} - \text{pre-event-DEM}$		0.17

Figure 12 shows the three DoDs derived from TLS for the erosion and deposition mapped areas. Figure 12a refers to the unthresholded DoD (no uncertainty analysis carried out), Figure 12b refers to thresholded DoD according to a spatially uniform $\min\text{LoD}$ (± 0.14 m, see Table VI), and finally Figure 12c refers to the thresholded DoD according to the two-input rule FIS system.

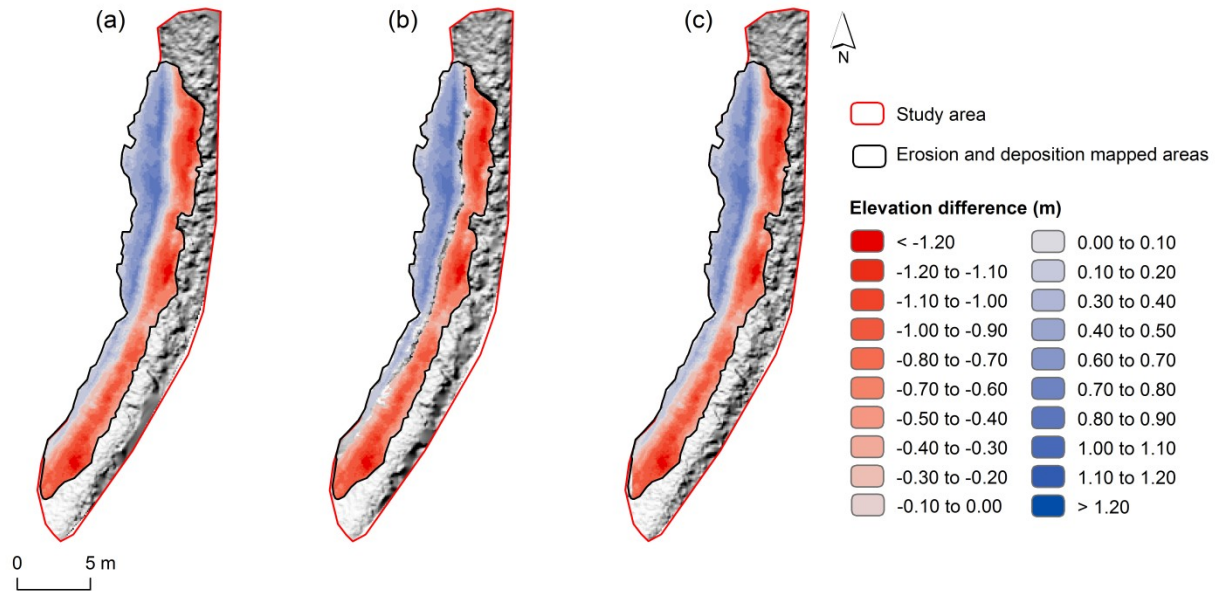


Figure 12. DEMs of difference (DoDs) derived from TLS: unthresholded DoD (no uncertainty analysis carried out) (a), thresholded DoD according to a spatially uniform minLoD (b), and thresholded DoD according to the two-input rule FIS system (c).

From Figure 12, as expected, it emerges that the thresholded DoDs entail a loss of information compared to the unthresholded DoD. However, this occurs at the expense of a better geomorphic plausibility (Wheaton et al., 2010). Volumes of eroded and deposited materials computed for each DoD considering (i) unthresholded DoDs (no uncertainty analysis), (ii) thresholded DoDs according to spatially uniform minLoD , and (iii) thresholded DoDs according to the two-input rule FIS system, are reported in Table VII.

Table VII. Volumes of eroded and deposited materials computed for each DoD considering (i) unthresholded DoDs (no uncertainty analysis), (ii) thresholded DoDs according to spatially uniform minLoD , and (iii) thresholded DoDs according to the two-input rule FIS system.

	Volume (m^3)		
	Erosion	Deposition	Net
<i>No uncertainty analysis (unthresholded)</i>			
DEM _{TLS} – pre-event-DEM	36.77	18.60	-18.17
DEM _{EOS} – pre-event-DEM	34.74	20.09	-14.66
DEM _{NKN} – pre-event-DEM	34.71	20.42	-14.29
DEM _{IPH} – pre-event-DEM	35.59	19.71	-15.88
<i>Spatial uniform minLoD</i>			
DEM _{TLS} – pre-event-DEM	36.53 (± 7.66)	18.16 (± 5.24)	-18.36 (± 9.28)
DEM _{EOS} – pre-event-DEM	34.26 (± 9.44)	19.40 (± 6.68)	-14.86 (± 11.60)
DEM _{NKN} – pre-event-DEM	34.25 (± 9.44)	19.83 (± 6.82)	-14.43 (± 11.64)
DEM _{IPH} – pre-event-DEM	35.20 (± 9.05)	19.09 (± 6.34)	-16.10 (± 11.05)
<i>Two-input rule FIS system</i>			
DEM _{TLS} – pre-event-DEM	20.91 (± 5.49)	12.96 (± 2.02)	- 7.95 (± 5.84)
DEM _{EOS} – pre-event-DEM	19.71 (± 5.41)	14.73 (± 2.24)	- 4.98 (± 5.85)
DEM _{NKN} – pre-event-DEM	20.08 (± 5.49)	14.78 (± 2.22)	- 5.30 (± 5.93)
DEM _{IPH} – pre-event-DEM	20.53 (± 5.88)	14.53 (± 2.25)	- 6.00 (± 6.02)

Table VII proves in quantitative terms what emerged visually from Figure 12: that uncertainties involve a loss of information, in terms of erosion and deposition volumes, when compared to the unthresholded DoD. This is valid for all the DEMs considered. As highlighted by Wheaton et al. (2010), the standard spatially uniform minLoD application generally produces a lower information loss compared to the FIS system. However, this loss of information, although low, is obtained by an overly simplistic model of uncertainty. In our work, this is confirmed for both the erosion and deposition volumes. For example, considering DEM_{IPH}, the erosion (35.20 m^3) and deposition (19.09 m^3) obtained from the standard spatially uniform minLoD application are greater than those obtained from the FIS system (20.53 and 14.53 m^3 , respectively). All the DoDs entail a net negative volume and the DEM_{IPH} is the one that achieves the values closest to those of the reference model (DEM_{TLS}), considering both the uncertainty analyses. Generally, for each DoD, the volume errors ($\pm \text{m}^3$) obtained from the FIS system are lower than the ones obtained from the standard spatially uniform minLoD application. In

this regard, the DEM_{TLS} is the one that achieves the lowest erosion ($\pm 7.66 \text{ m}^3$) and deposition ($\pm 5.24 \text{ m}^3$) volume errors for the spatially uniform $_{minLoD}$ application. Regarding the FIS system application, DEM_{TLS} is still the one that shows the lowest deposition volume error ($\pm 2.02 \text{ m}^3$) and an erosion volume error ($\pm 5.49 \text{ m}^3$) slightly greater than the one achieved by DEM_{EOS} ($\pm 5.41 \text{ m}^3$). However, generally, by considering the FIS system, and thus a more robust estimation of DEM quality and uncertainty (Wheaton et al., 2010), both the erosion and deposition volumes and their relative errors obtained for each DoD are similar to each other. These results prove the fact that SfM methodology can be successfully applied for the analysis of bank erosion and also for the estimation of the volumes of deposited and eroded material. Furthermore, the DEM_{IPH} is also confirmed to be a valid alternative to DEM_{TLS} for quantitative analyses. To test the effectiveness of the feature extraction methodology in quantitative terms, we computed the erosion volume by considering the eroded area that was automatically extracted according to the best σ_{RI} performance (Figure 11). We refer to it as the automatically extracted erosion volume (AE_{xtr}). As feature extraction also detected some pixels that were out of the mapped erosion area (Figure 11), we split the erosion volume AE_{xtr} into two parts: an interior (considering the pixels extracted within the mapped erosion area) and an exterior (considering the pixels outside the mapped erosion area) component (IAE_{xtr} and EAE_{xtr} , respectively). For this analysis, although we took into consideration both the uncertainty analyses, we consider it appropriate to report only the results thresholded according to the FIS system, as this is a more robust estimation of DEM quality and uncertainty (Wheaton et al., 2010) (Table VIII).

Table VIII. Comparison between the erosion volume AE_{xtr} according to the best σ_{RI} performance (Figure 11) and the erosion volume computed within the mapped area, considered as a reference. The erosion volume AE_{xtr} is split into two parts: an interior (IAE_{xtr}) and an exterior (EAE_{xtr}) component. The volumes considered are thresholded according to the FIS system.

	Erosion volume (reference) (m^3)	Erosion volume AE_{xtr} (m^3)	Erosion volume IAE_{xtr} (m^3)	Erosion volume EAE_{xtr} (m^3)
DEM_{TLS} – pre-event-DEM	20.91 (± 5.49)	15.37 (± 3.94)	15.06 (± 3.76)	0.31 (± 0.18)
DEM_{EOS} – pre-event-DEM	19.71 (± 5.41)	13.67 (± 3.62)	13.46 (± 3.50)	0.21 (± 0.12)
DEM_{NKN} – pre-event-DEM	20.08 (± 5.49)	13.68 (± 3.63)	13.46 (± 3.51)	0.22 (± 0.12)
DEM_{IPH} – pre-event-DEM	20.53 (± 5.88)	13.99 (± 3.68)	13.72 (± 3.52)	0.27 (± 0.16)

From Table VIII, it emerges that erosion volumes AE_{xtr} are lower than the erosion volumes computed within the mapped area, which were considered as a reference (reduction of about 5 to 6 m^3 considering each DoD). Although this methodology implies a non-negligible underestimation (about 30%), the fact that it is completely automatic must be stressed. On considering the erosion volumes IAE_{xtr} and EAE_{xtr} , the former

(about 13 to 15 m³) are much greater than the latter (about 0.2 to 0.3 m³). Rather, we can assert that the erosion volumes EAE_{extr} are almost insignificant. The values obtained from the DEM_{sfm} are comparable to the ones achieved by DEM_{TLS} and, among the DEM_{sfm} , DEM_{IPH} is still the one that achieves the closest values to those of the reference model (DEM_{TLS}), considering the IAE_{extr} . However, generally, although the feature extraction methodology underestimates erosion volumes, the results are in line with what was expected from a completely automatic procedure and, therefore, also prove its effectiveness in quantitative terms. Furthermore, the DEM_{sfm} are confirmed to be a valid alternative to the DEM_{TLS} .

4.6 Conclusions

This work proposed an analysis of bank erosion in agricultural drainage networks. High-resolution DEMs derived from the SfM photogrammetry technique are considered and compared to a DEM derived from TLS as a benchmark. SfM data are derived from two standalone digital reflex cameras and a built-in iPhone5® camera. We then applied the topographic parameter RI for the identification of eroded areas. After that, we quantitatively estimated the volumes of deposited and eroded material. Considering the DEM derived from the TLS as the reference model, we compared the DEMs derived from SfM in terms of: (i) accuracy, (ii) capability to depict erosion areas, and (iii) estimation of eroded and deposited volumes. The results highlight the fact that the SfM technique is a useful approach for the production of topographic data for the qualitative and quantitative analyses of geomorphic processes, with results that are comparable with those obtained by LiDAR technology. The high-resolution DEM_{sfm} revealed to be effective for the detection of erosion areas and the estimation of volumes of soil deposited. All of the DEMs considered in the analysis tend to present similar performances in term of geomorphic feature analyses. However, the most interesting results are those related to the data derived from a smartphone. The DEM derived from an iPhone, provided that the elevation points cloud had undergone a correct georeferentiation process, was revealed to be the most similar to the DEM derived from TLS in terms of both accuracy (Tables II and IV) and estimation of eroded and deposited soil volumes (Tables VII and VIII). This may suggest findings similar to those found when dealing with the scale of topographic parameters (i.e. Sofia et al., 2013). Given a high-resolution dataset, the rougher the surface (the smallest scale), the higher the error effect. In the case of photogrammetry, as in Westoby et al. (2012), this could imply that, as long as the sensor has a sufficient pixel resolution, using an average megapixel value (8 MP on the iPhone versus 22.3 MP on the Canon EOS) results in smoothing of the surface and of the errors. Regarding the feature extraction methodology based on the best σ_{RI} performance, this proved to be effective in both identifying eroded areas (Figure 11) and quantitatively estimating erosion volumes (Table VIII). Therefore, the SfM technique is also comparable to LiDAR technology in terms of automatic feature extraction. According to our results, the SfM technique offers a faster, simpler, and cheaper alternative to the LiDAR technology, especially if photographs are taken by a smartphone. Nowadays, smartphones are commonly available for any land-user or manager, from farmers to researchers to technicians.

Anyone who has a good smartphone can take photographs and, from them, could easily obtain a high-resolution DEM for the analysis of Earth surface processes. We are just at the beginning of geomorphological applications of SfM and there will be several opportunities and challenges in using the SfM technique with smartphones. For example, the 3G iPhone was the first consumer device to provide an integration of three positioning technologies: Assisted GPS (AGPS), WiFi positioning, and cellular network positioning (Zandberg, 2009). Furthermore, the accurate generation of 3D models of real-world environments has become a primary target in computer vision. For example, a complete on-device 3D reconstruction pipeline for mobile monocular hand-held devices, which generates dense 3D models with absolute scale on-site, has been proposed by Tanskanen et al. (2013). Therefore, smartphones will become increasingly important and, in the near future, may be more than enough for quick post-event analyses, as long as they are provided with a high-resolution built-in camera, an activated GPS module, and an Internet connection. In the same way, the SfM technique with smartphones can really be a strategic and easily usable tool for post-flood field surveys on agricultural floodplains to increase the existing knowledge of such events and to provide technical solutions for flood risk mitigation.

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5 Vineyards in terraced landscapes: new opportunities from lidar data ⁴

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5.1 Abstract

Vineyard landscapes are a relevant part of the European culture, and several authors concluded that they are the agricultural practice that causes the highest soil loss. Grape quality depends on the availability of water, and soil erosion is an important parameter dictating the vineyard sustainability; therefore, soil and water conservation measures are often implemented. Among them, the construction of terraces is the most widely used system. However, while favouring agricultural activities, terraces if not properly maintained can lead to local instabilities creating hazards for settlements and cultivations, and for the related economy. Terraced fields are also served by agricultural roads that can have deep effects on water flows triggering surface erosion. The goal of this research is to use lidar elevation data for a hydro-geomorphological analysis of terraced vineyards. The work is divided in two parts. At first, the Relative Path Impact Index is tested in two vineyards to identify terrace-induced and road-induced erosions. Statistical thresholds of the Relative Path Impact Index are then defined to label the most critical areas. On the second step, using the index and the defined thresholds, we simulate different scenarios of soil conservation measures, establishing the optimal solution to reduce erosion. The results highlight the effectiveness of high-resolution topography in the analysis of surface erosion in terraced vineyards, when the surface water flow is the main factor triggering the instabilities. The proposed analysis can help in scheduling a suitable planning to mitigate the consequences of the anthropogenic alterations induced by the terraces and agricultural roads.

5.2 Introduction

In Mediterranean and temperate climate regions, vineyards cover mountainous, hilly, coastal and floodplain areas. They represent one of the largest agro-ecosystem (Tonietto & Carbonneau, 2004), and they probably are the most important agricultural activity in terms of environmental impact and income (Raclot et al., 2009). One of the most distinctive components of the Mediterranean (Douglas et al., 1994, 1996; Gallart et al., 1994; Dunjo et al., 2003) and semi-arid (Ore & Bruins, 2012) mountain/hilly vineyards are terraces (Tarolli et al., 2014). Terraced vineyards witness historic viticulture and agricultural techniques, bearing a high historical and cultural value related to the cultivation of steep sites, and to the craftsmanship of dry-stone walling (Petit et al., 2012). Terraced vineyards also possess an inherent aesthetic, social and ecological value, providing diversity of species and biotopes (Höchtel et al., 2007). The purpose of terracing and its effect on hydrological processes depends on geology and soil properties (Grove & Rackham, 2003), but terraces are generally built to retain more soil and water, and to reduce both hydrological connectivity and erosion (Lasanta et al., 2001; Cammeraat, 2004). Terraces reduce the slope gradient and length, facilitating the cultivation on steep slopes, and they increase the infiltration of water in areas with a moderate to low soil permeability (van Wesemael et al., 1998; Yuan et al., 2003), with positive effects on agricultural activities. Despite their value, terraced vineyards are among the most endangered landscapes in Europe, and they are most commonly threatened by abandonment (Tarolli et al., 2014) and subsequent subjection to succession, or else they are transformed to desert vine steppes as part of intensive land consolidation measures (Höchtel et al., 2007). As well, in the past, terraced vineyards were planted also in areas that never had hosted vines, including dormant landslides (Corti et al., 2011), and hilly soils prone to runoff and erosion (Arnaez et al., 2007). As a consequence, they often present erosion and instabilities both on the terraces (Figure 1a and b), and on the nearby structures (Figure 1c).

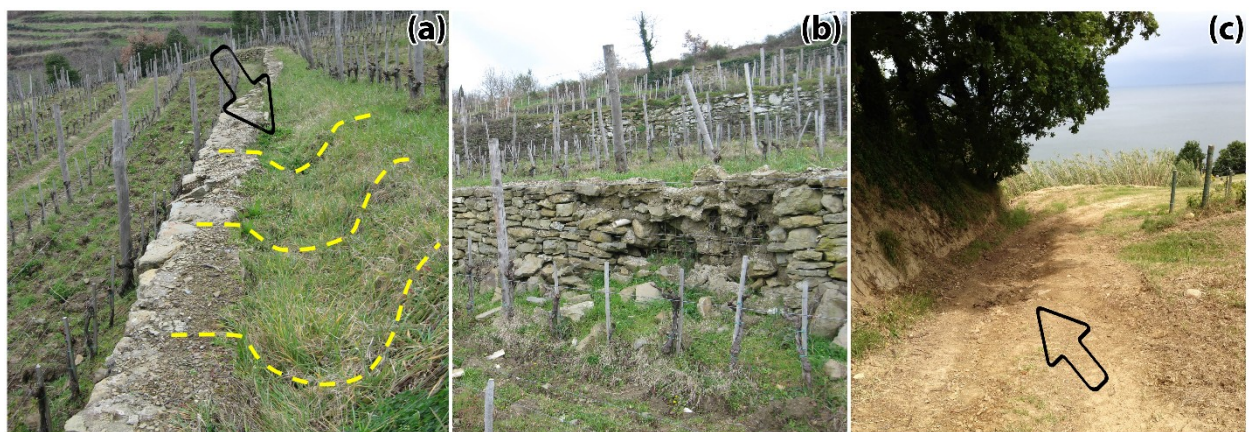


Figure 1. Erosion path covered by grass above a terrace wall (a), and related instability on the terrace (b), and example of erosion area on an agricultural road (c).

Literature underlined that there is a relation between the expansion of vineyards and the increase of hydraulic erosion processes (Tropeano, 1984; Costantini, 1992; Kosmas et al., 1997; Costantini et al., 2001; Pisante et al., 2005; Bazzoffi et al., 2006; Martínez-Casasnovas et al., 2013), with consequences on the loss of nutrients (Novara et al., 2013) and in the redistribution of chemicals (Fernández-Calviño et al., 2013). Several researches highlighted that vineyards, if compared with other crops, constitute for the Mediterranean areas the form of agricultural land use that causes the highest soil erosion (Tropeano, 1983; Martínez-Casasnovas & Sánchez-Bosch, 2000; Cerdan et al., 2002, 2006; Martinez-Casasnovas & Ramos, 2006; Cerdà & Doerr, 2007; García-Ruiz et al., 2010). Similar conclusions have been drawn also for other climatological regions (Lieskovský & Kenderessy, 2014). Terraced vineyards are also served by agricultural roads, and the construction of these anthropogenic features can have deep effects on water flows and instabilities (Tarolli et al., 2013). Roads present in fact a limited soil development (Jimenez et al., 2013), but with high rates of erosion on the roads themselves and on their embankments (Cerdà, 2007), requiring huge investments to reduce the erosion rates (Lee et al., 2013). The plane surface of roads, in fact, can intercept the overland and the subsurface flow (Reid & Dunne, 1984; Luce & Cundy, 1994; Luce & Black, 1999; Borga et al., 2004; Penna et al., 2014) and can modify the natural flow directions expanding the drainage network (Gucinski et al., 2001; Tarolli et al., 2013). The same issues could be induced also by the terrace benches, resulting in local instabilities and/or erosion. As well, the lack of maintenance or an incorrect planning can determine an increase of erosion that can cause the terraces to collapse (Gallart et al., 1994; Lasanta et al., 2001; Crosta et al., 2003; Canuti et al., 2004). The strict connection between vineyard and terraces management and erosion (Cerdà, 1997; Tarolli et al., 2014) brought the attention to the need of high-quality topographic information. Researchers and land managers, in fact, have called for the development of cost-effective and flexible methods and tools to gather detailed, updated, accessible and specific knowledge of terraced vineyards' characteristic features and of their present state (Tarolli et al., 2014).

5.2.1 Research and Background Motivation

Following this line of research, the aim of this study is to present an application of high-resolution lidar-derived topography for a first and rapid analysis of vineyards terraced landscape. The availability of high-resolution data sets has offered new tools for the mapping of erosion features (Desprats et al., 2013) and the understanding of erosion risk (Leh et al., 2013), and topography from laser scanning technologies provides new and accurate methodologies for land management and planning (Cavalli and Tarolli, 2011; Pirotti et al., 2012; Tarolli, 2014). This study is motivated by the fact that, recently, in many Italian regions, vineyards' terraced landscapes have acquired a special status influencing their management and planning (i.e. the area of Cinque Terre in Liguria, recognized by UNESCO as part of the World Heritage), and practice-oriented guidelines and methods are needed for their correct management. The proposed work focuses that the erosion in terraced vineyards is mainly connected to the water redistribution (Cammeraat et al., 2010). Erosion is, in fact, facilitated by the changes in the spatial

distribution of saturated areas and on the runoff concentration on pathways, and by the insufficient or incorrect drainage of retaining walls (Crosta et al., 2003). Terrace benches are the main sources for generation of runoff contributing to the increase of erosion (Llorens et al., 1992 and Lesschen et al., 2008), and to the increase in earth and water pressures behind the inner face of the retaining wall, main cause of terrace walls collapsing (Tarolli et al., 2014). The most important element to consider to prevent this type of erosion is to analyse properly and regulate overland flows; hence, we proposed to analyse the redistribution of the upslope area because of the presence of terrace benches and agricultural roads. The work considers a methodology previously and effectively applied to different environmental contexts and further explores the use of high-resolution topography and morphological indexes in the analysis and management of terraced vineyards.

5.3 Material and methods

5.3.1 Study Areas

In this work, we consider two terraced vineyards (Figure 2) in Central Italy, where an increasing of surface instabilities was witnessed during the last few years. The two case studies are related to the Chianti Classico Gallo Nero vines (for the Lamole case study, Fattoria Lamole, Tuscany, Figure 2a, c and d) and to the Pinot vines (for the Pesaro case study, Fattoria Mancini, Marche, Figure 2b, e and f). Dry-stone wall terraces are the main feature of the Lamole study. These terraces have a height of about 2 m, while the terrace benches are about 8 to 10 m wide, and they are slightly outward sloped for drainage. Starting in 2003, in Lamole, the restoring of the terraces and the planting of new vines follow an avant-garde project that aims at reaching an optimal level of mechanization as well as maintaining untouched the typical landscape elements. The Pesaro study area is a small privately owned terraced field. Here, terraces are built with compacted sand, and the terrace risers are generally covered with grass. The terraces generally reach at most 2 m in height, and they present benches about 5 m wide and with a reversed slope. For both study areas, several field surveys were carried out during spring–summer 2013, and terraced (T1 to T5 in Figure 2) and road (R1 to R3 in Figure 2) instabilities were mapped on the field with differential global positioning system (DGPS).

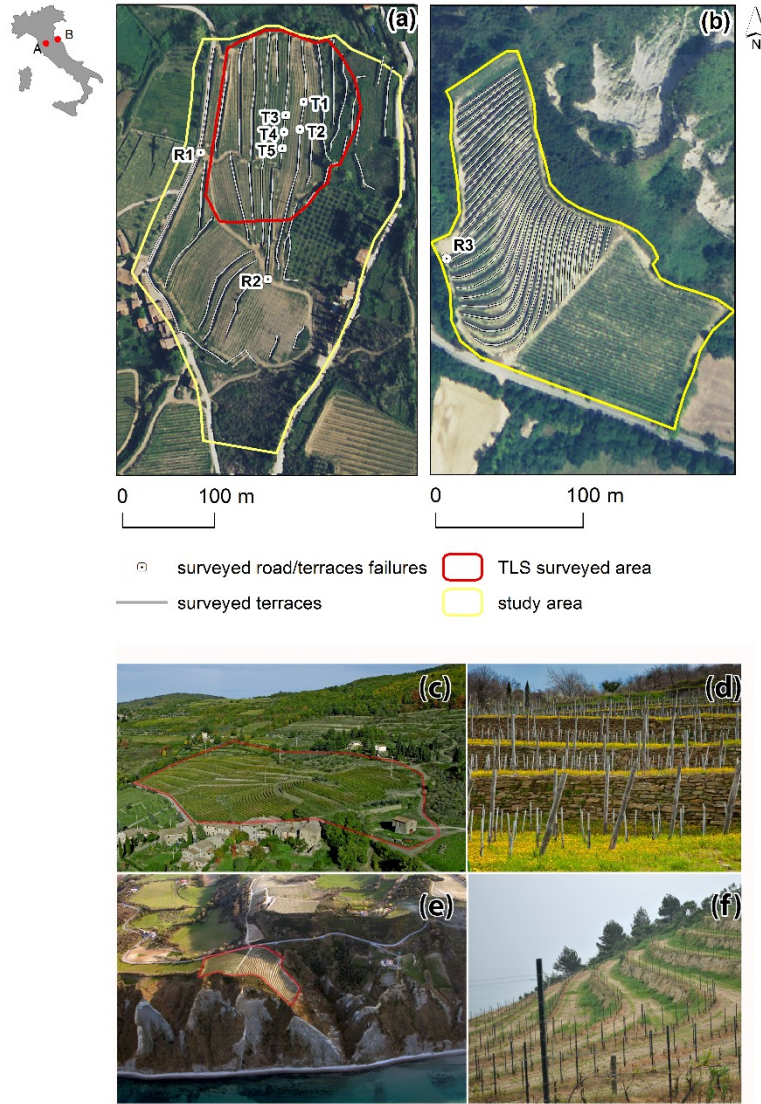


Figure 2. Considered study areas: Lamole case study (a, c and d) and Pesaro case study (b, e and f). Investigated failures surveyed with the DGPS are labelled with T (1 to 5) for terraces failures and R (1 to 3) for road erosion. The small portion of the Lamole area surveyed with TLS is also shown.

5.3.2 Lidar Data Sets

5.3.2.1 Aerial laser scanner digital terrain model

For both case studies, there is the availability of a digital terrain model (DTM) with a 1-m resolution derived from aerial laser scanner (ALS) (Figure 3a and b) (Cazorzi et al., 2013; Sofia et al., 2014a). The DTM has a horizontal accuracy of about ± 0.3 m and vertical accuracy of ± 0.15 m (root-mean-square error estimated using DGPS ground truth control points) (see the work by Sofia et al., 2014a for a more detailed description of the data set).

5.3.2.2 Terrestrial laser scanner digital terrain model

In March 2013, using a ‘time-of-fly’ terrestrial laser scanner (TLS) system Riegl® LMS-Z620 (maximum measurement range: 2 km; accuracy: 10 mm; speed of acquisition: up to 11,000 pts/s), we performed a detailed survey of a small portion of the Lamole area, where numerous failures were registered (T1 to T5 in Figure 2). The TLS survey was carried out from six scan positions, in order to have a full coverage of the area of interest. Once acquired, the elevation points were interpolated by the natural neighbour method (Sibson, 1981) to generate a 0.2-m resolution DTM (Figure 3d), with an absolute vertical error smaller than 0.1 m (root-mean-square error estimated using DGPS ground truth control points). This data set provides a more detailed DTM (Figure 3d), if compared with the available 1-m ALS DTM (Figure 3c).

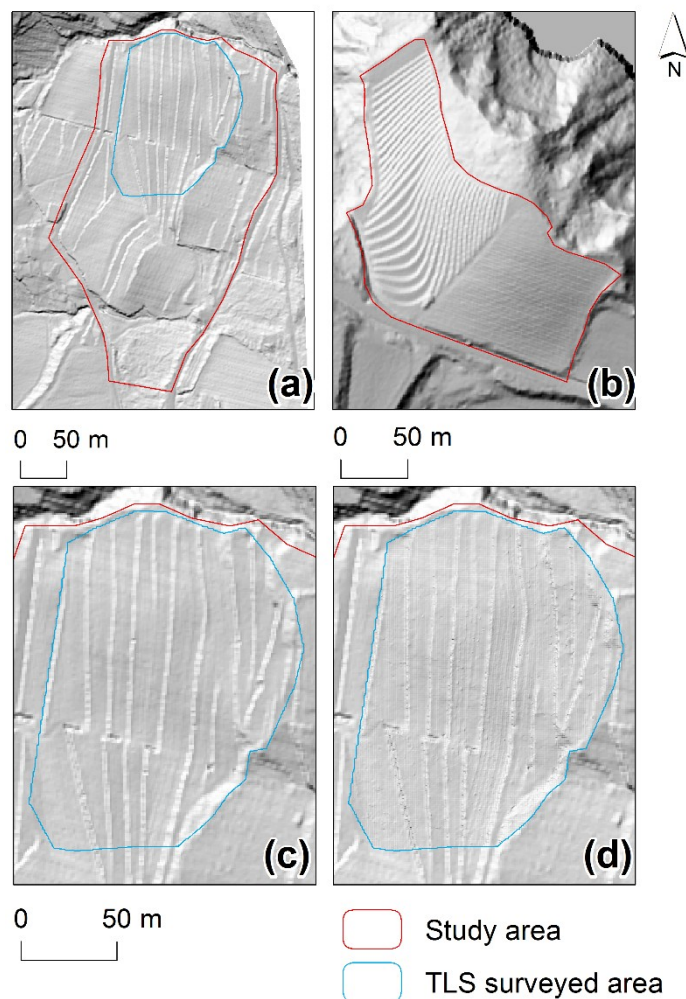


Figure 3. Available ALS DTMs at 1-m resolution for the Lamole (a) and the Pesaro (b) study areas, and comparison between the ALS 1-m DTM (c) and the TLS 0.2-m DTM (d) for the Lamole study area.

5.3.3 Relative Path Impact Index

To quantify the influence of terraces on the surface flow paths, we applied the Relative Path Impact Index (RPII, Equation 1) proposed by Tarolli et al. (2013). This index considers the contributing area as a proxy of the flow path distributions, and in a logarithmic form, it emphasizes and maps areas presenting an increased drainage area because of the presence of anthropogenic features.

$$RPII = \ln\left(\frac{A_r - A_{sm}}{A_{sm}}\right) \quad (1)$$

Where: A_r is the contributing area evaluated in the presence of terraces on the hillslopes, while A_{sm} is the contributing area evaluated in the absence of morphological alterations on the hillslopes. For the calculation of the drainage area, Tarolli et al. (2013) considered the D^∞ flow direction algorithm (Tarboton, 1997), while to simulate the absence of roads and trail, they considered a smoothed DTM based on the quadratic approximation of the original surface (Equation 2) as proposed by Evans (1979), solved within a local moving window, as modified by Wood (1996),

$$Z = ax^2 + by^2 + cxy + dx + ey + f \quad (2)$$

Where: x , y and Z are local coordinates, and a to f are quadratic coefficients. In this work, to eliminate the presence of terraces from the smoothed DTM, we applied a 41-m moving window to solve Equation 2. The higher the RPII, the higher the potential runoff-induced erosion is. However, in some case, some RPII high values derive from the smoothing of other morphological forms. As a consequence, the RPII should not be considered an 'absolute' map of erosion but rather as a map that can show the likely sections of a terrace/road subject to potential alteration (Tarolli et al., 2013), and this information can be considered for the identification of possible solutions (see section on Terrestrial Laser Scanner Data and Relative Path Impact Index: A Practical Application), or to provide a location for further and deeper analyses.

5.3.3.1 Critical values of Relative Path Impact Index

Equation 1 produces maps having widespread RPII values, depending on the amount of differences between the original upslope area and the upslope area derived without terraces. It might be useful, therefore, to identify a threshold discriminant enough for the goal of identifying the most critical areas. A recent literature underlined how statistics drawn from lidar topography carry the signature of some important geomorphic processes (Tarolli & Dalla Fontana, 2009; Sofia et al., 2011; Lin et al., 2013), artificial features (Cazorzi et al., 2013; Sofia et al., 2014a; Sofia et al., 2014b), or errors in the digital maps (Sofia et al., 2013). Starting from this literature, we tested different statistics to identify critical values of the RPII. The considered thresholds are taken from (i) Tarolli et al. (2012): multiples of the standard deviation (σ_{RPII}), interquartile range (IQR_{RPII}) and mean absolute deviation (MAD_{RPII}); (ii) Lashermes et al. (2007) and Passalacqua et al.

(2010): multiples of the deviation from the normal distribution in the quantile–quantile plot ($Q-Q \text{ plot}_{\text{RP II}}$); and (iii) Sofia et al. (2014a): the bounds defining the outliers in the box plot. See the referenced works for the detailed thresholds definition.

5.4 Results and Discussion

5.4.1 Relative Path Impact Index Analysis

Figure 4 shows the overall RPII map derived from the 1-m ALS DTM of the Lamole case study (a) and the Pesaro case study (b), with the field-surveyed instabilities (T1 to T5 for terraces instabilities and R1 to R3 for road erosion). As depicted by Figure 4, all the surveyed failures are related to high values of the RPII, underlining indeed how this index is highly effective in identifying the likely sections of areas at risk of collapsing or erosion.

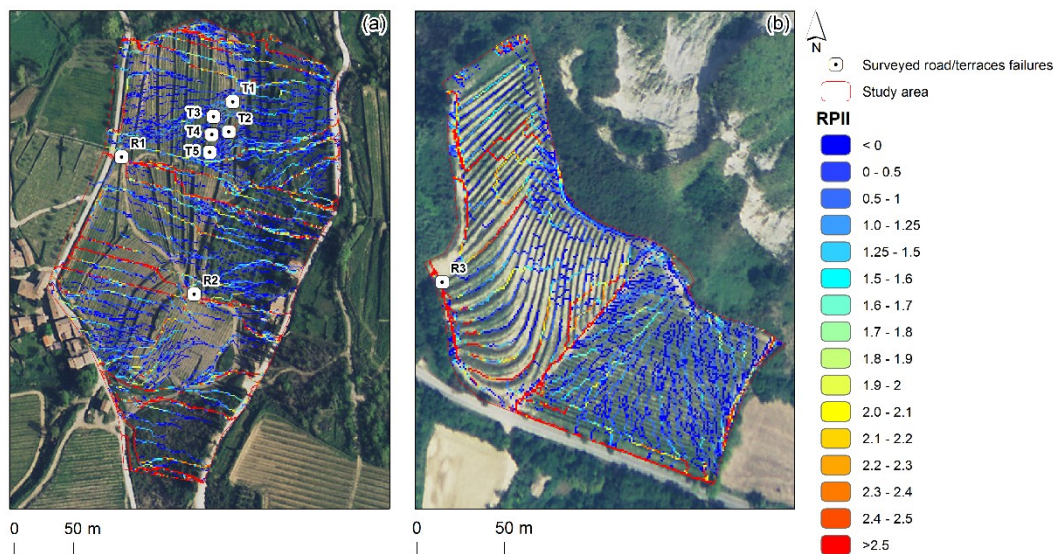


Figure 4. RPII maps derived from the 1-m ALS DTM for the Lamole case study (a) and the Pesaro case study (b), with surveyed terraces instabilities (T1 to T5) and eroded roads (R1 to R3).

Analysing more in detail the road failures for the Lamole case study (R1 and R2, in Figure 5), the 1-m ALS DTM is able to capture correctly the presence of the road that induces the flow deviation and the consequent flow concentration and erosion on the road itself. The index here is highly effective because the size of the road (about 2 m in width) is correctly captured using a 1-m DTM.

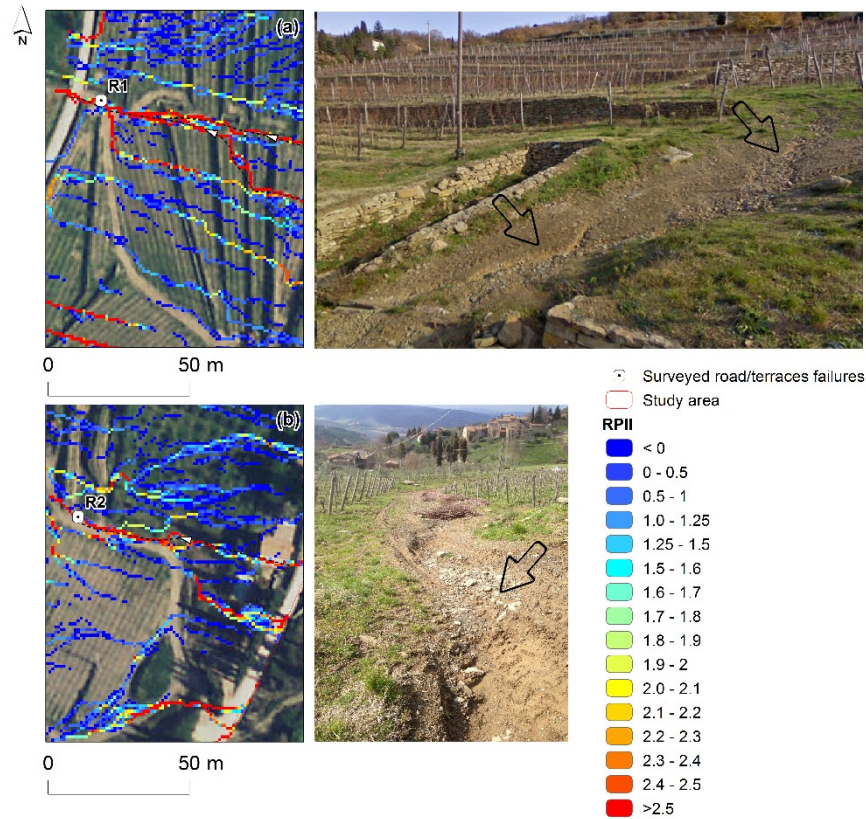


Figure 5. Lamole case study. RPII maps derived from the 1-m ALS DTM, highlighting the flow modifications induced by the road, resulting in road erosion.

However when considering the terrace failures (T1 to T5 in Figure 6), a 1-m ALS DTM does not seem accurate enough for the correct characterization of the flow alterations causing the problems. The flow alteration here are probably due to a morphological conformation whose size is best represented when considering as basic topographic information the 0.2-m TLS DTM (Figure 6b): at this resolution, the RPII results more accurate and can depict correctly all the surveyed failures (T1 to T5).

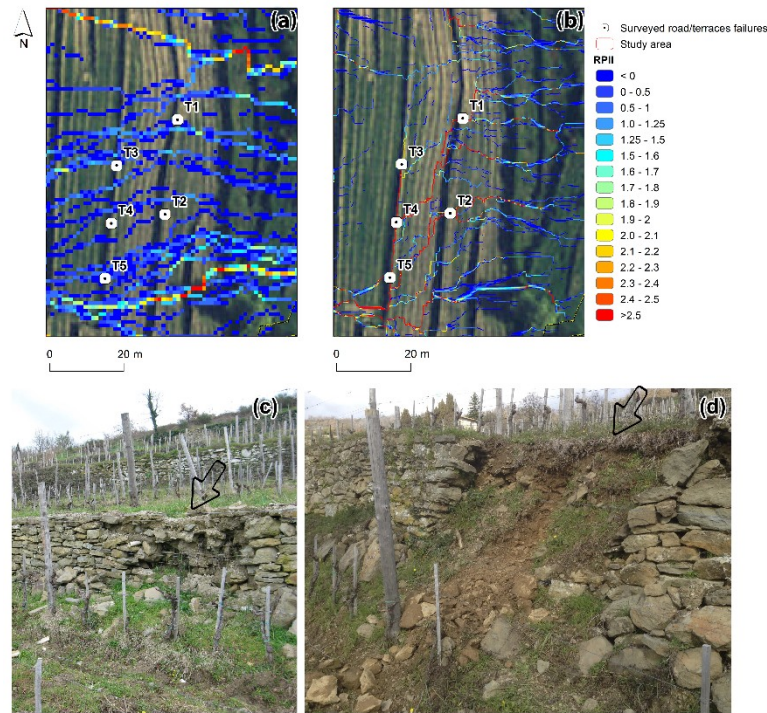


Figure 6. Lamole case study. RPII maps derived from the 1-m ALS DTM (a) and from the 0.2-m TLS (b), highlighting the flow modifications induced by the terraced structures, and the related terraces failures (T1 in (c) and T5 in (d)).

Moving on to the Pesaro case study, using a DTM with a 1-m grid cell size, it is possible to successfully intercept the flow directions' modifications induced by the terraces (Figure 7), which contributed to a deviation of the flows and to an accumulation of water within the agricultural road causing the road disruption.



Figure 7. Pesaro case study. RPII maps derived from the 1-m ALS DTM (a) highlighting the flow modifications induced by the terraces, causing water accumulation that erodes the road (b).

In both case studies, the methodology could be considered as the first and relatively fast approach to map water surface paths alteration due to roads or terraces presence, as an important factor triggering dry-stone wall instabilities or road erosion. A 1-m ALS DTM is able to capture correctly the terraces and road-induced flow modification determining the erosion surveyed on different agricultural roads. However, to capture the failures on dry-stone wall terraces having a height of 1 to 2 m maximum, a 1-m DTM is not accurate enough, while the availability of a 0.2-m DTM provides the best results.

5.4.2 Critical Values of Relative Path Impact Index

Figure 8 shows the RPII maps thresholded considering one, two and three times the defined threshold (thr). In the figure, the considered thresholds are applied to the RPII derived from the TLS DTM; they are the σ_{RPII} (Figure 8b), IQR_{RPII} (Figure 8c), MAD_{RPII} (Figure 8d), $Q-Q\ plot_{RPII}$ (Figure 8e) and outliers (Figure 8f).

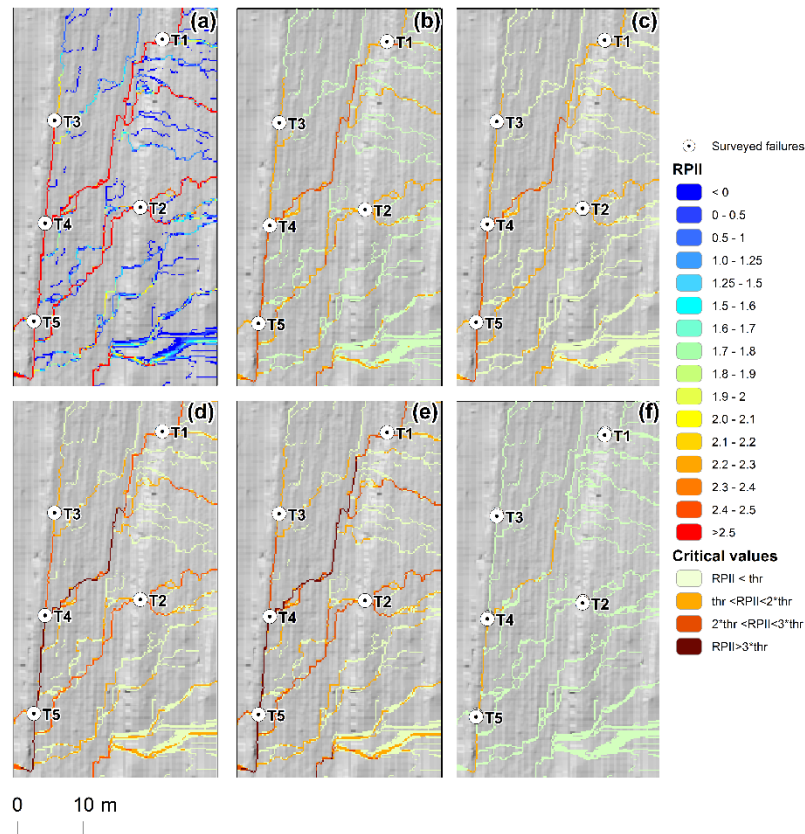


Figure 8. RPII for the TLS surveyed area (a) and extraction of critical values (b–f). The considered thresholds are the σ_{RPII} (b), IQR_{RPII} (c), MAD_{RPII} (d), $Q-Q\ plot_{RPII}$ (e) and outliers (f).

To test the extent at which the different thresholds can identify terraces/road erosions, we compared values using a Mann–Whitney test, which is a nonparametric test that proposes a null hypothesis where two populations are the same. We deemed the RPII values obtained for the erosion/failures the basic population, and we progressively tested these values against the RPII values larger than multiples of each considered threshold. From Figure 8, it appears that most of the terraces failures (T1 to T3) are not

characterized by outlier values of the RPII (Figure 8f), and the Sofia et al. (2014a) threshold only identifies two out of the five failures (T4 and T5). As well, the results of the Mann–Whitney test confirm the inapplicability of this threshold: RPII values related to the terraces failures are statistically different to RPII values larger than one time the Sofia et al. (2014a) threshold (Mann–Whitney test p-value 0.0001). The other statistics (Figure 8c–e) correctly label the potentially critical areas, whereas terraces collapsings seem to be generally related to an RPII higher than one time the value of the considered threshold. Although different statistical thresholds are able to correctly identify the most critical values of the RPII, a threshold value of a multiple of the standard deviation (Tarolli & Dalla Fontana, 2009; Tarolli et al., 2012) (Figures 8b and 9) or of the Q–Q plot_{RPII} (Tarolli et al., 2012) (Figure 8c) results to be more accurate identifying areas potentially at risk of erosion. The results of the Mann–Whitney test for these two thresholds underline a failure of rejection of the null hypothesis at the 5% significance (Mann–Whitney test p-value 0.46 and 0.43 for the σ_{RPII} and Q–Q plot_{RPII}, respectively). This means that the RPII values related to the terraces failures are statistically similar to RPII values larger than one time σ_{RPII} or Q–Q plot_{RPII}. As well, from the Mann–Whitney test, it appears that the σ_{RPII} better represents critical areas as a threshold (p-value is larger than the p-value obtained using the Q–Q plot_{RPII}). The results of the Mann–Whitney test for the MAD_{RPII} underlines that this threshold is able to identify values related to the terraces failures, however with a lower power than the σ_{RPII} (Mann–Whitney test p-value 0.23). When we increase the thresholds to multiples of the considered values, the Mann–Whitney test p-value indicates in all cases a rejection of the null hypothesis at the 5% significance, meaning that the RPII values measured where the terraces fail are statistically different from the values labelled as critical. Figure 9 shows the identification of critical areas (with RPII greater than the RPII standard deviation, σ_{RPII}) for the Lamole case study (Figure 9a and b) and for the Pesaro case study (Figure 9c). As depicted from the Figure 9, considering the σ_{RPII} as threshold, all the areas with the surveyed failures/erosion are correctly labelled.

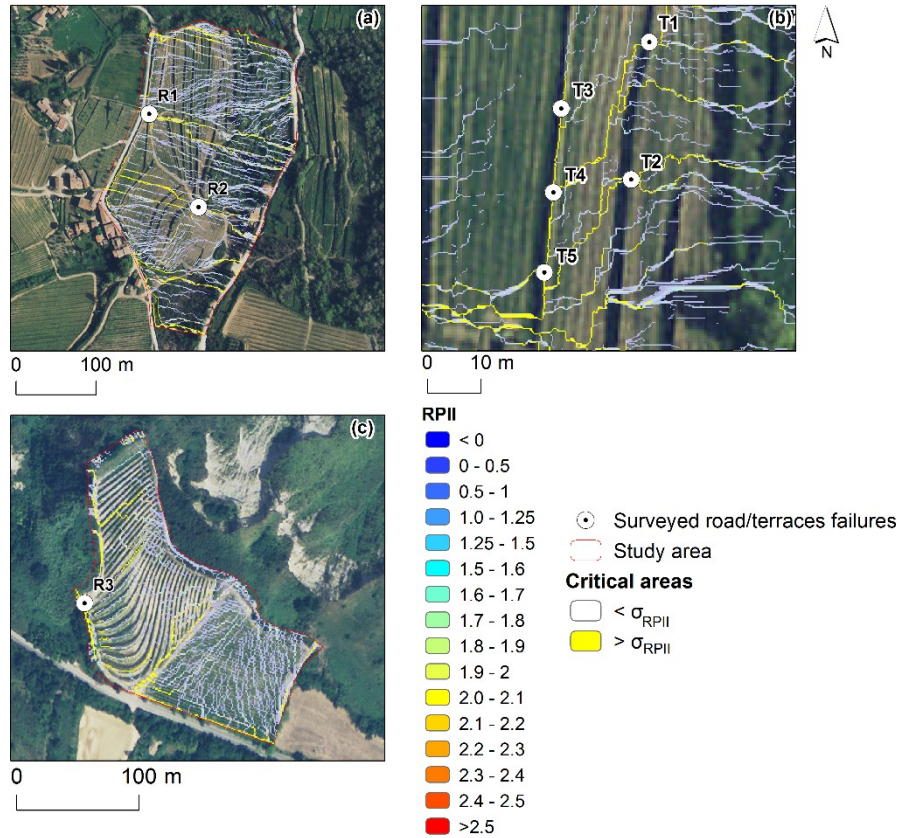


Figure 9. Identification of critical areas (with RPII greater of σ_{RPII}), for the Lamole case study (a and b) and the Pesaro case study (c).

5.4.3 Terrestrial Laser Scanner Data and Relative Path Impact Index: A Practical Application

When the surface water flow is the main factor triggering the dry-stone wall instabilities, a common soil conservation measure is to reduce erosion by building ditches at the bottom of the terrace risers, to accommodate all runoff created by the terrace itself as well as any tributary runoff that enters the terrace drain. For the Lamole case study, therefore, we simulated on the TLS DTM three different drainage ditch systems (Figure 10), trying to identify the optimal solution to reduce the critical areas (whereas critical areas are identified using σ_{RPII} as threshold).

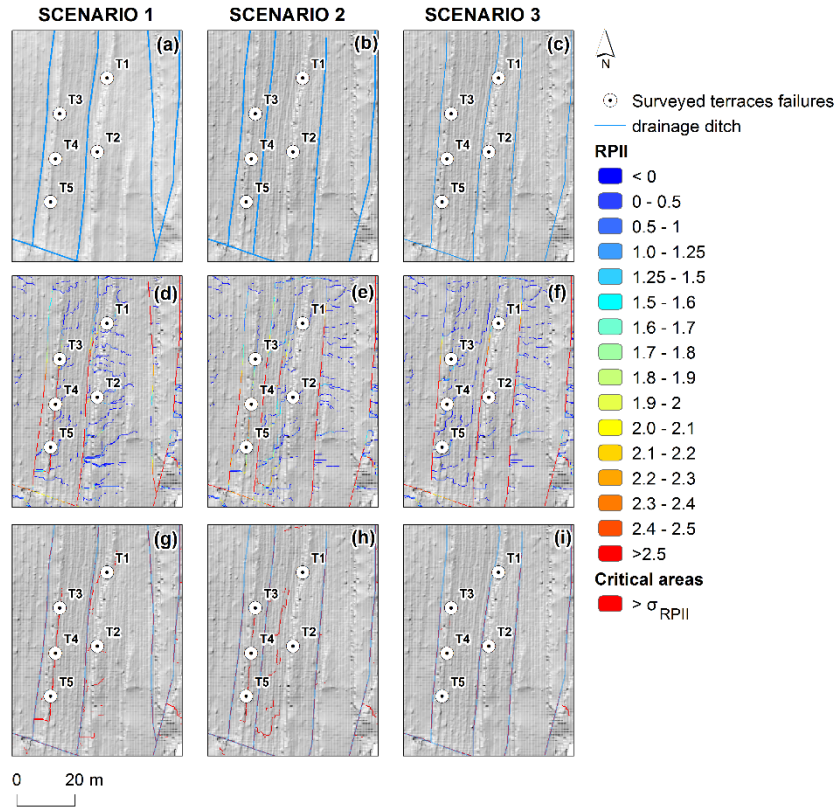


Figure 10. Lamole case study. Simulations of four different drainage ditch systems (a–c) and derived RPII maps (d and e). Figures from (f) to (i) show the critical areas, identified using σ_{RPII} as threshold.

The simulated ditches have a width of 0.2 m (corresponding to the DTM resolution), a depth of 0.2 m and are built to guarantee the hydrological connectivity along the ditch itself. In the first scenario (Figure 10a), ditches are created at the bottom of the main terrace risers. With this ditch network, the erosion is slightly reduced for the T1 and T2 areas; however, it is not reduced enough (Figure 10d and g). Moving on to scenario 2, the ditch is created in the middle of the bench of the terrace where the T1 and T2 erosions are surveyed. This is a feasible situation, for this peculiar case, because in that position, there is already a change in slope visible on the DTM and that corresponds to a small terrace wall (Figure 10b). By positioning the channel in this location, T1 and T2 do not present anymore high values of the RPII (Figure 10e and h). Focusing on the terrace failures in T3, T4 and T5, one must note that the scenario 1 and scenario 2 networks do not succeed in reducing the RPII. This is because these failures are caused by an accumulation of water eroding over time the ground on the back side of the dry-stone wall, creating a sort of erosion ditch (Figure 1a). By filling that erosion ditch on the DTM, and adding a drainage ditch at the bottom of each terrace riser (scenario 3, Figure 10c), the RPII is significantly reduced for all the considered terraces failures (Figure 10f and i). This example shows how the RPII can be used to help in scheduling correctly the soil conservation measures, to mitigate the consequences of the anthropogenic alterations induced by the terraces structures.

5.5 Conclusion

This research presented an application of lidar elevation data for a first high-resolution hydro-geomorphological analysis of terraced vineyards. The work is based on the analysis and practical application of the RPII, through the use of high-resolution DTMs and statistical thresholds. The results of this work underlined how the RPII is able to identify correctly terraced failures and road erosions in terraced vineyards, when the surface water flow is the main factor triggering the instabilities. Thanks to this type of support, it is possible to simulate different soil conservation measures scenario, identifying the optimal solution. The proposed approach, therefore, can help in scheduling a suitable planning to mitigate the consequences of the anthropogenic alterations induced by the terraces structures and agricultural roads. One must consider that the construction of dry-stone terraces has been based for centuries on the farmers' empirical knowledge; therefore, spatial databases compatible with modern land-management systems are not always present. Many authorities have nowadays an easy access to qualified and updated high-resolution lidar data, which can be used for the definition of strategies for the conservation of the environment, and to strengthen and improve the quality of the territorial knowledge, and this research underlined how these models can offer easy-to-access tool for land management in terraced landscapes.

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6 Mulching practices for reducing soil water erosion: A review ⁵

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6.1 Abstract

Among the soil conservation practices that are used, mulching has been successfully applied to reduce soil and water losses in different contexts, such as agricultural lands, fire-affected areas, rangelands and anthropic sites. In these contexts, soil erosion by water is a serious problem, especially in semi-arid and semi-humid areas of the world. Although the beneficial effects of mulching are known, further research is needed to quantify them, especially in areas where soil erosion by water represents a severe threat. In the literature, there are still some uncertainties about how to maximize the effectiveness of mulching to reduce the soil and water loss rates. Given the seriousness of soil erosion by water and the uncertainties that are still associated with the correct use of mulching, this study review aims to (i) develop a documented and global database on the use of mulching with vegetative residues; (ii) quantify the effects of mulching on soil and water losses based on different measurement methods and, consequently, different spatial scales; (iii) evaluate the effects of different types of mulches on soil and water losses based on different measurement methods; and (iv) provide suggestions for more sustainable soil management. The data published in the literature have been collected. The results showed the beneficial effects of mulching in combating soil erosion by water in all of the environments considered here, with reduction rates in the average sediment concentration, soil loss and runoff volume that, in some cases, exceeded 90%. However, the economic feasibility of mulching application was not readily available in the literature. Therefore, more research should be performed to help both farmers and land managers by providing them with evidence-based means for implementing more sustainable soil management practices.

6.2 Introduction

Mulching is referred to as the agronomic practice of leaving mulch on the soil surface for soil and water conservation and to favour plant growth (Jordán et al., 2011). The term 'mulch' refers to any material other than soil or living vegetation that performs the function of a permanent or semi-permanent protective cover over the soil surface (Jordán et al., 2011). For this purpose, different materials can be used, such as vegetative residues, biological geotextiles, gravel and crushed stones (Blavet et al., 2009; Cerdà, 2001; Gilley et al., 1986b; Jiménez et al., 2016; Jordán et al., 2010; Keizer et al., 2015; Mandal and Sharda, 2013; Robichaud et al., 2013a; Smets et al., 2008a; Xu et al., 2012; Zhao et al., 2015). Fig. 1 shows an example of three types of vegetative residues that were used as mulch to reduce soil water erosion in agricultural environments: straw mulching (a and b) and mulching with prunings (c) applied to vine inter-rows and mulching with chopped prunings (d) in an apricot orchard.

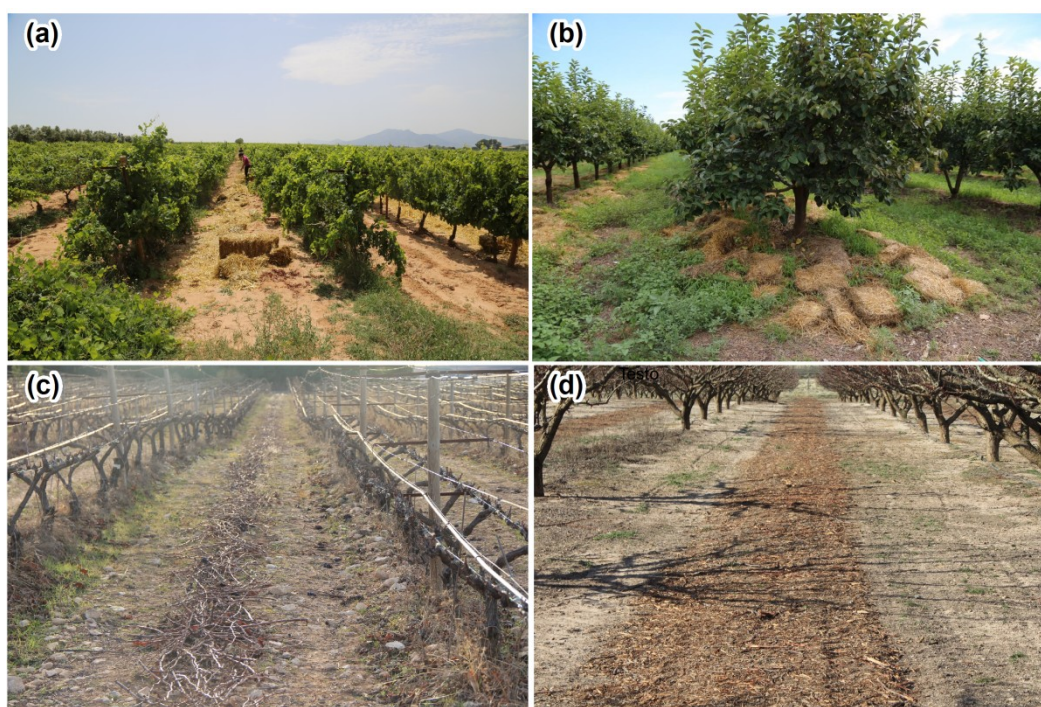


Fig. 1. Straw mulching (a and b) and mulching with prunings (c) applied along vine inter-rows, and mulching with chopped prunings (d) used in an apricot orchard. These pictures were taken at Celler del Roure and Casa Pago Gran in Les Alcusses de Moixent (Province of Valencia, Spain) (photos by A. Cerdà).

Mulching has been shown to confer several beneficial effects. First, it protects the soil against raindrop impact (Blavet et al., 2009; Jordán et al., 2010; Morgan, 1986; Sadeghi et al., 2015a; Smets et al., 2008a), thereby reducing the water and soil loss rates in different environments, such as agricultural lands (Cook et al., 2006; García-Orenes et al., 2009, 2012; Keesstra et al., 2016; Mwango et al., 2016; Prosdociimi et al., 2016a, 2016b; Tebrügge and Düring, 1999), rangelands (Fernández et al., 2012; Fernández and Vega, 2014; Sadeghi et al., 2015a), fire-affected areas (Bautista et al., 1996; Prats et al., 2014; Robichaud et al., 2013a, 2013b), and anthropic sites (Albaladejo Montoro

et al., 2000; Gilardelli et al., 2016; Hayes et al., 2005; Pereira et al., 2015). Second, it reduces both the overland flow generation rates and velocity by increasing roughness (Cerdà, 2001; Jordán et al., 2010), and it cuts the sediment and nutrient concentrations in runoff (Cerdà, 1998; Gholami et al., 2013; Poesen and Lavee, 1991). Furthermore, mulching allows improved infiltration capacity (Jordán et al., 2010; Wang et al., 2016) and increases water intake and storage (Cook et al., 2006; Mulumba and Lal, 2008). It enhances the activity of some species of earthworms as well as crop performance (Fonte et al., 2010; Tebrügge and Düring, 1999; Thierfelder et al., 2013; Wooldridge and Harris, 1991), interactions with nutrients (Campiglia et al., 2014; MovahediNaeni and Cook, 2000), the soil structure and the organic matter content within the soil (De Silva and Cook, 2003; Karami et al., 2012). The increase in the soil organic matter content can be particularly significant when vegetative residues are used as mulches, as shown by García-Orenes et al. (2009) and Jordán et al. (2010). Mulching has also been shown to reduce the topsoil temperature for more optimal germination and root development (Dahiya et al., 2007; Riddle et al., 1996) and to decrease evaporation (Qin et al., 2006; Uson and Cook, 1995; Vanlauwe et al., 2015). Among these beneficial mulching effects, the reduction of water and soil loss rates is one of the most significant and remarkable (Adekalu et al., 2007; Cerdà, 2001; Dahiya et al., 2007; Díaz-Raviña et al., 2012; Groen and Woods, 2008; Hayes et al., 2005; Jiang et al. 2011; Liu et al., 2012; Prats et al., 2014; Prosdocimi et al., 2016b; Robichaud et al., 2013b; Sadeghi et al., 2015a). In fact, soil erosion by water is a serious problem, especially in the semi-arid and semi-humid areas of the world, such as the Mediterranean (Cerdà et al., 2009; Cerdan et al., 2010; Garcia-Ruiz, 2010), central Asia (Dregne, 1992; Lal, 1995; Sadeghi et al., 2015a, 2015b), the USA (Morgan, 2005; Robichaud et al., 2013a, 2013b) and developing countries such as China and India (Barton et al., 2004; Bhatt and Khera, 2006; Lal, 2000; Zheng, 2006). Although soil erosion by water consists of physical processes that vary significantly in severity and frequency according to when and where they occur, they are also strongly influenced by anthropic factors, such as unsustainable farming practices and land-use changes on large scales (Boardman et al., 1990; Cerdà, 1994; Lal, 1984; Martínez-Casasnovas et al., 2016; Montgomery, 2007; Tebrügge and Düring, 1999). This research has led to the definition of 'accelerated' soil erosion as being the result of human impacts on the landscape (Morgan, 2005). The impact of soil erosion on modern society has raised the need for threshold values against which to assess the soil monitoring data, especially in agriculture (Montgomery, 2007). The agricultural sector is known to be affected by higher erosion rates than other sectors because of several factors, such as conventional ploughing, low vegetation cover, soil compaction and sealing by heavy machinery, an absence of soil erosion control measures and the use of pesticides and herbicides that damage biological activity in soils (Arnáez et al., 2015; Bakker et al., 2005; Cerdà et al., 2009; Ciampalini et al., 2012; Cots-Folch et al., 2009; Freemark and Boutin, 1995; Johnsen et al., 2001; Laudicina et al., 2015; Raclot et al., 2009; Rodrigo Comino et al., 2016; Tarolli et al., 2014, 2015; Tarolli and Sofia, 2016; Tebrügge and Düring, 1999). Post-fire soil erosion is another serious problem, and the subsequent increases in debris flows, sedimentation and flooding are well recognized (Bento-Gonçalves et al., 2012; Ferreira

et al., 2008; Kunze and Stednick, 2006; Lane et al., 2006; Moody and Martin, 2009; Moody et al., 2008a, 2008b; Nyman et al., 2011; Robichaud et al., 2013a, 2013b; Shakesby and Doerr, 2006; Silins et al., 2009). Reduced rainfall interception and soil exposure from the direct impact of raindrops are the primary reasons for fire-enhanced erosion rates (Ben-Hur et al., 2011; Fernández et al., 2011; Soto et al., 1998; Wagenbrenner et al., 2006). These erosion rates reach their maximum values immediately after a wildfire, and they tend to decrease with time (Cerdà and Doerr, 2005; Cerdà and Lasanta, 2005; Robichaud, 2009; Swanson, 1981; Shakesby and Doerr, 2006). Anthropogenic slopes that are present in quarries, waste disposal and construction sites are also known to enhance soil erosion by water, regardless of whether proper soil control measures are adopted (Albaladejo Montoro et al., 2000; Goff et al., 1993; Gray, 1986; Hayes et al., 2005; Muzzi et al., 1997). Given these enhanced erosion rates, there is a need to find and apply the appropriate soil management strategies to reduce the runoff and erosion rates to approximately the rates that would occur under natural conditions (Morgan, 2005). This management must be performed to protect public health and safety and to reduce the potential for resource damage resulting from erosion, sedimentation, and increased flooding (Brevik and Sauer, 2015; Galati et al., 2015; Marques et al., 2015; Mekonnen et al., 2015; Robichaud et al., 2010). Within this context, mulching has recently been implemented as a more conservation-minded soil management practice that can preserve the soil and water quality, and it is preferable to conventional soil management techniques such as tillage (mechanical weeding) and no-tillage (chemical weeding) operations (Cerdà et al., 2009; Jordán et al., 2011). Among the different types of mulch, mulching with vegetative residues has been considered one of the most effective at reducing the soil erosion rates and water losses in agricultural lands, rangelands, fire-affected areas and anthropic sites (Bautista et al., 1996; Cook et al., 2006; Fernández et al., 2012; Hayes et al., 2005; Prats et al., 2014; Prosdocimi et al., 2016a; Robichaud et al., 2013b; Sadeghi et al., 2015a; Shi et al., 2013). However, although the beneficial effects of mulching with vegetative residues are known, further research is needed to quantify these effects, especially in areas where soil erosion by water represents a severe threat. Furthermore, there are still some uncertainties in the literature about how to maximize the effectiveness of mulching to reduce soil and water loss rates. First, the choice of vegetative residue type is fundamental; this choice drives the application rate, cost and, consequently, effectiveness of mulching (Bautista et al., 2009; Beyers, 2004; Erenstein, 2003; Lal, 1976; Prats et al., 2012; Robichaud et al., 2013a; Smets et al., 2008a, 2008b). Second, the appropriate application rate is another significant factor that strongly influences the effectiveness of mulching in reducing soil and water losses (Bautista et al., 1996; Jordán et al., 2010; Lal, 1984; Lattanzi et al., 1974; Meyer et al., 1970; Mulumba and Lal, 2008; Prosdocimi et al., 2016a) as well as the percentage of area covered by mulch (Adekalu et al., 2007; Harold, 1942; Laflen and Colvin, 1981; Lal, 1977; Norton et al., 1985; Wischmeier, 1973). In addition, the effectiveness of mulch cover is believed to depend on the raindrop erosivity, soil condition, steepness and length of the slope (Francis and Thornes, 1990; Jin et al., 2009; Lattanzi et al., 1974; Sadeghi et al., 2015b; Smets et al., 2008b). Given the seriousness of soil erosion by

water and the uncertainties that still concern the correct use of mulching, this study review focuses on mulching with vegetative residues, particularly its effects on the soil erosion rates and water loss. Therefore, this study review aims to (i) develop a documented global database on the use of mulching with vegetative residues; (ii) quantify the effects of mulching on soil and water losses based on different measurement methods and, consequently, spatial scales; (iii) evaluate the effects of different mulch types on soil and water losses based on different measurement methods; and (iv) provide suggestions for more sustainable soil management. The data that have been published in the literature were collected using the Scopus, Thomson Reuters ISI and Google Scholar databases.

6.3 Methods

6.3.1 Data collection

Twenty-three experimental studies that reported the quantitative effects of mulching with vegetative residues on soil and water losses have been collected from the published literature. It was considered essential to select studies in which field and laboratory experiments were conducted on both control and mulched plots. This data collection process allowed a global database containing 296 records to be created. For each record, the following variables were collected, if available: (i) spatial location (Sl), (ii) spatial scale (Ss), (iii) measurement method (Mm), (iv) soil conservation techniques (SCTs), (v) mulch application rate (Ar) (g m^{-2}), (vi) cover mulch (Cm) (%), (vii) soil loss (SL) (g), (viii) erosion rate (ER) ($\text{Mg ha}^{-1} \text{ yr}^{-1}$), (ix) sediment concentration (Sc) (g L^{-1}), (x) runoff (R) in terms of runoff volume (L) and height (mm), (xi) runoff coefficient (RC) (%), (xii) slope gradient (Slo) (m m^{-1}), and (xiii) mean rainfall intensity (Rlmean) (mm h^{-1}). If the exact plot size was reported, the spatial scale is classified here as very fine (microplots) ($< 1 \text{ m}^2$), fine ($1\text{--}1000 \text{ m}^2$), hillslope ($1000 \text{ m}^2\text{--}1 \text{ ha}$) and field scale ($> 1 \text{ ha}$) (Boix-Fayos et al., 2006; Verheijen et al., 2009). Because the spatial scale is related to the measurement method, if the exact plot size was not reported, the spatial scale was defined according to the measurement method. This approach justifies the fact that for some studies, a spatial scale ranging from fine to hillslope (fine–hillslope) was assigned (Prosdocimi et al., 2016b). The measurement methods are (i) rainfall simulation (RS) (i.e., Prosdocimi et al., 2016a; Sadeghi et al., 2015a, 2015b), (ii) runoff plot (RP) (i.e., Fernández and Vega, 2014; Mwango et al., 2016), (iii) silt fence (SF) (i.e., Robichaud et al., 2013a; Rough, 2007) and (iv) sediment trap (SD) (i.e., Robichaud et al., 2013b). The RS method is associated with very fine scales and refers to splash and sheet erosion. The RP method is associated with both fine and hillslope scales because runoff plots may have different sizes and usually refer to sheet and rill erosion. SF is associated with fine scales and, thus, with sheet erosion processes, and SD is linked to field scales and may refer to gully erosion processes. The SCT classification is based on the information provided for each study and the categorization made by Maetens et al. (2012). The presence of both control and mulched plots has been considered essential for each work under study. The SCTs used in this work include (i) control (C), (ii) straw mulching (SM), (iii) grass mulching (GM), (iv) wood mulching (WM), (v) mulching with

prunings (MP), (vi) mulching with needle casts (MN), (vii) hydromulching (HM), (viii) hydromulching + mulching (HM + M), (ix) mulching + seeding (M + Sd), (x) mulching with prunings + grass cover (MP + GC), and (xi) mulching with prunings + tillage (MP + T).

6.3.2 Data organisation and analysis

Given the high variability of the collected data, the soil loss and erosion rate data that were measured using the same method were separated from those obtained by using a different method. This action was performed to reduce the uncertainty of comparing published data that were derived from different erosion measurement methods with one another (García-Ruiz et al., 2015; Prosdocimi et al., 2016b). Non-parametric Kruskal-Wallis and Mann-Whitney U tests were used to evaluate the presence of significant differences between the different measurement methods and types of mulch with respect to the soil loss and/or soil erosion rate because the assumptions of normality or homogeneity of variances were rejected. For all analyses, a value of $p < 0.05$ was considered statistically significant, and the open-source R software was used.

6.4 Results

6.4.1 Description of database

The data considered for the purposes of this work are summarized in Table 1. Two of the 23 reported articles are cited works. For each article, the minimum, maximum and mean values (in brackets) of the above-mentioned variables are noted, if available. The articles are listed in chronological order.

Table 1. Minimum, maximum and mean values (in brackets) of all the variables collected from published literature, if available: i) spatial location (SI), (ii) spatial scale (Ss), (iii) measurement method (Mm), (iv) soil conservation techniques (SCTs), (v) mulch application rate (Ar), (vi) cover mulch (Cm), (vii) soil loss (SL), (viii) erosion rate (ER), (ix) sediment concentration (Sc), (x) runoff (R) in terms of runoff volume and height, (xi) runoff coefficient (RC), (xii) slope gradient (Slo), and (xiii) mean rainfall intensity (Rlmean). For the Mm variable, the following types of measurement methods are reported: rainfall simulation (RS), runoff plot (RP), silt fence (SF) and sediment trap (SD). For the SCT variable, the following types of soil conservation techniques are reported: (i) control (C), (ii) straw mulching (SM), (iii) grass mulching (GM), (iv) wood mulching (WM), (v) mulching with prunings (MP), (vi) mulching with needle casts (MN), (vii) hydromulching (HM), (viii) hydromulching + mulching (HM + M), (ix) mulching + seeding (M + Sd), (x) mulching with prunings + grass cover (MP + GC), and (xi) mulching with prunings + tillage (MP + T). The articles are reported in chronological order.

References	SI	Ss	Mm	SCTs	Ar (g m ⁻²)	Cm (%)	SL (g)	ER (Mg ha ⁻¹ yr ⁻¹)	Sc (g L ⁻¹)	R (L)	RC (mm)	RC (%)	Slo (m m ⁻¹)	Rlmean (mm h ⁻¹)
Bekele and Thomas (1992)	Kenya	Fine-hillslope	RP	C-SM	50-225	-	-	149.5-203.5 (173.25)	-	-	194-240 (212.25)	-	5.32	-
Albaladejo Montoro et al. (2000)	Spain	Fine	RP	C-HM + M	1500	-	-	0.026-1.70 (0.606)	-	744-5316 (2511)	-	-	12.3	-
Barton et al. (2004)	China	Fine	RP	C-SM	400	-	-	0.46-7.5 (2.54)	-	-	-	-	1.72-15.11 (7.51)	-

Table 1. (continued)

References	SI	Ss	Mm	SCTs	Ar (g m ⁻²)	Cm (%)	SL (g)	ER (Mg ha ⁻¹ yr ⁻¹)	Sc (g L ⁻¹)	R (L)	R (mm)	RC (%)	Slo (m m ⁻¹)	RI mean (mm h ⁻¹)
Döring et al. (2005)	Germany	Fine	RS	C-SM	125-500	-	170-10357 (2370.6)	-	1.1-69 (17.24)	-	-	-	2.634	60
^a Wagebrenner et al. (2006)	USA	Hillslope	SF	C-SM	220	33-74 (57.75)	-	0.5-9.5 (6.25)	-	-	-	-	8.89	-
Adekalu et al. (2007)	Nigeria	Very fine	RS	C-GM	90-610	0-90	-	-	-	-	-	19-90 (51.64)	1.96-3.91 (2.94)	100
^a Rough (2007)	USA	Hillslope	SF	C-M + Sd	224	5-96	-	0.009-13.2 (3.65)	-	-	-	-	6.91	-
Groen and Woods (2008)	USA	Very fine	RS	C-SM	224	0-100	-	1-7.2 (3.65)	-	-	-	-	4.55-4.86 (4.71)	80

Table 1. (continued)

References	SI	Ss	Mm	SCTs	Ar (g m ⁻²)	Cm (%)	SL (g)	ER (Mg ha ⁻¹ yr ⁻¹)	Sc (g L ⁻¹)	R (L)	R (mm)	RC (%)	Slo (m m ⁻¹)	RI _{mean} (mm h ⁻¹)
García-Orenes et al. (2009)	Spain	Very fine	RS	C-SM- MP	50- 250	0-83.8	0-0.5 (0.2)	-	0-1.5 (0.63)	0-0.4 (0.167)	-	0-2.7 (1.23)	1.66	55
Jordán et al. (2010)	Spain	Very fine	RS	C-SM	100- 1500	-	-	-	0.1- 4.35 (1.83)	0.13- 3.55 (1.53)	-	0.97- 27.34 (11.86)	1.32- 1.54 (1.44)	65
Li et al. (2011)	China	Fine	RP	C-GM	-	0-100	-	-	-	-	-	2.13- 22.43 (12.28)	7.97	-
Liu et al. (2012)	China	Fine	RP	C-SM	600	-	-	0.77-1.02 (0.89)	-	-	-	16.7- 23.5 (20.0)	-	-
Fernández et al. (2012)	Spain	Very fine	RS	C-M + Sd	250	0-28.7	-	-	-	-	8.91- 12.66 (10.785)	25-37 (31)	8.531	67

Table 1. (continued)

References	SI	Ss	Mm	SCTs	Ar (g m ⁻²)	Cm (%)	SL (g)	ER (Mg ha ⁻¹ yr ⁻¹)	Sc (g L ⁻¹)	R (L)	R (mm)	RC (%)	Slo (m m ⁻¹)	Rlmean (mm h ⁻¹)
Díaz-Raviña et al. (2012)	Spain	Fine	RP	C-SM	250	-	-	0.22-2.04 (1.13)	-	-	-	-	9.481	-
Robichaud et al. (2013a)	USA	Fine	SF	C-SM- WM- HM	60- 1250	0-87	-	0-22 (1.158)	-	-	-	-	7.41- 19.08 (14.59)	-
Robichaud et al. (2013b)	USA	Field	SD	C-HM- SM	110- 220	-	-	0-46.3 (9.86)	-	-	0-119.1 (20.75)	-	5.82- 10.92 (7.93)	-
Fernández and Vega (2014)	Spain	Fine	RP	C-SM- WM	200- 350	0-70	-	0.5-5.4 (2.20)	-	-	-	-	17.203	-
Prats et al. (2014)	Portugal	Very fine	RP	C-WM	1100	0-85	-	0.63-8.48 (4.55)	-	-	378-785 (581.5)	-	14.036	-

Table 1. (continued)

References	SI	Ss	Mm	SCTs	Ar (g m ⁻²)	Cm (%)	SL (g)	ER (Mg ha ⁻¹ yr ⁻¹)	Sc (g L ⁻¹)	R (L)	R (mm)	RC (%)	Slo (m m ⁻¹)	RImean (mm h ⁻¹)
Sadeghi et al. (2015a)	Iran	Fine	RS	C-SM	500	0-90	54.24- 787.94 (300.06)	-	3.53- 10.71 (6.39)	15.66- 74.24 (43.77)	-	-	9.481	30-90
Sadeghi et al. (2015b)	Iran	Very fine	RS	C-SM	500	0-90	0-787.94 (181.5)	-	0-10.71 (3.725)	-	-	1.18- 79.42 (42.87)	-	50-90
Wang et al. (2016)	China	Fine	RS	C-MP- MP + GC- MP + T	-	0-98	-	-	13- 133.5 (42.12)	10.8- 51.9 (23.15)	-	-	8.531	-
Mwango et al. (2016)	Tanzania	Fine	RP	C-GM	360	-	-	5.08- 183.6 (49.61)	-	-	-	-	13.61- 14.87 (12.24)	-

Table 1. (continued)

References	SI	Ss	Mm	SCTs	Ar (g m ⁻²)	Cm (%)	SL (g)	ER (Mg ha ⁻¹ yr ⁻¹)	Sc (g L ⁻¹)	R (L)	R (mm)	RC (%)	Slo (m m ⁻¹)	RI _{mean} (mm h ⁻¹)
Prosdocimi et al. (2016a)	Spain	Very fine	RS	C-SM	75	0-90	6.13- 111.97 (47.48)	0.24-4.48 (1.89)	1.76- 14.2 (6.79)	3.48- 8.96 (6.37)	-	25.35- 65.15 (46.37)	0.57- 3.43 (1.87)	55

If the exact plot size was reported, the spatial scale is classified here as very fine (microplots) (< 1 m²), fine (1 – 1000 m²), hillslope (1000 m² – 1 ha) and field scale (> 1 ha). Because the spatial scale is related to the measurement method, if the exact plot size was not reported, the spatial scale was defined according to the measurement method. This approach justifies the fact that for some studies, a spatial scale ranging from fine to hillslope (fine – hillslope) was assigned.

^a in Groen and Woods (2008).

First, Table 1 shows that not every study included data for every variable. Based on the full database, the spatial location, spatial scale, measurement method and soil conservation techniques were the variables for which 100% of the data were accounted for. For the variables on the soil loss and/or erosion rate, application rate of mulch, slope and cover mulch, 97, 96, 91 and 82% of the data were accounted for, respectively. For the mean rainfall intensity, runoff volume and height, runoff coefficient and sediment concentration, 48, 42, 39 and 35% of the data were accounted for, respectively. Furthermore, Table 1 indicates that the USA, Spain, Iran and Nigeria are the countries in which most of the studies on the effects of mulching with vegetative residues on soil erosion by water have been performed. In fact, these countries together account for 87% of the full database. Fig. 2 shows the spatial distribution around the world for the records in our database.

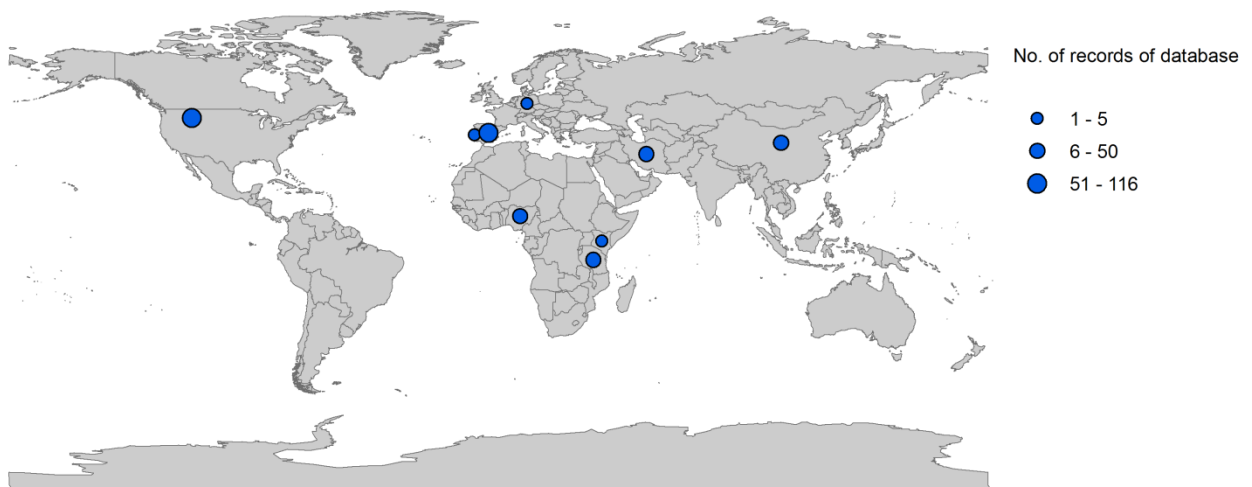


Fig. 2. Global spatial distribution of the collected database records. The USA, Spain, Iran and Nigeria were the countries in which most of the studies were performed.

With respect to the spatial scale, the fine scale is the most representative one, with 44% of the data accounted for in the full database. These data were followed by the categories very fine, field, hillslope and fine–hillslope scales, with 39, 13, 3 and 1%, respectively. With respect to the measurement methods, the RS is the most frequent method, with 50% of the data considered in the full database. RS was followed by the SF, RP and SD methods, with 25, 13 and 13%, respectively. Regarding the SCTs, the mulched plots account for 61.8% of the full database, and the remaining 38.2% is associated with control plots. This finding is related to the fact that most of the studies only rely on one control plot but a larger number of mulched plots, which can be distinguished according to their different application rates or types of mulches. Among the mulched plots, SM is the most often accounted-for variable, with 31.4% of the data. It was followed by GM, HM, WM, MN, M + Sd, HM + M, MP, MP + GC, and MP + T, with 12.2, 9.5, 4.4, 1.7, 1, 0.3, 0.7, 0.3 and 0.3%, respectively. Table 1 also shows that the soil losses and erosion rates are highly variable, to extents that may be even greater than the upper limit of the European tolerable soil erosion rate (Verheijen et al., 2009). This high variability is primarily caused by the different temporal and spatial scales of analyses and measurement methods employed in the studies (Prosdocimi et al.,

2016b). Regarding the environments, agricultural lands and fire-affected areas are represented by nearly the same percentage of data in the full database (39 and 41%, respectively). They were followed by rangelands and anthropic sites with 18 and 1%, respectively. Only 2% of data considered in the full database do not report the type of environment (Fig. 3).

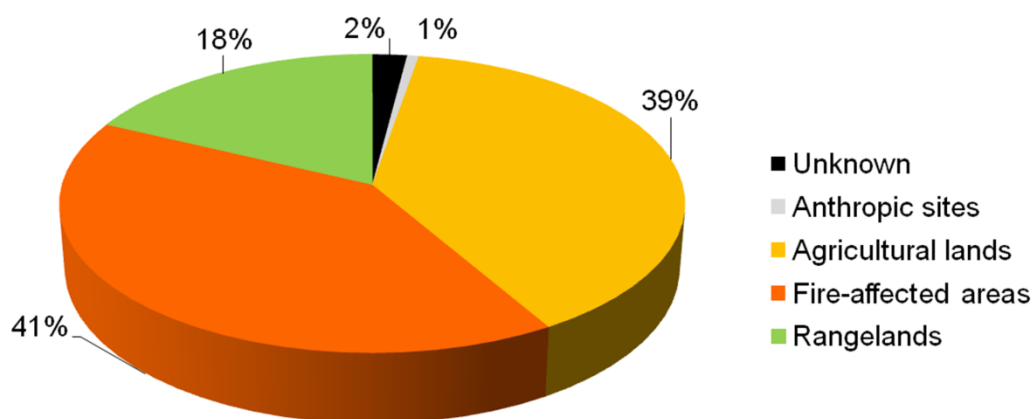


Fig. 3. This pie chart shows the relative frequency, as expressed in percentages, of the environments where the studies that were collected in our database were performed. Agricultural lands and fire-affected areas are represented by nearly the same percentage of data in the full database (39 and 41%, respectively). Those percentages were followed by rangelands and anthropic sites, with 18 and 1%, respectively. Only 2% of the data considered from the full database do not report the type of environment.

6.4.2 *Mulching compared with the control*

The effects of mulching are summarized in Table 2, where the reduction percentages in the average sediment concentration, average soil loss and/or erosion rate, runoff volume and height, and runoff coefficient resulted from the application of mulching are reported, if available, for each work cited in Table 1. In addition, the mean rainfall intensity and the application rate for the mulch and measurement methods are shown, if available.

Table 2. Reduction percentages in the average (avg) sediment concentration (Sc), average soil loss (SL) and/or erosion rate (ER), runoff volume (R (L)) and height (R (mm)), and runoff coefficient (RC) resulted from the application of mulching are reported, if available, for each work cited in Table 1. In addition, the mean rainfall intensity (Rlmean), mulch application rate (Ar) and measurement method (Mm) are shown, if available. For the Mm variable, the following types of measurement methods are reported: rainfall simulation (RS), runoff plot (RP), silt fence (SF) and sediment trap (SD). For the SCT variable, the following types of soil conservation techniques are reported: (i) control (C), (ii) straw mulching (SM), (iii) grass mulching (GM), (iv) wood mulching (WM), (v) mulching with prunings (MP), (vi) mulching with needle casts (MN), (vii) hydromulching (HM), (viii) hydromulching + mulching (HM + M), (ix) mulching + seeding (M + Sd), (x) mulching with prunings + grass cover (MP + GC), and (xi) mulching with prunings + tillage (MP + T). The articles are reported in chronological order.

References	SCTs	Variable	Rlmean (mm h ⁻¹)	Ar (g m ⁻²)	Mm	Reduction (%)
Bekele and Thomas (1992)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	203.50	0	RP	
		Avg R (mm)	240.00			
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	178.50	50		-12.29
		Avg R (mm)	214.00			-10.83
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	161.50	100		-20.64
		Avg R (mm)	201.00			-16.25
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	149.50	225		-26.54
		Avg R (mm)	194.00			-19.17
Albaladejo Montoro et al. (2000)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	1.70	0	RP	
		Avg R (L)	5316.00			
	HM + M	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.09	1500		-94.84
		Avg R (L)	1473.00			-72.29
Barton et al. (2004)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	4.17	0	RP	
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.91	400		-78.16

Table 2. (continued)

References	SCTs	Variable	Rlmean (mm h ⁻¹)	Ar (g m ⁻²)	Mm	Reduction (%)
Döring et al. (2005)	C	Avg Sc (g L ⁻¹)	69.00	0		
		Avg SL (g)	1606.00			
	SM	Avg Sc (g L ⁻¹)	3.40	125		-95.07
		Avg SL (g)	31.00			-98.07
	SM	Avg Sc (g L ⁻¹)	2.20	250	RS	-96.81
		Avg SL (g)	42.00			-97.38
	SM	Avg Sc (g L ⁻¹)	1.10	500		-98.41
		Avg SL (g)	26.00			-98.38
	SM	Avg Sc (g L ⁻¹)	10.50	250		-84.78
		Avg SL (g)	133.00			-91.72
^a Wagebrenner et al. (2006)	C	Avg ER Mg ha ⁻¹ yr ⁻¹)	7.85	0	SF	
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	4.65	220		-40.76
Adekalu et al. (2007)	C	Avg RC (%)	80	0	RS	
	GM	Avg RC (%)	59	90		-26.67
	GM	Avg RC (%)	41	240		-49.03
	GM	Avg RC (%)	27	610		-66.11
^a Rough (2007)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	6.93	0	SF	
	M + Sd	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.37	224		-94.59
Groen and Woods (2008)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	5.7	0	RS	
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	1.6	224		-71.93

Table 2. (continued)

References	SCTs	Variable	Rlmean (mm h ⁻¹)	Ar (g m ⁻²)	Mm	Reduction (%)
García-Orenes et al. (2009)	C	Avg Sc (g L ⁻¹)	1.50	0		
		Avg SL (g)	0.50	0		
		Avg R (L)	0.40	0		
		Avg RC (%)	2.70	0		
	SM	Avg Sc (g L ⁻¹)	0.00	250		-100.00
		Avg SL (g)	0.00	250	RS	-100.00
		Avg R (L)	0.00	250		-100.00
		Avg RC (%)	0.00	250		-100.00
	MP	Avg Sc (g L ⁻¹)	0.40	50		-73.33
		Avg SL (g)	0.10	50		-80.00
		Avg R (L)	0.10	50		-75.00
		Avg RC (%)	1.00	50		-62.96
	C	Avg Sc (g L ⁻¹)	4.35	0		
		Avg R (L)	3.55	0		
		Avg RC (%)	27.34	0		
Jordán et al. (2010)	SM	Avg Sc (g L ⁻¹)	3.46	100		-20.46
		Avg R (L)	2.73	100		-23.10
		Avg RC (%)	20.99	100		-23.23
	SM	Avg Sc (g L ⁻¹)	0.94	500		-78.39
		Avg R (L)	1.10	500	RS	-69.01
		Avg RC (%)	8.55	500		-68.73
	SM	Avg Sc (g L ⁻¹)	0.31	1000		-92.87
		Avg R (L)	0.18	1000		-94.93
		Avg RC (%)	1.45	1000		-94.70
	SM	Avg Sc (g L ⁻¹)	0.10	1500		-97.70
		Avg R (L)	0.13	1500		-96.34
		Avg RC (%)	0.97	1500		-96.45

Table 2. (continued)

References	SCTs	Variable	Rlmean (mm h ⁻¹)	Ar (g m ⁻²)	Mm	Reduction (%)
Li et al. (2011)	C	Avg RC (%)	22.43	0	RP	-90.50
	GM	Avg RC (%)	2.13	-		
Liu et al. (2012)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.98	0	RP	-20.02
		Avg RC (%)	23.05	0		
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.79	600		
		Avg RC (%)	9.00	600		
		Avg R (mm)	12.66	0		
Fernández et al. (2012)	C	Avg RC (%)	37.00	0	RS	-29.62
	M + Sd	Avg R (mm)	8.91	250		
		Avg RC (%)	25.00	250		
Díaz-Raviña et al. (2012)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	2.04	0	RP	-89.22
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.22	250		
		Avg ER (Mg ha ⁻¹ yr ⁻¹)	4.19	0		
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	3.62	220		
	WM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.90	1250		
	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.74	0		
Robichaud et al. (2013a)	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.21	220	SF	-71.70
		Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.82	0		
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.03	560		
	HM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.60	60		

Table 2. (continued)

References	SCTs	Variable	Rlmean (mm h ⁻¹)	Ar (g m ⁻²)	Mm	Reduction (%)	
Robichaud et al. (2013a)	MN	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.92		0	+12.19 [*]	
	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.41		0		
	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.10	-	220	SF	-76.36
	WM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.04		450		-89.70
	HM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.03		110		-92.73
	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	10.69		0		
Robichaud et al. (2013b)	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	4.08		220		-61.82
	HM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	13.54		200	SD	+26.74 [*]
	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	13.53	-	0		
	HM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	6.57		220		-51.48
	HM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	9.06		110		-33.05
	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	5.40		0		
Fernández and Vega (2014)	SM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.50	-	200	RP	-90.74
	WM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.70		350		-87.04
	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	8.48		0		
Prats et al. (2014)		Avg R (mm)	785.00		0	RP	
		Avg ER (Mg ha ⁻¹ yr ⁻¹)	0.63	-	1100		-92.57
	WM	Avg R (mm)	378.00		1100		-51.85

Table 2. (continued)

References	SCTs	Variable	Rlmean (mm h ⁻¹)	Ar (g m ⁻²)	Mm	Reduction (%)
Sadeghi et al. (2015a)	C	Avg Sc (g L ⁻¹)	6.60	0		
		Avg SL (g)	132.15	0		
		Avg R (L)	20.20	0		
	SM	Avg Sc (g L ⁻¹)	3.83	500		-41.99
		Avg SL (g)	60.58	500		-54.15
		Avg R (L)	17.01	500		-15.81
	C	Avg Sc (g L ⁻¹)	7.28	0		
		Avg SL (g)	265.24	0		
		Avg R (L)	36.53	0		
	SM	Avg Sc (g L ⁻¹)	4.09	500		-43.84
		Avg SL (g)	140.73	500	RS	-46.94
		Avg R (L)	33.90	500		-7.19
	C	Avg Sc (g L ⁻¹)	9.11	0		
		Avg SL (g)	491.67	0		
		Avg R (L)	53.93	0		
	SM	Avg Sc (g L ⁻¹)	5.61	500		-38.43
		Avg SL (g)	268.86	500		-45.32
		Avg R (L)	49.34	500		-8.52
	C	Avg Sc (g L ⁻¹)	10.38	0		
		Avg SL (g)	760.94	0		
		Avg R (L)	73.76	0		
	SM	Avg Sc (g L ⁻¹)	4.29	500		-58.67
		Avg SL (g)	280.30	500	RS	-63.16
		Avg R (L)	65.49	500		-11.21

Table 2. (continued)

References	SCTs	Variable	Rlmean (mm h ⁻¹)	Ar (g m ⁻²)	Mm	Reduction (%)
Sadeghi et al. (2015b)	C	Avg Sc (g L ⁻¹)	5.36	0	RS	
		Avg SL (g)	257.76	0		
		Avg R (L)	52.70	0		
	SM	Avg Sc (g L ⁻¹)	2.09	500		-60.90
		Avg SL (g)	105.24	500		-59.17
		Avg RC (%)	33.04	500		-37.30
Wang et al. (2016)	C	Avg Sc (g L ⁻¹)	133.50	0	RS	
		Avg R (L)	51.90	0		
	MP	Avg Sc (g L ⁻¹)	13.00	-		-90.26
		Avg R (L)	10.80	-		-79.19
	MP + GC	Avg Sc (g L ⁻¹)	15.90	-		-88.09
		Avg R (L)	11.50	-		-77.84
	MP + T	Avg Sc (g L ⁻¹)	18.10	-		-86.44
		Avg R (L)	18.40	-		-64.55
Mwango et al. (2016)	C	Avg ER (Mg ha ⁻¹ yr ⁻¹)	128.77	0	RP	
	GM	Avg ER (Mg ha ⁻¹ yr ⁻¹)	10.02	360		-92.22
		Avg Sc (g L ⁻¹)	10.31	0		
	C	Avg SL (g)	76.43	0		
		Avg R (L)	7.38	0		
		Avg RC (%)	53.70	0		
Prosdocimi et al. (2016a)		Avg Sc (g L ⁻¹)	3.29	75	RS	-68.05
	SM	Avg SL (g)	18.53	75		-75.75
		Avg R (L)	5.37	75		-27.31
		Avg RC (%)	39.03	75		-27.31

* Increase, instead of reduction, in average soil erosion rate (ER).

^a in Groen and Woods (2008).

Table 2 shows that except for a couple values obtained from Robichaud et al. (2013a, 2013b), mulching always entails a reduction in the average sediment concentration, soil loss and/or erosion rate, runoff volume and height, and runoff coefficient with respect to the control plots. The reduction rates usually increase with the increased mulch application rate. Bekele and Thomas (1992) tested three different application rates for straw mulch, i.e., 50, 100 and 225 g m⁻², to evaluate the effects of mulching on the erosion rate and runoff height. They found that an application rate of 225 g m⁻² minimised both the average erosion rates (-26.54%) and the runoff (-19.17%). Similarly, Döring et al. (2005) achieved reduction rates of up to -98.41 and -98.38% in terms of the average sediment concentration and soil loss, respectively, with a straw mulch application rate of 500 g m⁻². Adekalu et al. (2007) showed that a grass mulch application rate of 610 g m⁻² significantly reduced the average runoff coefficient (-66.11%). In the same way, Jordán et al. (2010) tested four different straw mulch application rates and found that the highest application rate (1500 g m⁻²) resulted in the maximum reduction in terms of average sediment concentration (-97.70%), runoff volume (-96.34%) and runoff coefficients (-96.45%). Another important finding that comes from Table 2 is the fact that various authors tested different types of mulch, rather than different application rates (Fernández and Vega, 2014; García-Orenes et al., 2009; Robichaud et al., 2013a, 2013b; Wang et al., 2016). Robichaud et al. (2013a) compared the effectiveness of straw mulching (SM), wood mulching (WM), hydromulching (HM) and mulching with needle casts (MN) in different fire-affected areas. They found that wood mulching entailed a greater reduction in terms of the average erosion rate than did straw mulching (-78.57% of reduction vs. -13.52% for the first study area and -89.70% of reduction vs. -76.36% for the second study area). However, Fernández and Vega (2014) obtained slightly better results with straw mulching than wood mulching in terms of reducing the average erosion rate in Spanish shrubland (-90.74% of reduction vs. -87.04%). For each variable considered in Table 2, the average and standard deviation values have been computed by grouping data according to the soil conservation techniques, control (C) and mulching (M), and the measurement methods (Table 3). Furthermore, the average reduction and/or increase (%) provided by mulching for each variable have been computed as well. It should be stressed that no distinction has been made among the different types of mulch at this point.

Table 3. Average (avg) and standard deviation (SD) values computed for the sediment concentration (Sc), soil loss (SL) and/or erosion rate (ER), runoff volume (R (L)) and height (R (mm)), by grouping data according to the soil conservation techniques, control (C) and mulching (M), and the measurement methods (RS = rainfall simulation, RP = runoff plot, SF = silt fence, and SD = sediment trap). The average reduction (%) induced by mulching for each variable has been computed as well. No distinction has been made among the different types of mulch at this point.

	Sc (g L ⁻¹)		SL (g)		ER (Mg ha ⁻¹ yr ⁻¹)						R (L)		R (mm)		RC (%)	
	RS		RS		RP		SF		SD		RS		SD		RS	
	C	M	C	M	C	M	C	M	C	M	C	M	C	M	C	M
Avg	12.06	3.75	433.25	107.62	53.62	27.43	2.73	0.98	12.00	8.71	21.62	15.84	19.80	21.27	56.67	35.97
SD	20.22	3.48	1516.18	152.03	75.21	57.24	5.21	3.06	14.89	9.56	22.86	19.66	28.84	31.13	19.87	20.05
Reduction (%)	-68.9		-75.2		-48.8		-64.2		-27.4		-26.7		+7.4 ^a		-36.5	

^a Increase, instead of reduction, in average runoff height (R (mm)).

Table 3 indicates that among all measurement methods, rainfall simulation is the usually the only one that allows researchers to consider the largest amount of variables related to hydrological and erosion processes. In fact, the sediment concentration, runoff volume and runoff coefficient were considered only in studies that relied on the rainfall simulation method. In terms of these variables, the use of mulching led to average reductions of -68.9, -26.7 and -36.5%, respectively (Table 3). Furthermore, the mulching applications caused an average reduction of -75.2% in terms of soil loss, and of -48.8, -64.2 and -27.4%, in the erosion rate, runoff plot, silt fence and sediment trap methods, respectively (Table 3). Only for the runoff height, which has been considered only in studies that applied the sediment trap measurement method, has mulching induced an increase, of +7.4%, rather than a reduction (Table 3). By considering the average reductions induced by mulching in terms of the soil loss and erosion rate, these findings follow the spatial scale effect. The average reduction induced by mulching at very fine scales, namely those that are represented by rainfall simulation experiments (-75.2%), is greater than the average reductions obtained at larger spatial scales, namely those that are represented by the runoff plot (-48.8%), silt fence (-64.2%) and sediment trap (-27.4%) experiments. On the same basis, the average reductions achieved at the field scale, namely those that are represented by sediment trap experiments, were the lowest ones. In accordance with this principle, the average reductions obtained at fine and fine-hillslope scales, namely those that are represented by silt fence and runoff plot experiments, are more similar to one another than they are to the others. To support this principle statistically, the relative percentage changes in terms of soil loss and erosion rate were computed for each measurement method. This calculation was performed to make the data comparable with one another because the soil loss and erosion rate are expressed according to different units of measurement. As an example, the results obtained for the runoff plot method are reported in Table 4.

Table 4. Relative percentage changes in terms of the soil erosion rate (ER) induced by the mulching (M) application with respect to control (C) plots, as computed for the runoff plot measurement method (RP).

References	ER (Mg ha ⁻¹ yr ⁻¹)		Reduction (%)
	C	M	
	203.50	178.50	-12.3
Bekele and Thomas (1992)	-	161.50	-20.6
	-	149.50	-26.5
Albaladejo Montoro et al. (2000)	1.70	0.09	-94.8
	0.83	0.46	-44.6
Barton et al. (2004)	4.17	0.90	-78.4
	7.50	1.37	-81.7
Liu et al. (2012)	0.94	0.77	-18.2
	1.02	0.80	-21.7
Díaz-Raviña et al. (2012)	2.04	0.22	-89.2
	5.40	0.50	-90.7
Fernandez and Vega (2014)	-	0.70	-87.0
	8.48	0.63	-92.6
Prats et al. (2014)	124.30	7.86	-93.7
	131.60	7.55	-93.9
	183.60	5.08	-96.1
	75.60	5.31	-96.0
	-	19.22	-89.5
Mwango et al. (2016)	-	19.50	-89.4
	-	7.57	-90.0
	-	8.10	-89.3
	-	-	-

Table 4 shows that the two soil conservation techniques are not represented by the same amount of data. There are fewer control data than mulching data because for each control plot, different application rates of the same type of mulch or different types of mulch are usually tested, as was also clear from Table 2. The relative percentage changes for each measurement method have been compared with one another by applying the non-parametric Kruskal-Wallis test because the ANOVA assumptions were seriously violated. This test revealed a statistically significant difference among the four measurement methods with regard to the mulching effect on the soil loss and erosion rate ($X^2(3) = 9.662$, $p < 0.05$). Post-hoc pairwise comparisons using Tukey and Kramer (Nemenyi) tests with a Tukey-Dist approximation for independent variables revealed that only the rainfall simulation and sediment trap variables were significantly different from one another.

6.4.3 Application rate, cover and types of mulches

Table 5 shows the average and standard deviation values of the application rate and cover mulches that have been computed for each measurement method.

Table 5. Average (avg) and standard deviation (SD) values of the application rate (Ar) and cover mulch (Cm) computed for each measurement method (RS = rainfall simulation, RP = runoff plot, SF = silt fence, and SD = sediment trap).

	RS		RP		SF		SD	
	Ar (g m^{-2})	Cm (%)	Ar (g m^{-2})	Cm (%)	Ar (g m^{-2})	Cm (%)	Ar (g m^{-2})	Cm (%)
Avg	326.04	69.9	431.19	-	401.20	60.0	186.67	-
SD	257.34	22.0	322.53	-	399.19	12.3	46.03	-

Table 5 shows that among the measurement methods under consideration, the runoff plot experiments are the ones that use, on average, the highest application rate (431.19 g m^{-2}). By contrast, the sediment trap experiments are those that use the lowest application rate, on average (186.67 g m^{-2}). For the cover mulch, only rainfall simulation and silt fence experiments also account for this important variable, which is usually associated with the corresponding application rate. From the rainfall simulation experiments, there was an average cover mulch of 69.9% and an average value of 60% for the silt fence experiments. The relative percentage changes in terms of the erosion rate have been computed for each type of mulch, and they have been compared with one another by applying either the non-parametric Kruskal-Wallis or Mann-Whitney U tests, depending on the number of mulch types. These non-parametric tests were applied after the assumption of normality or homogeneity of variances was rejected. These tests were performed only for the runoff plot, silt fence and sediment trap experiments. Rainfall simulation experiments have been excluded because, in this case, the soil loss was only associated with a type of mulching, namely straw mulching. Regarding runoff plot experiments, the Kruskal-Wallis test revealed a statistically

significant difference among the four types of mulches (SM, HM+M, GM and WM) with respect to their effects on the erosion rate ($X^2(3) = 12.704$, $p < 0.05$). Post-hoc pairwise comparisons using Tukey and Kramer (Nemenyi) tests with a Tukey-Dist approximation for independent variables revealed that only grass mulching (GM) and straw mulching (SM) were significantly different from one another. For the silt fence experiments, the Kruskal-Wallis test revealed a statistically significant difference among the five types of mulches (SM, M + Sd, WM, HM and MN) with respect to their effects on the erosion rate ($X^2(4) = 14.6157$, $p < 0.05$). Post-hoc pairwise comparisons using Tukey and Kramer (Nemenyi) tests with a Tukey-Dist approximation for independent variables revealed that hydromulching (HM) was significantly different from both mulching + seeding (M+Sd) and wood mulching (WM). Finally, also in the sediment trap experiments, there was a statistically significant difference ($p < 0.05$) between the straw mulching (SM) and hydromulching (HM) mulch types, and their effect on the erosion rate was confirmed by Mann-Whitney U test.

6.5 Discussion

6.5.1 Effectiveness of mulching in reducing soil and water losses

Mulching with vegetative residues has been shown to be an important global practice (Table 1 and Fig. 2); it is used efficiently in different environments (Fig. 3) to reduce soil and water losses. This reduction is clearly shown in Table 2, in which the average sediment concentration, soil loss and/or erosion rate, runoff volume and height, and runoff coefficient have been computed for the mulched plots, and they are lower than those obtained from the controls. However, a direct comparison among the considered studies, in terms of the reduction of soil and water losses induced by mulching, would be misleading because of the several different conditions that characterize each work, principally the measurement method that was applied and the type of mulch that was used. In this regard, the results reported in Table 3 are very interesting because they give an understanding of the effect of the measurement method and, consequently, the spatial scale of the analysis on the effectiveness of mulching in terms of reductions in soil and water losses. The Kruskal-Wallis test and the consequent post-hoc pairwise comparisons revealed that only rainfall simulation and sediment trap experiments were significantly different from one another. This finding is consistent with the spatial scale effect because the rainfall simulation method is associated with very fine scales, whereas the sediment trap is linked to field scales.

6.5.2 Appropriate application rate, cover and types of mulches

Regarding the average mulching application rate, our results ranged from a minimum of 186.67 g m^{-2} , which was associated with sediment trap experiments, to a maximum of 431.19 g m^{-2} for runoff plot experiments (Table 5). An appropriate application rate that was valid at any condition could not be found, although a higher application rate resulted in higher reduction rates in terms of the average sediment concentration, soil loss and/or erosion rate, runoff volume and height, and runoff coefficient (Table 2) (Adekalu et al., 2007; Bekele and Thomas, 1992; Döring et al., 2005; Jordán et al.,

2010). In fact, this review supports the statement that the appropriate application rate should be established for site-specific soil and environmental conditions (Mulumba and Lal, 2008). For example, Lal (1984) found that for slopes ranging between 2 and 20%, mulching rates of 600–800 g m⁻² were adequate if they were regularly maintained in the tropics, but at the same time, these rates were difficult to procure from a single crop. Lattanzi et al. (1974) showed that interrill erosion was reduced by approximately 40% when wheat straw mulch was applied at a rate of 600 g m⁻² and by an estimated 80% at a rate of 920 g m⁻². Meyer et al. (1970) reported that 50 g m⁻² of rice straw mulch reduced the soil loss by one-third of the soil with no mulch cover. Jordán et al. (2010) found that a mulching rate of 500 g m⁻² yr⁻¹ was sufficient to render the runoff flow and sediment concentration negligible in runoff from a no-tilled Fluvisol under semi-arid conditions in SW Spain. Mulumba and Lal (2008) determined an optimum mulch rate of 400 g m⁻² for increasing porosity and 800 g m⁻² to enhance the available water capacity, moisture retention and aggregate stability. Similarly, Bautista et al. (1996) showed that straw mulch applied at a rate of 200 g m⁻² to 16 m² plots reduced the soil loss by 91% in the 19 months following a wildfire in a semiarid pine forest in Spain. Further research should be performed to develop appropriate methods to procure adequate amounts of residue mulch and to show that the optimum mulch rate entails reasonable expenses that farmers and land managers can afford. In this regard, Prosdociimi et al. (2016a) showed that a barley straw cover that was applied at an average rate of 75 g m⁻² and cost approximately 155 € ha⁻¹ contributed to a significant reduction in the surface runoff (from 52.59 to 39.27%), sediment concentration in runoff (from 9.8 to 3.0 g L⁻¹), and soil loss rate (from 2.81 to 0.63 Mg ha⁻¹ h⁻¹) after its immediate application. Regarding the average percentage of area covered by mulch, our results ranged from 60%, which was associated with silt fence experiments, to 69.9% for rainfall simulation experiments (Table 5). These results are consistent with those in the literature. In fact, a mulch cover of 60% is usually considered the minimum threshold for a significant reduction in soil loss (Cerdà and Doerr, 2008; Pannkuk and Robichaud, 2003; Robichaud et al., 2010). In the case of straw mulch, this threshold cover was achieved by applying 200 g m⁻² (Badía and Martí, 2000; Bautista et al., 1996; Fernández et al., 2011; Groen and Woods, 2008; Miles et al., 1989; Wagenbrenner et al., 2006), with costs that can range from 600 to 1200 USD ha⁻¹ for aerial and manual application, respectively (Napper, 2006). In terms of the reduction of soil and water losses that are induced by different types of mulches, the Kruskal-Wallis and Mann-Whitney U tests confirmed some significant differences among i) grass mulching and straw mulching (for runoff plot), ii) hydromulching and mulching + seeding and wood mulching (for silt fence), and iii) straw mulching and hydromulching (for a sediment trap). However, there was no one specific type of mulch that was effective in any environment (Table 2). In agricultural land, straw and grass mulching and mulching with prunings have been found to achieve good results in reducing soil erosion rates (Barton et al., 2004; Cerdà et al., 2016; Khybri, 1989; Lal, 1976; Liu et al., 2012; Maene et al., 1979; Othieno, 1978; Sherchan et al., 1990). According to the field experiments of Gilley et al. (1986a, 1986b), maize residue was significantly more effective in reducing runoff coefficients than were soybean and sorghum residues. In fire-affected areas, although

straw mulching has been commonly used, it has been shown to have some disadvantages, such as a high cost, the potential introduction of non-native plants (Beyers, 2004), and susceptibility to wind-scattering (Bautista et al., 2009). In recent years, there has been increasing interest in alternative mulch types derived from forest residues, such as using fibres of different shapes and sizes (Prats et al., 2012, 2014; Robichaud et al., 2013a; Smets et al., 2008a, 2008b; Yanosek et al., 2006). With respect to this interest, Robichaud et al. (2013a) showed that wood mulching was the most long-lived of the mulch treatments in fire-affected areas of the USA and that straw mulching decreased nearly twice as fast as the wood strand mulch. Similarly, Prats et al. (2012) supported the effectiveness of long chopped eucalyptus bark fibres in reducing post-fire erosion during the first year after a fire. This type of mulch had the additional advantages of being readily available in the study region, not being susceptible to removal by wind, decaying more slowly than straw, and not introducing invasive weeds.

6.5.3 Future guidelines

This literature review confirmed the global importance of using mulch with vegetative residues to reduce soil and water losses in agricultural lands, rangelands, fire-affected areas and anthropic sites. However, when addressing the study of soil water erosion across the world, great variability in conditions can prevent the full comprehension of the factors that affect this phenomenon, as already noted by García-Ruiz et al. (2015) and Prosdocimi et al. (2016b). Given the enhanced erosion rates that can affect these environments, this review stresses the importance of applying appropriate soil management practices to reduce soil and water losses as much as possible. Indeed, in the agricultural sector, if no proper strategies were adopted to protect the soil, the increasing demand for food would further exacerbate soil water erosion processes (Ochoa-Cueva et al., 2015). The monitoring of the mulched plots over longer periods is, in our opinion, essential to enriching our understanding about the persistence of the beneficial effects of mulching. More research should be performed in other study areas across the world to help fill the knowledge gap about the most suitable and affordable type of mulch, the application rate and the cover. We stress the importance of using standardized procedures for evaluating the effectiveness of mulching on soil water erosion and reporting the results to enable data comparisons among different study areas, as also highlighted by García-Ruiz et al. (2015) and Prosdocimi et al. (2016b). The criteria we adopted in this review to collect and organize the published data can help guide future research in this topic. In our opinion, the cost of the mulch and its applications should also be reported as part of the economic evaluation of future research articles. Moreover, the effect of the removal from the original site of vegetative residues used as mulching should be another interesting factor to consider. Vegetative residues may derive from different sources according to their geographic location, from the savannah in the tropics (Nishigaki et al., 2016) to Mediterranean forests (Prats et al., 2014) and agricultural crops (Prosdocimi et al., 2016a). The displacement of these materials has an effect on the soil organic matter content and erosion processes that would be worth considering as well.

6.6 Conclusions

This work presents a review of published studies about the use of vegetative residue mulching to reduce soil and water losses in different environments. The complexity of the processes involved and the variability of the conditions under which studies were performed led us to separate the data according to measurement method, and consequently, to separate the spatial scale of analysis. We believe that our work confirmed the effectiveness of mulching with vegetative residues for reducing soil and water losses in different environments across the world, and we helped to fill the knowledge gap in this important topic. There are still some open questions about the most appropriate types of mulches as well as the application rate and cover that require further research. The economic feasibility of mulching application is another fundamental aspect that should be addressed in future research articles. Science must be of assistance to both farmers and land managers by providing them with evidence-based means for implementing more sustainable soil management practices.

6.7 References

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7 The immediate effectiveness of barley straw mulch in reducing soil erodibility and surface runoff generation in Mediterranean vineyards ⁶

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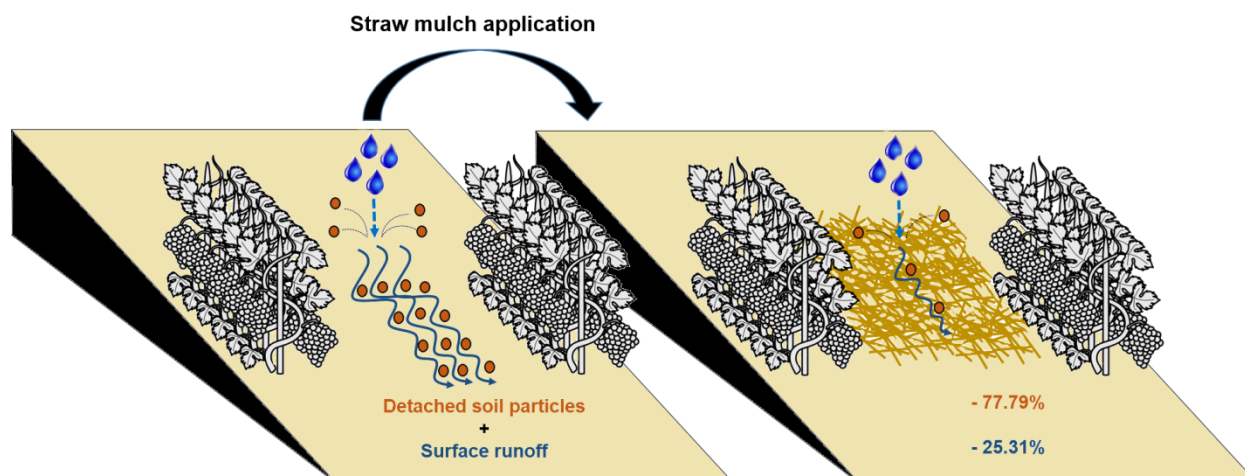
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7.1 Graphical Abstract



7.2 Abstract

Soil and water loss in agriculture is a major problem throughout the world, and especially in Mediterranean areas. Non-conservation agricultural practices have further aggravated the situation, especially in vineyards, which are affected by one of the highest rates of soil loss among cultivated lands. Therefore, it is necessary to find the right soil practices for more sustainable viticulture. In this regard, straw mulching has proven to be effective in other crop and fire affected soils, but, nonetheless, little research has been carried out in vineyards. This research tests the effect of barley straw mulching on soil erosion and surface runoff on vineyards in Eastern Spain where the soil and water losses are non-sustainable. An experiment was setup using rainfall simulation tests at 55 mm h^{-1} over 1 h on forty paired plots of 0.24 m^2 : twenty bare and twenty straw covered. Straw cover varied from 48 to 90% with a median value of 59% as a result of the application of 75 g of straw per m^2 . The use of straw mulch resulted in delayed ponding and runoff generation and, as a consequence, the median water loss decreased from 52.59 to 39.27% of the total rainfall. The straw cover reduced the median sediment concentration in runoff from 9.8 to 3.0 g L^{-1} and the median total sediment detached from 70.34 to 15.62 g per experiment. The median soil erosion rate decreased from 2.81 to $0.63 \text{ Mg ha}^{-1} \text{ h}^{-1}$ due to the straw mulch protection. Straw mulch is very effective in reducing soil erodibility and surface runoff, and this benefit was achieved immediately after the application of the straw.

7.3 Introduction

Soil erosion by water is considered one of the major threats to soil resources in Mediterranean areas due to their climatic, edaphic and geomorphologic conditions (Boardman et al., 1990; Cerdà and Doerr, 2007; Cerdà et al., 2009; Cerdan et al., 2010; Novara et al., 2011, 2015; Verheijen et al., 2009). Rainfall-induced soil erosion risk in the Mediterranean is especially high during summer storms or the early wet season, when plant cover is low (García-Orenes et al., 2009; Taguas et al., 2015). Soil loss is enhanced in cropped soils due to soil management and tillage practices (Blavet et al., 2009; Boardman et al., 1990; Boix-Fayos et al., 2005; Cerdan et al., 2010; Gómez et al., 1999; Vanwalleghem et al., 2011). Indeed, this is due to several reasons such as conventional plowing, removal of the original vegetation, use of pesticides and herbicides that damage biological activity in soils (Freemark and Boutin, 1995; Johnsen et al., 2001; Pelosi et al., 2013), low overall vegetation cover, soil compaction and sealing due to machinery traffic, depletion of organic matter and absence of soil erosion control measures (Arnáez et al., 2015; Bakker et al., 2005; Carr et al., 2015; Cerdà et al., 2009; Ciampalini et al., 2012; Cots-Folc et al., 2009; Laudicina et al., 2015; Raclot et al., 2009; Tarolli et al., 2014, 2015). The effect of intensive agricultural practices on soil erosion is now well known and is concerning given evidence that civilizations have collapsed throughout human history due to erosion (Brevik and Hartemink, 2010) and that erosion continues to negatively affect civilizations in all regions of the world (Brevik, 2009a; Brevik et al., 2015; Cerdà and Doerr, 2007; O'hara et al., 1993; Pimentel et al., 1987; Shi and Shao, 2000; Smith et al., 2015). Therefore, there is a need to find best management practices that will make agriculture sustainable. Among the cultivated lands, Mediterranean vineyards are recognized to be affected by high soil erosion rates because of a combination of natural and anthropogenic factors (Brillante et al., 2015; Cerdà and Doerr, 2007; Cerdan et al., 2002, 2010; Martínez-Casasnovas and Sánchez-Bosch, 2000; Raclot et al., 2009). The main reason for their high erodibility are practices that keep the soil between the vines bare during the entire year (Arnáez et al., 2007; Lasanta and Sobrón, 1988) and these bare surfaces are affected by intense storms that induce severe water erosion and runoff processes (Borga et al., 2011; Poesen and Hook 1997; Santos, 2000). Moreover, vineyards are often planted on steep-sloping soils (Arnáez et al., 2007; Wichereck, 1993) with poor nutrient and organic matter content (Cerdà, 1996; Corti et al., 2011; Novara et al., 2011, 2013). Changes in land use and farming practices or land abandonment have also negatively affected Mediterranean vineyards (Cerdà, 1994; Porta et al., 1994; Tarolli et al., 2014, 2015). Vineyards represent one of the most important crops in terms of income and employment, especially for three of the world's top ten grape-producing countries found in the Mediterranean region, namely Spain, Italy and France, where the total grape area in 2009 amounted to 11,000, 8020 and 8100 km² and thus 2.2, 2.6 and 1.5% of the respective national land areas (Anderson and Norman, 2011). For these reasons, adequate soil management practices are needed to contribute to a more sustainable viticulture, which includes evaluation to determine whether they are acceptable to the farmers who will have to utilize them (Galati et al., 2015; Marques et al., 2015). The

most common soil management techniques in Mediterranean countries consist of tillage (mechanical weeding) and no-tillage (chemical weeding) operations (García-Orenes et al., 2009), and both of them generally result in bare soils during the entire year (Cerdà et al., 2009; Lasanta and Sobrón, 1988; Vaudour et al., 2015). However, alternative and more conservation-minded soil management practices have also been used like catch crops (Bonfante et al., 2015; García-Orenes et al., 2009), mulching (Jordán et al., 2011; Costantini et al., 2015), hydromulching (Prats et al., 2013), geotextiles (Giménez-Morera et al., 2010), natural grassing (Raclot et al., 2009) and rock fragments (Blavet et al., 2009). In particular, the use of catch crops, the implementation of no-tillage or reduced tillage systems, the addition of chipped pruned branches and the use of straw mulches are some of the soil management practices that have been applied on rain-fed experimental orchards in eastern Spain to reduce the high erosion rates (García-Orenes et al., 2009). Staff of the Soil Erosion and Degradation Research (SEDER) group have been studying soil erosion processes due to agriculture and cooperating with farmers to find possible solutions to soil erosion issues in vineyards, olive groves, and fruit and almond orchards. Among the soil conservation practices that have recently been implemented, mulching has proven to be effective in reducing water and soil loss rates and improving soil condition (Cerdà et al., 2015; Cook et al., 2006; García-Orenes et al., 2009; Jordán et al., 2010; Mwango et al., 2015; Mulumba and Lal, 2008; Sadeghi et al., 2015a; Tebrügge and Düring, 1999; Winteraeken and Spaan, 2010). According to Jordán et al. (2011), mulch is any material, other than soil, placed or left on the soil surface for soil and water management purposes. Mulching involves maintaining a permanent or semi-permanent protective cover on the soil surface that can be composed of different materials such as vegetative residues, biological geotextiles, gravel and crushed stones (Cerdà, 2001; Gilley et al., 1986; Jordán et al., 2010; Mandal and Sharda, 2013; Smets et al., 2008; Xu et al., 2012; Zhao et al., 2013). The beneficial effects of mulching can be summarized as follows: i) increased water intake and storage (Cook et al., 2006; Mulumba and Lal, 2008), ii) protection of soil against raindrop impact, reducing erosion rates (Blavet et al., 2009; Jordán et al., 2010; Sadeghi et al., 2015a), iii) decreased sediment and nutrient concentrations in runoff (Cerdà, 1998; Gholami et al., 2013; Poesen and Lavee, 1991), iv) decreased runoff generation rates and surface flow velocity by increasing roughness (Cerdà, 2001; Jordán et al., 2010), v) improved infiltration capacity (Jordán et al., 2010; Wang et al., 2014), vi) increased activity of some species of earthworms and crop performance (Wooldridge and Harris, 1991), vii) enhanced soil physical conditions such as soil structure and organic content (De Silva and Cook, 2003; Jordán et al., 2010; Karami et al., 2012), viii) reduced topsoil temperature for more optimum germination and root development (Dahiya et al., 2007; Riddle et al., 1996) and decreased evaporation (Uson and Cook, 1995), and xix) enhanced interactions with nutrients (Campiglia et al., 2014; Movahedi Naeni and Cook, 2000). Among the different types of mulching, straw mulch is considered one of the most effective in achieving the above-mentioned benefits (Blavet et al., 2009; Dahiya et al., 2007). In addition, it is easy to retrieve and transport and is relatively inexpensive if used at optimum application rates. Jordán et al. (2010) found that a mulching rate of 5 Mg ha⁻¹ yr⁻¹ was sufficient to make runoff flow and sediment concentration in runoff

negligible in a no-tilled Fluvisol under semiarid conditions in SW Spain. Similarly, Mulumba and Lal (2008) determined an optimum mulch rate of 4 Mg ha⁻¹ for increasing porosity and 8 Mg ha⁻¹ for enhancing available soil water capacity, moisture retention and aggregate stability. Although the beneficial effects of straw mulch are known, their quantification needs further research, especially within the context of rainfall-induced soil erosion in vineyards. Furthermore, the critical mulch rate needs to be established for site-specific soil and environment conditions (Mulumba and Lal, 2008). Jin et al. (2009) suggested that the relation between mulching rate and interrill soil detachment is not unique and can vary depending on rainfall intensity. They observed that sparse straw mulching on silt loam soils might favor soil loss compared to the bare soil under certain rainfall intensities (i.e. 65 mm h⁻¹). Given the importance of vineyards in the Mediterranean region, further research should be carried out to meet farmers' needs too. Indeed, it is also necessary to find the optimum mulch rate that entails reasonable expenses that farmers can afford. Therefore, the objective of this paper is to assess the effect of barley straw mulch on soil water erosion and water losses in a vineyard in a Mediterranean area affected by intensive erosion rates under simulated rainfall.

7.4 Materials and methods

7.4.1 Description of the study area

The study area consists in a 25-year-old vineyard located at El Celler del Roure in Les Alcusses de Moixent, within the Canyoles river watershed in the province of Valencia (La Costera District, eastern Spain) (38° 48' 30.33" N, 0° 48' 57.88" O). Wine making has a long tradition in this area which can be shown to go back to 400 BC in the nearby Iberian settlement of La Bastida of the Moixent municipality. The soil in inter-rows of the vineyard is managed by using a cultivator. The parent materials in this area are Cretaceous limestones and Tertiary Marly deposits that develop Typic Xerothent soils (Soil Survey Staff, 1998). The soils of the studied vineyards show a mean depth of 40 cm and are sandy, with 8% rock fragment content. The soils are characterized by low levels of soil organic matter (~1%) due to the millennia of agricultural use and soil disturbance (plowing), basic pH (8) and sandy soil textures in the foot-slope where the vineyards are located and, thus, the experiments were carried out. Slope angle was similar among the bare (average value of 3.3°) and straw (average value of 3.2°) plots. Climate is typical Mediterranean with 3–5 months of summer drought (June–September). Mean annual rainfall is about 350 mm yr⁻¹. Rainfall is distributed among autumn, winter and spring, with maximum peak rainfall intensities during the autumn season, when values higher than 200 mm day⁻¹ were recorded during the last 50 years. Mean annual temperature is about 13.8 °C while the hottest month (August) has average temperatures of about 23 °C. Extreme storm events occur in this area, which is located about 100 km from the Mediterranean Sea. Recurrent rainfall events of more than 100 mm day⁻¹ or 30 mm h⁻¹ are able to induce significant soil water erosion processes (Fig. 1).



Fig. 1. Two visual perspectives of rilling and gully processes (white arrows) caused by a 40 mm in 30 min thunderstorm occurred in mid-June 2015 in the study area.

7.4.2 Materials and instruments

The rainfall simulation experiments were carried out during July 2013 under very dry soil moisture contents ranging from 4.4 to 8.3% in the top soil (0–2 cm depth). A nozzle-type rainfall simulator was used to reproduce rain storms at 55 mm h^{-1} rainfall intensity for one hour on 40 paired circular plots of 0.24 m^2 . The paired plots were bare ('bare'; $n = 20$) and covered with straw ('straw'; $n = 20$). The rainfall simulator used was described by Cerdà et al. (1997), is effective in rugged terrain conditions, and has proven to give good results in semi-arid environments. Its main components consist of a nozzle, a structure to hold the nozzle, a pumping system, the connections with the water supply and a tarpaulin to protect the rainfall simulation from wind. By keeping the nozzle at about 2 m height over a plane surface, the wetted area was slightly larger than 1 m^2 . Therefore, to avoid border interference, only the 0.24 m^2 of the central part of the sprinkled area is recommended to be used for measurements. Readers are referred to Cerdà et al. (1997) and Iserloh et al. (2013) for further details (Fig. 2).

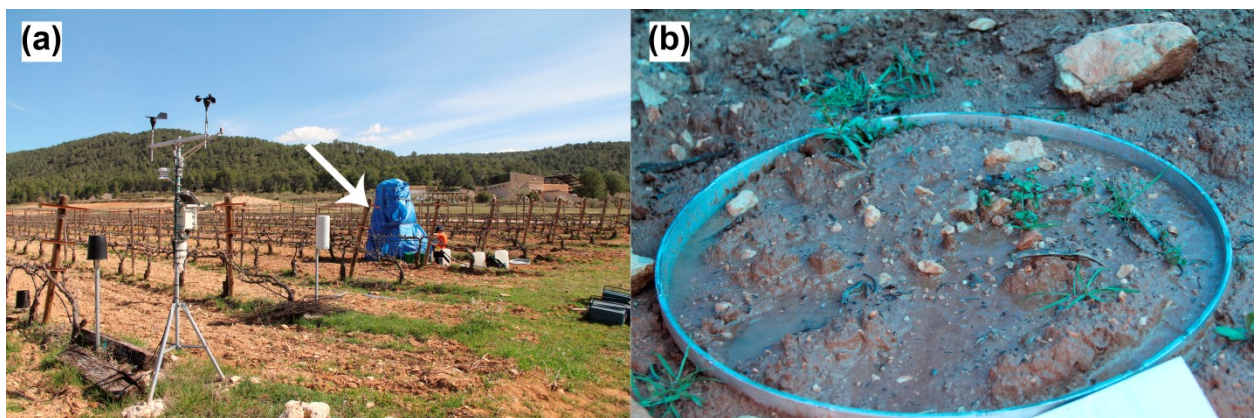


Fig. 2. A view of the study area, meteorological station and rainfall simulator (white arrow) (Fig. 2a) and of the circular plots of 0.24 m^2 (Fig. 2b) used for the measurement of the surface runoff and sediment yield.

Storms similar to the ones simulated have a return period of 10 years in the study area (Cerdà, 1996; Elías and Ruiz, 1977) and are those that contribute to the majority of surface runoff and soil erosion in the study area. The barley straw cover (75 g of straw per m²) was applied before the rainfall experiments at doses that provided a soil cover ranging from 48 to 90% of the soil surface area, with an average value of 62.2%. This application rate represents a reasonable and affordable expense for the farmers and, furthermore, it resulted in enough cover to protect the soil from weeds and water erosion. Barley straw was produced in nearby fields, located about 3.2 km from the study area, and transported by the farmers. The bare plots had a negligible surface cover that ranged from 0 to 4% with an average value of 0.8%. Soil surface cover was determined with vertical point frames of 10 sliding pins spaced about 5 cm apart (Hofmann et al., 1983). This led to a total of 100 points read within each 0.24 m² plot to discriminate between bare and straw covered soil. Erosion measurements are representative of splash and initial interill soil erosion processes and shed light about the impact on runoff initiation and sediment detachment. Detailed information on the distribution of rainfall parameters can be found in Cerdà (1996; 1997). Overland flow from the plot areas was measured at 1-min intervals. Every tenth 1-min runoff sample was collected for laboratory analysis in order to determine sediment concentration. Runoff rates and sediment concentration were used to calculate the sediment yield, total runoff, runoff coefficient, and erosion rates. Parameters such as time to ponding (Tp), time to runoff (Tr), time to runoff– time to ponding (Tr–Tp), time to runoff in outlet (Tro), and time to runoff in outlet–time to runoff (Tro–Tr) were also analyzed through statistical tests as described below. Tp was measured when about 40% of the surface showed ponds on flat or concave microsurfaces. To ensure uniformity, one person made these assessments for all the experiments. Tr–Tp and Tro–Tr are important to understand the mechanisms of Hortonian runoff generation because they represent the delay and the velocity of the overland flow. Soil moisture was measured by means of the desiccation of soil samples collected before the simulated rainfall experiment that were dried at 105 °C for 24 h. Soil samples were not taken directly from the plots, but next to them, to avoid affecting erosion processes within the plots. Sediment concentration in the runoff was calculated after the desiccation of the samples in the laboratory.

7.4.3 Data analysis

The normality of distribution and the homogeneity of variances of rainfall simulation data sets were checked using the Shapiro–Wilk's and the Levene's test, respectively. After the assumptions of normality or homogeneity of variances was rejected in most cases (Table 1), non-parametric tests were used.

Table 1 Results of the Shapiro-Wilk's test for normality of data distributions and Levene's test for homogeneity of variances of groups (bare soil, B; straw covered soil, S) for each variable considered: organic matter (OM), soil water content (SWC), cover, time to ponding (Tp), time to runoff (Tr), time to runoff - time to ponding (Tr-Tp), time to runoff in outlet (Tro), time to runoff in outlet - time to runoff (Tro-Tr), runoff coefficient, sediment concentration and sediment yield. Significance values (p) not shown are < 0.05.

Variable	Treatment	Shapiro-Wilk's test			Levene's test			
		Statistic	df	p	Statistic	df1	df2	p
OM (%)	B	0.940	20	0.244	2.328	1	38	0.135
	S	0.896	20					
SWC (%)	B	0.763	20		0.544	1	38	0.465
	S	0.868	20					
Cover (%)	B	0.705	20		27.370	1	38	
	S	0.883	20					
Tp (s)	B	0.937	20	0.206	27.663	1	38	
	S	0.956	20	0.463				
Tr (s)	B	0.963	20	0.606	27.994	1	38	
	S	0.898	20					
Tr-Tp (s)	B	0.961	20	0.564	10.427	1	38	
	S	0.878	20					
Tro (s)	B	0.933	20	0.175	35.141	1	38	
	S	0.906	20	0.052				
Tro-Tr (s)	B	0.949	20	0.356	7.562	1	38	
	S	0.969	20	0.729				
Runoff coefficient (%)	B	0.975	20	0.851	0.067	1	38	0.797
	S	0.971	20	0.771				
Sediment concentration (g L ⁻¹)	B	0.869	20		16.241	1	38	
	S	0.872	20					
Sediment yield (g)	B	0.872	20		14.677	1	38	
	S	0.901	20					

The Mann–Whitney U test was performed to assess differences between results (Tp, Tr, Tr–Tp, Tro, Tro–Tr, runoff coefficient, total runoff, sediment concentration in runoff, sediment yield, and soil erosion) coming from 'bare' and 'straw' treatments. All statistical analyses were computed with the SPSS 23.0 software package (IBM Corp., 2015).

7.5 Results

Soil cover was 0–4% in bare plots and from 48 to 90% in straw plots. No significant differences were found between median values of organic matter content (1.02 and 0.99%) and soil water content (4.69 and 4.85%) determined in the bare and straw plots, respectively (Table 2).

Table 2 Median value and range (between parentheses) of soil organic matter content (OM, 0-2 cm), soil water content (SWC, 0-2 cm) and soil cover (vegetation, litter and straw) for different treatments and results of the Mann-Whitney U test for differences between groups. N = 20 at each case.

Treatment	Soil cover (%)	OM (%)	SWC (%)
Bare soil	0 (0, 4)	1.02 (0.65, 1.90)	4.69 (3.56, 9.12)
Straw covered soil	59 (48, 90)	0.99 (0.65, 1.84)	4.85 (3.45, 9.58)
Mann-Whitney U, p	-	> 0.05	> 0.05

During rainfall simulations on bare plots, mean time required for ponding was 39 s, surface runoff appeared after 101.5 s and runoff in the outlet was observed after 208 s (median values; Table 3).

Table 3 Median value and range (between parentheses) of time to ponding (Tp), time to runoff (Tr), delay between the time to ponding and time to runoff (Tr-Tp), time to runoff in outlet (Tro) and delay between the time to runoff in outlet and time to runoff (Tro-Tr) for different treatments and results of the Mann-Whitney U test for differences between groups. N = 20 at each case.

Treatment	Tp (s)	Tr (s)	Tr-Tp (s)	Tro (s)	Tro-Tr (s)
Bare soil	39 (32, 51)	101.5 (56, 129)	61.5 (20, 87)	208 (145, 249)	107.5 (63, 134)
Straw covered soil	133.5 (90, 185)	208 (156, 314)	80 (50, 176)	346 (256, 514)	138 (74, 206)
Mann-Whitney U, p	0.000	0.000	0.000	0.000	0.001

The delayed times between ponding and runoff (61.5 s) and between runoff in outlet and runoff (107.5 s) were very short (Table 3). On the contrary, straw plots showed delayed ponding (133.5 s), surface runoff initiation (208 s) and runoff in outlet (346 s). The transition from ponding to runoff in the soil surface and then to runoff in outlet was delayed 80 and 138 s, respectively (Table 3). The relationships between time to ponding, runoff initiation and runoff in outlet are shown in Fig. 3.

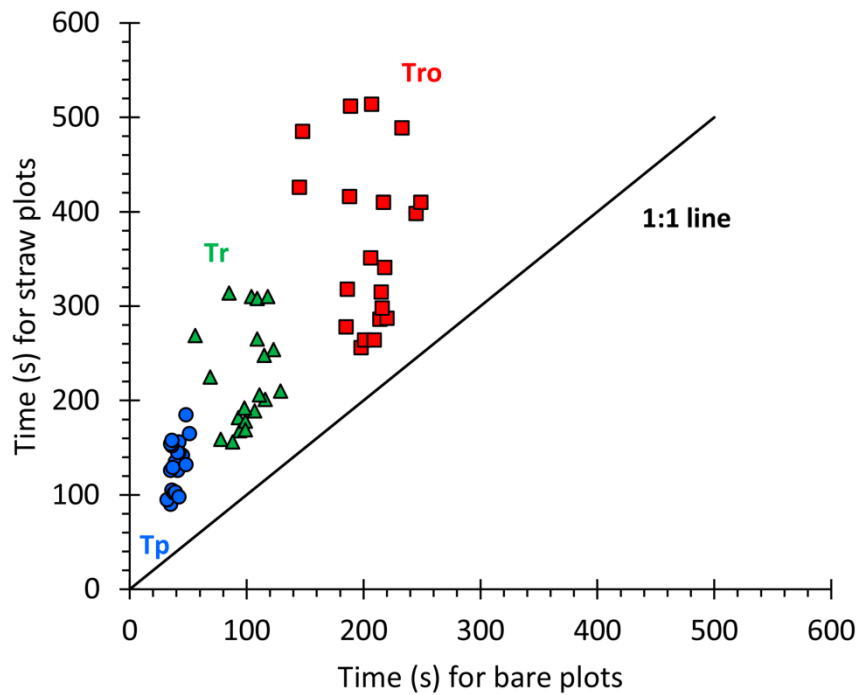


Fig. 3. Relations between time to ponding (T_p), time to runoff (T_r) and time to runoff in outlet (T_{ro}) for bare and straw plots. The effect of straw mulching in delaying the times of ponding, runoff initiation and runoff in outlet is shown by the fact that the points do not align following the 1:1 line.

The effectiveness of straw mulching in delaying the mentioned times is shown by the fact that the points corresponding to straw mulch do not follow the 1:1 line (Fig. 3). With respect to runoff coefficients and sediment concentrations, the application of straw mulch was revealed to be effective as well. Recorded runoff coefficients are linearly related to the proportion of soil cover in the straw plots (Fig. 4).

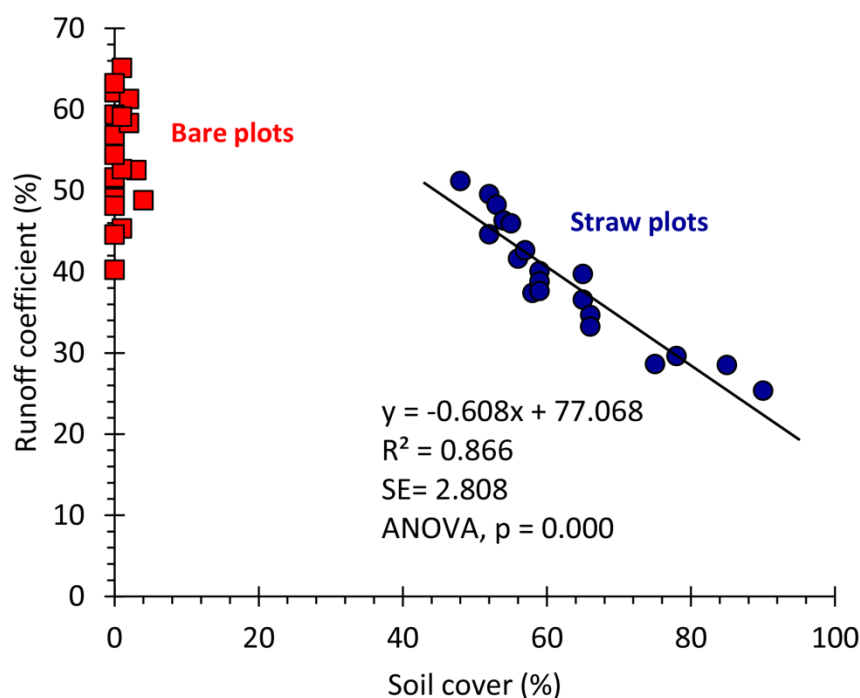


Fig. 4. Relation between soil cover and runoff coefficient for bare and straw plots. Regression line for runoff coefficient/soil cover in straw plots is shown. SE: standard error for estimate.

The application of straw mulch induced a reduction of the median value of total runoff from 7.23 to 5.40 L and of runoff coefficient from 52.59 to 39.27% (Table 4).

Table 4 Median value and range (between parentheses) of total runoff (R), runoff coefficient (Rc), sediment concentration in runoff (Sc), sediment yield (Sy) and soil erosion (Se) for different treatments and results of the Mann-Whitney U test for differences between groups. N = 20 at each case.

Treatment	R (L)	Rc (%)	Sc (g L ⁻¹)	Sy (g)	Se (Mg ha ⁻¹ h ⁻¹)
Bare soil	7.23 (5.53, 8.96)	52.59 (40.25, 65.15)	9.8 (7.8, 14.2)	70.34 (53.42, 111.98)	2.81 (2.14, 4.48)
Straw covered soil	5.40 (3.49, 7.04)	39.27 (25.35, 51.19)	3.0 (1.8, 5.9)	15.62 (6.13, 41.72)	0.63 (0.25, 1.67)
Mann- Whitney U, p	0.000	0.000	0.000	0.000	0.000

A significant decrease in the median value of sediment concentration in runoff flow from 9.8 to 3.0 g L⁻¹ was also achieved after the application of straw mulch (Table 4). Therefore, the median value of sediment yield decreased from 70.34 to 15.62 g with a consequent reduction of the soil erosion rates from 2.81 to 0.63 Mg ha⁻¹ h⁻¹. Sediment concentration in runoff increased with runoff coefficient in straw plots (Fig. 5). In contrast, no significant correlation was observed between these variables in bare plots.

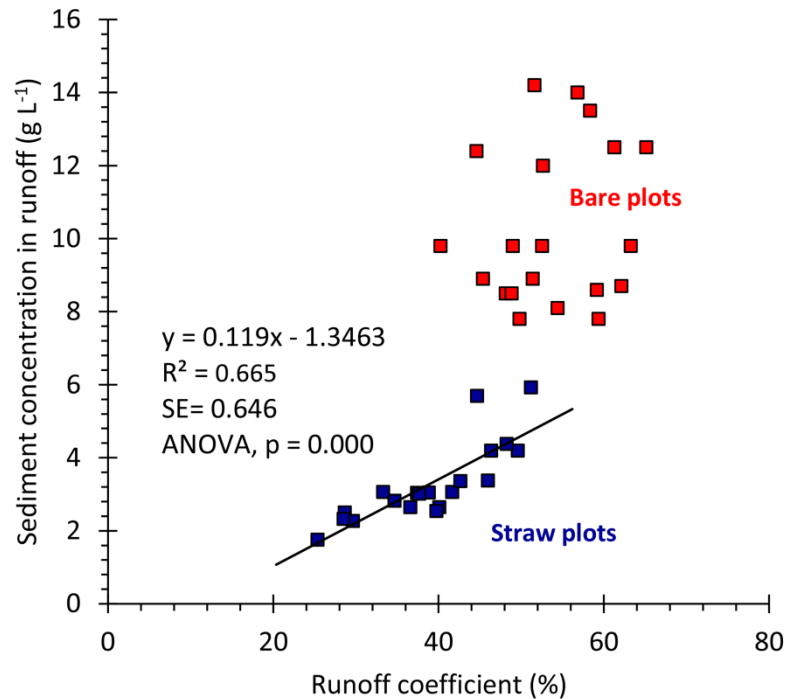


Fig. 5. Relation between sediment concentration in runoff and runoff coefficient from bare and straw plots. The regression line for straw plots is shown. SE: standard error for estimate.

7.6 Discussion

The experiments carried out at El Celler del Roure in the Les Alcusses of Moixent show that the use of a straw mulch cover (75 g m⁻²) contributes to positive effects on surface runoff generation and soil loss. No significant differences between organic matter (median values 1.02% and 0.99%, for bare and straw covered soil, respectively) and water contents (median values 4.69% and 4.85%, for bare and straw soils, respectively) from soils under different treatments were found because soil plots were covered by straw just a few hours before the beginning of the experiments (Table 2). Under low frequency-high magnitude rainfall events, such as the ones simulated here (return period of 10 years), and under dry summer conditions, the straw cover delayed the time to ponding (about 242.31%), time to runoff (about 104.93%), and time to runoff in outlet (about 66.35%). A dense straw cover protects soil from the direct impact of raindrops and contributes to increase surface roughness (Jordán et al., 2010), that, in turn, increases time to ponding and reduces sheet flow velocity, enhancing infiltration (Giménez and Govers, 2001; Guzha, 2004). The straw cover delayed the connectivity (Parsons et al., 2015) of the surface runoff, or at least the connectivity efficiency, as

both Tr–Tp and Tro–Tr were much delayed (Table 3). Runoff rates were significantly reduced by the straw cover (Table 4). The bare plots converted 52.59% of the rainfall into runoff, while this value decreased to 39.27% in the straw plots. This is a direct effect of the straw and other organic fibers that reduced raindrop impact on the soil, delayed runoff initiation and increased infiltration. These findings were also confirmed by Döring et al. (2005) and Edwards et al. (2000) in straw-mulched potato-cropped soils, García Moreno et al. (2013) in straw-mulched fruit orchards, Jordán et al. (2010) in straw mulched fallow soils, and Cerdà et al. (2015) in persimmon plantations. Our results confirm that straw is very efficient at reducing water losses under low frequency-high magnitude rainfall events, and we contribute to increased knowledge with the finding that straw cover can be a key factor in reducing the amount of runoff generated. Although straw mulching has significant effects on semiarid soil properties in the short (Jordán et al., 2010) and long term (García-Moreno et al., 2013), our results show that after a recent application (so that no effects on soil physical properties exist due to the straw application) a straw cover was efficient at reducing soil erosion risk. Although no significant relationship has been observed between the cover and runoff coefficient in bare plots, Fig. 4 shows a clear linear relationship between the straw cover and runoff coefficient for soil plots treated with straw. However, a non-answered question emerges from our results: is a lighter straw cover as efficient? The answer will require a new experiment as little has been done to find the most efficient straw mulch cover to reduce expenses. The total amount of straw applied was about 750 kg ha⁻¹ that resulted in an average cover of 62.2% in our experiments. This cost about 155 € ha⁻¹: 0.08 € kg⁻¹ to acquire 37 bales that were about 20 kg each; 0.06 € kg⁻¹ for transport and 0.07 € kg⁻¹ to apply the straw. A reduction in the straw cover would reduce the expenses, but would it also be less efficient at controlling erosion and water losses? Therefore, there is a need to find a balance between the cost and the efficiency of the straw mulch cover. In this regard, there are other researchers who studied the efficiency of mulches on fire affected land and found that the mulch cover of mulches does not need to be high to be efficient (Fernández et al., 2012; Prats et al., 2014; Robichaud et al., 2013a; 2013b). In these cases, straw mulch was confirmed to have a positive effect on water quality (Faucette et al., 2007) and the mitigation of wind-induced soil erosion (Nelson, 2002). Low mulch cover was also found to be efficient by researchers who dealt with road embankments protection (Lee et al., 2013), afforested land (Jiménez et al., 2013) and agricultural soils (Jordán et al., 2010). In addition, García-Moreno et al. (2013) reported that excessive mulch rates (9–12 Mg ha⁻¹ yr.⁻¹) may enhance runoff rates and soil erosion risk due to the development of soil water repellency. Furthermore, there is a need to determine which type mulching is most efficient on agriculture land, on which research has been focused over the last decade (Liu et al., 2012; García-Moreno et al., 2013). New materials are also being used in comparison with straw or in combination with it (Fernández and Vega, 2014; Mahmoud and Abd El-Kader, 2014; Moreno-Ramón et al., 2014; Tejada and Benítez, 2014). Moreover, it is necessary to investigate the scale effect, as straw reduces the connectivity of the runoff and, therefore, this can be the key question to determine the amount of straw to be used and the way it should be applied at different scales (Sadeghi et al., 2015b). Finally, we know that straw will have

long-term impacts on soils characteristics (Edwards et al., 2000; Tejada and Benítez, 2014), soil microbial biomass (Tu et al., 2006) and macro invertebrates in vineyards (Thomson and Hoffmann, 2007), and this will also contribute to improved soil quality and reduced soil erosion rates. In our study, the most important impact of straw mulch was reduction of the median value of sediment concentration in runoff from 9.8 to 3.0 g L⁻¹ (Table 4) in the bare plots versus straw plots. Again, there was a clear relationship between straw cover and sediment concentration that confirms that straw mulch can be a sustainable management strategy in vineyards located in semiarid areas and can immediately reduce the high erosion rates of vineyards. However, more research should be carried out in other vineyards to better understand whether a standardized straw mulch application rate exists, and consequently the ideal mulch cover that can lead to a balance between tolerable soil erosion rates and expenses. To this purpose, researchers should be at farmers' disposal to find out the best possible sustainable solutions. We stress here that the effect of the straw on runoff and soil loss control was immediate. This then is a quick and efficient strategy to reduce soil losses and increase infiltration. The straw should also have a positive effect on soil quality as it will improve soil organic matter, soil biological activity and infiltration rates (Brevik, 2009b; Lieskovský and Kenderessy, 2014; Mahmoud and Abd El-Kader, 2015; Mwangi et al., 2015).

7.7 Conclusions

This research demonstrated that under low frequency-high magnitude rainfall events in Mediterranean vineyards a barley straw cover of about 59% (median value) applied at a rate of 750 kg ha⁻¹ resulted in delayed times to ponding, runoff and runoff in outlet. In addition, it contributed to a reduction of i) surface runoff rates from 52.59% (bare) to 39.27% (straw), ii) sediment concentration in runoff from 9.8 to 3.0 g L⁻¹ and iii) soil loss rates from 2.81 to 0.63 Mg ha⁻¹ h⁻¹. This reduction of soil and water losses was reached immediately after the straw application. Barley did not grow in the same field where our plots were located, rather it was produced in the nearby fields, mulched and then transported. This entailed a cost that farmers had to bear, however, based on the results obtained, straw mulch was confirmed to be a relatively inexpensive and effective soil conservation practice that can be adopted by winegrowers to reduce the high erosion rates in semiarid areas.

7.8 References

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8 Conclusions

This thesis proposed an integrated approach, by means of innovative remote-sensing technologies, especially those relying on low-cost and fast techniques, field activities, and quantitative analyses to the investigation of soil erosion processes caused by water in agricultural landscapes, especially Mediterranean vineyards. Furthermore, this thesis also suggested mulching with vegetative residues as an effective soil management technique to mitigate soil and water losses in the before-mentioned environments.

The effects of the triggering factors on soil water erosion on Mediterranean vineyards were clarified, by identifying and confirming some general trends that can help to fill the gap in knowledge of these processes. The importance of a continuous and prolonged monitoring of soil erosion processes, as long as it is based on standardized procedures to allow the comparison of data derived from different study areas, was stressed.

Both experimental plots under simulated rainfall and the structure-from-motion (SfM) technique, which was applied to obtain high-resolution DEMs, were applied and compared with each other to quantify soil losses on a Mediterranean vineyard. SfM data were derived from one standalone digital reflex camera and a smartphone built-in camera. Soil loss data estimated with the surface change-based method were comparable to those derived from the rainfall simulation method, as long as the sediment connectivity (Index of Connectivity - IC) within the plot was taken into account. This proved the fact that high-resolution topography, derived from SfM, revealed to be essential in the sediment connectivity analysis and, consequently, in the estimation of eroded materials, if compared them to those derived from another methodology such as the rainfall simulation. In addition, the fact that smartphones built-in cameras produced as much satisfying results as those derived from standalone digital reflex cameras is undoubtedly a high value added. Nowadays, smartphones are commonly available for anyone, from farmers to researchers, and will become increasingly important for fast and cheap post-event analyses, as long as they are provided with a high-resolution camera. This was also proved in the case study of a bank erosion affecting an agricultural channel in a land reclamation territory, where high-resolution DEMs derived from SfM were compared to a DEM derived from lidar (TLS) considered as a benchmark. Also in this case, SfM data were derived from standalone digital reflex cameras and a smartphone built-in camera. The results highlighted the fact that the SfM technique was a useful approach for the production of topographic data for the qualitative and quantitative analyses of geomorphic processes, and for automatic features extraction, with results that were comparable with those obtained by lidar technology.

Still, high-resolution DEMs revealed to be essential for a first high-resolution hydro-geomorphological analysis of Mediterranean terraced vineyards. In this case, high-resolution DEMs were derived only from lidar technology and were used as basis for computing a topographic index (Relative Path Impact Index - RPII), which was able to correctly identify terraced failures and road erosions. Thanks to this type of support, it was possible to simulate different drainage ditch systems, identifying the optimal solution. The proposed approach results to be helpful in scheduling a suitable and

possible planning to mitigate the consequences of the anthropogenic alterations induced by the terraces structures and agricultural roads. Still to help to to mitigate the problem of soil erosion by water, mulching practice with vegetative residues was deeply studied and experimentally tested in a Mediterranean vineyard. The results demonstrated that under low frequency-high magnitude rainfall events a barley straw cover resulted in delayed times to ponding, runoff and runoff in outlet. In addition, it contributed to a significant reduction of surface runoff rates, sediment concentration in runoff, and soil erosion rates. The effectiveness of mulching with vegetative residues for reducing soil and water losses was also proved in different environments across the world.

In conclusion, the presented methodologies and results may be of assistance to both farmers and land managers by providing them with evidence-based means for implementing more sustainable soil management practices. High-resolution topographic data, especially those obtained from cheaper and faster remote-sensing technologies may provide new insights into the knowledge of erosion processes that affect agricultural lands and can serve as basis for the definition of strategies for the conservation of the environment. We are just at the beginning of geomorphological applications of SfM and, although it has still limits in highly vegetated areas, there will be several opportunities and challenges in using the SfM technique, especially with smartphones, thanks to the increasing development of computer vision technologies and digital camera sensors, which makes the process of taking good pictures quite easy. A farmer would require few hours of training to learn how to take good pictures of a specific case study located in its own land, whether it is a rill, a gully or an agricultural channel affected by evident bank erosion. Afterwards, he would be completely independent during the whole field survey, and then he could send the pictures taken to a researcher for further analyses. By doing this, he could easily keep monitoring some of the erosion processes that occur in his land and the researcher could provide him quantitative information about net erosion and deposition rates. Yet from this perspective, additional rainfall simulation experiments, coupled with SfM surveys, should be carried out in other study areas, characterized by different types of soil, to better understand the effects of the soil spatial variability. Furthermore, this thesis also aims to stress the importance of adopting proper measures to protect the soil, such as mulching, and monitoring soil erosion by field measurements over longer periods. Further research is needed to answer some still open questions about the most appropriate types of mulches, application rate and cover and economic feasibility, by taking also into consideration that mulches, other than reducing soil and water losses, can contribute to other important environmental functions.

The viticulture sector requires attention to reduce its impact on soil erosion problems and terraced landscapes need to be maintained, well managed, and protected. These actions can help to overcome the critical issues related to erosion risk and landslides and to make this important cultivation more sustainable.

9 References

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