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Conventional and organic farming: Soil erosion and conservation potential for row crop cultivation

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A B S T R A C T

The cultivation of row crops on mountainous farmland can generate severe soil erosion due to low ground cover, especially in the early growth stages. Organic farming, due to the absence of herbicides, can support the development of weeds and increase the ground cover compared to conventional farming. However, the benefits towards soil erosion, and the conservation potential of organic farming systems, in terms of herbicide application and weed growth, have not been investigated. Aim of this study was to identify how conventional and organic farming influence the erosion rate of soil, due to row crops cultivated on mountainous farmland in the presence or absence of agricultural chemicals. We measured multiple vegetation parameters of crops and weeds of conventional and organic farms cultivated with bean, potato, radish, and cabbage in a mountainous watershed in South Korea. We simulated the long-term soil erosion rates with the Revised Universal Soil Loss Equation (RUSLE) by using 13 years of recorded rainfall data in order to account for the temporal variability of monsoonal rainfall. We determined average annual erosion rates for the study area to be between 30.6 t ha^{-1} yr^{-1} and 54.8 t ha^{-1} yr^{-1}, with maximum values when radish was grown, due to the shorter growing period, higher soil disturbance at harvest, and low amounts of residue. Organic farming reduced soil loss for radish by 18% as a result of a high weed biomass density and cover at the end of the growing season. For potato, organic farming increased soil loss by 25% due to a reduced crop coverage, which is suspected to have been a consequence of crop–weed competition or increased herbivory associated with the absence of agricultural chemicals. Our results demonstrate that organic farming can potentially decrease the soil erosion risk for row crops because it supports weed development in the furrows, but it can also produce higher erosion rates when crop yields are reduced as a consequence, outweighing the protective effect of the weeds. However, the simulated erosion rates under both farming systems exceeded by far any tolerable soil loss. We conclude that organic farming alone cannot be used to effectively control erosion, and that both farming systems require additional conservation measures, such as winter cover crops and residue mulching, to sufficiently prevent soil loss for row crop cultivation.

1. Introduction

Intensive agriculture in mountainous landscapes can cause high soil erosion with negative impacts on farmland productivity and sustainability, as well as downstream water quality. The severity of erosion is strongly affected by the specific nature of cultivation within such areas. Vegetation above the surface protects the soil from the impact of raindrops and runoff, while the root system contributes to the internal stabilization of the soil (Morgan, 2005). Therefore, the crop type and management system applied by farmers plays a critical role in erosion control on steep agricultural land. The cultivation of row crops generally results in more serious erosion problems due to the high ratio of exposed ground, especially in the early growth stages, and due to the need for seedbed preparations (Morgan, 2005). More extensive groundcover can be provided by weeds, helping to further reduce soil erosion (Bennett, 1939), and Brock (1982) reported that weed control by the use of herbicides significantly increases soil erosion rates. Several other studies have also shown that a developed weed

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cover can effectively reduce soil loss compared to manual weeding or the application of herbicides (Afandi et al., 2002; Blavet et al., 2009; García-Orenes et al., 2009; Weil, 1982). Environmentally friendly farming systems rely on the minimization of chemical use, such as herbicides and pesticides, and can therefore play an important role in erosion control. Especially for row crops, the percentage of ground cover can be altered by weed growth, which could provide additional soil protection on organic farmland. Nevertheless, organic farming can also result in reduced crop yields due to crop–weed competition and herbivory.

Several authors have already described the potential effects of organic versus conventional farming on soil erosion control (Erhart and Hartl, 2010; Goh, 2011; Gomiero et al., 2011; Lotter et al., 2003). However, the individual studies used different methodologies to assess the erosion potential, and they observed very different impacts of the two farming systems. Lockeretz et al. (1981) calculated potential soil loss of organic and conventional farms by using the Universal Soil Loss Equation (Wischmeier and Smith, 1978) and found about one-third less erosion where organic farming was practiced, due to the different crop rotation systems in place. Reganold et al. (1987) investigated the long-term effects by comparing erosion measurements and topsoil thickness of two farms, and found an almost four times lower erosion on the organic farm as a result of different crop rotations and less tillage operations. Fleming et al. (1997) used soil samples from organic and conventional fields and calculated the soil erodibility, finding that organic farming could potentially reduce erosion for some soils. Also Siegrist et al. (1998) found, in a long-term field experiment, that organic farming increased the aggregate stability of the soil. However, organic farming did not sufficiently reduce soil erosion in their study. Also during a long-term field experiment, Eltun et al. (2002) observed lower erosion on plots with organic arable crops, but higher erosion on plots with organic forage crops. Auerswald et al. (2003) investigated the soil erosion potential also by using the Universal Soil Loss Equation, based on cropping statistics of conventional and organic farms, finding a slightly lower soil loss where organic farming was practiced, but concluding that there was no general effect, due to the large variability within both farming systems. Pacini et al. (2003) modeled soil erosion using GLEAMS (Leonard et al., 1987) on different farms, and they found that organic farming dramatically increased erosion compared to conventional farming, because of different crops and more intense tillage operations. In another study using rainfall simulations, Kuhn et al. (2012) reported lower erosion rates from organic compared to conventional soils.

Although the erosion control potential of organic farming could be identified in many of these studies, a general conclusion of the impact of the farming systems can still not be drawn. Soil stabilization might be an effect of long-term organic farming and may not apply for recently established organic farms. Large differences between both farming systems were primarily observed where farms used different crops and tillage operations. The effects of weed development associated with the two farming systems for the same crop as a specific consequence of the application or absence of agricultural chemicals has not been investigated.

The aim of this study was to identify the erosion control potential of conventional and organic farming systems on mountainous farmland in South Korea, which is highly susceptible to soil erosion due to the steep slopes and the cultivation of row crops. In the Kangwon Province in the northeast of South Korea, for instance, primarily radish and cabbage are cultivated (Kim et al., 2007), having short growing periods, thus leaving the farmland with low protection against rainfall and runoff (Y. Park et al., 2010). Conventional farmland management in South Korea is characterized by an intensive use of agricultural chemicals, including herbicides and pesticides (Kang and Kim, 2000; Kim and Kim, 2004). However, environmentally friendly farming systems (organic farming and no-chemical farming), which do not use agricultural chemicals are becoming more popular (Choo and Jamal, 2009; Kim et al., 2001). Due to governmental support, the number of organic farms in South Korea has strongly increased within recent years (Kim and Kim, 2004; Kim et al., 2012). The effect of different row crops on soil erosion in Korea has previously been studied over many years by the National Academy of Agricultural Science (NAAS) (Jung et al., 2003). Other studies investigated the effect of planting time and vegetation cover (Cho et al., 2010) or the erosion control potential of cover crop cultivation together with row crops (Kim et al., 2008; Ryu et al., 2010), but the impact of vegetation development associated with conventional and organic farming needs further investigation.

We formulated the following hypotheses: (1) organic farming increases weed coverage for row crops due to the absence of herbicides, and (2) the protective effects of weeds control soil erosion for organic farming. To test the hypotheses, we measured multiple vegetation parameters of four major row crops and the associated weeds on both conventional and organic farms in a watershed in the Kangwon Province of South Korea, and we determined the potential resultant soil loss amounts using the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997). To better understand the long-term effects of the farming systems, we considered the regional climate development, as soil loss rates associated with crops and farming systems are highly variable depending on the planting and harvest times and the occurrence of erosive rainstorm events. The majority of annual rainfall on the Korean peninsula is concentrated in the summer monsoon between June and August (J. Park et al., 2010) and hence, the annual soil erosion rate may be dependent on only a few extreme events. Choi et al. (2008) observed an intensification of extreme rainfall events in Korea, and found a strong change in temporal distribution over the years, and Kim et al. (2009) reported a large variability in precipitation during the monsoon season. Hence, the variation in rainfall patterns and intensities can therefore result in highly variable erosion rates for similar crops and farming systems between different years. The severity of erosion is also controlled by other factors, such as the level of soil disturbance during harvest and the amount of residue remaining on the field (Toy et al., 2002). Therefore, we used long-term weather station data to account for the variability of monsoonal rainstorm events, and we simulated different scenarios to include variable planting dates and harvest operations for the different row crops and farming systems.

2. Materials and methods

2.1. Study area

This study was conducted in the Haean-Myeon catchment in the Kangwon Province of South Korea (Fig. 1). The catchment is located within the watershed of the Soyang Lake, which is the largest reservoir in South Korea (Kim et al., 2000). The reservoir is affected by high amounts of nutrients from the Soyang River largely due to eroded soils from agricultural areas within the watershed (Kim and Jung, 2007; J. Park et al., 2010). The Haean catchment is a major agricultural hotspot area, which substantially affects the trophic state of the reservoir (J. Park et al., 2010). The total area of the catchment is 64 km^2, of which 58% are covered with forest and 30% by agricultural lands (22% dryland fields, 8% rice paddies). The remaining 12% consist of residential areas and seminatural areas, which include grassland, field margins, riparian areas, and farm roads. The topography of the study area is characterized by flat areas and moderately steep slopes in the center of the catchment, and high slopes at the forest edges. The terrain is highly complex with a variety of different hillslopes and flow directions. The soils of the Haean catchment are dominated by *Cambisols* formed from weathered granite. They are highly influenced by human disturbances. Especially dryland fields were modified by the replenishment of excavated material from nearby mountain slopes in order to compensate for annual erosion losses (J. Park et al., 2010). The average annual precipitation in the Haean catchment is 1599 mm (1999–2011), of which more than 65% are concentrated in July, August, and September.

For this study, 25 field sites were selected, which included the four major dryland row crops, bean (*Glycine max*), potato (*Solanum tuberosum*), radish (*Raphanus sativus*), and cabbage (*Brassica rapa* and

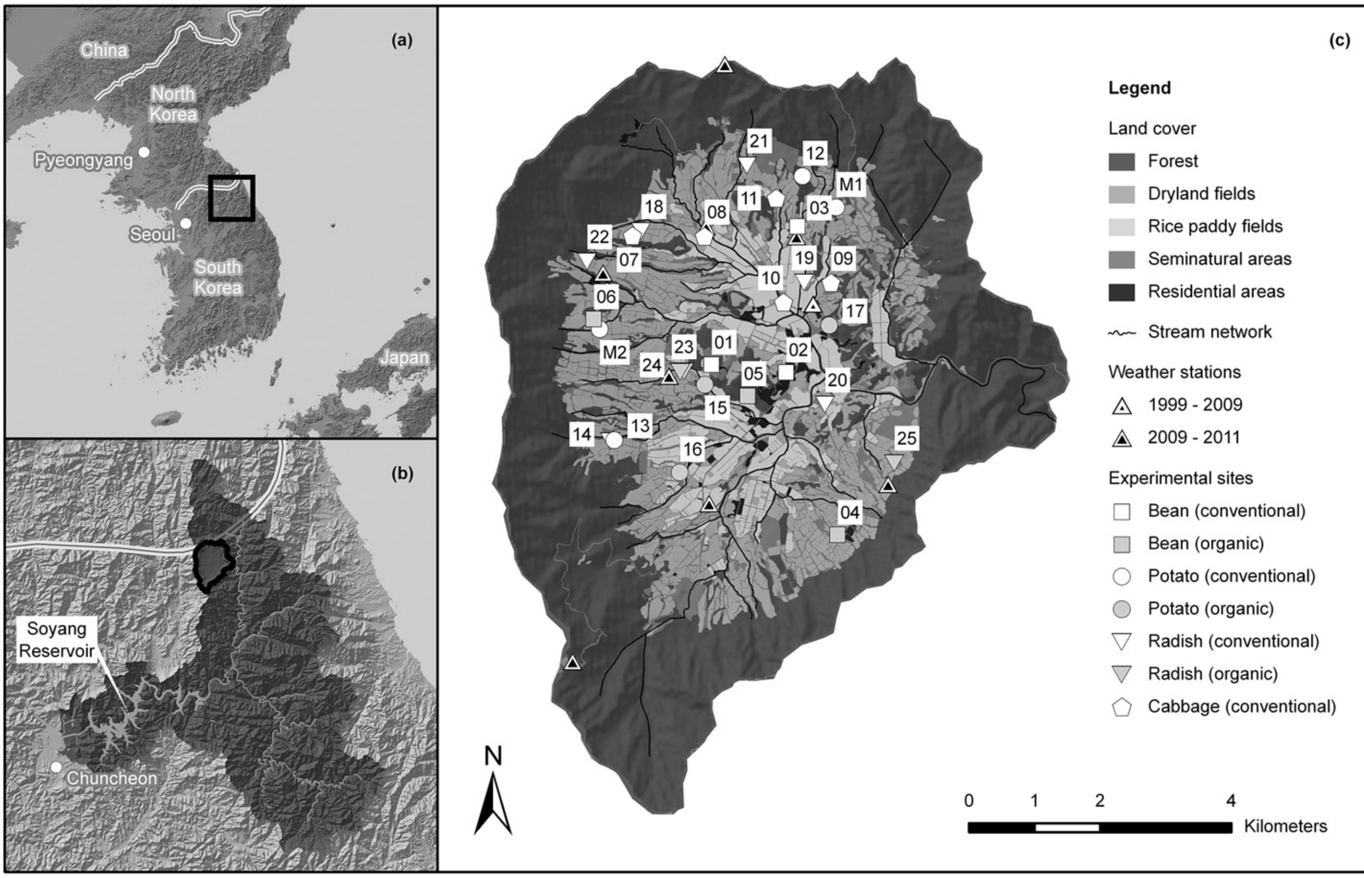

Fig. 1. Location of the study area (Haean-Myeon catchment) on the Korean peninsula (a) and within the watershed of the Soyang Lake (b) with the locations of the weather stations and 25 experimental sites (01 to 25) selected for this study (c). The sites M1 and M2 indicate the position of two additional fields where soil loss was measured in 2010, which was used to evaluate model plausibility.

Brassica oleracea) cultivated by conventional and organic farmers in this region (Fig. 1c). Organic cabbage fields were not available for this study. Therefore, we could only differentiate between organic and conventional management for bean, potato, and radish. For both potato and bean, six fields were selected, each with three conventional and three organic sites. For radish, five conventional and three organic field sites were used, and for cabbage only five conventional sites were available. The field sites were distributed over the entire Haean catchment, some of which were located in the center, and some upon steep slopes near the forest edges. They represented the variety of different field sizes, hillslopes, and soil conditions of the mountainous agricultural land in this region. For all crops investigated, the fields were cultivated by a ridge–furrow system covered with plastic film (plastic mulch). The spacing between two ridges was approximately 70 cm, and the ridge height was about 15 cm. The plastic film covered approximately 50% of the soil surface.

2.2. Erosion simulation with the Revised Universal Soil Loss Equation

We used the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997) to calculate the average annual soil erosion rates for the 25 selected field sites, and to compare the effects of the four row crops and the applied farming systems. RUSLE is an empirical soil erosion model founded on the Universal Soil Loss Equation (USLE) described by Wischmeier and Smith (1978) (Renard et al., 1997). The factor approach of the model allowed us to identify and directly compare the effects of crop management on soil erosion. RUSLE provides the possibility to enter multiple parameters that can be measured in the field, to describe the crop conditions and surface properties associated with a specific management practice. RUSLE calculates the average annual erosion from a given field slope as follows (Renard et al., 1997):

$$A = R \cdot K \cdot L \cdot S \cdot C \cdot P \qquad (1)$$

where A is the average annual soil loss (t ha^{-1} yr^{-1}), R the rainfall and runoff erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}), K the soil erodibility factor (t h MJ^{-1} mm^{-1}), L and S the slope length and slope steepness factors (−), C the cover-management factor (−), and P the support practice factor (−).

Both the USLE and RUSLE model have been widely applied in many different regions all over the world (Onori et al., 2006; Tiwari et al., 2000). However, they were originally developed for the application within the United States (Kinnell, 2010; Sonneveld and Nearing, 2003), and thus, the applicability should be limited to the range of experimental conditions from which they were derived (Renard and Freimund, 1994). They were designed to estimate the average annual soil erosion for agricultural land with medium textured soils and field slopes between 3% and 18% that are no longer than 122 m (Kinnell, 2010; Rapp, 1994; Risse et al., 1993). However, when appropriate values for the individual factors are determined, the models can be also transferred to other locations outside of the United States (Foster et al., 1982; Renard and Freimund, 1994). Nevertheless, especially in mountainous landscapes, such as Korean watersheds with highly complex topographies, steep slopes, and a strong variability of rainfall, the applicability of RUSLE can be problematic (Millward and Mersey, 1999). Therefore, special care should be taken when determining the individual empirical factors for specific local conditions in order to ensure that they lie within the scope of the model design and produce reliable results (Amore et al., 2004).

In the following sections, we describe the calculation routines and data collection procedures for each of the individual RUSLE factors, and we discuss their applicability for the field sites in the Haean catchment.

2.2.1. Rainfall and runoff erosivity factor (R)

The *R*-factor quantifies the effect of raindrop impact on soil erosion and reflects the amount and rate of runoff associated with the rainfall (Renard et al., 1997; Wischmeier and Smith, 1978). The *R*-factor for a given year is computed from recorded weather station data sets by adding the total kinetic energy multiplied by the maximum 30-minute intensity of erosive rainstorm events (EI_{30}) within that year (Renard et al., 2011). The total energy of a rainstorm event is the sum of the rainfall energies of all individual recording time intervals. The energy for each time interval is the product of the unit energy and the rainfall amount within that interval. The unit energy is calculated as follows (Brown and Foster, 1987):

$$e = 0.29 \cdot [1 - 0.72 \cdot \exp(-0.05 \cdot i)] \tag{2}$$

where *e* is the unit energy (MJ ha^{-1} mm^{-1}), and *i* the rainfall intensity (mm h^{-1}) for each time interval.

The *R*-factor calculation procedure was developed from a large data base of weather stations throughout the United States with variable climatic conditions (Renard and Ferreira, 1993; Renard et al., 1994). However, it has been argued that an adequate representation of other regions is questionable (Schönbrodt-Stitt et al., 2013), for example, that the rainfall energy–intensity relationship given by Eq. (2) may be associated with uncertainties when applied to other areas, for example, in tropical regions (Hoyos et al., 2005; Van Dijk et al., 2002). Van Dijk et al. (2002) evaluated a new general relationship derived from different locations all over the world and found that the differences to the EI_{30} approach are modest, especially when compared to the natural variation of rainstorms and the uncertainty associated with other RUSLE factors. Moreover, they concluded that a recalculation of the RUSLE *R*-factor seems not warranted (Van Dijk et al., 2002). Also in other studies, it has been found that the EI_{30} index produced better results than other erosivity indices (Hoyos et al., 2005; Ruppenthal et al., 1996). Therefore, we could assume that the above relationship of Brown and Foster (1987) was an appropriate approach in our study.

The most accurate values for the *R*-factor are obtained when using local high resolution rainfall intensity records (Kinnell, 2010; Renard and Freimund, 1994). When such data sets are not available, alternative approaches (e.g., Arnoldus, 1977; Hudson, 1995; Lal, 1976) can be applied using daily and monthly rainfall records which can also provide reasonable estimates (Renard and Freimund, 1994; Schönbrodt-Stitt et al., 2013). In this study, two types of data sets of high resolution weather station records were available from the Haean catchment, providing a total of 13 years of precipitation and temperature data. The first data set was derived from a weather station located in the center of the Haean catchment (Fig. 1c), which recorded precipitation and temperature from January 1999 to May 2009 with 1 hour resolution. The second data set was derived from nine weather stations installed in May 2009 in Haean (Fig. 1c) recording weather data with 30 minute resolution until December 2011. Due to technical problems, only four of the weather stations in 2010, and only two weather stations in 2011 could be used for the *R*-factor calculations. We developed an algorithm by using the R programming language that automatically identified the erosive events and calculated *R*-factor and rainfall erosivity for every half-month period from the weather station data sets. According to Renard et al. (1997), small rain showers with less than 12.7 mm of rain and rainfall intensities of less 25.4 mm h^{-1} were excluded from the calculations, and periods with six hours of less than 1.27 mm of rainfall were used to divide one rain event into two (Meusburger et al., 2012). Precipitation occurring at temperatures of less than or equal to 0.0 °C was considered as snow, or solid precipitation, and hence, was excluded from the calculations (Leek and Olsen, 2000; Meusburger et al., 2012). By using this algorithm (performed by RStudio ver. 0.95.258), we calculated the *R*-factors for the years 1999 to 2011 and the temporal distribution of the rainfall erosivity in half-month periods, which was required for the calculation of the *C*-factor (see Section 2.2.4). Subsequently, we corrected the *R*-factors for 1999 to 2009 data sets, as the maximum 30-minute intensity was underestimated due to the lower resolution. Therefore, we calculated the rainfall erosivity for the 2009 to 2011 data sets, first by using the original 30 minute resolution, and second by using aggregated 1 hour resolution data sets. We correlated the calculated erosivity values and derived the slope of the regression line, which was used as a correction factor for the data sets of 1999 to 2009. Finally, the average annual *R*-factor was calculated as the mean of the 13 years individual *R*-factors.

An important issue that needs to be considered when the *R*-factor is applied to a mountainous landscape is the influence of the topography that indirectly controls the spatial variability of rainfall erosivity (Schönbrodt-Stitt et al., 2013; Wang et al., 2002b). The applicability of the *R*-factor in highly mountainous areas may be problematic, because it has been shown in different studies that there is a strong relationship between rainfall erosivity and elevation (Hoyos et al., 2005; Millward and Mersey, 1999; Schönbrodt-Stitt et al., 2013; Van Dijk et al., 2002). Also in the Haean catchment, we have observed differences in precipitation due to elevation. However, strong differences occurred especially between the agricultural areas in the center and the high forested mountain peaks along the catchment boundary. In this study, we focused on the agricultural field sites only, which were located below the high mountain slopes within the catchment (Fig. 1c) with a maximum elevation difference of 240 m. The differences in rainfall between the agricultural areas in the Haean catchment were moderate, and therefore the use of a single *R*-factor for all 25 field sites was adequate.

2.2.2. Soil erodibility factor (K)

The *K*-factor represents the effects of soil properties and soil profile characteristics on soil erosion (Renard et al., 2011). It is usually obtained from the soil erodibility nomograph of Wischmeier et al. (1971) as a function of the soil texture (content of clay, silt, sand, and very fine sand), the organic matter content, and the soil structure and permeability. An algebraic approximation of the nomograph is given by the following equation for those cases where the silt content of the soil does not exceed 70% (López-Vicente et al., 2008, modified after Renard et al., 1997):

$$K = 0.1317 \cdot \left[0.00021 \cdot (12 - OM) \cdot M^{1.14} + 3.25 \cdot (s - 2) + 2.5 \cdot (p - 3)\right] / 100 \tag{3}$$

where *K* is the soil erodibility factor (t h MJ^{-1} mm^{-1}), *OM* the content of organic matter (%), *M* the product of the primary particle size fractions (−), *s* the soil structure code (−), and *p* the soil permeability code (−). The factor 0.1317 is used for unit conversion to SI units (Foster et al., 1981). *M* is calculated as (modified after Renard et al., 1997):

$$M = (silt + vfs) \cdot (silt + sand) \tag{4}$$

where *silt* is the percentage of silt (0.002–0.05 mm) (%), *vfs* the percentage of very fine sand (0.05–0.1 mm) (%), and *sand* the percentage of sand (0.05–2 mm) (%).

The erodibility nomograph was originally developed from erosion plots in the United States as the ratio of measured soil loss and erosivity (Kinnell, 2007; Rapp, 1994), and may therefore not always be applicable for soils in other regions in the world (Onori et al., 2006; Renard and Ferreira, 1993). It has been derived for medium textured soils with low aggregate stability and may produce large uncertainties for other soils, especially with high clay contents (Hammad et al., 2005;

Römkens et al., 1977; Wang et al., 2013). Zhang et al. (2004, 2008) reported strong overestimations of the K-factor when applying the nomograph to soils in China. Also other studies outside of the United States have shown overpredictions (Hammad et al., 2005; Victoria et al., 2001). Therefore, several other equations have been developed to calculate the K-factor (Kinnell, 2010), for example by El-Swaify and Dangler (1976) for tropical soils, or an equation based on the average particle diameter developed from a global data set (Renard and Ferreira, 1993; Renard et al., 1997) that has been already used in different studies (e.g., Fu et al., 2005; Onori et al., 2006; Yue-qing et al., 2009). However, the latter global relationship gives estimates for the K-factor based on limited data only and produces less accurate values than those obtained from regression data such as the erodibility nomograph (Renard et al., 1997). For the agricultural soils in the Haean catchment, which were characterized by low clay contents and a poor soil structure, we assumed that their properties were within to the range for which the nomograph has been developed. When comparing the K-factors for those soils computed with the nomograph to the K-factors calculated with the global relationship after Renard et al. (1997), we found only small differences of about 20%. We could therefore presume that the nomograph produced reliable results and might be more accurate for our study sites than other equations for the K-factor mentioned above.

To obtain the parameters required for Eqs. (3) and (4), we took samples of top soils (0 to 30 cm depth) of the 25 sites (mixed samples from five sampling locations distributed over the field) and determined soil texture (wet sieving for sand, and laser particle measurement for silt and clay), and the organic matter contents in the laboratory. The soil structure code was estimated from field observations. The dominant portion of the dryland fields in the study area was characterized by artificially deposited structureless, sandy top soil horizons (Ruidisch et al., 2013), and the structure code was therefore set to 1 (very fine granular). Profile descriptions and tracer experiments on dryland fields in the study area indicated a relatively low infiltration capacity of most of the subsoil horizons due to repeated plowing and compaction (Ruidisch et al., 2013). The permeability code for all 25 fields was therefore set to 4 (moderate to slow). By using the analyses data of the soil samples and the assumptions for s and p, we calculated the K-factor with Eq. (3). Potential seasonal variations of the K-factor due to soil freezing, soil water, and soil surface conditions (López-Vicente et al., 2008) were not considered in this study.

2.2.3. Slope length and steepness factors (L and S), and support practice factor (P)

The L-factor and the S-factor describe the effect of the field topography on soil erosion. The L-factor considers the higher erosion potential with increasing slope length and the S-factor reflects the influence of the slope steepness on erosion (Renard et al., 1997). The L-factor is calculated as (modified after Renard et al., 1997):

$$L = (3.2808 \cdot \lambda/72.6)^{\beta/(1+\beta)} \tag{5}$$

where L is the slope length factor (−), λ the slope length (m), and β the ratio of rill erosion to interrill erosion, which itself is calculated as (Renard et al., 1997):

$$\beta = (\sin\theta/0.0896)/\left[3.0 \cdot (\sin\theta)^{0.8} + 0.56\right] \tag{6}$$

where θ is the slope angle (°). The factor 3.2808 in Eq. (5) is used to insert slope length as SI unit. The S-factor is calculated as (modified after McCool et al., 1987; Renard et al., 1997):

$$S = 10.8 \cdot \sin\theta + 0.03 \qquad \theta < 5.14^\circ \tag{7}$$

$$S = 16.8 \cdot \sin\theta - 0.50 \qquad \theta \geq 5.14^\circ \tag{8}$$

where S is the slope steepness factor (−), and θ the slope angle (°).

The topographic factors, especially the S-factor, are associated with the highest uncertainty in the model (Falk et al., 2009; Wang et al., 2002a). Especially slope steepness has been identified to be the most sensitive in soil loss predictions with RUSLE (Renard and Ferreira, 1993; Wang et al., 2002a). Therefore, special attention has to be paid to the appropriate use of those factors and the identification of their input parameters. The L-factor and S-factor have been originally developed in USLE for relatively gently sloping cropland, but have been refined in RUSLE for the application on steeper and also short slopes (Rapp, 1994; Renard et al., 1994; Van Remortel et al., 2004). However, Eqs. (7) and (8) have been derived from erosion plot studies of slopes up to 18% and may strongly underestimate soil loss from very steep slopes that can occur in mountainous watersheds (Liu et al., 1994; Nearing, 1997; Shi et al., 2004). In addition, slope lengths should be no shorter than 4.5 m and do practically not exceed 122 m in most situations, although longer slope lengths are possible (McCool et al., 1987; Renard et al., 1997). The individual field sites used in this study generally showed slope lengths that were within the practical range for RUSLE, but slope steepness exceeded the limit of 18% in some cases up to a maximum of 27%. Nearing (1997) developed an equation that fits the RUSLE equations for slopes up to 25% but produces more realistic results for steeper slopes. However, up to the maximum slope of the fields in this study, we have found that the approach of Nearing (1997) produced very similar outputs than Eqs. (7) and (8). We could therefore conclude that the RUSLE approach for computing the L-factor and S-factor was appropriate.

The P-factor reflects the positive impact of management through the control of runoff by practices such as contour tillage, strip cropping, terracing, or subsurface drainage (Renard et al., 2011). The ridge–furrow cultivation system on the dryland fields in South Korea can be regarded as contouring support practice. The effectiveness for a given ridge height is controlled by the field slope steepness and the steepness along the furrows when ridges are not parallel to the contours. First, the P-factor for on-grade contouring (P_0) is calculated and subsequently adjusted for off-grade contouring (Renard et al., 2011). For the high ridges (approximately 15 cm) on Korean row crop fields, P_0 is calculated as (modified after Renard et al., 1997):

$$P_0 = 18051 \cdot (0.0797 - \sin\theta)^4 + 0.27 \qquad \sin\theta < 0.0797 \tag{9}$$

$$P_0 = 10.24 \cdot (\sin\theta - 0.0797)^{1.5} + 0.27 \qquad \sin\theta \geq 0.0797 \tag{10}$$

$$P_0 = 1.0 \qquad \sin\theta \geq 0.2516 \tag{11}$$

where P_0 is the on-grade contouring support practice factor (−), and θ the slope angle (°). The adjusted contouring P-factor is calculated as (modified after Renard et al., 1997):

$$P = P_0 + (1 - P_0) \cdot \left(\sin\theta_f / \sin\theta\right)^{0.5} \tag{12}$$

where P is the off-grade contouring P-factor (−), P_0 the on-grade contouring P-factor (−), θ_f the slope angle along the furrows, and θ the average slope angle of the field (°).

The P-factor is the most unreliable factor in the model due to limited experimental data (Rapp, 1994; Renard and Ferreira, 1993), but little information can be found in the literature about its applicability and appropriate use. It has been discussed that the P-factor for contouring in USLE may not be suitable to accurately describe the conservation effect because of the small size of runoff plots (Risse et al., 1993; Sonneveld and Nearing, 2003). In RUSLE, the P-factor equations presented above have been developed not only based on plot studies, but also from watershed observations, and calculations using detachment and transport theory (Renard and Ferreira, 1993; Renard et al., 1997), and may therefore provide a broader applicability. The ridge–furrow cultivation system found in the study area can be compared in terms of minimum ridge height and spacing to the contour farming practice described by the NRCS (2007), for which the *P*-factor has been developed. We have therefore assumed that Eqs. (9) to (12) could sufficiently reflect this cultivation system although uncertainties were expected due to the plastic film covers that involve additional effects, which cannot be adequately described by the *P*-factor.

Before the above factors could be calculated, the representative hillslope profile for each individual field site had to be identified. Commonly, the values for the *L*-factor and *S*-factor are derived by manual measurements using a tape or GPS and an inclinometer (Angima et al., 2003; Rapp, 1994; Zhang et al., 2013). However, an infinite number of slopes exists in a field (Angima et al., 2003; Renard et al., 1997), and especially for the highly complex terrain found in the study area, where fields can have various flow paths in different directions, it was difficult to identify the representing hillslope profiles and to determine slope lengths and the slope angles of the field and along the furrows. Motivated by the work of Cochrane and Flanagan (2003), we developed an algorithm using the R programming language that automatically identified all possible flow paths within one field site and extracted the mean slope length and slope angle from three provided ArcGIS raster grids of a given field site. From an available 30 m resolution digital elevation model of the study area, we developed a 0.25 m resolution DEM by using bilinear interpolation (performed by ArcGIS ver. 10.0), from which we extracted individual digital terrain models for the 25 field sites. Based on our field observations in 2009 and from photographs taken from all field sites, we created 25 additional digital terrain models, which included the height, dimension, and orientation of the ridges and furrows. For each of those 50 DTMs, we developed three raster grids which were used for the R algorithm: a depression-filled elevation raster, a flow direction raster, and a flow accumulation raster (performed by ArcGIS ver. 10.0). The depression-filled elevation raster contained the elevation model of the field site without topographical sinks. The flow direction raster contained for each cell the information to which neighboring cell water would flow. The flow accumulation raster contained for each cell the number of cells that would drain into it, and was used to identify the starting cells of flow paths (cell value equals 0). We extracted the mean slope length and slope angle by using the R algorithm (performed by RStudio ver. 0.95.258) for the raster grids without ridges and furrows, and calculated *L*-factor and *S*-factor for the 25 fields with Eqs. (5) to (8) and P_0 with Eqs. (9) to (11). Subsequently, we used the same algorithm for the raster grids including ridges and furrows to extract the mean slope angle along the furrows, which was used to calculate the off-grade contouring *P*-factor with Eq. (12). Finally, we changed the *P*-factor to 1.0 for those field sites for which the slope length considerably exceeded the critical slope length according to the slope steepness, as described by Wischmeier and Smith (1978). When slope length increases a critical length, ridge breakovers can occur resulting in a higher erosion rate that makes contouring ineffective (Wischmeier and Smith, 1978).

2.2.4. Cover-management factor (C)

The cover-management factor represents the effects of crop and management practices on soil erosion and is used to compare the relative impacts of the different crops and management types (Renard et al., 1997). It includes the impact of previous management, the soil surface protection of vegetation cover, and the reduction in erosion due to surface cover and surface roughness (Renard et al., 1997). Because these conditions change over the course of the year, a time-varying *C*-factor approach is used in RUSLE based on half-month time steps (Renard et al., 1997). For each of the half-month periods within the year, a soil loss ratio (*SLR*) is calculated, for which the conditions are assumed to remain constant, and is weighted by the percentage of rainfall erosivity associated with that period (see Section 2.2.1) to obtain the annual *C*-factor (modified after Renard et al., 1997):

$$C = (SLR_1 \cdot EI_1 + SLR_2 \cdot EI_2 + \ldots + SLR_{24} \cdot EI_{24})/100 \tag{13}$$

where *C* is the cover-management factor (−), SLR_i the soil loss ratio for the half-month period *i*, and EI_i the percentage of the total rainfall erosivity (EI_{30}) within the half-month period *i* (%). The soil loss ratio for each half-month period is calculated as the product of five subfactors (Renard et al., 1997):

$$SLR = PLU \cdot CC \cdot SC \cdot SR \cdot SM \tag{14}$$

where *SLR* is the soil loss ratio (−), *PLU* the prior land use subfactor (−), *CC* the canopy cover subfactor (−), *SC* the surface cover subfactor (−), *SR* the surface roughness subfactor (−), and *SM* the soil moisture subfactor (−).

The prior land use subfactor is calculated as (modified after López-Vicente et al., 2008; Renard et al., 1997):

$$PLU = C_f \cdot C_b \cdot \exp - \left[(c_{ur} \cdot 8.9219 \cdot B_{ur}) + \left(c_{us} \cdot 8.9219 \cdot B_{us} / C_f^{\,c_{uf}} \right) \right] \tag{15}$$

where *PLU* is the prior land use subfactor (−), C_f the surface–soil consolidation factor (−), C_b represents the relative effectiveness of subsurface residue in consolidation, B_{ur} is the mass density of live and dead roots in the upper 2.54 cm of soil (g m^{-2}), B_{us} is the mass density of incorporated surface residue in the upper 2.54 cm of soil (g m^{-2}), c_{ur} and c_{us} are coefficients indicating the impact of the subsurface residues, and c_{uf} represents the impact of soil consolidation on the effectiveness of incorporated residue. The factor 8.9219 is used to insert root and residue mass density as SI units. The soil consolidation factor for freshly tilled soil is 1.0 and decreases to 0.45 when soil is left undisturbed for seven years (Renard et al., 1997). Because in the study area fields are usually tilled every year, soils are disturbed by harvest activities, and short term consolidation rates were not known, we used a value of 1.0 for all 24 half-month periods throughout the year. For the coefficients C_b, c_{ur}, c_{us}, and c_{uf}, the values 0.951, 0.00199, 0.000416, and 0.5 were used, respectively (Renard et al., 1997). The canopy cover subfactor is calculated as (modified after Renard et al., 1997):

$$CC = 1 - F_c \cdot \exp(-0.1 \cdot H \cdot 3.2808) \tag{16}$$

where *CC* is the canopy cover subfactor (−), F_c the fraction of the land area covered by canopy (−), and *H* the distance that raindrops fall after striking the canopy (m), calculated as (modified after USDA, 2008):

$$H = H_b + 0.29 \cdot (H_t - H_b) \tag{17}$$

where *H* is the raindrop fall height (m), H_b the height to the bottom of the canopy (m), and H_t the height to the top of the canopy (m), assuming a round canopy shape and a uniformly distributed canopy density. The factor 3.2808 in Eq. (16) is used to insert *H* as SI unit. The height to the bottom of the canopy was assumed to be 0.15 m (ridge height). The surface cover subfactor is calculated as (modified after Renard et al., 1997):

$$SC = \exp\left\{ -b \cdot S_p \cdot [0.24/(0.3937 \cdot R_u)]^{0.08} \right\} \tag{18}$$

where *SC* is the surface cover subfactor (−), *b* an empirical coefficient, which is 0.035 for typical cropland erosion conditions (Renard et al., 1997), S_p the percentage of land area covered by surface cover (%), and R_u is the surface roughness (cm). The surface roughness subfactor is calculated as follows (modified after Renard et al., 1997):

$$SR = \exp[-0.66 \cdot (0.3937 \cdot R_u - 0.24)] \tag{19}$$

where *SR* is the surface roughness subfactor (−), and R_u the surface roughness (cm). The factor 0.3937 in Eqs. (18) and (19) is used to insert surface roughness as SI unit. The soil moisture subfactor is only used in the Northwest Wheat and Range Region of the United States (Renard et al., 2011) and was therefore set to 1.0 for this study.

RUSLE is very sensitive to crop and management parameters (Benkobi et al., 1994; Renard and Ferreira, 1993), and it has been reported that the *C*-factor has the highest uncertainty, besides the topographic factors (Rapp, 1994; Wang et al., 2002a). Therefore, care should be taken when selecting input parameters, because regional crop and management conditions may strongly differ from those in the United States. The majority of studies that applied RUSLE, especially at larger scales, for example for watersheds, obtained the *C*-factor from literature values that were often derived from local experimental data (e.g., Cohen et al., 2005; Fu et al., 2005; Hoyos, 2005; Park et al., 2011; Yue-qing et al., 2009), or they computed it as a function of ground cover (e.g., Falk et al., 2009; Shi et al., 2004). However, little is known about the general applicability of the RUSLE subfactor approach presented in Eqs. (13) to (19) outside of the United States. Renard and Ferreira (1993) and Kinnell (2010) stated that the subfactor approach for the determination of the *C*-factor enables the application also to crops and management practices beyond the range of experimental data from which it was developed, by using field measurements. Angima et al. (2003), for instance, used local crop growth data and cropping patterns in Kenya to create a crop data base for the RUSLE program and reported reasonable results for both *C*-factors and simulated soil loss. Also other studies have shown that the RUSLE subfactor approach can be successfully applied outside of the United States when local crop and management data is used (e.g., Gómez et al., 2003; López-Vicente et al., 2008; Millward and Mersey, 1999; Renschler et al., 1999). We therefore assumed that the above equations were also applicable in this study, since we determined the relevant crop and management parameters for calculating the soil loss ratios solely from field measurements in the Haean catchment.

To obtain these parameters, we measured the development of biomass density, cover, and canopy height for the four major crops and the associated weeds during the growing season of 2009 on four of the 25 sites (sites 03, 07, 16, and 18). At three (radish and cabbage) and four (bean and potato) different dates between planting and harvest, we sampled the crops and weeds from nine subplots, separated the different plant parts of crops (roots with radishes or potatoes, stems or cabbage cores, leaves, seeds, and dead plant material) and weeds (below-ground and above-ground), and determined the dry biomass of the different components. Based on the number of crops and weeds per m^2, we calculated the average biomass density of each component. From photographs of the different subplots taken on the day of sampling, we estimated the associated crop cover, the weed cover, and the canopy height for the different sampling dates. We created growth charts of the four major crops, including weeds, for biomass density (separated by plant components), canopy cover, and canopy height. The growth charts were completed by biomass, cover, and height measurements of the four sites before harvest. From three subplots, we further sampled all crops and weeds, separated the different plant parts, determined their biomass densities, and estimated crop cover, weed cover, and canopy height either in the field, or from additional photographs. Subsequently, we adjusted those growth charts to fit to the real planting and harvest dates, as the last biomass sampling was carried out before harvest. On the remaining 21 field sites including organic and conventional farming, we also sampled all crops and weeds from three subplots before harvest, and determined the biomass density of the different plant parts. We additionally estimated crop cover, weed cover, and canopy height from three subplots in the field, and from additional photographs. Based on these data, we calculated the average yield, cover, and canopy height of all four row crops and both farming systems at harvest. The four base growth charts were finally adjusted to those values, resulting in growth charts containing the crop and weed biomass density separated by plant parts, crop cover, weed cover, and canopy height for conventional and organic bean, potato, radish, as well as for conventional cabbage production.

The associated soil loss ratios for the 24 individual half-month periods of 2009 were then calculated with Eqs. (14) to (19) underlying the following assumptions. Because farmers did not cultivate their crops according to fixed rotation systems, we had no information of potential residual biomass and cover from previous crops. To compare the farming system for the individual crops, we decided to focus only on the current growing season without considering effects of previous years. Prior to planting, it was therefore assumed that fields did not contain plastic film covers, and that the biomass density of roots and residues, as well as crop cover, surface cover and canopy height, were zero. The surface roughness R_u was estimated as 1.65 cm, by comparing soil surface photographs of dryland fields in the study area to roughness plot photographs by Renard et al. (1997). During the growing season, between planting and harvest, only root biomass density of the weeds was relevant (assuming 10 cm rooting depth), because weeds were growing in the furrows, whereas crop roots were only concentrated in the ridges, which were covered by plastic film. The application of plastic mulch provided 50% surface cover for the entire growing season, but surface roughness was also reduced to 0.83 cm, assuming that the roughness of the plastic film covering 50% of the soil surface was 0.0 cm. Canopy cover was the combination of crop cover and the cover of weeds, assuming that weeds covered both, ridges and furrows, and crops covered primarily ridges. Canopy cover of crops was reduced by the amount of dead biomass (as the ratio of dead biomass to total biomass), because dead plant parts fell to the ground, became residues, and were therefore add to the surface cover. After harvest, all crop biomass died and became residue, and the canopy cover was then only determined by weed cover. The canopy height was set to zero. The amount of biomass density remaining in the field was depending on the crop type. For bean, the crop yield accounted only for a relatively small fraction of the plant, and almost the whole biomass remained on the field, and for potato, only the potatoes were harvested, whereas for radish, most of the biomass was harvested, and for cabbage, everything except the roots and the outer leaves (approximately 15% of the leaf biomass) was harvested. The density of incorporated root and residue biomass after harvest, as well as the percentage of canopy and surface cover were depending on the degree of soil disturbance at harvest. Bean and cabbage could be harvested above the soil surface without plastic film removal and soil disturbance. The harvesting of potato required the removal of the plastic film and a complete disturbance and mixing of the soil in the ridges (50% of the field surface–soil). Also radish was harvested by the removal of at least 50% of the plastic film and soil disturbance and mixing of the underlying ridges (25% of the field surface–soil). However, the different farmers in the study area used different techniques and machinery for harvesting their crops, which could produce different levels of disturbance and mixing. To include the variety of those different harvesting procedures, two scenarios were simulated: a low disturbance scenario representing the minimum required disturbance for manual harvest (described above), and a high disturbance scenario representing a maximum disturbance such as that created by using machinery. For bean and cabbage, 50% of the plastic film was removed and 25% of the surface–soil was mixed. For potato, 100% of the plastic film was removed and 100% of the surface–soil was mixed, and for radish 100% of the plastic film was removed and 50% of the surface–soil was mixed. According to those ratios of plastic film removal and soil surface-mixing, the canopy cover and surface cover were reduced, the surface roughness was increased, and the biomass density of incorporated roots and residue was calculated based on the remaining biomass density of dead crops and weeds. The average depth of disturbance was assumed to be 10 cm, which was estimated from field observations. For the periods after harvest, it was assumed that the cover and biomass density status remained stable. Additional growth of weeds after harvest and the decomposition of residue could not be included in this study, because we did not have data of growth and decomposition rates after the cropping period.

In order to account for different schedules of planting and harvesting over different years, we simulated two additional scenarios, one scenario representing an early planting year by shifting all calculated *SLR* values to the previous half-month period, and one scenario representing a late planting year shifting all *SLR* values to the next half-month period. Subsequently, we calculated the *C*-factor for the major crops, for conventional and organic farming, and for each of the 13 years of available rainfall data and each of the scenarios by using Eq. (13). Finally, we calculated the average annual *C*-factor for conventional and organic bean, potato, radish, as well as conventional cabbage, as the mean *C*-factor over all years and scenarios.

2.2.5. Calculation of soil erosion rates

The average annual soil erosion rates associated with the four major crops and two different farming systems were calculated according to Eq. (1) by using the average annual *C*-factors and combining them with all 25 field sites. Using the average annual *R*-factor and the fields' individual factors *K*, *L*, *S*, and *P*, we calculated the average annual erosion rate for conventional and organic bean, potato, radish, as well as conventional cabbage, for each field site.

2.3. Model plausibility

In order to evaluate, if RUSLE produced reliable results, we compared the simulated erosion estimates with measured soil loss information in the study area. Only a few soil loss observations were available in the Haean catchment that could be used to verify the simulated erosion rates. We used two field sites, M1 and M2 (Fig. 1c), where Arnhold et al. (2013) have measured sediment for a one month period during the monsoon season of 2010. Between the 5 July and 9 August 2010, the amount of surface runoff and eroded sediment was measured with three runoff collectors designed according to Bonilla et al. (2006) within both field sites. The amount of rainfall was continuously measured with two rain gauges, soil samples of both fields were analyzed, and the cover and dimensions of the crop (*S. tuberosum*) canopy were determined. Additionally, digital terrain models with 0.25 m resolution were developed of both fields, both with and without ridges and furrows. Details about the experimental setup and measurement procedures are given in Arnhold et al. (2013). Based on the information from this study, we calculated the individual RUSLE factors and the soil loss rates for both fields for the observation period from 5 July to 9 August 2010, using the methods described above. The calculated soil loss rate was then compared to the observed erosion.

Because the observation period in 2010 was relatively short and only two field sites were measured, we additionally compared the average annual erosion rates computed with RUSLE to the long-term annual erosion rates estimated using the fallout radionuclide Caesium-137 (^{137}Cs) for two sloping dryland fields in the Haean catchment (Meusburger et al., in preparation). One site was located in an area of the catchment, which was recently converted from forest to farmland, and the other site was characterized by long-term agricultural use. Soil core samples were taken from three locations within each site and were analyzed for ^{137}Cs activity. Fallout ^{137}Cs was generated as a product of nuclear weapons tests between the mid 1950s and the early 1970s (Meusburger et al., 2013). It is redistributed in soils by processes like erosion and deposition, and can be used for assessing the amount of soil loss (Mabit et al., 2008). When comparing model results to ^{137}Cs estimates, attention must be paid to the spatial scale to which the model was applied. Erosion estimates with ^{137}Cs on hillslopes, for instance, may not be representative for soil loss from large areas, for example watersheds, because of changes in amount and quality of sediment due to selective transport and deposition of soil particles from fields over the landscape (e.g., Kuhn et al., 2009). In this study, RUSLE was applied to individual hillslopes, i.e., the 25 field sites, similar in size and topography to those where the ^{137}Cs measurements were conducted, without consideration of field margins or surrounding land use. Meusburger et al. (2013) have shown that average soil loss rates estimated with the ^{137}Cs method on forest slopes in the Haean catchment can be adequately reflected by RUSLE. Model results and ^{137}Cs estimates should be therefore also comparable for cropland when applied at the same scale.

However, it should be noted that these comparisons could be used only to verify the overall performance of the model in reproducing the average magnitude of soil loss from agricultural fields in the study area rather than testing absolute accuracy at specific field sites. RUSLE is used as a conservation management tool, where the relative differences between the cultivation strategies are more critical than predicting highly precise absolute soil loss values for individual locations (Millward and Mersey, 1999).

3. Results and discussion

3.1. Rainfall and runoff erosivity factor (R)

As expected, the rainfall and runoff erosivity for the period from May 2009 to December 2011, calculated on the basis of 30 minute resolution records, was higher than for the aggregated 1 hour resolution data sets (Fig. 2). Both data sets exhibited a high linear correlation ($R^2 = 0.998$). The slope of the regression line was 1.391, which was used as correction factor for rainfall erosivity calculated on the basis of the 1 hour resolution records between January 1999 and May 2009. The resulting corrected rainfall erosivity factor for the years 1999 to 2011 is shown in Fig. 3. It was estimated to be highly variable over the 13 year period with a maximum in 2006 and minima in the years 2000 and 2002. Fig. 3 also shows that the *R*-factor did not closely follow the amount of rainfall that occurred. Although the highest *R*-factors were calculated from 2006 to 2008, where also the highest rainfall was recorded, other years showed very low erosivity values relative to their rainfall amounts. This demonstrates the importance of rainfall intensity for the soil erosion process. Although having similar rainfall amounts, years with storm events of higher intensity produced much higher erosivities, because of the high kinetic energy of raindrops striking the soil surface, but especially due to higher runoff generation when the soil's infiltration capacity is exceeded (Morgan, 2005). Runoff excess could lead to soil detachment by overland flow and rill erosion in concentrated flow paths that accounts for a major part of soil removed from field sites due to its high erosive power (Morgan, 2005; Renard et al., 1997).

The erosive rain events were concentrated in the monsoon season between June and September, but the temporal distribution within the individual years showed very different seasonal rainfall patterns (Fig. 4). In the years 2001, 2009, and 2011 erosive rain events were

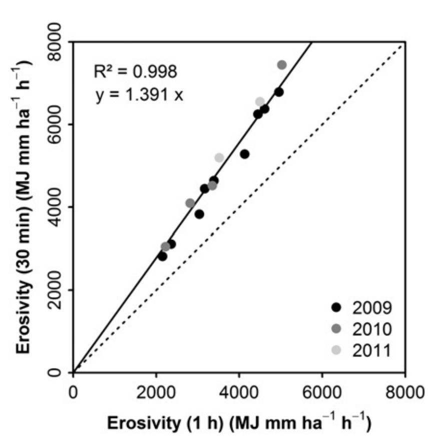

Fig. 2. Correlation between rainstorm erosivity calculated on the basis of 1 hour and 30 minute resolution rainfall records of nine weather stations in the Haean catchment from May 2009 to December 2011. The solid line shows the line of the linear regression through the origin, and the 1:1 line (dotted) is included as reference.

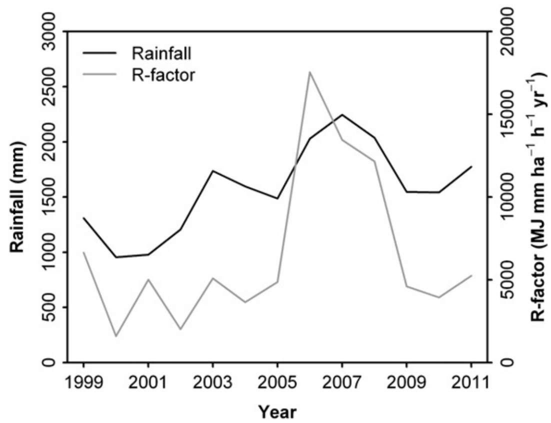

Fig. 3. Annual rainfall and *R*-factor for the Haean catchment for the years 1999 to 2011.

found relatively early in June and July, whereas in the years 2000, 2003, 2007, and 2010 most of the erosive rainfall was calculated for August and September. In the years 1999 and 2006 almost all of the erosive rain events were concentrated within one single month, whereas in the years 2004 and 2005 the erosive rainfall was spread over a relatively long period from June to September.

The average annual *R*-factor for the Haean catchment was 6599.1 MJ mm ha^{-1} h^{-1} yr^{-1}, which is about 50% higher than the *R*-factor found in previous studies of this region (Chuncheon) (Jung et al., 1983; Park et al., 2000). This might be a result of recent extreme years, for example that of 2006 with Typhoon Ewiniar, which saw the highest daily rainfall recorded in Korea (Park et al., 2011), and which could not be considered by these authors. These studies, additionally, used hourly precipitation data with limited utility to calculate the actual rainfall erosivity (Lee and Heo, 2011; Park et al., 2000). In another study, Lee and Heo (2011) presented rainfall erosivity for Chuncheon of 6076 MJ mm ha^{-1} h^{-1} yr^{-1} based on long-term high resolution rainfall data. It indicates that our calculations were plausible, although only 13 years of weather station data were available in the Haean catchment, instead of the 20 to 25 years recommended by Wischmeier and Smith (1978).

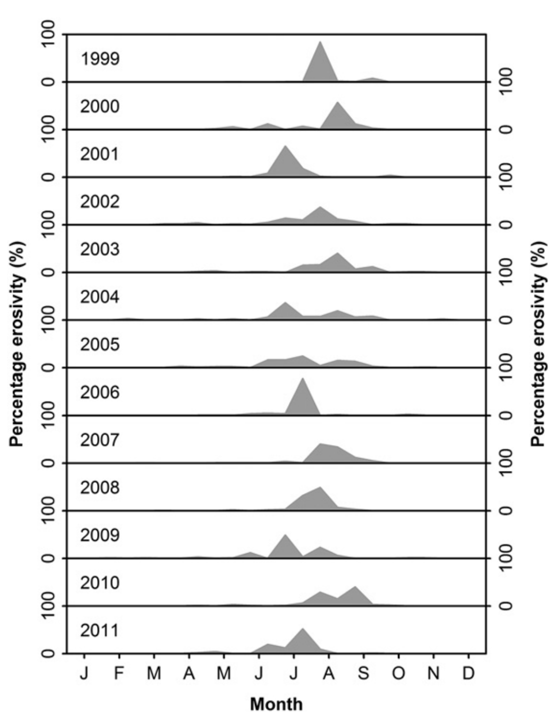

Fig. 4. Temporal distribution of rainstorm erosivity (percentage of the half-month period erosivity) within the individual years from January to December for 1999 to 2011.

3.2. Soil erodibility factor (K)

The soils of the 25 field sites selected for this study were characterized by a high sand content (41% to 90%) and a low amount of organic matter (0.3% to 2.2%) resulting in an average soil erodibility of 0.0211 t h MJ^{-1} mm^{-1} (Table 1). Soil texture was predominantly sandy loam or loamy sand. Due to the generally low clay contents, soils were more susceptible to detachment as the adhesive and chemical bonding forces of clay particles that form stable soil aggregates (Morgan, 2005) were missing. However, the high sand contents increased the resistance against detachment by rainsplash and overland flow, because of the higher weight of particles that require a greater force to be entrained from the soil surface (Morgan, 2005).

The minimum *K*-factor was therefore calculated for site 06 (0.0092 t h MJ^{-1} mm^{-1}), which had the highest sand content among the 25 fields. The maximum *K*-factor was calculated for site 2 (0.0367 t h MJ^{-1} mm^{-1}), which was characterized by a finer texture (loam) compared to the other field sites. Fields having loam and fine sandy soils were the most erodible due to their lack of both the cohesiveness of clay minerals and the weight of large particles (Morgan, 2005).

The average *K*-factor for fields under conventional farming management systems (chemical usage) was 0.0219 t h MJ^{-1} mm^{-1}, and 0.0199 t h MJ^{-1} mm^{-1} for the organic fields. The lower calculated soil erodibility for the organic fields was primarily due to higher sand and lower silt contents. The average organic matter content was slightly higher for soils of conventionally managed fields (1.1%) than for organic fields (0.7%). The high variability of texture and organic matter within conventional and organic fields indicated that the different erodibility factors resulted from the spatial variation of soil properties within the study area, and not from the farming systems employed, as described by Fleming et al. (1997). Organic farming is still a relatively new management practice in the study area. The contents of organic matter were generally very low and did therefore not contribute considerably to soil stability, especially due to the lack of clay minerals that combine with the organic compounds to form stable aggregates (Morgan, 2005). In addition, the artificial layering and relatively low permeability of subsoil horizons due to past soil replenishments and frequent plowing activities (Ruidisch et al., 2013) might have contributed to accelerated runoff generation and erosion by overland flow for both conventional and organic fields. The positive effects of organic farming on soil properties, for example, the increased infiltration capacity and improved soil stability by the addition of organic matter (Erhart and Hartl, 2010), may only become visible after many years of organic management.

3.3. Slope length and steepness factors (L and S), and support practice factor (P)

The 25 field sites were highly variable in their slope length and steepness (Table 1) representing the topographical variability of the agricultural land in the study area (see Section 2.1). The slope length varied from 4.7 m to 124.6 m, resulting in *L*-factors between 0.380 and 2.479. The average slope length among all 25 fields was 38.4 m and the average *L*-factor was 1.341. It has been discussed that an increasing slope length may not always result in higher soil loss rates from a field, for example, when deposition occurs (Morgan, 2005). However, when runoff volume is accumulated and concentrated flow occurs, erosion could strongly increase with slope length, particularly when a dense network of rills has been formed (Meyer et al., 1975; Morgan, 2005). The disturbed, structureless topsoil horizons of the fields in this study were more susceptible to rill formation than consolidated, structured soils (Renard et al., 1997). When concentrated rill

Table 1
Soil characteristics (organic matter and texture) and topography (slope angle and slope length) of the 25 field sites with the calculated *K*-factors, *L*-factors, *S*-factors, and contouring *P*-factors for the Revised Universal Soil Loss Equation.

Site	Area (m^2)	Org. matter (%)	Clay (%)	Silt (%)	Sand (%)	*K*-factor ($t\ h\ MJ^{-1}\ mm^{-1}$)	Slope angle (°)	Slope length (m)	*L*-factor (−)	*S*-factor (−)	*P*-factor (−)
01	2246	2.2	5.2	18.3	76.5	0.0155	7.0	47.7	1.525	1.539	1.000
02	6228	1.1	16.9	41.9	41.2	0.0367	3.6	36.4	1.244	0.701	0.949
03	3056	0.6	3.6	17.1	79.3	0.0219	4.6	21.8	0.992	0.898	0.742
04	5787	1.6	9.3	26.6	64.1	0.0251	8.6	106.5	2.479	2.000	1.000
05	5447	1.1	4.9	21.0	74.1	0.0225	1.9	16.7	0.911	0.393	0.994
06	5712	0.3	1.6	8.3	90.1	0.0092	14.9	57.1	1.845	3.828	1.000
07	11,347	0.8	2.2	12.4	85.3	0.0153	3.7	54.0	1.489	0.729	0.858
08	10,536	2.1	5.6	22.1	72.4	0.0230	0.0	15.7	0.996	0.036	1.094
09	1767	1.3	6.2	23.0	70.9	0.0246	7.5	24.4	1.056	1.687	0.730
10	4878	0.4	4.1	13.0	82.8	0.0155	0.0	11.3	0.998	0.031	1.049
11	13,764	1.2	5.0	20.6	74.4	0.0220	5.6	124.6	2.440	1.153	1.000
12	6452	0.7	5.4	21.1	73.5	0.0262	5.8	35.5	1.280	1.207	0.832
13	2711	1.3	7.6	25.0	67.3	0.0247	5.1	31.3	1.188	0.984	0.901
14	5643	1.3	6.4	23.5	70.1	0.0250	10.9	45.6	1.553	2.683	1.000
15	1143	0.3	2.8	13.3	83.8	0.0173	0.0	6.3	1.000	0.030	1.000
16	10,981	0.4	3.4	16.6	80.0	0.0212	6.9	50.7	1.572	1.509	1.000
17	72	0.3	2.8	15.9	81.3	0.0204	12.0	4.7	0.380	3.005	1.009
18	3779	0.8	3.3	15.3	81.4	0.0180	6.4	37.1	1.318	1.369	1.148
19	3967	0.7	2.5	13.6	83.8	0.0182	2.5	24.6	1.041	0.497	1.088
20	2408	1.2	6.3	23.6	70.1	0.0243	10.1	22.0	0.996	2.452	0.999
21	14,843	1.0	4.1	16.7	79.2	0.0181	11.8	44.5	1.540	2.923	1.000
22	16,578	1.3	6.4	19.8	73.9	0.0204	11.4	52.1	1.693	2.816	1.000
23	1913	0.4	4.3	18.7	77.0	0.0216	0.0	9.0	1.000	0.030	1.000
24	1978	0.4	3.6	17.0	79.4	0.0201	0.0	10.6	1.000	0.030	1.000
25	11,652	1.7	5.7	21.6	72.7	0.0217	10.6	68.9	1.990	2.584	1.000
Mean	6196	1.0	5.2	19.4	75.4	0.0211	6.0	38.4	1.341	1.405	0.976

flow occurs, the erosion process becomes non-selective regarding particle size, and also heavy particles, such as course sand and rock fragments, can be removed (Meyer et al., 1975; Morgan, 2005; Poesen, 1987), which could result in high potential soil losses from fields with long slopes. However, slope steepness had a much stronger effect on erosion than slope length (Renard and Ferreira, 1993). It ranged from 0.0° (sites 08, 10, 15, 23, and 24) to 14.9° (site 06) and the associated *S*-factors were 0.030 and 3.828, respectively. The average slope steepness among all 25 field sites was 6.0° resulting in an average *S*-factor of 1.405. The high *S*-factor values for steep field sites can be attributed to the downslope movement of soil particles by rainsplash, but primarily to the elevated overland flow velocity with increasing slope steepness (Morgan, 2005). High flow velocities involve a higher sediment transport capacity and a greater flow shear stress that could strongly accelerate erosion in concentrated flow paths by deeper incision of rills and advance of headcuts (Toy et al., 2002).

The initiation and severity of rill erosion on the field sites were also controlled by the orientation of the ridges–furrow cultivation system. The 15 cm high ridges modified the flow patterns by redirecting the runoff along the furrows instead of moving along the main slope direction of the fields. Therefore, flow distance and slope angle along the furrows primarily controlled rill formation, erosion rate, and with that, the effectiveness of the ridge–furrow system as conservation management practice. Rill erosion is initiated only at a critical slope distance where overland flow becomes channeled (Morgan, 2005) and is assumed to be insignificant for slopes shorter than 4.5 m (Renard et al., 1997). However, flow distances in the furrows were generally higher, and the calculated slope angle along the furrows ranged from 0.9° to 12.9°. For most of the sites, the slope along the furrows was smaller than the average steepness of the hillslope. However, the slope angle along the furrows was still relatively high, because ridges were generally not oriented along the contours, or the slope length exceeded the critical length, which resulted in *P*-factors close to 1 for most of the sites. The smallest *P*-factor was calculated for site 09 (0.730). For some field sites, the slope calculation for the ridge–furrow system resulted in higher slope angles along the furrows compared to the steepness of the hillslope resulting in *P*-factors larger than 1. The highest *P*-factor was calculated for site 18 (1.148). The average *P*-factor among all 25 field sites was 0.976. The generally high *P*-factors show that the contouring control effect provided by the ridge–furrow system in the study area was not very effective, because furrow slope angles were too high to considerably reduce overland flow velocity. At high furrow grades, the flow shear stress can exceed the critical shear stress of the soil, and rill erosion can occur in the furrows (Toy et al., 2002) that, eventually, could produce similar high soil loss rates as fields without ridges and furrows. Moreover, Arnhold et al. (2013) and Ruidisch et al. (2013) have shown that the impermeable plastic film covers of the ridges generated more surface runoff that could additionally increase the erosive power of overland flow in the furrows.

3.4. Cover-management factor (C)

The growth charts of the four major dryland row crops in 2009 show a highly variable development of biomass, cover, and canopy height (Fig. 5). The main difference was the duration of the growing period between bean (157 days, planted on 27 May and harvested on 31 October 2009), potato (123 days, planted on 30 April and harvested on 31 August 2009), radish (82 days, planted on 2 June and harvested on 23 August 2009) and cabbage (61 days, planted on 20 May and harvested on 20 July 2009). The highest leaf biomass at the end of the individual growing periods was measured for bean (253.3 g m^{-2}) and cabbage (134.0 g m^{-2}), resulting in a higher crop cover compared to potato and radish. For potato and radish, the highest portion of crop biomass was represented by the below-ground parts, and for potato, approximately after the first half of the growing period, we observed a strong decrease in leaf biomass and crop cover. At the same time, weed biomass and weed cover increased until the end of the growing period to 184.8 g m^{-2} and 44%, respectively. For the other three crops, weed biomass and cover remained negligible compared to crop biomass and cover throughout the growing season. The canopy height curves show a similar shape to the curves of crop cover development. Radish and cabbage had their maximum canopy height at the end of the growing period. For bean and potato, canopy height was decreasing at the end of the growing period.

The yield measurements of the 25 fields before harvest showed a higher total crop biomass density for conventional management

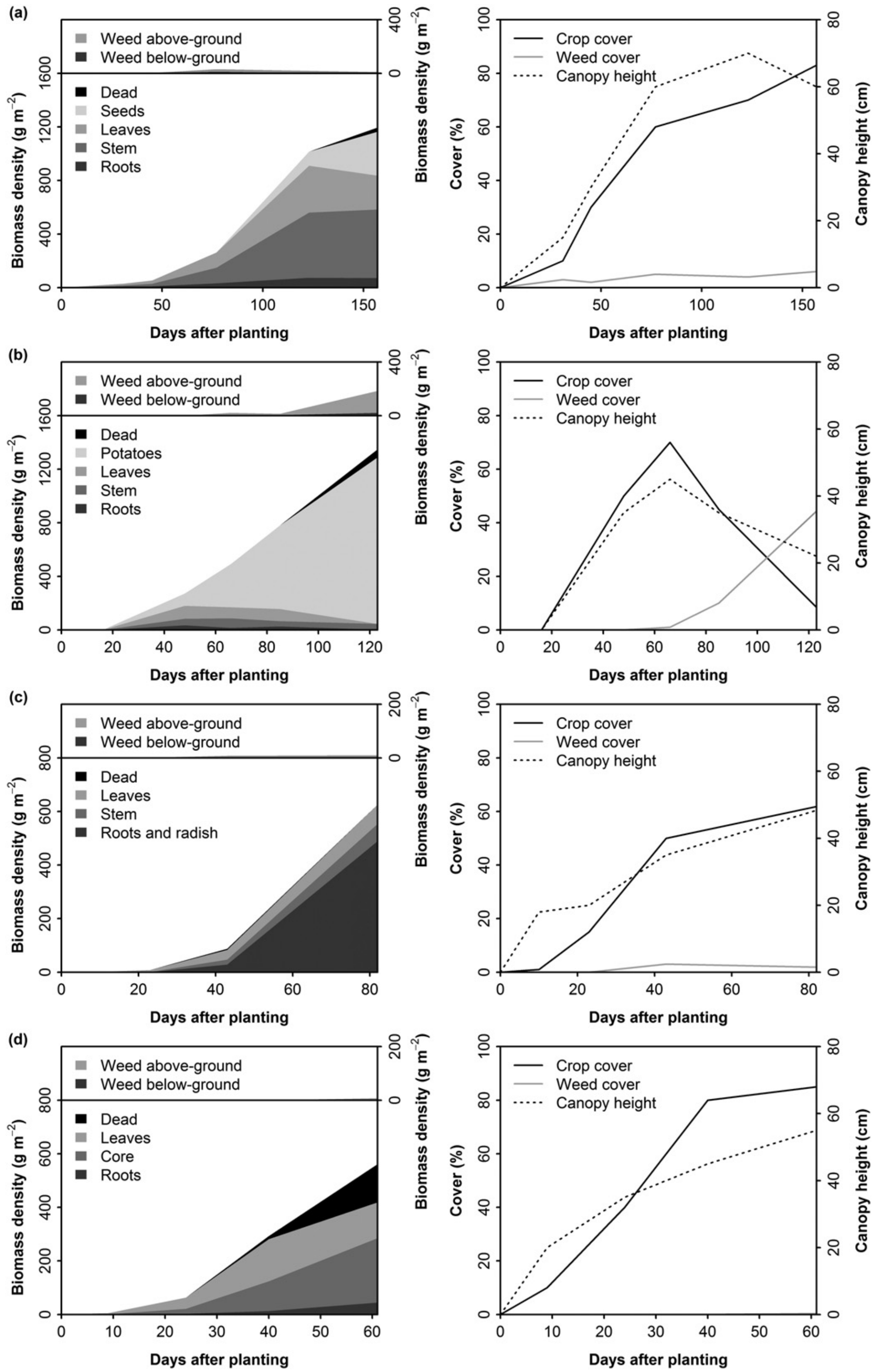

Fig. 5. Growth charts of the four major row crops, bean (a), potato (b), radish (c), and cabbage (d) with crop and weed biomass density (left), and crop cover, weed cover, and canopy height (right). The lower segment of the biomass density plot shows the development of the different crop components and the upper segment shows the development of the associated weeds.

compared to organic management for bean and potato (Fig. 6a). The mean crop biomass density for conventional farming for bean and potato was 1205.5 g m^{-2} and 1976.0 g m^{-2}, respectively. The mean crop biomass density for organic farming was 995.3 g m^{-2} and 1270.9 g m^{-2}, respectively. In contrast, radish showed a slightly higher mean crop biomass for organic farming (669.7 g m^{-2}) compared to conventional farming (568.0 g m^{-2}). The mean crop cover at harvest for radish was also higher for organic (71.2%) than for conventional farming (61.7%) (Fig. 6c). However, the crop cover for potato was much higher for conventional (26.8%) than for organic farming (12.1%). Bean showed approximately the same values under both management systems. Weed biomass and cover were consistently higher for organic than conventional

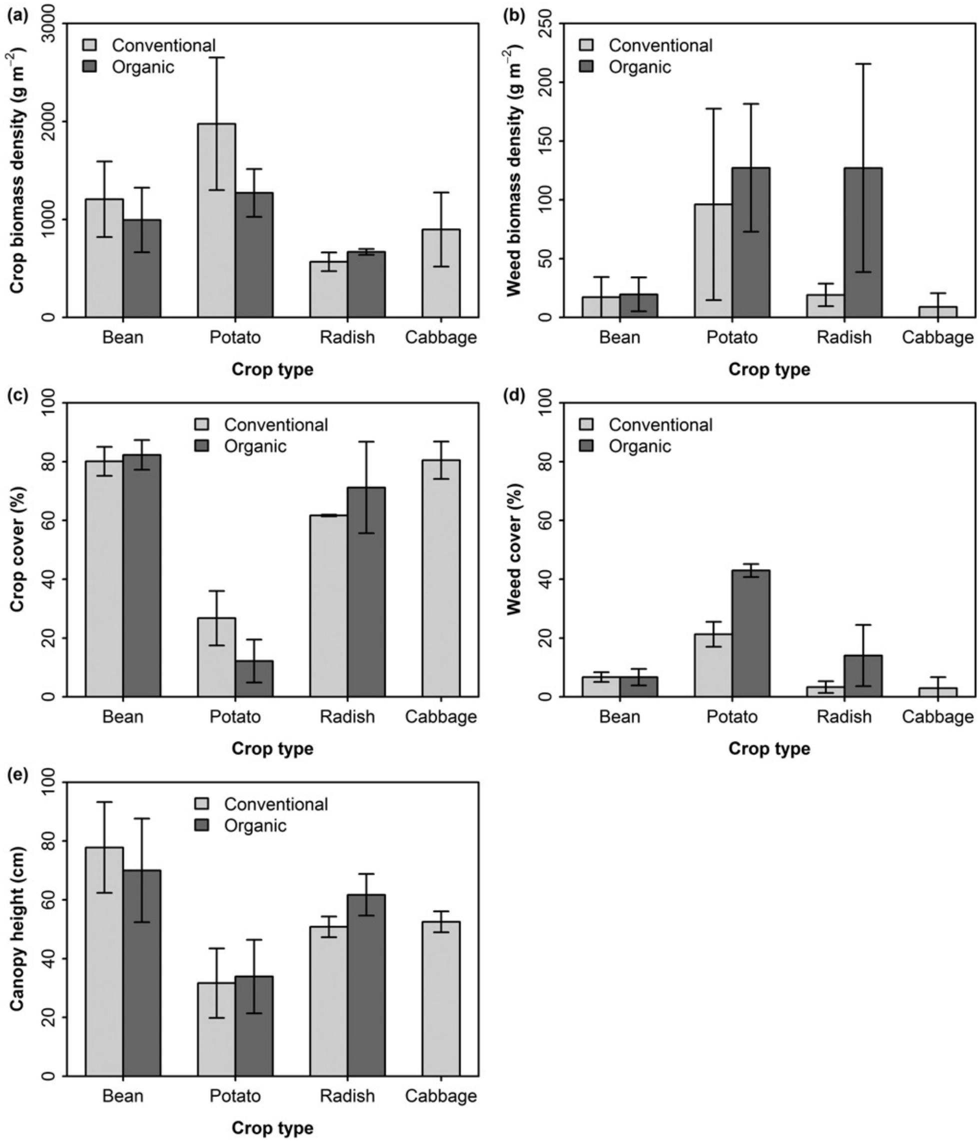

Fig. 6. Vegetation parameters of crops and weeds measured before harvest for conventional and organic farming of four major row crops. Crop and weed biomass density (a and b), crop and weed cover (c and d), and canopy height (e). The bars show the mean value and the error bars the standard deviation of the associated field sites.

farming, except for bean, which showed similar values for weed development under both farming systems (Fig. 6b and d). For potato, the mean weed biomass density for conventional and organic farming was 96.1 g m^{-2} and 127.2 g m^{-2}, respectively. Mean weed cover for conventional and organic potato was 21.3% and 43.0%, respectively. Radish showed a high difference in weed biomass densities between conventional (19.1 g m^{-2}) and organic (127.1 g m^{-2}) farming methods. Weed cover for conventional radish farming was 3.3%, and 14.0% for organic radish farming. Conventionally grown cabbage showed the lowest values for weed biomass density and weed cover among all four crops. The canopy height did not change much between both farming systems (Fig. 6e). For bean, the canopy height was slightly lower for organic (70.0 cm) compared to conventional farming (77.8 cm), and for radish the canopy height was higher for organic (61.7 cm) than for conventional farming (50.8 cm). Potato showed approximately the same canopy height for both systems.

These results demonstrate that weed biomass and especially the ground cover provided by weeds can be highly increased by the absence of herbicides associated with organic farming. For bean, the low weed biomass and cover under organic farming might be explained by the high crop coverage of the plant throughout the growing period, and the resultant constriction of weed development. Although organic farming supported the development of weeds, we also found that for potato, organic farming resulted in a lower crop yield and crop cover, which might be a consequence of crop–weed competition or herbivory due to the absence of herbicides and pesticides.

The calculated *C*-factors for the four main crops and the two management types showed a high variability over the 13 year period in terms of the level of disturbance, and the timing of planting and harvesting (Fig. 7).

For bean (Fig. 7a), maximum *C*-factors were calculated for the years 2002, 2009, and 2011, when rain events occurred in April and May. Bean did not show a considerably different *C*-factor between low and high levels of disturbance at harvest, as it was harvested at the end of October when the monsoon season was already over. Effects of surface cover by residues remaining on the field or the incorporation of plant material due to harvest activities did therefore not play an important role for erosion. Bean fields were more susceptible to erosive rain events early in the year, when fields were not yet cultivated and soils were without a protective cover. For all 13 years, a higher *C*-factor was therefore calculated for late planting and harvesting, rather than for an earlier

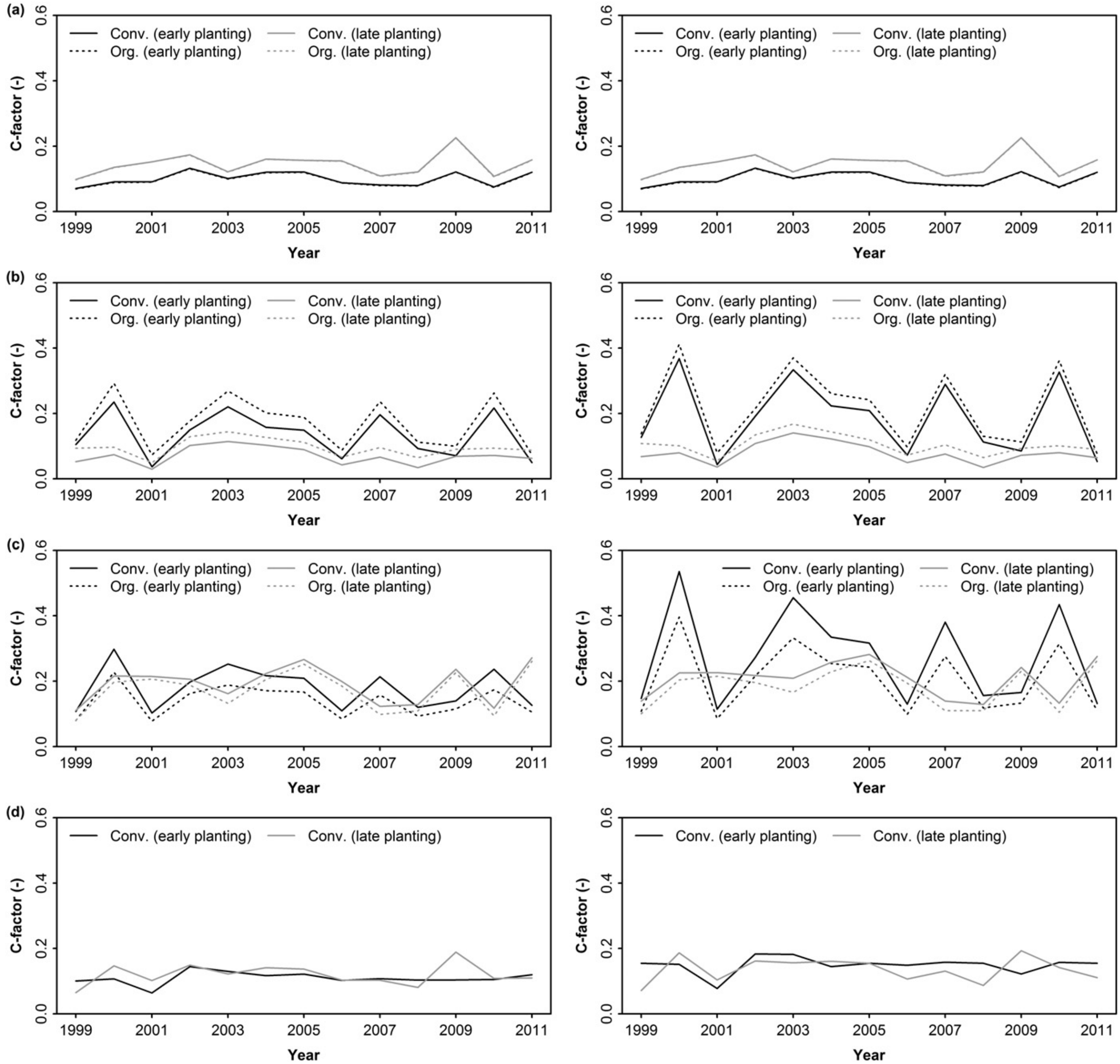

Fig. 7. Variation of the *C*-factor between 1999 and 2011 for conventional (conv.) and organic farming (org.) of the four major row crops, bean (a), potato (b), radish (c), and cabbage (d) for a low degree of disturbance (left) and a high degree of disturbance at harvest (right), and variable planting and harvest times. Early planting means two weeks before, and late planting two weeks after the observed planting and harvest dates of 2009.

schedule. No difference between conventional and organic bean farming was detected, in accordance with the similar measured crop and weed parameters.

For potato (Fig. 7b), maximum *C*-factors were calculated for the years 2000, 2003, 2007, and 2010, when erosive rain events occurred late in the year, at which stage the potato was already harvested and the canopy cover that protected the soil from raindrop impact was lost. For a high level of disturbance at harvest, a higher *C*-factor was calculated. High disturbance resulted in a reduction of surface cover due to the incorporation of plant residues. Although incorporated residues help to stabilize the soil by binding soil particles and increase soil aggregation (Renard et al., 1997), the lower surface cover had a much stronger effect on erosion. Surface cover affects erosion by reducing the area that is exposed to the impact of raindrops (Renard et al., 1997), but also by reducing the transport capacity of surface runoff and by causing deposition in ponded areas (Foster, 1982; Laflen, 1983). When residue cover was lost by disturbance through the harvest activities, soils became more susceptible to detachment of rainsplash and overland flow. A high level of disturbance affected primarily the early planting and harvest scenario. Potato was generally planted early in the year, which made potato fields more susceptible to late rainstorm events. Therefore, early planting and harvest resulted in much higher *C*-factors for all 13 years than a late schedule. Organic farming generally showed slightly higher *C*-factors than conventional farming, which can be explained by the lower crop biomass and surface cover by crop residue, which had a stronger effect than the higher weed cover and biomass of the organic system. The difference between organic and conventional farming was higher where a low level of disturbance occurred at harvest, as less crop residue, which can act as surface cover, was incorporated.

For radish (Fig. 7c), the years with maximum *C*-factors were different for the early and late planting and harvest scenarios. For early planting and harvest, peaks were calculated for the same years as for potato (2000, 2003, 2007, and 2010). For late planting and harvest, the highest *C*-factors were calculated for the years 2005, 2009, and 2011, when erosive rain events occurred early in the year. Also for radish, a high level of disturbance resulted in higher *C*-factors, affecting primarily the early planting and harvest scenario. Radish had a relatively short growing period compared to bean and potato, which made radish fields susceptible for those years with late rain events, if planting and harvesting occurred earlier, as well as for those years with early rain events, if planting and harvesting occurred late. On average, early planting and harvesting resulted in higher *C*-factors than a later planting and harvesting schedule. Contrary to potato, for radish, lower *C*-factors were calculated for organic than for conventional farming for all 13 years, due to the slightly higher crop biomass and cover, but especially as a result of the higher weed biomass and cover. The higher canopy cover of weeds for the organic system reduced the kinetic energy of raindrops falling onto the soil surface (Morgan, 2005; Renard et al., 1997). It has been discussed that raindrops that fall from leaves after interception could locally increase soil detachment for high growing canopies (Morgan, 2005). Weeds however were growing relatively low and closer to the ground compared to the crops and could therefore absorb some of the rainfall energy (Morgan, 2005), especially in the furrows. In addition, weeds could stabilize the soil surface against overland flow in the furrows by acting as mechanical barriers and binding soil particles with their roots (Benkobi et al., 1994; Renard et al., 1997). The difference between organic and conventional farming for radish was considerably higher where a high level of disturbance occurred at harvest and for the early planting and harvesting scenarios. The advantage of higher weed cover for organic farming was reduced by a high disturbance, but at the same time a large amount of residue was added to the soil from the high weed biomass pool, which increased the soil stability. Additionally, by the removal of the plastic film associated with the higher disturbance, a larger amount of soil was exposed, but the ratio of the remaining surface cover for organic farming became higher than before, due to higher residue coverage. For early planting and harvesting, this had a much stronger impact on the *C*-factor, because it affected only the late rainstorm events. On the contrary, for low disturbance levels in combination with late planting and harvesting, the differences between organic and conventional farming practices were almost negligible.

For cabbage (Fig. 7d), the years with the highest *C*-factors were also different between the early and late planting and harvesting scenarios. Cabbage had the shortest growing period of all, and was therefore affected by both early and late rain events. To what degree the rain events affected the *C*-factor was depending again on the planting and harvesting schedule. For early planting and harvesting, the highest values were calculated for the years 2002 and 2003, when rain events occurred late in the year. For late planting and harvesting, the maximum values were found for 2000, 2002, 2004, and 2009, when rain events occurred earlier (similar to bean). On average, the scenarios of early and late planting and harvest did not result in considerably different *C*-factors. The higher level of disturbance at harvest resulted in higher *C*-factors for cabbage as a result of the reduced surface cover.

The average annual *C*-factor calculated over all scenarios and years was the highest for radish with 0.202 for conventional farming, and 0.166 for organic farming. For bean, average annual *C*-factors of 0.121 and 0.120 were calculated for conventional and organic farming, respectively. For potato, the calculated *C*-factors were 0.113 for conventional, and 0.141 for organic farming. For conventional cabbage, the average annual *C*-factor was 0.128.

3.5. Soil erosion rates

According to the highest *C*-factor, radish also showed the highest average annual soil erosion rate over all 25 field sites (Table 2). The high erosion for radish can be explained by the relatively short growing period, the higher disturbance and lower amount of crop residue remaining on the field after harvest compared to the other crops. The growing period of cabbage was shorter than that for radish, but less disturbance took place due to above-ground harvesting, and a higher residue cover reduced the erosion risk. Potato required the highest disturbance at harvest, but due to the longer growing period, it provided a better soil protection than radish. Bean provided a high coverage due to a very long growing period, but because of the relatively late planting, fields were still vulnerable to early rainstorm events, which resulted in soil loss rates similar to those of potato and cabbage.

The mean annual soil loss of radish was reduced by 18% by organic farming (45.0 t ha^{-1} yr^{-1}) compared to conventional farming (54.8 t ha^{-1} yr^{-1}) due to the higher weed biomass density and weed cover at the end of the growing season, as a consequence of the absence of agricultural chemicals. Also the slightly higher crop biomass and coverage contributed to the lower soil loss rate. Nevertheless, our results demonstrate that the protective effect of weeds cannot sufficiently counteract the negative effects of the short growing period in combination with low residue and high disturbance, because the average erosion for organic radish was still higher than those of the other three crops. For potato, the soil loss rate was increased by 25% by organic farming (38.2 t ha^{-1} yr^{-1}) compared to conventional farming (30.6 t ha^{-1} yr^{-1}) due to a reduced crop biomass density and cover. Although, weed biomass and cover was increased by the absence of agricultural chemicals, the negative effects of a reduced crop yield had a more significant impact. However, our results also demonstrate that a reduced crop yield for potato as a possible consequence of crop–weed competition or herbivory associated with organic farming, does not dramatically increase erosion, because the average soil loss did not strongly exceed those of bean or cabbage, and was still lower than those of radish. For bean, no considerable difference between organic farming (32.5 t ha^{-1} yr^{-1}) and conventional farming (32.8 t ha^{-1} yr^{-1}) could be identified according to similar vegetation characteristics of crops and weeds for both farming systems.

The highest soil erosion rates among the 25 field sites were calculated for site 04 with values between 93.0 t ha^{-1} yr^{-1} (conventional potato) and 166.4 t ha^{-1} yr^{-1} (conventional radish). Site 04 was characterized by a relatively steep hillslope in combination with a high slope length (Table 1). The lowest erosion was calculated for site 10 with rates between 0.4 t ha^{-1} yr^{-1} and 0.7 t ha^{-1} yr^{-1}. Also the sites 08, 15, 23, and 24 showed similarly low soil loss rates. These sites were located in the center of the catchment and did not have considerable slope angles (Table 1).

3.6. Model plausibility

The comparison between the simulated and observed soil loss amounts from July to August 2010 showed a strong underestimation for site M1 and a slight overestimation for site M2 (Table 3). The rainfall erosivity calculated from rain gauge records during the measuring

Table 2
Simulated average annual soil loss for conventional and organic farming of the four major row crops in the Haean catchment. Mean, maximum, and minimum refer to the simulated soil loss over all 25 field sites.

Crop type	Management system	Average annual soil loss (t ha^{-1} yr^{-1})		
		Mean	Maximum	Minimum
Bean	Conventional	32.8	99.6	0.4
	Organic	32.5	98.7	0.4
Potato	Conventional	30.6	93.0	0.4
	Organic	38.2	116.1	0.5
Radish	Conventional	54.8	166.4	0.7
	Organic	45.0	136.7	0.6
Cabbage	Conventional	34.7	105.4	0.4

period in 2010 was lower for site M1 (363.9 MJ mm ha^{-1} h^{-1}) than for site M2 (588.2 MJ mm ha^{-1} h^{-1}). However, the simulated soil loss for site M1 (1.27 t ha^{-1}) was almost twice as much as the simulated soil loss of site M2 (0.71 t ha^{-1}) as a result of the higher *S*-factor and *C*-factor. Both observation sites had similar soil conditions, and soil texture was sandy loam, and organic matter content was 3.0% for both sites. The average slope lengths were also very similar for site M1 and M2 with 23.9 m and 25.1 m, respectively. Therefore, the calculated *K*-factor and *L*-factor were very similar for both sites. Site M1 (9.6°) was slightly steeper than M2 (8.1°), which resulted in a higher *S*-factor. The main difference between both sites was the lower crop cover during the observation period on site M1 (72%) compared to M2 (94%), which resulted in highly varying *C*-factors, and therefore a higher simulated soil loss for site M1. However, even though the RUSLE model produced a much higher erosion rate for site M1 compared to M2, the actual soil loss on M1 was still highly underestimated. This insufficient performance might be partially explained by the higher runoff generation associated with the plastic film covers of ridges, which cannot be adequately modeled by RUSLE. The model does not contain parameters that can be used to control the infiltration capacity as a result of an impermeable surface cover. Effects of the plastic mulch cultivation could be therefore only incorporated in the surface cover subfactor (*SC*) and roughness subfactor (*SR*). Arnhold et al. (2013) however showed that plastic mulch can considerably increase runoff generation and soil erosion. On site M1, they observed severe gully erosion generated by ridge breakovers as a consequence of accumulated surface runoff. Runoff was concentrated in the furrows and drained to the center of the field, where it formed a gully that was primarily responsible for the high soil loss rate on this field site (Arnhold et al., 2013).

The ^{137}Cs analyses carried out by Meusburger et al. (in preparation) revealed long-term soil loss rates of 9.1 t ha^{-1} yr^{-1} on the recently deforested site, and 41.8 t ha^{-1} yr^{-1} on the long-term farmland site. Although, our study was carried out on different field sites, the simulated erosion rates with RUSLE were within the range of the ^{137}Cs estimates. The lower erosion rate on the recently deforested site may be partially attributed to higher infiltration capacities and greater soil stability due to high organic matter contents that can be found in forest soils in the Haean catchment (Ruidisch et al., 2013). The long-term farmland site however showed an erosion rate that is very similar to the mean average annual soil losses from the fields in our study (Table 2). Also other erosion studies on dryland fields in the Kangwon Province show similar values. Jung et al. (2003) found an average erosion rate of 47.5 t ha^{-1} yr^{-1}, and Choi et al. (2005) reported erosion rates between 4.2 t ha^{-1} yr^{-1} and 29.6 t ha^{-1} yr^{-1} for potato, and 3.3 t ha^{-1} yr^{-1} and 81.6 t ha^{-1} yr^{-1} for radish.

Although the RUSLE model cannot accurately reproduce erosion processes associated with plastic mulch cultivation, the comparison to other studies in the Haean catchment and Kangwon Province show that the long-term simulated erosion rates were plausible. We had to assume a number of simplifications during the model parameterization, most notable prior and after the growing season. However, the simulated soil loss rates adequately reflected the actual annual erosion in this region, as erosive rain events were concentrated only in the monsoon season, and hence, the effects of weed growth after harvest and residue decomposition played only a marginal role.

4. Summary and conclusions

In this study we analyzed the effect of conventional and organic farming on soil erosion of row crop cultivation on mountainous farmland in South Korea. We measured multiple vegetation parameters of four major row crops (bean, potato, radish, and cabbage), as well as those of weeds from different fields of conventional and organic farms, and simulated the long-term soil loss using the Revised Universal Soil Loss Equation (RUSLE). The comparison of the model results to the observed soil erosion rates demonstrated an acceptable performance of RUSLE for row crop cultivation in this region. We found the highest erosion rate for radish due to the shorter growing period in combination with high soil disturbance at harvest and low amounts of remaining residue. Nevertheless, the simulated erosion rates for the other three crops were not considerably lower. Organic farming reduced soil loss for radish due to higher weed coverage, but increased erosion for potato due to lower crop yield.

These results demonstrate that the absence of agricultural chemicals, especially herbicides, in organic farming systems can reduce soil erosion for row crops due to the development of weeds in the furrows. However, our results also show that a reduced crop yield associated with crop–weed competition or herbivory outbalances the positive effects of weeds, and can therefore produce higher erosion rates in organic farming systems. Nevertheless, in both cases the difference in soil loss between the farming systems were relatively small, and the effects of weed coverage and crop yield were highly variable depending on the timing of planting and harvest in relation to the occurrence of rainstorm events, and the degree of soil disturbance. The simulated average annual soil loss for both management systems exceeded, by far, any tolerable soil loss rates. The OECD (2001) defined soil loss as tolerable when it is less than 6.0 t ha^{-1} yr^{-1}, and severe when it exceeds 33.0 t ha^{-1} yr^{-1}. The average annual erosion rate for all four row crops in this study was at least at the limit of severe erosion, and well above in many cases. Our results also show that the maximum erosion rates can be three to four times higher than the average values depending on field topography.

We can therefore conclude that neither farming system sufficiently lowers the amount of soil erosion of row crop cultivation on mountainous farmland. Although we identified a protective effect of a high weed coverage associated with the absence of herbicides, organic farming alone cannot be used to effectively control soil erosion. Both farming systems require additional conservation measures to prevent soil loss from row crop fields in this region. Especially after harvest, when soil is disturbed and ground cover is low, fields are very susceptible to erosion. The work of Kim et al. (2007) suggests that winter crop cultivation with ryegrass can be used to protect the soil after the growing season. However, the development of a high coverage that effectively reduces soil loss takes time and requires early sowing (Morgan, 2005). Soil protection may therefore be more effective in the following year, before seed bed preparation is carried out. The incorporation of ryegrass residue into the soil may provide additional beneficial effects on soil properties and crop yields, but requires further investigation (Kim et al., 2007). To improve the protection of the furrows during the growing period, Rice et al. (2007) suggested cereal grass cultivation to

Table 3
Rainfall erosivity, factors for the Revised Universal Soil Loss Equation, and simulated soil loss for the sites M1 and M2 in comparison to the observed soil loss measured during the monsoon season of 2010.

Site	EI_{30} (MJ mm ha^{-1} h^{-1})	*K*-factor (t h MJ^{-1} mm^{-1})	*L*-factor (−)	*S*-factor (−)	*C*-factor (−)	*P*-factor (−)	Simulated Soil loss (t ha^{-1})	Observed Soil loss (t ha^{-1})
M1	363.9	0.0286	1.047	2.310	0.055	0.917	1.27	3.65
M2	588.2	0.0275	1.075	1.856	0.024	0.911	0.71	0.63

increase the infiltration capacity and reduce runoff flow velocity. However, cultivating cover crops during the growing season could involve competition with the main crop, which could result in lower yields. Another very effective measure to prevent erosion is mulching with plant residues (Morgan, 2005). Our results show that surface cover by plant residue is more effective than the canopy cover provided by weeds. Plant residue can therefore be used to cover furrows instead of cultivating cover crops that may have negative impact on crop yield. Also Kim et al. (2007) found in their study that ryegrass residue mulching significantly reduces soil loss on row crop fields. We therefore recommend residue mulching during the growing season in combination with winter cover crops after harvest as additional conservation measures that would help for both conventional and organic farming to sufficiently prevent soil erosion for row crop cultivation on mountainous farmland.

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