Effectiveness of land use and soil conservation techniques in reducing runoff and soil loss in Europe and the Mediterranean

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“We can only see a short distance ahead,
but we can see plenty there that needs to be done.”

Turing, A.M., 1950. Computing Machinery and Intelligence.
Mind 59(236): 433-460.

Voorwoord

Beste lezer, zoals enkelen onder jullie waarschijnlijk weten moest ik regelmatig teruggefloten worden wanneer de teksten die na dit voorwoord volgen al te prozaïsch werden. Voor deze gelegenheid haal ik er echter toch een stukje proza (en zelfs fictie) bij om mijn wedervaren met het doctoreren op te tekenen.

> *Captain Robert Walton in "Letter 2":*
>
> *"There is something at work in my soul which I do not understand. I am practically industrious - painstaking, a workman to execute with perseverance and labour - but besides this there is a love for the marvellous, a belief in the marvellous, intertwined in all my projects, which hurries me out of the common pathways of men, even to the wild sea and unvisited regions I am about to explore."*
>
> *Mary Wollstonecraft Godwin - Shelley,*
> *Frankenstein; or, The Modern Prometheus*

Naast het feit dat een doctoraat schrijven inderdaad een wonderlijke en uitdagende, maar bij wijlen flink lastige ontdekkingsreis is, heb ik tijdens mijn doctoraatsonderzoek soms om nog andere redenen aan Mary Shelley's boek moeten denken: publicaties werden opgegraven uit hun wetenschappelijk graf in een archief, plotdata aan stukken gezaagd en weer aan elkaar genaaid, en het geheel leven ingeblazen gebruik makend van de ondoorgrondelijke krachten der statistiek. Ook het resultaat vertoonde bij wijlen gelijkenissen met het monster van Frankenstein: soms waren duidelijk de naden (lees bias) van het aan elkaar naaien zichtbaar, en soms was er zelfs sprake van een onherkenbaar gedrocht (lees scatter). Maar kijk, het leeft, en u heeft het nu in handen.

Tot zover de prozaïsche vergelijkingen, in tegenstelling tot een monster creëer je een doctoraat niet alleen. Ik wil dan ook de vele mensen bedanken die hebben bijgedragen tot dit doctoraat. In de eerste plaats wil ik mijn promotor, Jean Poesen, bedanken die me de kans gaf om onder zijn auspiciën onderzoek te doen en te doctoreren, en ook steeds bereid was mee ervoor te zorgen dat het resultaat nog net iets beter werd (en dan steeds opnieuw nog wat beter). Daarnaast was er ook Matthias, die de laatste vier jaar veelvuldig het lot beschoren was om voornoemd proza mee om te bouwen tot iets wetenschappelijks, maar ook goed was voor enkele dolle avonturen op conferenties en projectmeetings. I'd also like to thank Dino and Mauro who have been instrumental in the analysis in Chapter 5 of this manuscript and with whom I had a very nice time and discussions in Perugia. There are also the colleagues in the DESIRE project of which this PhD. research was part and with whom I shared many memorable moments at project meetings and conferences. Tot slot wil ik zeker ook nog de leden van de jury en An Carbonez van L-Stat bedanken voor het helpen dichten van de overgebleven spleten en kieren in dit manuscript.

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Dan zijn er nog de hele resem mensen die zeker een eervolle vermelding verdienen als zeer gewaardeerde "compagnons de route": de vrienden van de fiets, de mensen van de touwen, iedereen waarmee ik mij al eens regelmatig rond de (vergader)tafel terugvind, de Scoutties in het algemeen en een kuifke, iemand met een dakpannenfixatie, en een rabiate vegetariër in het bijzonder.

En tot slot, maar zeker niet in het minst, mijn familie aan wie ik plechtig beloof eindelijk eens (aanstalten te maken) om een eigen wasmachine te zoeken.

Willem
Leuven, maart 2013

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Abstract

Runoff and soil loss caused by water erosion are major drivers of soil and land degradation throughout Europe and the Mediterranean. This study aims to better understand and quantify (1) plot-scale annual runoff (R_a) and annual soil loss (SL_a), (2) the factors that control R_a and SL_a over the wide range of environmental conditions in Europe and the Mediterranean, and (3) the effectiveness of different land use types and soil and water conservation techniques (SWCTs) in reducing R_a and SL_a. A more detailed discussion of the knowledge gaps with respect to plot-measured R_a and SL_a in Europe and the Mediterranean, and the specific objectives for this research is given in chapter 1.

A database of R_a and SL_a data measured throughout Europe and the Mediterranean on bounded runoff and soil loss plots under natural rainfall, with a measuring period (representative for) at least one year was compiled from the literature. The resulting database contains plot data from 1 409 plots, corresponding to 9 297 plot-years from 239 plot-measuring stations throughout Europe and the Mediterranean. The database contains R_a data for 804 plots (corresponding to 5 327 plot-years) and SL_a data for 1 056 plots (corresponding to 5 327 plot-years) under conventional land management practice (see chapter 2). Furthermore, also R_a data for 287 plots (corresponding to 1 713 plot-years) and SL_a data for 356 plots (corresponding to 2 035 plot-years) where SWCTs were tested were collected (see chapter 7). This study is both the largest compilation of plot SL_a data in Europe and the Mediterranean to date, and the first to systematically include R_a data and data on plots where SWCTs are tested. A detailed discussion of the data included in the database is given in chapter 2 and chapter 7.

Several knowledge gaps with respect to the effect of land use type on R_a and SL_a over the whole of Europe and the Mediterranean are addressed in chapter 3. The analysis confirmed the important control of vegetation cover on R_a and SL_a rates, with marked differences in both R_a and SL_a between cultivated land (i.e. cropland, fallow plots, vineyards, tree crops), and semi-natural vegetation

(i.e. shrubland, rangeland, forest, post-fire and grassland) for the whole of Europe and the Mediterranean. Generally, there is a good correspondence between R_a and SL_a for the different land use types, but at the regional scale, differences were found between R_a and SL_a rates for different climatic zones. Mean SL_a values were smaller in the Mediterranean than in temperate and cold climatic zones, and mean annual runoff coeffient (RC_a) rates were generally higher in the cold climatic zone than in the temperate and Mediterranean zones for similar land use types. Nevertheless, each land use type also comprises a wide variability in plot-measured R_a and SL_a, and only weak relations were found between R_a and SL_a and other environmental factors that are generally considered important determinants of R_a and SL_a at the local scale such as plot length and slope gradient, indicating that these factors explain only a small part of the large variability in R_a and SL_a that is observed at the continental scale.

Part of the large uncertainty associated with plot R_a and SL_a rates is associated with temporal variability. To explore the inter-annual variability in plot R_a and SL_a rates, an analysis of 234 R_a and 307 SL_a time-series with measuring periods equal to or longer than five years is performed in chapter 4. Temporal variability of RC_a and SL_a rates were shown to be related, but temporal variability in RC_a is generally smaller than temporal variability in SL_a. This was confirmed by a Monte-Carlo analysis which indicated that compared to SL_a, shorter measuring periods are needed for plot measurements of RC_a to allow the expected long-term mean RC_a to be estimated with a specified degree of certainty. Nevertheless, uncertainties with respect to the estimation of long-term mean RC_a and SL_a can be large even after long measuring periods (i.e. 30 years). Closer examination of several environmental factors (i.e. climatic zone, land use type, plot length, slope gradient and annual precipitation) showed that these factors explain little temporal variability, and indicate that a large portion of the observed variability may indeed be random. Furthermore, there are substantial differences between temporal variability in plot-measured SL_a and catchment sediment yield, and a better understanding of these differences can improve our understanding of differences in erosion processes between these spatial scales.

Plot-measured R_a (804 plots corresponding to 5 327 plot-years) has received much less attention than SL_a (1 056 plots corresponding to 7 204 plot-years) in Europe and the Mediterranean, both with respect to the reported data, as with respect to the analysis of R_a data at a continental scale. Therefore, a closer analysis of the annual rainfall (P_a) - annual runoff (R_a) relation is presented in chapter 5. In this assessment, two simple models were used; a linear mixed effects model and a modified Curve Number Method, adapted for annual data. Fitting of the models showed the important controlling effect of both land use type and soil texture (as expressed by the Hydrologic Soil

Group) on the P_a-R_a relation. Contrary to expectations, fine-textured soils generally did not show the highest runoff response, which was attributed to the cracking behaviour of some clayey soils. An important effect of intra-annual precipitation distribution was expected, but this could only be demonstrated through simulation and not in the plot-measured data.

A confrontation of the plot SL_a data with two models for the estimation of interrill and rill erosion at a continental scale (i.e. the process-based PESERA model and empirical SEM model) in chapter 6 showed that there is a large variability in the relation between predicted and observed SL_a data. This variability is partly attributed to temporal variability due to the fact that these models predict long-term mean SL_a. Both models tend to under-predict SL_a values for the Continental climatic zone, which is attributed to the fact that important processes such as freeze-thaw cycles and snowmelt erosion are not accounted for. Furthermore, improvements to both models can be made by using a land cover classification that is specifically designed for erosion studies, rather than the more general CORINE land covers.

While it was shown in chapter 3 that land use management can be a very effective means of controlling both R_a and SL_a rates, this is not always possible. In these instances, specific soil and water conservation techniques (SWCTs) are used to reduce R_a and SL_a. A review of the effectiveness of different SWCTs in reducing plot-scale R_a and SL_a in Europe and the Mediterranean is presented in chapter 7. This analysis showed that most SWCTs are on average more effective in reducing SL_a than in reducing R_a. Furthermore, the importance of vegetation cover as a factor controlling R_a and SL_a was further confirmed by the finding that crop and vegetation management (i.e. buffer strips, mulching, cover crops) are more effective in reducing R_a and SL_a than soil management techniques (i.e. no-tillage, reduced tillage, contour tillage). However, the effectiveness of individual SWCTs in reducing R_a and SL_a was found to be highly variable, suggesting several controlling factors that are unaccounted for. An important effect of the R_a and SL_a rate measured on control plots with conventional treatment was found, and especially for smaller R_a and SL_a rates, effectiveness of the SWCTs was more variable. Effects of environmental factors such as plot length, slope gradient or P_a on SWCT effectiveness could not be clearly identified. Analysis of the temporal variability of SWCTs showed that there is considerable inter-annual variability in the effectiveness of conservation tillage techniques. With respect to runoff reduction, the effectiveness of no-tillage techniques tends to decrease over the years.

Finally, chapter 8 gives a synthesis of this research, along with a discussion of possibilities for further research.

Samenvatting

Waterafvoer (Eng.: runoff) en bodemverlies (Eng.: soil loss) door erosie zijn overal in Europa en het Middellandse Zeegebied belangrijke oorzaken van bodem- en landdegradatie. Dit onderzoek heeft tot doel het beter begrijpen en kwantificeren (1) van de natuurlijke processen die leiden tot jaarlijkse afvoer (R_a) en jaarlijks bodemverlies (SL_a), (2) van de factoren die een invloed hebben op deze processen in Europa en het Middellandse Zeegebied, en (3) van de effectiviteit van verschillende landgebruikstypen en bodem- en waterconserverende maatregelen (Eng.: Soil and Water Conservation Techniques, SWCTs) in het reduceren van R_a en SL_a. Hoofdstuk 1 geeft een gedetailleerde discussie van de onderzoeksvragen en objectieven in dit onderzoek.

Door middel van een uitgebreid literatuuronderzoek werd voor Europa en het Middellandse Zeegebied een database samengesteld van R_a en SL_a data gemeten op afvoer- en bodemverliesplots onder natuurlijke neerslag met een meetperiode die representatief is voor minstens één jaar. Deze database bevat data voor 1 409 plots (9 297 plot-jaren) afkomstig van 239 plot-meetstations in Europa en het Middellandse Zeegebied. De database bevat R_a data voor 804 plots (5 327 plot-jaren) en SL_a data voor 1 056 plots (5 327 plot-jaren) onder conventioneel landgebruik (hoofdstuk 2). Daarnaast werden ook R_a data verzameld voor 287 plots (1 713 plot-jaren) en SL_a data voor 356 plots (2 035 plot-jaren) op plots waar SWCTs werden toegepast (hoofdstuk 7). Een gedetailleerde bespreking van de databases wordt gegeven in hoofdstuk 2 en hoofdstuk 7.

De effecten van verschillende landgebruikstypes op R_a en SL_a voor Europa en het Middellandse Zeegebied worden besproken in hoofdstuk 3. Deze analyse bevestigt het belangrijke effect van vegetatiebedekking op R_a en SL_a, met duidelijke verschillen in zowel R_a als SL_a tussen landbouwpercelen (i.e. akkerland, braakliggende percelen, wijngaarden en boomgaarden), en percelen met een semi-natuurlijke bedekking (i.e. struikgewas, graaslanden, bos, percelen waar een (bos)brand plaatsvond, en grasland). Over het algemeen werd een goede correlatie gevonden tussen R_a en SL_a voor de

verschillende landgebruikstypes. Op regionale schaal werden evenwel verschillen gevonden tussen verschillende klimaatzones. Gemiddelde SL_a was lager in het Middellandse Zeegebied dan in de gematigde en koude streken, en de gemiddelde jaarlijkse afvoercoëfficiënten (Eng: annual runoff coefficient, RC_a) voor gelijkaardige landgebruikstypes waren doorgaans groter in de koude klimaatzones dan in de gematigde zone en het Middellandse Zeegebied. Desalniettemin is er een grote variabiliteit in gemeten R_a en SL_a waarden tussen verschillende landgebruikstypen. Toch werden er slechts zwakke relaties gevonden tussen R_a en SL_a en andere factoren zoals plotlengte en hellingsgraad die algemeen beschouwd worden als lokaal belangrijke controlerende factoren voor R_a en SL_a. Dit wijst erop dat op de continentale schaal deze lokale factoren slechts een deel van de waargenomen variabiliteit kunnen verklaren.

Om de inter-jaarlijkse variabiliteit in R_a en SL_a metingen nader te onderzoeken werd in hoofdstuk 4 een analyse uitgevoerd van 234 R_a en 307 SL_a tijdsreeksen met meetperioden langer dan of gelijk aan vijf jaar. De temporele variabiliteit in RC_a en SL_a waren gerelateerd aan elkaar, maar de temporele variabiliteit in RC_a is kleiner den de temporele variabiliteit in SL_a. Een Monte-Carlo analyse toonde verder aan dat om plotmetingen te bekomen met een bepaalde afwijking op het verwachtte lange-termijngemiddelde een kortere meetperiode nodig was voor RC_a metingen dan voor SL_a metingen. Desondanks kan de onzekerheid met betrekking tot de geschatte lange-termijngemiddelde RC_a en SL_a waarden zelfs voor lange meetperiodes (i.e. 30 jaar) nog steeds groot zijn. Verschillende onderzochte omgevingsvariabelen zoals klimaatzone, landgebruikstype, plotlengte, hellingsgraad en jaarlijkse neerslag konden slechts weinig variabiliteit in de geobserveerde data verklaren. Dit kan erop wijzen dat een groot gedeelte van de jaarlijkse variabiliteit willekeurig is. Temporele variabiliteit in SL_a op plotschaal vertoont enkele belangrijke verschillen met temporele variabiliteit waargenomen op bekkenschaal. Een beter begrip van deze verschillen kan ook een beter inzicht geven in de aandelen van verschillende erosieprocessen op verschillende ruimtelijke schalen.

Op plotschaal heeft R_a (804 plots, 5 327 plot-jaren) minder aandacht gekregen in Europa en het Middellandse Zeegebied dan SL_a (1 056 plots, 7 204 plot-jaren), zowel wat betreft het aantal gemeten data en de analyse die op de beschikbare data gebeurd is. Daarom werd de jaarlijkse neerslag (P_a) - jaarlijkse afvoer (R_a) relatie nader onderzocht in hoofdstuk 5. Daartoe werden twee relatief eenvoudige modellen gebruikt; een lineair gemengde-effectenmodel en een variant van de “Curve Number Method”, aangepast voor jaarlijkse data. Toepassing van deze modellen op de verzamelde data toonde de belangrijke effecten van landgebruik en bodemtextuur (uitgedrukt als Hydrologische Bodemgroep) op de P_a-R_a relatie aan. In tegenstelling tot de verwachtingen waren kleiige bodems niet altijd geassocieerd met de sterkste afvoerrespons, wat kan toegeschreven worden aan de eigenschap van sommige kleibodems om “cracks” te vormen wanneer ze uitdrogen. Ook een belangrijk effect van de neerslagverdeling binnen de individuele jaren werd verwacht, maar dit kon enkel aangetoond worden door middel van simulaties, en werd niet waargenomen in de verzamelde data.

De confrontatie van de verzamelde SL_a-data met twee modellen die intergeul- en geulerosie voorspellen op een continentale schaal (i.e. het proces-gebaseerde PESERA model en het empirische SEM model) in hoofdstuk 6 toont aan dat er een grote variabiliteit bestaat tussen het voorspelde en het waargenomen SL_a. Dit wordt ten dele verklaard door het feit dat beide modellen gemiddelde SL_a waarden voor de lange termijn voorspellen. Beide modellen onderschatten SL_a waarden voor de continentale klimaatzone wat kan toegeschreven worden aan het ontbreken van belangrijke processen zoals vries-dooi-cycli en erosie door smeltende sneeuw in beide modellen. Beide modellen kunnen verbeterd worden door het gebruik van een landgebruikskaart specifiek voor het voorspellen van erosie, in plaats van de meer algemene CORINE landgebruikskaart die nu in de modellen gebruikt wordt.

Het potentieel van landgebruiksbeheer om R_a en SL_a te reduceren werd aangetoond in hoofdstuk 3, maar drastische ingrepen in het landgebruik zijn niet altijd mogelijk. In deze gevallen kunnen bodem- en waterconserverende maatregelen (SWCTs) gebuikt worden om R_a en SL_a te reduceren. In hoofdstuk 7 werd een studie gemaakt van de effectiviteit van verscheidene SWCTs in het reduceren van R_a en SL_a op plotschaal in Europa en het Middellandse Zeegebied. Deze analyse toonde aan dat de meeste SWCTs effectiever zijn in het reduceren van SL_a dan in het reduceren van R_a. Ook het belang van vegetatiebedekking werd verder geïllustreerd in deze studie: SWCTs die gebruik maken van een verbeterde vegetatiebedekking (i.e. grasbufferstroken, mulsen, groenbedekkers) zijn effectiever in het reduceren van R_a enSL_a dan technieken met (enkel) gewijzigde bodembewerking (i.e. niet-kerende bodembewerking, contourploegen). De variabiliteit in de effectiviteit van al deze technieken in het reduceren van R_a enSL_a is echter sterk variabel, wat erop wijst dat nog andere factoren een belangrijke rol spelen in de effectiviteit van deze SWCTs. Er werd een belangrijk effect van de absolute hoeveelheid gemeten SL_a gevonden, en de variabiliteit in de effectiviteit van SWCTs was vooral hoog bij kleine gemeten SL_a waarden. Andere factoren zoals plotlengte, hellingsgraad of P_a vertoonden slechts een zwakke, of helemaal geen relatie met de effectiviteit van de verschillende SWCTs. Verder werd aangetoond dat de effectiviteit van niet-kerende bodembewerkingsmethoden en contourploegen sterk variabel is in de tijd. De effectiviteit van niet ploegen in het reduceren van R_a neemt af in de loop van de jaren wanneer deze techniek continu wordt toegepast.

Tot slot wordt in hoofdstuk 8 een synthese van dit onderzoek gegeven, samen met een bespreking van mogelijkheden voor verder onderzoek.

Abbreviations and symbols

abbreviation/ symbol	unit	explanation
PL	dimensionless	number of plots
PY	dimensionless	number of plot-years
(R)USLE		(Revised) Universal Soil Loss Equation
A		Atlantic climatic zone (LANMAP2)
$a_{2,cl}$	dimensionless	climate-specific parameter for the modified CN method regression
ARS		Agricultural Research Service (USDA)
B		Boreal climatic zone (LANMAP2)
Ba		bare
Bs		buffer strips
C		Continental climatic zone (LANMAP2)
Cb		contour bunds
Cc		cover crops
C-factor	dimensionless	(R)USLE cover management factor
CN	dimensionless	Curve Number (value)
CN_a	dimensionless	annual Curve Number (value)
CN_d	dimensionless	daily Curve Number (value)
CORINE		Coordination of information on the environment
Cr		cropland
Cs		construction sites
Ct		contour tillage
CV		coefficient of variation
Dr		drainage
Dt		deep tillage
Ex		exclosure
Fa		fallow
Fo		forest
Gr		grassland
Gt		geotextile
HC		Hydrologic Condition (NEH4, 2004)

abbreviation/ symbol	unit	explanation
HSG		Hydrologic Soil Group (NEH4, 2004)
k_1, k_2	dimensionless	modified CN method regression parameters
$k_{2,cl}$	dimensionless	climate-specific parameter for the modified CN method regression
LANMAP2		Landscapes of Europe Map, version 2
LISEM		LImburg Soil Erosion Model
L_{PL}	dimensionless	slope length factor (Wischmeier and Smith, 1978)
LS_{PL}	dimensionless	topographic factor (product of LPL and SPL)
LU		land use type
M		Mediterranean climatic zone (LANMAP2)
MFI	$mm^2 \cdot mm^{-1}$	Modified Fournier Index (Gabriëls, 2006)
MP	years	measuring period
Mu		mulching
MS		plot measuring site
NA		not applicable / not available
NRCS		Natural Resource Conservation Service (USDA)
Nt		no-tillage
P	mm	precipitation
P_a	$mm \cdot yr^{-1}$	annual precipitation depth
PCI	dimensionless	precipitation concentration index (Martin-Vide, 2004)
P_d	$mm \cdot day^{-1}$	daily precipitation depth
PESERA		Pan-European Soil Erosion Risk Assessment
Pf		post-fire
P-factor	dimensionless	(R)USLE support practice factor
PL		plot
p_m	$mm \cdot month^{-1}$	monthly precipitation depth
P_T	$mm \cdot year^{-1}$	threshold annual precipitation
PY	years	plot-years
R	mm	runoff
Ra		rangeland
R_a	$mm \cdot yr^{-1}$	annual runoff depth
RC	%	runoff coefficient
RC_a	%	annual runoff coefficient
RC_d	%	daily runoff coefficient
R_d	$mm \cdot day^{-1}$	daily runoff depth
R_{diff}	dimensionless	Relative difference between model-predicted SL_a and plot-measured SL_a (Nearing et al., 1999)
RR	dimensionless	runoff ratio
r_p	-	Pearson's correlation coefficient
r_s	-	Spearman's rank correlation coefficient
Rt		reduced tillage
S		Steppic climatic zone (LANMAP2)
Sa		soil amendment
S_a	dimensionless	annual S-number (value)
S'	dimensionless	area enclosed by the Lorentz-curve and equidistribution line

abbreviation/ symbol	**unit**	**explanation**
Sc		strip cropping
S_{cl}	dimensionless	climate-specific S-number (value) for the modified CN method regression
SCS		Soil Conservation Service (now USDA-NRCS)
S_d	dimensionless	daily S-number (value)
SEM		Soil Erosion Map (Cerdan et al., 2010)
Sh		shrubland
SL	$Mg{\cdot}ha^{-1}$	soil loss
SL_a	$Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$	annual soil loss
SLM		sustainable land management
SLR	dimensionless	soil loss ratio
SL_u	$Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$	unit plot soil loss
SOM	%	soil organic matter
S_{PL}	dimensionless	slope gradient factor (Nearing, 1997)
SWAT		Soil and Water Assessment Tool
SWCT		soil and water conservation technique
T		Anatolian climatic zone (LANMAP2)
Tc		tree crops
Te		terraces
USDA		United States Department of Agriculture
Vi		vineyard
WaTEM/SEDEM		Water and Tillage Erosion Model / Sediment Delivery Model
WEPP		Water Erosion Prediction Project
Z		Alpine climatic zone (LANMAP2)
α	dimensionless	significance level
γ	dimensionless	$\log_{10}$-ratio of model-predicted to plot-measured SL_a (Eq. 6.2)
ΔRC_a	%	absolute reduction of RC_a by SWCT application
ΔSL_a	$Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$	absolute reduction of SL_a by SWCT application
θ	degrees	slope gradient
λ	dimensionless	Curve Number method Lambda value

Glossary

terminology	explanation
annual	The value of a measured variable (runoff, soil loss, precipitation,...) corresponding to yearly data. Either the measurement period is one year, or the value has been averaged or extrapolated to correspond to yearly data.
(runoff and soil loss) plot	Experimental set-up consisting of a sediment source area that is hydrologically isolated by a border that is not permeable for runoff and transported sediment. The runoff water and transported sediment are collected at the bottom of the plot.
plot measuring station	Geographic location of the experimental station with a single plot or a set of different (adjacent) plots.
plot-year	Data corresponding to the measurement of runoff and/or soil loss for a measuring period of one year on a single plot.
measuring period	The period during which plot measurements have been carried out, i.e. the length of time (years) between the start and end dates of data recording on active plots.
replicate plots	Plots at the same plot measuring station and hence the same environment (slope gradient, soil, depth of natural precipitation) and where the experimental conditions (plot length, treatment,...) are replicated. Nevertheless, some variability can not be excluded (e.g. small variations in plot microtopography, soil,...)
conventional practice	Treatment or operations that are the local customary practice for that specific land use and crop type, without application of SWCTs. Plots with conventional practice are used as reference against which the effectiveness of SWCTs is evaluated.
unit plot	Standard (R)USLE plot; a runoff and soil loss plot having a plot length of 22.13m and a slope gradient of 9%

Contents

Chapter 1

Introduction

1.1 Land degradation and desertification

1.1.1 Definitions and impacts

Soil and land degradation

The ongoing and worldwide rise in population pressure and living standards has prompted increasingly intensive and large-scale use of all natural resources, including the soil (Millenium Ecosystem Assessment, 2005a; Schröter et al., 2005). Soil is the mainstay of agriculture as it acts as a plant growth medium and repository for nutrients and water, but it is also closely linked to several other ecosystem services (e.g. climate regulation, buffer to several disturbances such as drought and flooding, production of raw materials such as lumber and fodder, refugium for several species) that are essential for peoples' livelihood across the globe (Costanza et al., 1997; Dominati et al., 2010). The intensive use of these soil functions causes widespread soil degradation, which in turn is an important driver of land degradation and desertification. In this respect, soil degradation is defined as "a decline in the productive capacity of the soil as a result of soil erosion and changes in the hydrological, biological, chemical and physical properties of the soil" (Douglas, 1994). These processes have both environmental and socio-economic consequences far beyond the soil alone (Matson et al., 1997; Swinton et al., 2007). In addition to causing a decline in the productive capacity of the soil, soil degradation will also have a negative impact on other ecosystem services provided by the soil. While land degradation has no generally accepted definition, an irreversible decline of biological potential of the land and an important anthropogenic cause are essential aspects of land degradation (Eswaran et al., 2001). While these processes can and do have natural drivers, it is the unprecedented anthropogenic impact on the environment which accelerates soil and land degradation (Montgomery, 2007a), which will finally result in a loss of productive capacity if the process remains unchecked for long periods of time.

Based on remote-sensed NDVI data, Bai et al. (2008b,a) found that in the last 24 years, 24% of global land suffered from land degradation, affecting 1.5 billion people (Bai et al., 2008b). Analysis of remote-sensed data showed that land degradation currently mainly occurs in southern Africa, Indo-China, Myanmar, Malaysia and Indonesia, south China, north-central Australia, the Pampas and high-latitude forests. This is considerably different from the traditional picture of land degradation based on the GLASOD approach (e.g. Oldeman et al., 1991), which pinpointed the Mediterranean, Middle East and south and central Asia as land degradation hotspots (Bai et al., 2008b). GLASOD has long been the only global assessment of land degradation, therefore having been very influential in the development of the general perception of land degradation. The limitations of the GLASOD expert-based approach were recognised by the original authors, but this caveat was lost in later use of the approach. In recent-years, GLASOD has been shown to be flawed (Sonneveld and Dent, 2009) as compounding the effects of land degradation from recent centuries and ongoing processes (Bai et al., 2008b,a). Nevertheless, alternative global assessments of land degradation are currently still lacking, although recent advances have been made using remote-sensed data (Bai et al., 2008b,a; Lobell, 2010; King et al., 2005).

Hence, degraded land is indeed a global problem, but the process is not necessarily active around the world and there is still a need for an assessment of the global extent of the problem. Nevertheless, land degradation from previous centuries can still have a considerable impact on soil productivity and ecosystem services today, even if there is no ongoing soil degradation.

Desertification

The exact definition and interpretation of desertification has long been debated (Herrmann and Hutchinson, 2005; Hutchinson, 1996; Thomas, 1997; Thomas and Middleton, 1994) and the definition, severity and even the very existence of desertification has been criticized (e.g. Thomas and Middleton, 1994). The concept has evolved from the perception of continuously expanding deserts, especially a southwards encroachment of the Sahara into the Sahel (Lamprey, 1988), to a temporally and spatially complex process affecting drylands worldwide (D'Odorico et al., 2012; Reynolds et al., 2007; Thomas and Middleton, 1994).

Currently, the most widely used definition of desertification is: "land degradation in arid, semi-arid and dry subhumid areas resulting from various factors, including climatic variations and human activities." (Millenium Ecosystem Assessment, 2005b; UNCCD, 2011). Hence, what sets desertification apart within the more general term land degradation is that it occurs in drylands, which cover about 41% of the Earth's land (Reynolds et al., 2007). These drylands have been shown to be highly dynamical ecosystems (e.g. Helldén, 1991; Nyssen et al., 2009; Tucker et al., 1991) with strong vegetation responses to climatic variations (Evans and Geerken, 2004) and feedbacks between vegetation and land degradation processes (D'Odorico et al., 2012). Degradation of vegetation during drier years caused by natural climatic variations should not be considered desertification, but drylands may be more sensitive to degradation during these dry years as otherwise sustainable land use can have degrading effects in these periods by reducing the recovery potential of the land after drought (Thomas, 1997). Furthermore, strong population growth has decreased the margin to cope with declines in the agricultural production capacity of the land, irrespective of whether these declines are caused by direct human impact (i.e. overexploitation of natural resources), natural climatic variations intrinsic to drylands or human induced climatic change.

Where mitigation of land degradation in temperate regions mostly focuses on ensuring that the human impact does not compromise the sustained long-term productive capacity of the land, mitigation of desertification additionally includes taking an important natural variability of this productive capacity into account. This means that addressing land degradation in dryland regions should not limit itself to only reduce currently existing human impacts on the land to sustainable levels. In addition, also the resilience against both short- to medium-term variations in production capacity due to natural climatic variations, as well as the longer term effects of anthropogenic climate change (e.g. Nearing et al., 2004) needs to be built up.

To assess desertification risks in Europe and the Mediterranean, several sets of indicators have been developed (Kosmas et al., 2003, 1997; Rubio and Bochet, 1998). An extensive set of 148 indicators was compiled in the DESERTLINKS project, resulting in the DIS4ME indicator system (DESERTLINKS, 2004) which has been used in several scientific projects (DESERTNET, 2008; DESIRE, 2007; LADA, 2010; LUCINDA, 2008). This indicator system is designed to provide information in simple form, which can be collected easily, and can be used to map the extent and severity of ongoing desertification and provide insight in the causative processes. Of the 148 indicators used in the DIS4ME database, 71 relate to soil erosion by water and mostly focus on on-site erosion (Vanmaercke et al., 2011a). The large number of indicators related to on-site soil erosion by water underlines the importance of this problem, but also fails to address the complex effects of desertification at larger scales than the plot or hillslope (i.e. the catchment scale, Vanmaercke et al., 2011a).

Soil erosion by water

Soil erosion by water encompasses several often related processes of soil degradation caused by the detachment and transport of soil particles by rainfall, overland flow or subsurface flow (Boardman and Poesen, 2006). These processes include splash erosion (e.g. Eldridge and Greene, 1994; Moeyersons and De Ploey, 1976; Poesen, 1986a), interrill and rill erosion (Auerswald et al., 2009; Cerdan et al., 2006, 2010), gully erosion (e.g. Poesen et al., 2003, 2006) (e.g. Poesen et al., 2003; Poesen et al., 2006) and piping erosion (e.g. Faulkner, 2006; Verachtert et al., 2011).

Soil erosion by water can be greatly aggravated by human activity as it is tightly linked with agriculture (Cerdà et al., 2009; Montgomery, 2007a). It is one of the main causative processes of soil degradation and hence also land degradation and desertification. Soil erosion by water has important environmental and socio-economical impacts, both on-site and off-site. On-site impacts range from loss of nutrients and associated productivity decline (Bakker et al., 2004, 2007; Eswaran et al., 2001; Pierce et al., 1983) to land losing its ecosystem service functions altogether (e.g. becoming impassable or impossible to cultivate due to gully development (Poesen et al., 2006). Off-site, soil erosion by water is a major source of non-point source pollutants and causes several problems such as sedimentation of reservoirs, deterioration of water quality and flooding (Owens et al., 2005; Vanmaercke et al., 2011a; Verstraeten and Poesen, 1999; Verstraeten et al., 2006a). Through its effects on soil structure and (micro)topography, soil erosion by water also affects surface storage capacity of water, infiltration rates and runoff rates (Connolly, 1998). In most erosion studies, these runoff processes are studied primarily to better understand or predict soil loss (SL) or export of pollutants (e.g. nutrients or pesticides). Nevertheless, with respect to land degradation and desertification, water loss through runoff is an important issue in its own right as water is a key resource (Rockström et al., 2010; Wallace, 2000), especially in drylands facing desertification.

1.1.2 Status quo in Europe and the Mediterranean

The extent of soil erosion by water in Europe and the Mediterranean

Europe and the Mediterranean (i.e. the southern part of the European continent and the countries bordering the Mediterranean Sea) have some of the most agriculturally productive soils in the world (European Commission, 2012), but nevertheless face a considerable problem of soil and land degradation (Louwagie et al., 2009b). There are strong indications that the Mediterranean region has been subject to severe land degradation caused by soil erosion for long periods during the Holocene. This is attributed to the combination of a seasonal climate, steep topography, and a long history of intensive human disturbance (Collins et al., 2010; Montgomery, 2007b; Vita-Finzi, 1969). Currently, the Mediterranean is characterised by often shallow soils with a high rock fragment cover (Poesen and Lavee, 1994; Poesen et al., 1994; Yaalon, 1997) and sclerophyllous vegetation. Nevertheless, large areas in the Mediterranean are intensively cultivated (e.g. olive groves and vineyards), often with land use types that are prone to land degradation (Cerdan et al., 2010).

The most prominent driver of soil degradation in Europe and the Mediterranean is soil erosion by water. An estimated 115 million ha or 12 % of the European land area is affected by soil degradation through water erosion (EEA, 1995). In this study, the focus is on on-site soil erosion by interrill and rill erosion. While other processes may be a more important source of sediment in specific regions (Poesen et al., 2003; Vanmaercke, 2012; Vanmaercke et al., 2011b; Verachtert et al., 2011), interrill and rill erosion occur to some extent throughout Europe (Fig. 1.1) and are often important contributing factors to, or the initial stages of the other water erosion processes (e.g. gully development).

Several attempts have been made to assess or quantify interrill and rill erosion rates for the whole of Europe and (parts of) the Mediterranean (e.g. Cerdan et al., 2006, 2010; Jagu et al., 2007; Kirkby et al., 2004; Pimentel et al., 1995; Van Oost et al., 2007; Verheijen et al., 2009). Cerdan et al. (2010) estimated mean annual soil loss (SL_a) by interrill and rill erosion for the whole of Europe to be 1.2 $Mg \cdot ha^{-1} \cdot yr^{-1}$ on average, and 3.6 $Mg \cdot ha^{-1} \cdot yr^{-1}$ in arable land (Fig. 1.1). SL_a rates in the Mediterranean were found to be generally less than in the temperate regions of Europe. These values are much lower than the alarmingly

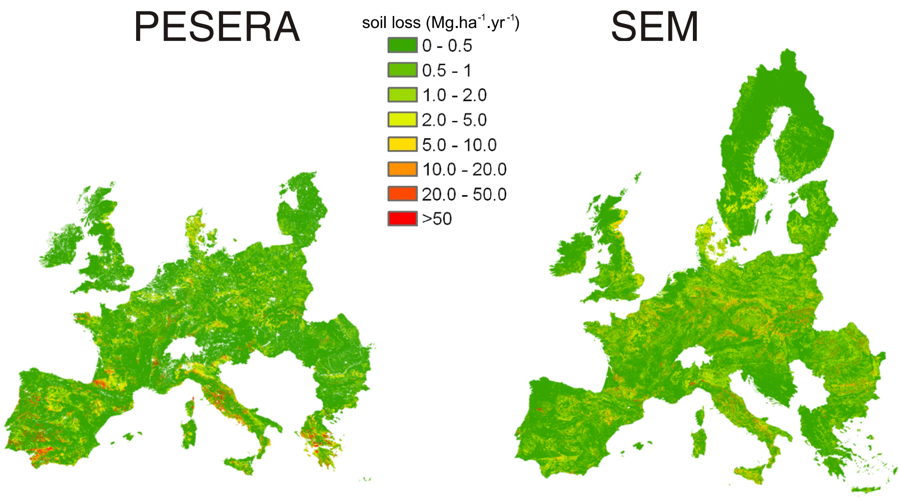


Figure 1.1: Pan-European estimates of soil loss (Mg·ha^{-1}·yr^{-1}) by interrill and rill erosion; the Pan-European Soil Erosion Risk Assessment (PESERA) map (Kirkby et al., 2004), and the European Soil Erosion Map (SEM) (Cerdan et al., 2010). Both maps have been redrawn to the same scale and use the same classification.

high rate of 17 Mg·ha^{-1}·yr^{-1} proposed in Pimentel et al. (1995). This was based on the extrapolation of only a few data however, and cannot be held representative as a mean SL_a value for the whole of Europe (Boardman, 1998). Nevertheless, SL_a rates above 15 Mg·ha^{-1}·yr^{-1} are no exception in Europe and the Mediterranean (Fig. 1.1, Cerdan et al., 2010, Kirkby et al., 2004).

In summary, severe SL_a rates by interrill and rill erosion are relatively frequently observed in Europe and the Mediterranean, but they appear to be localised in time and space and are the result of a concurrence of specific conditions, e.g. an exceptional storm occurring on land vulnerable to erosion (e.g. bare soil after tillage). On the other hand, large areas in Europe face continuously small to moderate rill and interrill erosion rates due to intensive land use. If these processes are allowed to continue over long periods of time, they can cause soil degradation and affect soil productivity, especially in certain regions of Europe where the buffer capacity of the soil is limited (Bakker et al., 2004, 2007). Hence, Europe and the Mediterranean are indeed faced with a problem of soil degradation through interrill and rill erosion. The resulting declines in agricultural productivity can be masked by an increase in fertiliser use and/or technological innovation (e.g. irrigation techniques, more drought-resistant plant varieties: Tilman et al., 2002), thereby also masking the impact of soil degradation. This is especially true for developed industrial regions such as Europe. In addition, large areas in Europe are covered by thick loess soils which have a large buffer capacity before SL will start to affect soil productivity. However, in the Mediterranean, an erosion-induced decline in soil productivity can happen in the foreseeable future (Bakker et al., 2007). Furthermore, on-site SL does increase production costs, causes off-site damage and creates other environmental problems such as sediment, nitrate and phosphorous contamination in ground- and surface waters (e.g. Puustinen et al., 2005, 2007; Ulén and Kalisky, 2005; Uusi-Kämppä, 2005).

1.2 Reducing runoff and soil erosion

The development of Sustainable Land Management (SLM) practices to mitigate problems caused by soil erosion by water has been the subject of extensive research and policies at all levels of government in Europe and the Mediterranean, often within the wider topics of soil and land degradation and desertification (e.g. Bowyer et al., 2009; European Commission, 2012; Millenium Ecosystem Assessment, 2005b; UNCCD, 2011; UNEP, 1994). Within the European Commission's Sixth Framework Programme, the DESIRE project (Desertification Mitigation and Remediation of Land: FP6, sub-priority 1.1.6.3: research on desertification) addresses land degradation and desertification by developing "a global approach for local solutions" (DESIRE, 2007). The approach for the mitigation of the effects of desertification taken by this project is that while remediation strategies are implemented locally and need to be suited to local needs and conditions, the global nature and consequences of land degradation and desertification require a common global framework to achieve better and integrated policies that will eventually be more effective on a large scale.

1.2.1 Soil and water conservation through land use change

Land use as a factor controlling runoff and soil loss

At field scale, land use is an important controlling factor of runoff (R) and SL (Kosmas et al., 1997), which is moreover strongly affected by human activity (Foley et al., 2005). The land use type affects R and SL in various ways. One of the most important controlling factors of R and SL associated with land use type is the vegetation cover associated with different land use types (Bochet et al., 2006; Gyssels et al., 2005). Vegetation cover acts as a buffer between the

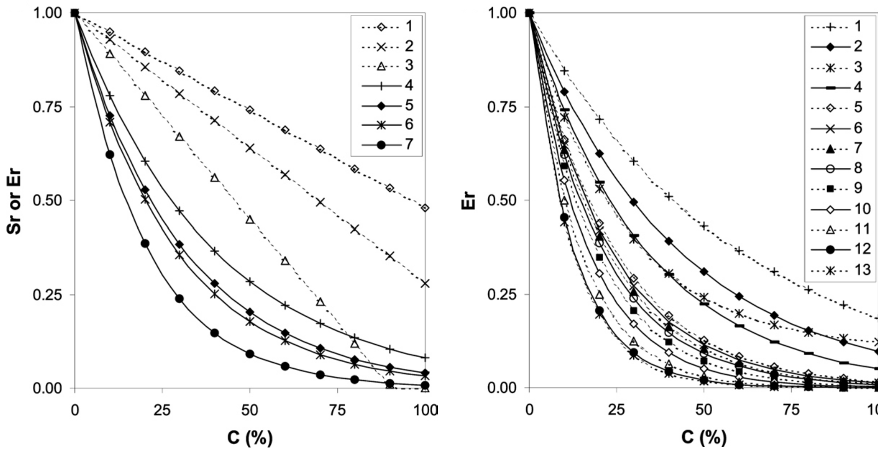


Figure 1.2: Relation between relative soil loss compared to bare soil due to **(left)** splash erosion (Sr) or **(right)** interrill and rill erosion (Er) and vegetation cover (C) according to various authors (source: Gyssels et al., 2005)

soil and raindrop impact or runoff and has been shown to be strongly correlated with R and SL (Eq. 1.1, Fig. 1.2, Fig. 1.3).

$$SLR = e^{-j \cdot C} \qquad \text{(Eq. 1.1)}$$

Where: SLR soil loss ratio; the ratio between soil loss under vegetation cover and that on bare ground, j= constant ranging between 0.025 and 0.06, C= vegetation cover (Morgan, 2005).

In addition to vegetation cover, the land use type is also related to several other factors that have an effect on R and SL such as rooting density and soil cohesion (e.g. De Baets et al., 2006; Gray and Leiser, 1982; Gyssels and Poesen, 2003), infiltration capacity (e.g. Thompson et al., 2010), tillage practices (Van Oost et al., 2006), soil compaction (e.g. Hamza and Anderson, 2005), surface storage capacity and evapotranspiration (e.g. Foley et al., 2005; Harbor, 1994; Kosmas et al., 1997; Niehoff et al., 2002).

Effects of land use change on runoff and soil loss

Over the last 3000 years, the main land use change trend for Europe and the Mediterranean has been one of deforestation of land that was turned into cropland and pasture (Kaplan et al., 2009) along with an intensification of the land use on existing agricultural land (Ewert et al., 2005). This evolution

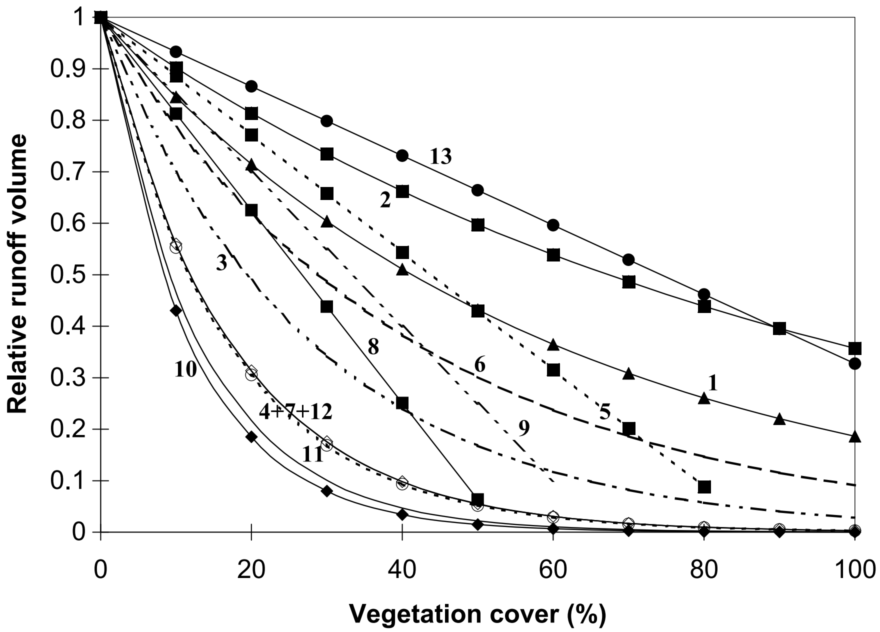


Figure 1.3: Relation between relative runoff volume and the vegetation cover according to various authors (source: Bochet et al., 2006)

led to increased soil erosion and depletion of soil resources, especially in the Eastern and Southern Mediterranean (Kaplan and Vanwalleghem, 2012). Since the 1990's however, important trends in land abandonment have been noted in Europe, which are expected to increase in the future (Verburg et al., 2006). This trend in land abandonment can be caused by a decline in land suitability for agriculture due to erosion. Bakker et al. (2005) showed that between 1886 and 1996, 53% of cropland cultivated with cereals in western Lesvos, Greece was abandoned and turned into rangeland on account of an erosion-induced decline in productivity. During the same period, neighbouring rangeland regions were converted to cropland however, showing that this land use change was not caused by a declining need for cereal production. Other reasons for current trends in land abandonment include economic drivers that make agricultural activity unprofitable or less profitable than the products of more extensive land use types. For instance, Rudel et al. (2005) showed that for several countries worldwide, a transition from a deforestation trend to an increase in forest area is observed that can be tied to the development path of these countries. Furhtermore, also deliberate policies to increase vegetation cover through reforestation or to reduce the area of cultivated land such as the European Union's Common Agricultural

Policy (e.g. MacDonald et al., 2000) can bring about considerable land use changes. Such de-intensification of land use often results in a strong decrease of water erosion. Using the WaTEM\SEDEM Bakker et al. (2008) showed that over ca. 50 years, sediment production more than halved in two de-intensified areas (48.81 to 20.52 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ in Amendoeira, Portugal, and 60.66 to 28.34 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ in Lautaret, France), was reduced somewhat in another de-intensified area (14.28 to 12.65 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ in Lagadas, Greece), all of which were also accompanied with an important decrease in sediment export. In a continuously cultivated rea in Hageland, Belgium, sediment production also decreased somewhat (15.34 to 11.14 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$), but contrary to the other study sites, this was not associated with a decrease in sediment export.

Hence, directing the abovementioned processes of land use change or planned changes in land use type towards less intensive land use can have a strong mitigating effect on R and SL in problem areas. Using a rainfall simulator, (Lasanta et al., 2000) found that when cultivation of terraced land was abandoned a decrease in the runoff response time from 897 s. to 210 s. and a after 60 months of land abandonment and concurrent increase in runoff coefficient from 8.5 to 50.2%. Sediment concentration decreased from 6.8 to 1.1 $g{\cdot}l^{-1}$ however. Nevertheless, other studies also reported an increase in R and/or SL on terraced lands after abandonment due to terrace failure. (e.g. Koulouri and Giourga, 2007; Lasanta et al., 2001; Lesschen et al., 2008).

These examples show that quantitative knowledge on processes of R and SL generation under different land use types and in different situations is a prerequisite to the implementation of successful sustainable land management practices.

1.2.2 The tools to reduce field-scale runoff and soil loss within a given land use type

A complete change of land use is not always possible as not all agricultural land (i.e. cropland, vineyards, orchards) can be turned into more erosion-resistant land use types such as forests, grassland or shrubland. Other land use types such as roadcut sites (e.g. Agassi and Benhur, 1991) or industrial sites (e.g. Biemelt et al., 2005; Kleeberg et al., 2008) often feature unconsolidated, bare soil with a low structural stability on steep slopes, and are hence inherently sensitive to interrill and rill erosion (e.g. Borselli et al., 2006). In these instances, specific techniques are needed to reduce both R and SL to tolerable levels. As old and universal as the problem of land degradation by water erosion is, so are specific techniques used to mitigate it. Soil conservation structures survive which date to the ancient civilisations in the Eastern Mediterranean and the

Mayans (Montgomery, 2007b), along with many examples of contemporary 'indigenous' soil conservation techniques that are still being used (e.g. Wakindiki and Ben-Hur, 2002). As agricultural revolutions transformed agriculture to become more large-scale, intensive and mechanised, also the land degradation by water erosion changed and soil conservation techniques had to evolve. Perhaps the most famous example of this is the Dust Bowl in the U.S.A. in the 1930's, which was due to a combination of a naturally occurring period of drought on land that had seen an unprecedented land use intensification (Thomas and Middleton, 1994). Although mostly caused by wind erosion, it also led to the initiation of extensive research on soil erosion by water and soil conservation (Baveye et al., 2011; Laflen and Moldenhauer, 2003). All this has led to an extensive body of experience with and literature on a set of different various techniques to mitigate the effects of soil erosion by water through the reduction of field-scale R and SL. These techniques are jointly referred to as soil and water conservation techniques (SWCTs).

In this research, the focus is on those SWCTs relevant for application in Europe and the Mediterranean, which can be broadly classified into three groups (Morgan, 2005): (1) crop and vegetation management, (2) soil management and (3) mechanical methods. For a detailed discussion of each of the individual techniques, see the appendix to chapter 7.

1.3 Research gaps

As shown in section 1.1, there has been a long-standing awareness of the problems of desertification, land and soil degradation in Europe and the Mediterranean. In response, significant research efforts have been made to quantify rates of plot-scale annual runoff (R_a) and annual soil loss due to interrill and rill erosion (SL_a) and to identify its controlling factors. Many studies have also addressed the possibilities of controlling R_a and SL_a through land use change and the application of SWCTs, as was shown in section 1.2. Nevertheless, several research gaps still remain with respect to a comprehensive and quantitative overview of R_a and SL_a and its mitigation measures for Europe and the Mediterranean.

1.3.1 A continental perspective on the assessment of runoff and soil loss rates

As shown in section 1.1, desertification and land and soil degradation are increasingly seen as global problems that require an approach that combines a global coordination and knowledge base with the ability to implement solutions adapted to regional or local conditions (e.g. DESIRE, 2007; European Commission, 2012; Karlen, 2008). In response to this need for a global approach, several overviews of available R_a and SL_a data have been made in recent years (e.g. Auerswald et al., 2009; Boardman and Poesen, 2006; Cerdan et al., 2006, 2010; de Vente, 2009). Table 1.1 shows that many R_a and SL_a plot studies exist throughout the whole of Europe and the Mediterranean. However, none of these review studies (Table 1.1) include both R_a and SL_a at a continental scale (i.e. Europe and the Mediterranean) and there is no comprehensive overview of R_a and SL_a plot data in Europe and the Mediterranean. In contrast to the U.S.A., where research into interrill and rill erosion started as a large coordinated effort with the development of the USLE equation (Laflen and Moldenhauer, 2003; Wischmeier and Smith, 1978), plot-scale erosion research in Europe and the Mediterranean was mostly carried out as individual studies with little coordination, explaining the lack of a comprehensive overview. This lack of overview also limits the possibilities of a continental-wide assessment of R_a and SL_a at the plot scale. Furthermore, Table 1.1 shows that R has received considerably less attention in the literature. Nevertheless, R_a in itself is related to several important problems like flooding (e.g. Poesen and Hooke, 1997) and agricultural productivity (e.g. Rockström et al., 2010; Wallace, 2000).

Table 1.1: Overview of compilation studies reporting annual runoff and soil loss data from experimental plots, including both conventional practices and the application of soil and water conservation techniques (SWCTs), collected at country, regional or subcontinental scale for Europe and the Mediterranean. R_a: annual runoff, SL_a: annual soil loss, SWCT: soil and water conservation technique, #PY= number of plot-years. +: reported in study, -: not reported in the study, Cr: cropland, Vi: vineyard, Sh, shrubland, Tc: tree crops, Pf: post-fire, Fo: forest, Ba: bare, Fa: fallow, Gr: grassland, NA: not available. PY= number of plot-years.

region/country	R_a	SL_a	SWCT	land use types	SWCT	PY	source
Mediterranean:							
MEDALUS project field sites; Spain, France, Italy, Greece	+	+	-	Cr, Vi, Tc, Sh, Eucalyptus	NA	NA	Kosmas et al., 1997
Portugal, Spain, France, Italy, Greece, Israel	-	+	-	Cr, Vi, Tc, Sh	NA	ca. 150	Wainwright and Thornes, 2004
Spain, Portugal, Greece, Syria	+	+	+	olive orchards	no-tillage, cover strips	54	Fleskens and Stroosnijder, 2007
Spain, France, Italy, Morocco	-	+	+	Cr, Sh	terraces	59	González-Hidalgo et al., 2007
Italy, Spain	-	+	+	Sh, Pf, Fo, Cr, Ba, Fa, Gr	sludge application, burnt mulch, no-tillage, conservation tillage, contour tillage	74	de Vente, 2009; de Vente et al., 2007
Portugal, Spain, France, Italy, Greece, Israel, Croatia	-	+	-	Pf, Cr, Vi, Sh, Eucalyptus	NA	NA	Shakesby, 2011
Germany	-	+	+	Cr, Fa, Gr, Vi, Fo	cover crop, mulch tillage, grass strips	1 078	Auerswald et al., 2009
Europe	-	+	+	Ba, Ba, Fo, Gr, Sh, Vi, Tc	cover crops	2 741*	Cerdan et al., 2006, 2010
Europe	-	+	+	Cr	mulch	NA	Smets et al., 2008a
Europe	+	+	+	Cr	conservation tillage	NA	Leys et al., 2010
Europe	-	+	-	Ba, Cr, Fa, Fo, Gr, Pf, Sh, Tc, Vi	NA	NA	Boardman and Poesen, 2006

* from 81 different plot-measuring stations.

Nevertheless, a comprehensive dataset of available plot-scale R_a and SL_a data for Europe and the Mediterranean is needed in the framework of developing a coordinated mitigation strategy for land degradation. Estimates of either annual or event R and SL are mostly obtained by the application of water erosion models. These models fall into two broad categories; physical process-based models (e.g. PESERA (Kirkby et al., 2004), WEPP (Flanagan and Livingston, 1995), LISEM (De Roo et al., 1996)) and empirical models (e.g. (R)USLE(2): Renard et al., 1997; USDA-ARS, 2008; Wischmeier and Smith, 1978). Some models employ both process-based and empirical components (e.g. WaTEM/SEDEM (Van Oost et al., 2000; Van Rompaey et al., 2001; Verstraeten et al., 2002) or SWAT (Neitsch et al., 2011)). Most physical process-based models lack validation with field-measured data. Furthermore, some of these models require very detailed data on several variables that may be impossible to obtain for the end users. A comprehensive dataset of field-measured R_a and SL_a in Europe and the Mediterranean on the other hand provides a quick way to assess R_a and SL_a rates in various situations and can serve as base to evaluate model output. Furthermore, plot data allow to evaluate relations between precipitation, R and SL, and factors controlling these relations based on data that are 1) measured in field conditions under natural rainfall and 2) representative for Europe and the Mediterranean, rather than drawing on relations established in other parts of the world.

Recent studies by Kirkby et al. (2004) and Cerdan et al. (2006, 2010) have presented an assessment of SL_a at a continental scale through the application of the PESERA model and extrapolation of existing plot data, respectively (Fig. 1.1). To date, no continental-scale assessments of R_a rates have been presented however. Furthermore, the validation of the PESERA map is restricted to a limited validation of the PESERA model itself (Kirkby et al., 2004; Licciardello et al., 2009; Tsara et al., 2005). The study by Cerdan et al. (2010) relies on the extrapolation of average SL_a values for different land use types by using equations derived from other datasets and publications. The PESERA map does not incorporate the effect of several key factors such as surface rock fragment cover (e.g. Poesen and Lavee, 1994; Poesen et al., 1994), while the SEM map (Cerdan et al., 2010) does not incorporate climatic data due to lack of a clear climatic trend in the data. The correspondence of these model predictions with each other (Fig. 1.1) and with field-measured data is largely unknown, and several authors have noted discrepancies between different model predictions and the general lack of validation of erosion models (e.g. Favis-Mortlock, 1998; Jetten et al., 1999; Vanmaercke et al., 2012a). Hence, continental scale studies quantifying the correspondence between model-predicted soil loss values and field-measured soil loss data to determine model accuracy and identify deviations between models and field-measured data can help improve these models.

1.3.2 Runoff and soil loss, and their controlling factors in Europe and the Mediterranean

The objectives of R and SL plot studies in Europe and the Mediterranean were often to study R_a and SL_a and their relationship with conditions that are thought to be important controls (e.g. different local crops). Hence, very detailed insights into the effect of local factors controlling R_a and SL_a have been gained (e.g. Bagarello and Ferro, 2010), but little information is available on the representativeness of these findings for the whole of Europe. Most research of the effect of controlling variables on R_a and SL_a over a wide range of conditions are restricted to the U.S. (e.g. Renard et al., 1997; Nearing, 1997) and only a limited number of studies are available specifically for Europe and the Mediterranean, or even including Europe and the Mediterranean. Examples of these include Torri et al. (1997), who review soil erodibility based on a global dataset of plot measurements. Kosmas et al. (1997) illustrate the important control of land use type and the effect of annual rainfall on R_a and SL_a for different land uses in 8 sites distributed over the Northern Mediterranean. For shrubland, they found a similar relation between annual rainfall (P_a) and R_a, and P_a and SL_a as the one proposed by Langbein and Schumm (1958) (Fig. 1.4). Cerdan et al. (2010) explored controlling factors of plot scale SL_a for different land uses at a continental scale but found only limited effects of plot length, slope gradient and soil erodibility. de Vente (2009) and Vanmaercke (2012) respectively made a meta-analysis of the effect of plot area and plot length on SL_a distribution in comparison to catchment sediment yield. These studies consider only one or a few controlling variables however. With respect to controlling factors of R_a at regional or continental scale, the study of controlling factors is restricted to the effect of land use type on R_a rates in the Northern Mediterranean (Fig. 1.4, Kosmas et al., 1997). A more comprehensive analysis of the controlling factors of R_a and SL_a at a continental scale can therefore strongly contribute to better models and risk assessment.

With respect to spatial variability, factors controlling R_a and SL_a are widely different throughout the study area, i.e. Europe and the Mediterranean. A lot of knowledge on this topic has been gained in the past, but remains mostly restricted to regional or country-wide reviews (e.g. Auerswald et al., 2009; Boardman and Poesen, 2006). At continental scale, Cerdan et al. (2010) found that the correlation between slope gradient and SL_a was significant outside the Mediterranean, while the correlation for plots within the Mediterranean was not significant. Furthermore, smaller mean SL_a were observed in the Mediterranean compared to the non-Mediterranean, which was attributed to the presence of more stony soils which are protected from SL (Poesen and Lavee, 1994; Poesen et al., 1994). Such assessments of the dominant controlling factors of

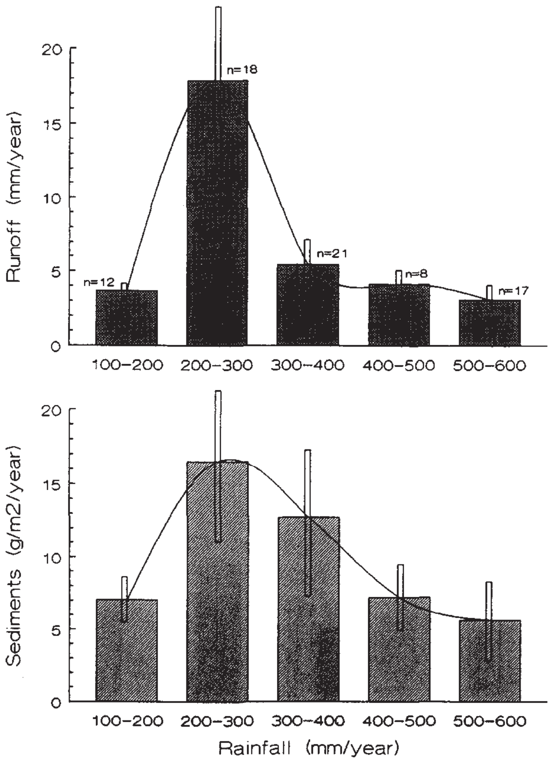


Figure 1.4: Relation between annual rainfall and annual runoff (upper graph) and annual rainfall and annual soil loss (lower graph) for shrubland in four Northern Mediterranean sites (Kosmas et al., 1997).

R_a and SL_a are rare on a continental scale however. A better understanding of the spatial variability of R_a and SL_a and spatial variability in the effect of controlling factors of R_a and SL_a can significantly increase to our capabilities to apply research results to other areas and improve modelling at a continental scale.

1.3.3 Uncertainty, spatial and temporal variability of runoff and soil loss

Many studies have pointed to the large variability in plot R_a and SL_a measurements and associated uncertainties (e.g. Evans, 1995; Nearing et al., 1999; Stroosnijder, 2005). Several causes of variability in R_a and SL_a plot measurements have been put forward. Firstly, several aspects of the experimental methodology such as the use of open or closed plots, the type of plot borders, collection tanks and flow splitters and sampling protocols influence measurement results (e.g. Bagarello and Ferro, 1998; Blanco-Canqui et al., 2004b; Boix-Fayos et al., 2007; Hudson, 1993; Zöbisch et al., 1996). Secondly, even between replicated plots (i.e. plots sharing the same experimental methodology and plot characteristics) large variations in measured R and SL are observed (Nearing et al., 1999; Wendt et al., 1985). For a more detailed discussion of these uncertainties, see chapter 4. Uncertainties can be partly explained by variations in soil roughness, infiltration capacity or vegetation density and pattern between the replicated plots, even when these factors are tightly controlled. In addition, a significant measurement error is likely, especially when small amounts of R_a and SL_a are measured. To address the uncertainty caused by small unavoidable differences between plots and measurement error, the average R_a and SL_a from two or more replicate plots is commonly used to obtain more reliable average values and gain insight in the variability of the measured R_a and SL_a. Furthermore, even within the plot-scale (a few m^2 to several hundred m^2), there is a scale dependency with respect to R_a and SL_a (e.g. Wainwright et al., 2008), with different processes (i.e. splash erosion, interrill and rill erosion) becoming significant or dominant at different spatial scales (Boix-Fayos et al., 2006, 2007). The abovementioned sources of uncertainty in R_a and SL_a plot measurements are hard to account for as the quantification of these uncertainties is rarely included in the objectives of the studies. Nevertheless, it is important to keep these uncertainties in mind and even a rough quantification of their magnitude (e.g. Nearing et al., 1999; Wendt et al., 1985) greatly improves the interpretation of results and decision making with respect to land degradation.

In addition to scale effects in interrill and rill erosion, the spatial scale of runoff and soil loss plots is also limited, and most plots are no longer than 30m). Hence, upscaling of of plot-measured soil loss beyond this spatial scales adds additional uncertainty and other erosion-related processes such as gullying and sediment deposition are rarely or never assessed in plot studies, but can become dominant at larger spatial scales (e.g. de Vente and Poesen, 2005). When plot-measured SL values are extrapolated to larger scales (e.g. the field or catchment scale), these processes need to be accounted for, for which different measuring techniques will be needed (e.g. ^{137}Cs measurements or catchment sediment yield measurements).

Furthermore, a large part of uncertainty in R_a and SL_a rates comes from a poorly understood temporal variability. R and SL plot studies under natural rainfall are typically conducted for a period between one season and several years (Cerdan et al., 2006). Often there are only limited indications on the representativeness of the measurements for long-term mean R_a and SL_a rates. P_a in the measuring period can be compared to the climatological (i.e. long-term, 30 to 50 years) mean annual rainfall to assess how representative the observed conditions are for the average year with respect to precipitation. Nevertheless, several authors (Baffaut et al., 1998; de Figueiredo et al., 1998; Gonzalez-Hidalgo et al., 2012) showed that SL_a is often largely determined by a limited number of low-frequency, high-intensity events, and the capturing of such an event greatly influences the measurements. However, an analysis of the recurrence periods of the observed erosive events is rarely included in plot studies. Therefore, it is often not known how long measurements should be made to be representative for the long-term mean R_a and SL_a rate at a specific site. A method for the estimation of the measuring period required with respect to the desired level of certainty on the long-term average catchment sediment yield was developed by Vanmaercke et al. (2012b), but so far no detailed estimate has been made for R_a and SL_a plot studies.

A better knowledge on the temporal variability in R_a and SL_a could also contribute to models that are able to predict R_a and SL_a better at different temporal resolutions. While models exist with a high temporal resolution (single-storm or even within-storm predictions; e.g. WEPP (Flanagan and Livingston, 1995)), running them for long periods is computationally time-consuming and requires detailed data that are not always available. Conversely, models that are optimised for prediction of medium to long-term annual values often fail to accurately predict short-term variation well (e.g. Licciardello et al., 2009). Nevertheless, accurate estimation of temporal variation in R_a and SL_a rates has important applications in e.g. conservation planning (e.g. Bagarello et al., 2011).

1.3.4 Soil and water conservation techniques

A continental perspective on soil and water conservation

Few reviews of plot R_a and SL_a rates in Europe and the Mediterranean consider the effect of SWCTs (Table 1.1). Hence, the knowledge gaps discussed in section 1.3 for plot-scale R_a and SL_a such as the lack of a Pan-European overview of available data also apply to plot studies where SWCTs are applied.

The limited number of SWCT plot studies in any country or region means that a comprehensive review of the effectiveness of SWCTs in reducing R_a and SL_a that encompasses different SWCTs and a wide range of environmental conditions needs to be conducted at a continental scale for a sufficient number of plots to be available. Furthermore, as indicated in section 1.2, policies on combating soil degradation are made at all levels of government, including the European level. Hence, the continental perspective on soil and water conservation is certainly relevant, but up to now understudied. While also socio-economic factors play an important role in effective and efficient SWCT implementation (Boardman et al., 2003), these aspects fall beyond the scope of this research.

Effectiveness of SWCTs in reducing runoff and soil loss

Currently, the effect of SWCTs on R_a and SL_a is either derived from laboratory studies or results from field studies with limited information on the applicability to other regions and conditions (e.g. Smets et al., 2011b). Models for the evaluation of SWCTs effectiveness in reducing R_a and SL_a have often been validated with limited field data for specific sites or are not validated at all (e.g. Hessel and Tenge, 2008). Hence, their range of applicability is uncertain. Therefore, a comprehensive approach to assess the effects of SWCTs on R_a and SL_a over a wide range of conditions needs to be developed, which can also be used to further develop erosion models that can better account for the application of SWCTs.

Several approaches for the quantification of R_a and SL_a reduction by the application of SWCTs have been followed. A general approach is illustrated by Montgomery (2007a) (Fig. 1.5). While this approach can be applied to a large range of measured soil loss data over different environmental conditions and clearly shows the potential of SWCTs to reduce SL_a rates to sustainable rates, it is less suited to quantify the effectiveness of individual SWCTs or to assess the effects of specific environmental conditions.

Another widely applied approach is the use of a runoff ratio (RR) and soil loss ratio (SLR), which are defined as the ratio of R and SL from a plot with SWCT application to R and SL from a reference plot with the same characteristics but without SWCT application. (e.g. Castillo et al., 1997; Cogo et al., 1984; Gilley and Risse, 2000; Smets et al., 2008a). The use of RR and SLR allows a more detailed analysis of SWCT effectiveness, but the additional data on measured R_a and SL_a rates on a reference plot that is required limits the amount of available data.

SLR values are similar to the widely used (R)USLE cover management (C) and support practice (P) factor (Renard et al., 1997). However, the calculation of

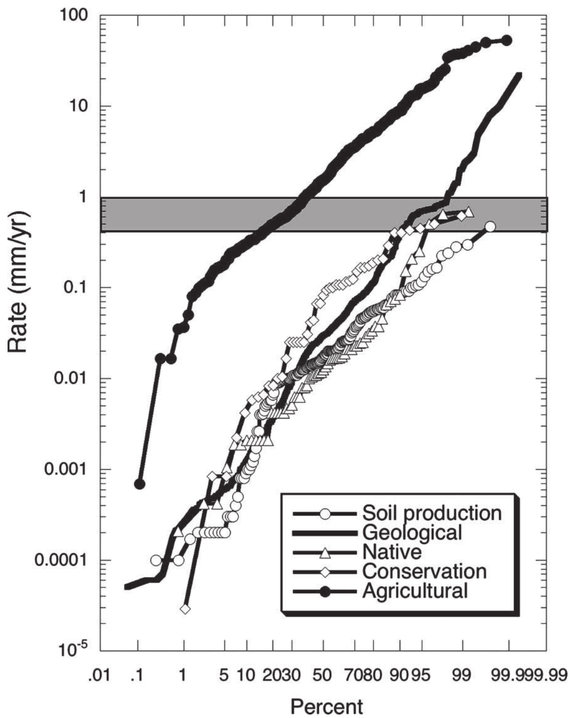


Figure 1.5: Comparison of the probability distributions of published annual soil loss rates (mm·yr^{-1}) under conventional (e.g. tillage) and conservation agriculture (e.g. terracing, no-tillage), with annual soil loss rates under plots with native vegetation, geologic soil erosion rates and soil production rates in a worldwide study. The shaded grey area indicates the range of USDA tolerable annual soil loss rates (T-values) (0.4-1.0 mm·yr^{-1}, corresponding to 5-12 Mg·ha^{-1}·yr^{-1}) (source: Montgomery, 2007a)

C- or P-factors for specific soil conservation techniques is not straightforward. Hessel and Tenge (2008) showed the need for local measurements of SWCT effectiveness, but such information is often not available. Furthermore, C- and P-factors apply only to SL and not to R. While RR have been used in some studies (e.g. Gilley and Risse, 2000), quantification of SWCT effectiveness remains mainly oriented at SL. Nevertheless, the term 'soil *and water* conservation techniques' implies that also an effect on R is expected or desired. However, R reduction by SWCTs is less studied and the relations between R reduction and SL reduction are rarely considered in these studies.

Quantifications of the reduction of plot-scale R_a and SL_a rates by the application of SWCTs have obvious applications in on-site conservation planning.

Furthermore, both land use change and the application of SWCTs also have important effects at catchment scale with respect to sediment yield (e.g. de Vente, 2009; Kondolf et al., 2002; Potter, 1991; Trimble, 1999; Vanmaercke, 2012) and flood peaks (e.g. Brath et al., 2006; Nyssen et al., 2010; Potter, 1991). Better data on the local effects of land use change and SWCT application can contribute to a more accurate prediction of SWCT effects on catchment scale hydrology and sediment yield for Europe and the Mediterranean. In addition, accurate data on SWCT effectiveness can also contribute to the development of field-scale erosion models that are able to better incorporate the application of SWCTs.

Factors controlling SWCT effectiveness in reducing runoff and soil loss

There are also strong indications that the effectiveness of individual SWCTs depends on environmental factors such as land use, saturated conductivity and storm size (Hessel and Tenge, 2008) or plot slope length (e.g Gilley and Risse, 2000; Smets et al., 2008b,a) and plot slope gradient (e.g Renard et al., 1997; Syversen, 2005). Nevertheless, very few quantitative assessments of the effects of these environmental factors on SWCT effectiveness in reducing R_a and SL_a have been made. Limited understanding of environmental effects on SWCT effectiveness in reducing R_a and SL_a also limits application of existing knowledge to other environments and the incorporation of SWCT application in erosion models (e.g. Hessel and Tenge, 2008).

Similar to R_a and SL_a under conventional practices (cf. section 1.3.3), SWCT effectiveness is likely subject to a significant temporal and spatial variability. With respect to temporal variability, both the variability in SWCT effectiveness over different years (i.e. how reliable is the SWCT effectiveness assessment in any given year?), and the evolution of SWCT effectiveness in the years after the first application (i.e. does the effectiveness of the SWCT increase or decrease over the years?) is of interest. With respect to spatial variability, very little is known on the variation in SWCT effectiveness between different regions of Europe and the Mediterranean. Region-specific characteristics such as precipitation distribution and intensity are known to have a strong effect on R_a and SL_a, and hence they can be assumed to have also an impact on SWCT effectiveness. However, no studies exist on the temporal and spatial variability of the effectiveness of SWCTs in reducing R_a and SL_a for Europe and the Mediterranean.

1.4 Research objectives

To address the knowledge gaps outlined in section 1.3, the overall objective of this research is to assess R_a and SL_a rates due to interrill and rill erosion, their controlling factors and variability, and to assess the potential of different land use types and SWCTs to reduce R_a and SL_a on a continental scale for Europe and the Mediterranean (Fig. 1.6).

Therefore, following objectives are formulated:

1. to compile a database of R_a and SL_a plot measurements under natural rainfall for Europe and the Mediterranean;
2. to quantify R_a and SL_a rates for different land use types, quantify the effect of controlling factors on R_a and SL_a rates, and assess the spatial variability in R_a and SL_a rates for Europe and the Mediterranean;
3. to analyse the temporal uncertainties with respect to measured R_a and SL_a for Europe and the Mediterranean and to assess the uncertainty on average R_a and SL_a rates due to inter-annual variability;
4. to analyse important factors that control the P_a-R_a relationship in Europe and the Mediterranean and determine whether plot-scale R_a in Europe and the Mediterranean can be predicted using simple models;
5. to determine how well some other continental-wide assessments of SL_a for Europe correspond with field-measured data;
6. to quantify the effectiveness of different soil and water conservation techniques in reducing R_a and SL_a for Europe and the Mediterranean and to provide a detailed analysis of major factors affecting the effectiveness of these different SWCTs.;

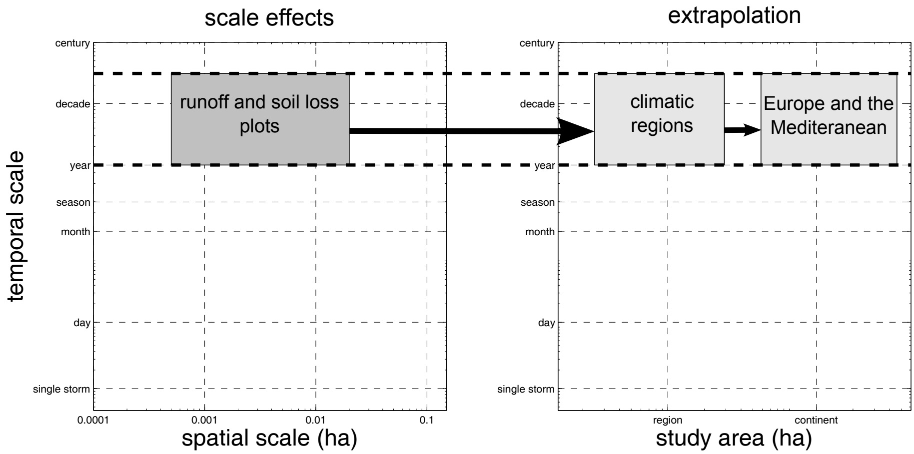


Figure 1.6: (left) Scope of this study, visualised as the range of temporal and spatial scales represented by runoff and soil loss plot measurements (dark grey box). **(right)** Study areas of this research; a runoff and soil loss plot database for Europe and the Mediterranean is compiled in order to compare plot measurements at the regional (i.e. climatic regions as defined by the LANMAP2 map (Metzger et al., 2005; Mücher et al., 2010), light grey box) and the continental scale (i.e. Europe and the Mediterranean, light grey box).

1.5 Thesis structure

Each of the research objectives stated in section 1.4 is addressed in different chapters of this thesis (Fig. 1.7). After this introductory chapter, a field plot database of annual runoff and soil loss data for Europe and the Mediterranean is presented and discussed in chapter 2, along with a discussion of data availability and recommendations for future data collection and analysis. The effect of different land use types on R_a and SL_a rates and controlling factors is explored in chapter 3. In chapter 4 the inter-annual variability on R_a and SL_a data is quantified and discussed. In chapter 5, the annual rainfall-runoff relationship is examined by means of a modified SCS Runoff Curve Number Method. In chapter 6, measured plot SL_a data are confronted with results from continental-scale, spatially distributed models of SL_a. The effectiveness of these SWCTs in reducing R_a and SL_a is analysed in chapter 7, along with detailed description of the various types of SWCTs that have been tested on runoff and soil loss plots in Europe and the Mediterranean section A. Finally, chapter 8 presents the general conclusions and recommendations of this research.

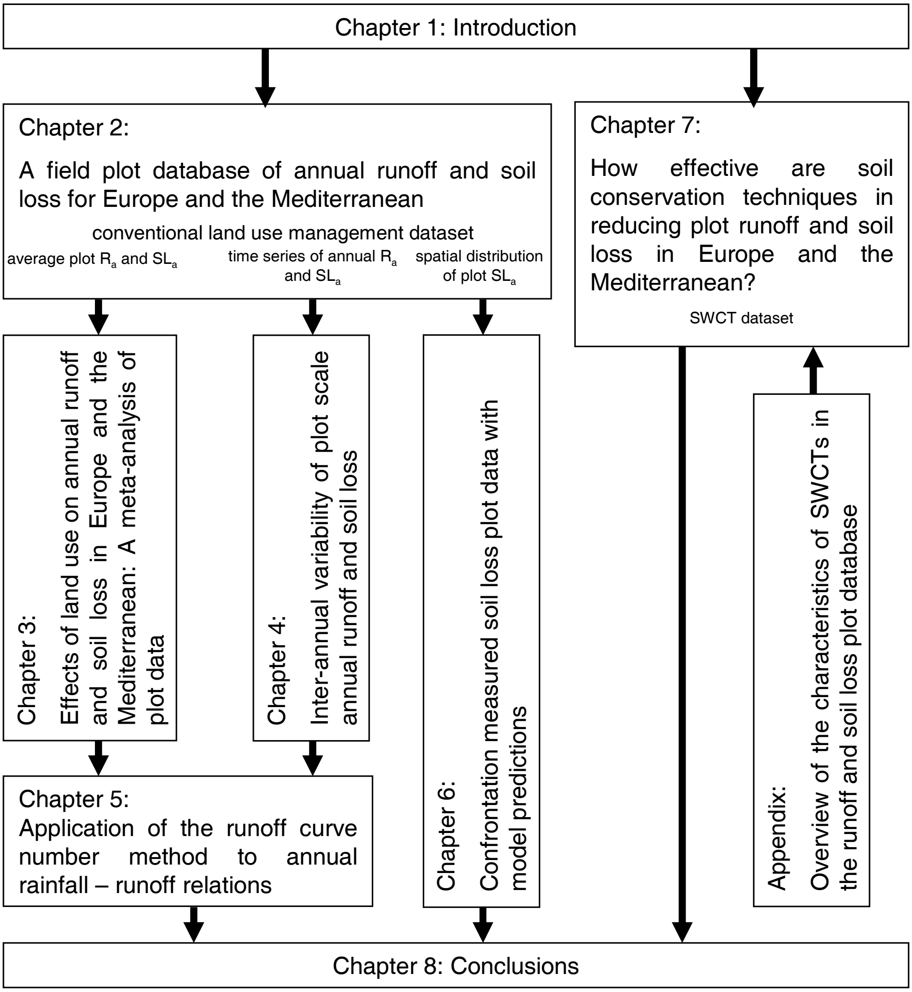


Figure 1.7: General outline of the thesis structure.

Chapter 2

A field plot database of annual runoff and soil loss for Europe and the Mediterranean

This chapter is based on: Maetens, W., Vanmaercke, M., Poesen, J., Jankauskas, B., Jankauskiene, G. and Ionita, I., 2012. Effects of land use on annual runoff and soil loss in Europe and the Mediterranean: A meta-analysis of plot data. Progress in Physical Geography 36(5): 597 - 651. doi:10.1177/0309133312451303

2.1 Database compilation

2.1.1 Runoff plot selection criteria

A database was constructed with annual runoff (R_a) and annual soil loss (SL_a) data, measured on bounded runoff plots under natural rainfall conditions in Europe and the Mediterranean region (Fig. 2.1). Data were collected from scientific papers, books (Boardman and Poesen, 2006), project reports, PhD. theses and through personal communication with various researchers. Only runoff and/or soil loss measurements conducted on bounded runoff plots under

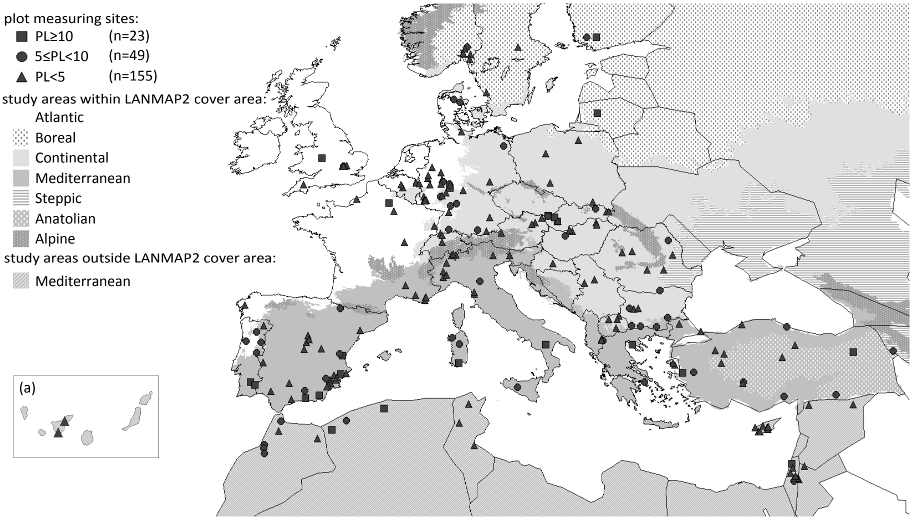


Figure 2.1: Geographical distribution of plot runoff and soil loss measuring stations over Europe and the Mediterranean with indication of the climatic zones derived from the LANMAP2 classification (Mücher et al., 2010; Metzger et al., 2005). Inset: (a) Canary Islands. PL= number of plots, n= number of plot measuring stations

natural rainfall conditions with a known land use, a minimum plot length of 5 m, and for a measuring period (MP) that is representative (cfr. section 2.1.2 for more details) for at least one year were considered. Results from runoff plots that were treated with soil and water conservation techniques were not included in this analysis since they do not represent prevailing field conditions. A list of the excluded soil and water conservation techniques is given in Table 2.1. While plots without any soil cover throughout the year, i.e. bare plots (Table 2.2), are not a common land use practice, they are often used in soil erosion studies as reference plots, representing maximum potential SL_a for the study conditions. Hence, plots with bare soil were included in the database for reference purposes.

While most plots use collection tanks or flow samplers to determine the total runoff and soil loss by interrill and rill erosion, in a small number of studies, soil loss is determined by measuring the rill volume (Govers and Poesen, 1988; Feiza et al., 2007; Jankauskas and Fullen, 2002; Jankauskas and Jankauskiene, 2003a; Jankauskas et al., 2004). Based on a literature survey by Govers and Poesen (1988), total soil loss for these studies were calculated by adding 25% to the measured rill soil loss to account for interrill soil loss.

Each runoff and soil loss plot in the database represents measurements of R_a and/or SL_a at a particular measuring station for a specific combination of land use, soil type, plot length and slope gradient. For each of these plots, the corresponding number of plot-years was also recorded, which indicates the number of years represented by the data for that plot. 1 plot-year corresponds to a measuring period of one year on a single runoff plot. When measurements were conducted on several replicate plots with identical experimental setup and results were reported individually for the different replicates, they were included as different plots in the database. If the average R_a and SL_a value for the replicate plots was reported (e.g. Bagarello and Ferro, 2010; Bagarello et al., 2010a,b; Mohammad and Adam, 2010; Lopes et al., 2002) the average values for all replicates were counted as one plot in the database, while the number of plot-years was considered to be the sum of all plot-years of the replicates.

Table 2.1: List of soil and water conservation techniques (SWCTs) tested on runoff and soil loss plots in Europe and the Mediterranean. Ba: bare soil, Cr: cropland, Cs: construction sites, Fa: fallow, Fo: forest, Gr: grassland, Pf: post-fire, Sh: shrubland, Tc: tree crops, Vi: vineyard

SWCT		Land use type	Description
Crop and vegetation management			
Cc:	cover crops	Ba, Cr, Tc, Vi	cover crop during the intercropping season, established after harvest (e.g. Laloy and Bielders, 2010).
Mu:	mulching	Cr, Gr, Pf, Ra, Sh, Vi	application of stone or organic mulch (crop residue or straw) (e.g. Brown, 1996).
Bs:	buffer strips	Ba, Cr, Fa, Tc, Vi	Strips of perennial vegetation (usually grasses) used to increase infiltration, slow down runoff and increase sediment deposition(e.g. Uusi-Kämppä, 2005).
Sc:	strip cropping	Cr	drilling or planting in strips of alternating crop types (e.g. Köse et al., 1996; Köse and Taysun, 2002).
Ex:	exclosure	Sh	closing of areas for grazing and agriculture (e.g. Bruggeman et al., 2005; Masri et al., 2005; Mazour, 1992).
Soil management			
Nt:	no-tillage	Ba, Cr, Fa, Tc, Vi	no tillage operations when the traditional practice is tillage (e.g. Turtola et al., 2007).
Rt:	reduced tillage	Ba, Cr, Fa, Tc, Vi	different forms of reduced tillage resulting in a smaller disturbance of the plough layer than conventional cultivationKwaad et al. (1998).
Ct:	contour tillage	Ba, Cr, Fa, Tc, Vi	tillage operations parallel the contour (e.g. Quinton and Catt, 2004).
Dt:	deep tillage		deep non-inversion tillage to improve infiltration (e.g. Chomanicová, 1988; Suchanic, 1987).
Dr:	drainage	Cr, Gr	application of subsurface drainage pipes (e.g. Øygarden, 1996; Øygarden et al., 1997) .
Sa:	soil amendment	Ba, Cr, Fa, Gr, Ra, Sh, Cs	application of soil conditioners or fertiliser to improve soil soil structural stability: i.e. phosphogypsum (e.g. Agassi and Benhur, 1991), polymers(e.g. Chomanicová, 1988; Lopez-Bermudez et al., 1991; Romero-Díaz et al., 1999), sewage sludge (e.g. Ingelmo et al., 1998) and NPK fertiliser (e.g. Kroumov and Malinov, 1989).
Mechanical methods			
Te:	terraces	Ba, Cr, Fo, Gr, Pf, Sh, Tc, Vi	construction of earthen or stone terraces parallel the contour (e.g. Koulouri and Giourga, 2007).
Cb:	contour bunds	Vi	stone or earthen bunds constructed parallel to the contour (e.g. Pinczés, 1982)
Gt:	geotextile	Ba, Cs, Tc, Vi	application of geotextile mats (e.g. Bhattacharyya et al., 2008, 2009; Jankauskas et al., 2008; Mitchell et al., 2003).

Table 2.2: Land use types considered in the plot database for Europe and the Mediterranean.

land use type	abbr.	crop	description
bare	Ba		continuously bare soil without crops or natural vegetation, sometimes tilled annually
fallow	Fa		plot with natural regrowth of grass and herbaceous spiecies, or sowing of those species in a rotation scheme
cropland	Cr	cereals	cereal (wheat, barley, oats, rye) cultivation
		maize	silage or grain maize cultivation
		sunflower	sunflower cultivation
		sugar beet	sugar beet cultivation
		potato	potato cultivation
		leguminous	leguminous (beans, peas, vetch, lentil, alfalfa, clover, yellow lupine) cultivation
		other	cultivation of other annual crops
tree crops	Tc		olive, almond or fruit (apple, citrus) cultivation
vineyards	Vi		vineyards, rows may have different orientation with respect to the contour
grassland	Gr		permanent grassland
rangeland	Ra		grass- or shrubland browsed by cattle
forest	Fo		natural vegetation or plantation with predominance of tree species
shrubland/matorral	Sh		natural vegetation or plantation with predominance of shrub species
post-fire	Pf		forested land or shrubland, burnt in the recent past (0-30 years)
construction sites	Cs		areas where urban-industrial activity (roadcut sites, mine areas) is the primary source of disturbance

Based on the description given by the authors, all plots were assigned to a land use type (Table 2.2). When different land use types were present on a single runoff plot (e.g. cropland-fallow rotation or cropland-grassland rotation), the years having the same land use were grouped together as one plot. Hence, data from a runoff plot with a rotation of cropland and fallow is entered in the database as 2 plots, one for cropland and one for fallow. This approach does not take into account possible effects of crop cultivation and soil treatment prior to the measuring period (i.e. carry-over effects), which may persist during the following months or years (Fiener and Auerswald, 2007; Hjelmfelt and Burwell, 1984; Wischmeier and Smith, 1978) and hence may explain part of the observed variability in R_a and SL_a rates. Furthermore, the location of each of the plot measuring stations was determined, either from coordinates given by the authors or from maps and descriptions in the publications. Some studies report data for plots that were installed at different sites close to each other (e.g. different slope aspects or gradients in the same valley).

Whenever it was possible to accurately distinguish between these different sites, they were incorporated as separate plot-measuring stations in the database but if this was impossible, the location of the study area where plots were located was included as one plot-measuring station. Subsequently, the climatic zone of each of the plot-measuring stations was determined according to the LANMAP2 classification (Metzger et al., 2005; Mücher et al., 2010). Plot-measuring stations in the Near East and in Northern Africa that fall outside the LANMAP2 cover area were classified as Mediterranean (Fig. 2.1). While soil properties like texture, rock fragment cover, soil organic matter content and soil erodibility are recognised as important determinants of runoff and soil loss (e.g. Cerdan et al., 2006, 2010; Poesen et al., 1994; Poesen and Lavee, 1994; Sanchis et al., 2008; Torri et al., 1997), quantitative data on these properties were not systematically reported in the literature from which plot data were extracted for the database. Hence, a quantitative analysis of the effect of these soil properties on R_a and SL_a could not be made.

2.1.2 Annual data and data extrapolation procedures

All data used in this analysis are annual data. More than 84% of plots (corresponding to >90% of the number of plot-years) reported R_a and SL_a that were obtained during a measuring period of one or more years and are reported as annual values by the author. In some studies, however, measurements were not carried out for full years, but the authors indicated that the data were representative for full years because no or negligible runoff and soil loss occurred during the period that the runoff plots were not in operation. This was the case for some studies during the dry season in the Mediterranean region (e.g. Roxo et al., 1996; Mohammad and Adam, 2010) or during permanent snow cover and frozen soil in colder climates (e.g. Fulajtár and Janský, 2001). In these cases, the measuring period was considered to be full years when calculating the corresponding number of plot-years for that plot. Other studies where the measuring period is shorter than 1 year were only included in the plot database when the authors explicitly report data as annual (extrapolation by the authors), or a reasonable extrapolation could be made. This was only done if measurements were conducted for a period during which at least two thirds of the P_a depth was recorded and rainfall is distributed uniformly throughout the year. In these cases R_a and SL_a were estimated by linear extrapolation according to the corresponding P_a. If no P_a data were available, data were extrapolated linearly to annual values according to the corresponding number of days if measurements continued for a period of at least 80% of the year and long-term average daily rainfall is distributed uniformly throughout the year (i.e. no distinctive dry and wet seasons). Uniformity of daily rainfall was assessed visually using the long-term average daily rainfall distribution for the plot measuring station, as given by the New_LocClim program (FAO, 2006).

2.2 Description of the plot database

The plot database contains data from 227 plot-measuring stations throughout Europe and the Mediterranean region (Fig. 2.1), compiled from 213 individual publications. Annual SL_a data are available for a total of 1 056 plots, corresponding to a total of 7 204 plot-years. R_a data were available for 804 plots (5 327 plot-years). For 766 of these plots, representing 5 013 plot-years, both R_a and SL_a data are available. For 673 plots (corresponding to 4 583 plot-years), both R_a and P_a are reported, allowing the calculation of annual runoff coefficients (RC_a). The distribution of the number of plots and plot-years over the different countries in the study area and the references to the data sources are given in Table 2.3. This database is substantially larger than any previously published plot runoff and soil loss database for Europe and the Mediterranean (Table 1.1). It is the first database to consider both R_a and SL_a as well as the relations between R_a and SL_a at a continental scale. Furthermore, it is also the most detailed database to date as it includes individual annual data when these are available, along with data on several factors controlling R_a and SL_a such as P_a, plot length, plot slope gradient, soil texture and soil organic matter content. Along with the database of plot runoff and soil loss data for the application of SWCTs described in chapter 7, it offers a tool to quantitatively evaluate R_a and SL_a rates for the whole of Europe and the Mediterranean, as well as the factors that control R_a and SL_a, and means of reducing R_a and SL_a rates through land use change or the application of SWCTs.

The first recorded soil loss measurements in the database started in 1950 at Cean-Turda, Romania (Motoc et al., 1998) and later on the number of plots increased until 1994, after which the number of plots started to decline. The earliest publication discussing plot measurements that could be found dated from 1968 (Dubber, 1968), although most of the publications date from 1986 onwards (Fig. 2.2). The average measuring period of all runoff and soil loss plots is 6.0 yrs. (median: 4 yrs., mode: 1 yr.) with a minimum of 1 and a maximum of 42 yrs. at Podu-Iloaiei, Romania (Bucur et al., 2007). (Fig. 2.3). For most plots (>84% of plots and >90% of the number of plot-years), measurements continued throughout the year for at least one full year. $R_a$and SL_a data calculated for a measuring period less than 1 yr which were extrapolated to a full year account for less than 3% of all plots and less than 1% of the number of plot-years (Table 2.4).

Table 2.3: Overview per country in Europe and the Mediterranean of the number of plots (PL), number of plot-years (PY) and sources included in the plot database.

Country	PL	PY	Source
Albania	14	66	Grazhdani et al., 1996; Grazhdani, 2006; Grazhdani et al., 1999; Grazhdani, personal communication
Algeria	60	233	Arabi and Roose, 1993; Mazour, 1992; Mazour et al., 2008; Morsli et al., 2004
Austria	3	33	Klik, 2003, 2010; Klik, personal communication; Strauss and Klaghofer, 2006
Belgium	2	17	Bollinne, 1982; Govers and Poesen, 1988; Verstraeten et al., 2006d
Bulgaria	43	377	Kroumov and Malinov, 1989; Rousseva et al., 2006
Croatia	2	10	Basic et al., 2001; Basic et al., 2004
Cyprus	7	14	Lenthe et al., 1986; Lüken, personal communication
Denmark	10	41	Schjønning et al., 1995; Veihe and Hasholt, 2006
Finland	20	102	Puustinen et al., 2005, 2007; Tattari and Rekolainen, 2006; Turtola and Paajanen, 1995; Turtola et al., 2007; Uusi-Kämppä, 2005
France	37	277	Auzet et al., 2006; AREDVI, 2003; Ballif, 1989; Brenot et al., 2006, 2008; Clauzon and Vaudour, 1969, 1971; Le Bissonnais et al., 2004; Martin, 1990; Martin et al., 1997; Messer, 1980; Viguier, 1993; Wicherek, 1986, 1988, 1991
Germany	102	330	Ammer et al., 1995; Auerswald, 2006; Auerswald et al., 2009; Barkusky, 1990; Biemelt et al., 2005; Botschek, 1991; Deumlich and Frielinghaus, 1994; Deumlich and Gödicke, 1989; Dikau, 1983, 1986; Dubber, 1968; Emde, 1992; Emde et al., 2005; Engels, 2009; Felix and Johannes, 1993; Fleige and Horn, 2000; Frielinghaus, 1998; Jung and Brechtel, 1980; Kleeberg et al., 2008; Richter, 1985, 1991; Richter and Kertesz, 1987; Saupe, 1990, 1992; Voss, 1978
Greece	36	84	Arhonditsis et al., 2000; Diamantopoulos et al., 1996; Dimitrakopoulos and Seilopoulos, 2002; Kosmas et al., 1996; Kosmas et al., 2006
Hungary	14	56	Hudek and Rey, 2009; Kertész, personal communication; Kertész and Centeri, 2006; Kertész and Huszár-Gergely, 2004; Kertesz et al., 2007; Pinczés, 1982; Richter and Kertesz, 1987; Richter, 1987
Israel	29	140	Agassi and Benhur, 1991; Inbar et al., 1997, 1998; Kutiel and Inbar, 1993; Lavee, personal communication; Lavee et al., 1998
Italy	80	609	Bagarello et al., 2010a,b; Bagarello and Ferro, 2010; Basso et al., 2002; Basso et al., 1983a,b; Bini et al., 2006; Caredda et al., 1997; Caroni and Tropeano, 1981; Chisci and Zanchi, 1981; Chisci, 1989; De Franchi and Linsalata, 1983; de Vente et al., 2007; Ollesch and Vacca, 2002; Porqueddu and Roggero, 1994; Postiglione et al., 1990; Rivoira et al., 1989; Torri et al., 2006; Tropeano, 1984; Vacca, personal communication; Vacca et al., 2000; Zanchi, 1983, 1988a,b
Jordan	2	4	Abu-Zreig, 2006; Abu-Zreig et al., 2011
Lithuania	103	792	Feiza et al., 2007; Jankauskas, personal communication; Jankauskas and Fullen, 2002, 2006; Jankauskas and Jankauskiene, 2003a,b; Jankauskas et al., 2004, 2007, 2008
Macedonia	8	36	Blinkov and Trendafilov, 2006; Jovanovski et al., 1999
Morocco	29	164	Chaker et al., 2001; Heusch, 1970; Laouina et al., 2003; Moufaddal, 2002; Yassin et al., 2009; Yassin, personal communication
Norway	10	82	Børresen, personal communication; Grønsten and Lundekvam, 2006; Lundekvam, 2007; Øygarden, 1996; Øygarden et al., 2006
Palestinian territories	9	42	Abu Hammad et al., 2004, 2006; Al-Seekh and Mohammad, 2009; Mohammad and Adam, 2010
Poland	10	79	Gil, 1986, 1999; Rejman and Rodzik, 2006; Rejman et al., 1998; Skrodzki, 1972; Stasik and Szafranski, 2001; Szpikowski, 1998
Portugal	52	406	Coelho, 2006; de Figueiredo, personal communication; de Figueiredo and Gonçalves Ferreira, 1993; de Figueiredo and Poesen, 1998; de Figuiredo et al., 2004; Lopes et al., 2002; Nunes and Coelho, 2007; Roxo et al., 1996; Shakesby et al., 1994
Romania	22	568	Bucur et al., 2007; Ene, 1987; Ionita, 2000; Ionita et al., 2006; Motoc et al., 1998; Nistor and Ionita, 2002; Teodorescu and Badescu, 1988
Serbia	6	74	Đjorović, 1990; Kostadinov et al., 2006; Sekularac and Stojiljkovic, 2007
Slovakia	62	104	Chomanicová, 1988; Fulajtár and Janský, 2001; Gajdová et al., 1999; Stankoviansky et al., 2006; Suchanic, 1987
Slovenia	4	19	Horvat and Zemljic, 1998; Hrvatin et al., 2006

Table 2.3: *Continued*

Country	PL	PY	Source
Spain	156	876	Albaladejo and Stocking, 1989; Albaladejo et al., 2000; Andreu et al., 1998a,b, 2001; Aspizua, 2003; Bautista et al., 1996, 2007; Bienes et al., 2006; Campo et al., 2006; Castillo et al., 1997, 2000; Cerdà and Lasanta, 2005; Chirino et al., 2006; de Vente and Poesen, 2005; Durán Zuazo et al., 2004, 2008; Francia Martínez et al., 2006; García-Ruiz et al., 1995; Gimeno-Garcia et al., 2007; Gómez et al., 2004, 2009; Gómez Plaza, 2000; González-Pelayo et al., 2010; Guerra et al., 2004; Ingelmo et al., 1998; Lasanta et al., 2006; Lopez-Bermudez et al., 1991; Martinez-Mena et al., 1999, 2001; Martínez-Murillo and Ruiz-Sinoga, 2007; Martinez Raya et al., 2006; Nadal Romero, personal communication; Puigdefábregas et al., 1996; Rodríguez Rodríguez et al., 2002, 2006; Romero-Díaz and Belmonte Serrato, 2008; Romero-Díaz et al., 1999; Rubio et al., 1997; Sanchez et al., 1994; Schnabel et al., 2001; Solé Benet, 2006; Solé Benet, personal communication; Soler et al., 1994; Soto and Díaz-Fierros, 1998; Williams et al., 1995
Sweden	6	52	Ulén, 1997, 2006; Ulén and Kalisky, 2005
Switzerland	12	218	Marxer, 2003; Schaub, 1998; Weisshaidinger and Leser, 2006
Syrian Arab Republic	7	20	Bruggeman et al., 2005; Masri et al., 2005; Shinjo et al., 2000
The Nether-lands	3	19	Kwaad, 1991, 1994; Kwaad, personal communication; Kwaad et al., 1998, 2006
Tunisia	9	76	Ben Chaabane and Hamrouni, 2008; Bourges et al., 1973, 1975; Kaabia, 1995
Turkey	84	1233	Erpul, personal communication; Kara et al., 2010; Köse and Taysun, 2002; Köse et al., 1996; Oguz, personal communication; Oguz et al., 2006; Ozhan et al., 2005
United Kingdom	46	293	Bhattacharyya et al., 2008, 2009; Boardman and Poesen, 2006; Brown, 1996; Fullen, 1992; Fullen et al., 2006; Fullen and Brandsma, 1995; Fullen and Reed, 1986; Fullen, 1998; Fullen and Booth, 2006; Mitchell et al., 2003; Morgan and Duzant, 2008; Quinton and Catt, 2004

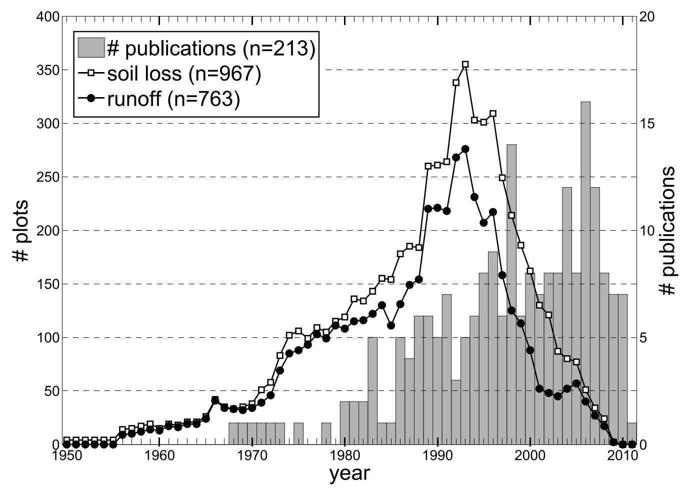


Figure 2.2: Evolution of the total number of plots (# plots) in operation per year for which annual soil loss and/or runoff were recorded in Europe and the Mediterranean as well as the total number of publications from which plot data were extracted in this study (# publications) per publication year.

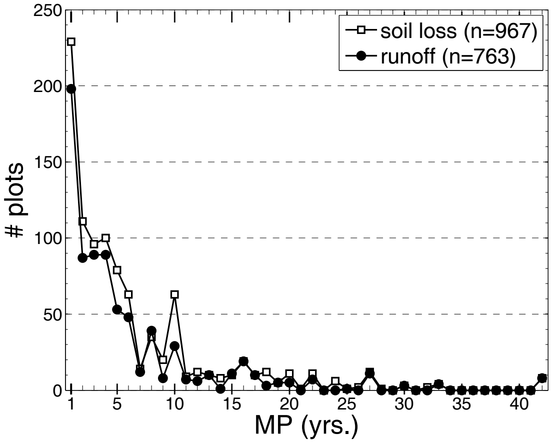


Figure 2.3: Frequency distribution of the number of plots (# plots) as a function the measuring period (MP) for which annual runoff or annual soil loss were measured continuously in Europe and the Mediterranean.

Table 2.4: Data collection period of plot data in the plot database. R_a: annual runoff, RC_a: annual runoff coefficient, SL_a= annual soil loss. Number of plots (PL) and number of plot-years (PY) for which plot data were collected during (1) full years (Full Year), (2) during a representative part of the year during which almost all of the annual rain was recorded and the authors considered the data to be representative for a full year (Repr. for Full Year), (3) a measuring period less than one year during which at least 67 % of the annual rain was recorded and for which the data were linearly extrapolated to 100% of the annual rainfall (Extrapol. to Full Year).

data collection period	PL (% of total)			PY (% of total)		
	R_a	RC_a	SL_a	R_a	RC_a	SL_a
Full Year	695 (84.7%)	594 (84.3%)	928 (86.2%)	4971 (91.0%)	4495 (90.6%)	6848 (93.1%)
Repr. for Full Year	115 (14.0%)	100 (14.2%)	122 (11.3%)	464 (8.5%)	437 (8.8%)	459 (6.2%)
Extrapol. to Full Year	11 (1.3%)	11 (1.6%)	26 (2.4%)	30 (0.5%)	30 (0.6%)	45 (0.6%)
Total	821 (100%)	705 (100%)	1076 (100%)	5465 (100%)	4962 (100%)	7352 (100%)

The distribution of the number of plots and plot-years over the different land use types and climatic zones is given in Table 2.5, while the distributions of the number of plots and plot-years according to P_a, plot length and plot slope gradients are given in Fig. 2.4. The different land use types in the database also show different frequency distributions (Fig. 2.6), with cropland and fallow mainly occurring on gentler slopes (<20%), while forest, vineyards, construction sites and tree crop plots are generally situated on steeper slopes (>20%).

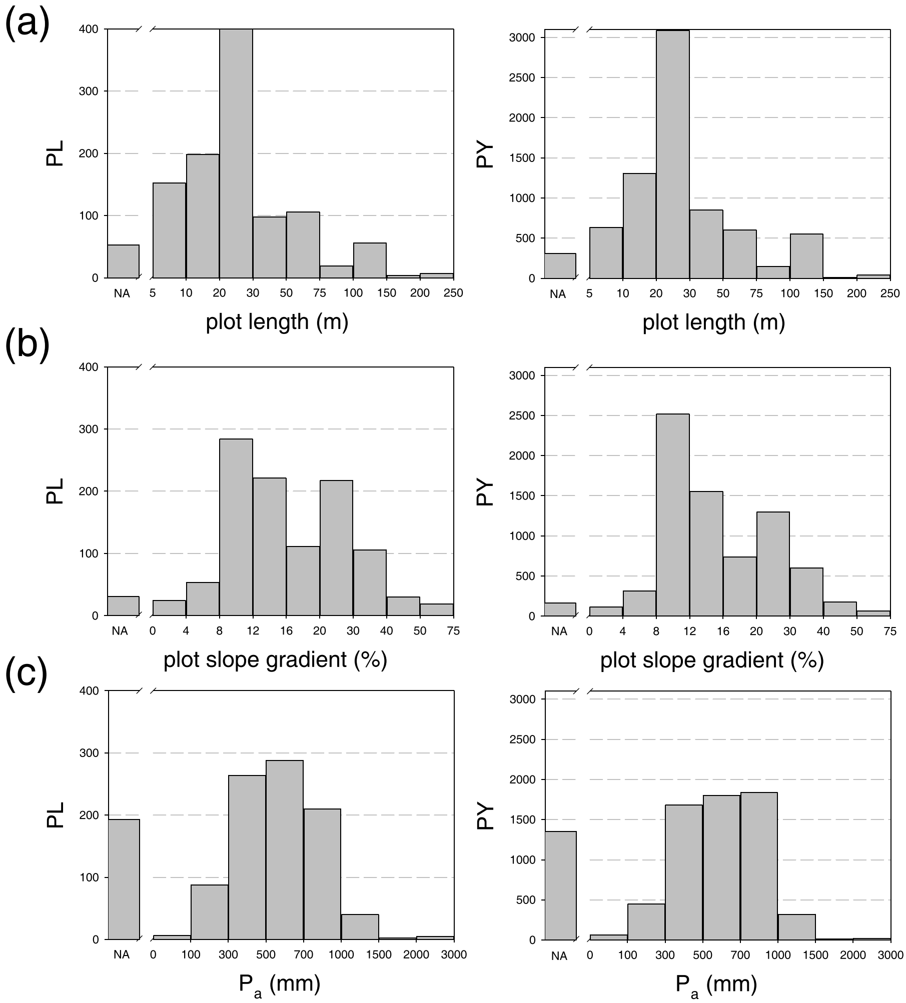


Figure 2.4: Frequency distribution of the number of plots (PL) and the number of plot-years (PY) in Europe and the Mediterranean for which annual runoff and/or annual runoff coefficient, and/or annual soil loss data are available with respect to: (a) plot length, (b) plot slope gradient and (c) annual precipitation (P_a). Total number of plots= 1096, total number of plot-years= 7533. NA: not available; i.e. plot length, slope gradient or P_a not reported.

Table 2.5: Overview of the number of plots (PL) and the number of plot-years (PY) for which annual runoff (R_a), annual runoff coefficient (RC_a) and/or annual soil loss (SL_a) data for Europe and the Mediterranean are available. NA: not available.

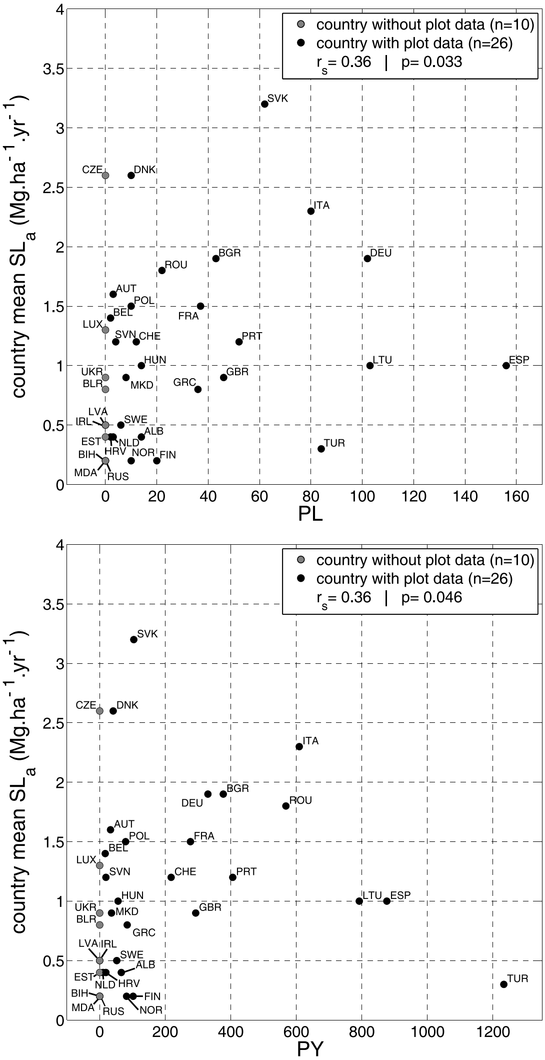
land use type	climatic zone	R_a PL (PY)	RC_a PL (PY)	SL_a PL (PY)
bare	**all data**	**133 (1 362)**	**95 (1 058)**	**182 (1 740)**
	pan-Mediterranean	72 (857)	58 (656)	100 (1 129)
	temperate	59 (490)	35 (387)	80 (596)
	cold	2 (15)	2 (15)	2 (15)
construction sites	**all data**	**3 (11)**	**3 (11)**	**3 (11)**
	pan-Mediterranean	2 (10)	2 (10)	2 (10)
	temperate	1 (1)	1 (1)	1 (1)
cropland	**all data**	**302 (2 018)**	**244 (1 737)**	**397 (2 749)**
	pan-Mediterranean	161 (1 136)	145 (1 035)	175 (1 232)
	temperate	114 (683)	79 (563)	147 (874)
	cold	27 (199)	21 (139)	76 (644)
fallow	**all data**	**47 (221)**	**46 (216)**	**60 (281)**
	pan-Mediterranean	23 (165)	23 (165)	25 (173)
	temperate	21 (46)	21 (46)	26 (86)
	cold	3 (10)	2 (5)	9 (22)
forest	**all data**	**59 (301)**	**55 (277)**	**59 (334)**
	pan-Mediterranean	41 (238)	41 (238)	40 (217)
	temperate	17 (58)	14 (39)	18 (113)
	cold	1 (5)	NA	1 (5)
grassland	**all data**	**69 (506)**	**52 (431)**	**109 (779)**
	pan-Mediterranean	30 (196)	17 (145)	29 (192)
	temperate	34 (296)	31 (277)	24 (233)
	cold	5 (14)	4 (9)	56 (355)
post-fire	**all data**	**54 (223)**	**46 (188)**	**56 (224)**
	pan-Mediterranean	49 (202)	43 (179)	51 (203)
	temperate	3 (9)	3 (9)	3 (9)
	cold	2 (12)	NA	2 (12)
rangeland	**all data**	**14 (59)**	**14 (59)**	**17 (69)**
	pan-Mediterranean	13 (56)	13 (56)	15 (64)
	temperate	1 (2)	1 (2)	2 (6)
shrubland	**all data**	**84 (372)**	**79 (351)**	**111 (589)**
	pan-Mediterranean	77 (357)	72 (336)	101 (559)
	temperate	7 (15)	7 (15)	10 (30)
tree crops	**all data**	**13 (133)**	**13 (133)**	**23 (154)**
	pan-Mediterranean	10 (59)	10 (59)	20 (80)
	temperate	3 (74)	3 (74)	3 (74)
vineyard	**all data**	**26 (123)**	**26 (123)**	**39 (272)**
	pan-Mediterranean	12 (90)	12 (90)	18 (107)
	temperate	14 (33)	14 (33)	21 (165)
Database Total		**804 (5 327)**	**673 (4 583)**	**1 056 (7 204)**

2.3 How representative are the available plot runoff and soil loss data for Europe and the Mediterranean?

The assessment of R_a and SL_a rates based on the review of data measured on runoff and soil loss plots is inherently biased since runoff plot experiments are generally set up to answer specific research questions and not to be representative for the entire range of actual hillslope conditions (Cerdan et al., 2006; Auerswald et al., 2009; Vanmaercke et al., 2012a). Many plot measuring stations are located in areas that experience interrill and rill erosion. Furthermore, no plots are located in Iceland or in Scandinavia above 61 degrees latitude (Fig. 2.1), probably due to logistic problems in cold environments and the low population density in these areas. A comparatively small number of runoff plots was found for North-Eastern Europe (Table 2.3), since many of these data have not been published in international journals and are not easily accessible. While even more plot data are likely existing, the database compiled in this study is currently the largest compilation of field-measured R_a and SL_a data at plot scale for Europe and the Mediterranean region.

If runoff and soil loss plot measurements are representative for the interrill and rill erosion problem in Europe and the Mediterranean, it can be expected that countries with higher interrill and rill erosion rates or with larger areas affected by these processes have made a larger effort to measure R_a and SL_a on runoff and soil loss plots. A comparison between the number of plots and plot-years for each country in Table 2.5 and the estimated mean SL_a for each country (derived from Cerdan et al. (2010)) is given in Fig. 2.5. The Spearman rank correlation coefficients (r_s) for the relations between the number of plots and plot-years, and the mean country SL_a are 0.36 and 0.33, respectively, and both correlations are significant at α=0.05 (Fig. 2.5). Hence, there is a significant positive correlation between the mean SL_a for individual countries and the number of plots and plot-years, but it is subject to a large scatter (Fig. 2.5). Furthermore, for some countries with a relatively large estimated mean SL_a (>2.5 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$, i.e. Czech Republic, Denmark and Slovakia) comparatively few soil loss plot data were reported (number of plots= 0, 10 and 62, respectively).

The lack of runoff and soil loss plot studies for countries with a relatively large mean SL_a is attributed to a variety of reasons; for the Czech Republic, erosion plot measurements are known to have existed (Dostál et al., 2006), but no data could be obtained for this study. For Denmark and Slovakia, more runoff and soil loss plot data are available (Table 7.1), but as they focus on the application of SWCTs, they are not included in the data for Fig. 2.5. For several countries such as Belgium, more runoff and soil loss plot data are available



Abbreviation	**Country**
ALB	Albania
AUT	Austria
BEL	Belgium
BGR	Bulgaria
BIH	Bosnia and Herzegovina
BLR	Belarus
CHE	Switzerland
CZE	Czech Republic
DEU	Germany
DNK	Denmark
ESP	Spain
EST	Estonia
FIN	Finland
FRA	France
GBR	United Kingdom
GRC	Greece
HRV	Croatia
HUN	Hungary
IRL	Ireland
ITA	Italy
LTU	Lithuania
LUX	Luxembourg
LVA	Latvia
MDA	Moldova
MKD	Macedonia
NLD	The Netherlands
NOR	Norway
POL	Poland
PRT	Portugal
ROU	Romania
RUS	Russian Federation
SVK	Slovakia
SVN	Slovenia
SWE	Sweden
TUR	Turkey
UKR	Ukraine

Figure 2.5: Comparison between the number of plots (PL) and plot-years (PY) and estimated mean annual soil loss (SL_a) per country. Annual soil loss estimates for each country as reported by Cerdan et al. (2010), while the number of plots and plot-years is taken from Maetens et al. (2012b). r_s= Spearman's rank correlation coefficient, p= correlation p-value.

but these data were not included as they were not measured over full years (e.g. Leys, 2008). On the other hand, in some countries such as Lithuania, Spain, and Turkey, a large number of plot runoff and soil loss data are available while the mean SL_a for those countries are relatively small (Fig. 2.5). This indicates that a disproportionately large effort to quantify R_a and SL_a through interrill and rill erosion is made, while this is not the largest erosion problem in those countries (e.g. de Vente and Poesen, 2005; Vanmaercke et al., 2012a). Another possible reason for the disproportionately large number of plots and plot-years in comparison to mean SL_a is that interrill and rill erosion problems in those countries are concentrated in e.g. regions with erodible soils or steep topography and the country mean SL_a is not a good indicator for the erosion problems. Hence, in this study the focus is on subdivisions that are related to interrill and rill erosion processes (e.g. land use types, hydrologic soil groups or climatic zones), rather than on a subdivision into countries that may not provide meaningful further insights. In addition, an effort is made to assess the way R_a and SL_a rates are distributed, rather than to estimate only mean R_a and SL_a rates.

The relatively short measuring periods for the plots (mean: 6 yrs., median: 4 yrs.) can be attributed to the relatively short duration of research or PhD. projects during which most of the plots are established, and the high cost of maintaining runoff and soil loss plots. This also means that it is difficult to assess temporal variability in R_a and SL_a on the longer term for a specific plot, which is nevertheless an important aspect for e.g. conservation planning (Bagarello et al., 2011). Figure 2.2 shows a clear decline of the number of plots in operation after 1996. This can be attributed to the fact that plot measurements are time- and labour-consuming and field-measured plot experiments are abandoned in favour of modelling studies (e.g. Merritt, 2003), which have become more prevalent with increasing and cheaper computing power over the last 20-30 years. A similar decline in catchment sediment yield studies was observed by Vanmaercke et al. (2011b), but already starting between 1970-80 when many large state-led monitoring schemes were abandoned. Most plot-scale studies are conducted by individual researchers however, and the number of publications on runoff and soil loss plots has not decreased in recent years (Fig. 2.2). Moreover, recent publications review previously published data to assess erosion rates and variability under field conditions at national or regional scales (Table 1.1), to use these data in new spatial analyses (e.g. Cerdan et al., 2010) or to validate erosion models (e.g. Amore et al., 2004; Licciardello et al., 2009; Tsara et al., 2005; Quinton, 1994). Hence, there is an ongoing interest in field-measured R_a and SL_a data and a review and regular update of existing databases offers opportunities to use these data in new analyses and thus to give added value to previously published studies (Baade and Rekolainen, 2006).

A large proportion of research has been carried out on bare and cropland plots (Table 2.5). Runoff and soil loss after wildfires have received considerable attention, despite R_a and SL_a rates still being low in comparison to those witnessed on some other land uses (Shakesby, 2011, Table 3.2). Permanent crops (tree crops and vineyards) have received more attention in plot studies in comparison to the areal percentage they represent in the CORINE land cover map (Vanmaercke et al., 2012a). Despite the relatively small area occupied by these land use types, they are important contributors to total soil loss in Europe (Cerdan et al., 2010) and can be the dominant land use type in certain regions. Furthermore, some of the highest RC_a (up to 40.2 %) and SL_a rates (up to 151 $Mg \cdot ha^{-1} \cdot yr^{-1}$) recorded in the database occurred on permanent crops (i.e. vineyards and tree crops) (Fig. 3.2, Table 3.2). The quantitative measurements currently available for these land use types may still not be sufficient to make a comprehensive assessment of erosion risk in these land use types (e.g. Gómez et al., 2008).

Also the distribution of plot lengths and slope gradients of the plots shows a research bias (Fig. 2.4). Plots with a plot length between 20 and 30m, i.e. being close to the standard RUSLE plot (22.1m, Renard et al., 1997), are by far the most frequent. Plot length is mainly determined by logistic limitations of the studies and the relation to actual field slope lengths, for which very few data are available, is unknown. On the other hand, observed frequency distributions of the plot slope gradients for different land use types (Fig. 2.6) show both a difference between slope gradient distributions for different land use types as they occur in the field as well as a research bias towards steeper slopes. This was demonstrated by Cerdan et al. (2006) who found that mean slope gradient of plots on grassland, forest and shrubland corresponded well to mean slope gradient for these land use types on the reclassified CORINE land cover map, while mean slope gradients for arable, vineyards, orchard and post-fire plots were found to be steeper than the CORINE average. On the whole, the plots show a relatively steep slope gradient, with the majority of plots having a slope gradient above the 9% of the standard RUSLE plot (figure 4). As was shown by Boardman (1998), careless extrapolation or generalization of these data can therefore lead to overestimations of SL_a rates. Therefore, the controlling environmental variables should always be accounted for when evaluating or extrapolating rates of R_a, RC_a and SL_a (Cerdan et al., 2010; de Vente et al., 2011). Hence, the rates of R_a and SL_a presented in this study may not be representative for the flat regions in Europe and the Mediterranean (Cerdan et al., 2010). Nevertheless, the extent of the database compiled in this study allows for the best representation of the relations between measured R_a, SL_a and P_a for different land use types currently available.

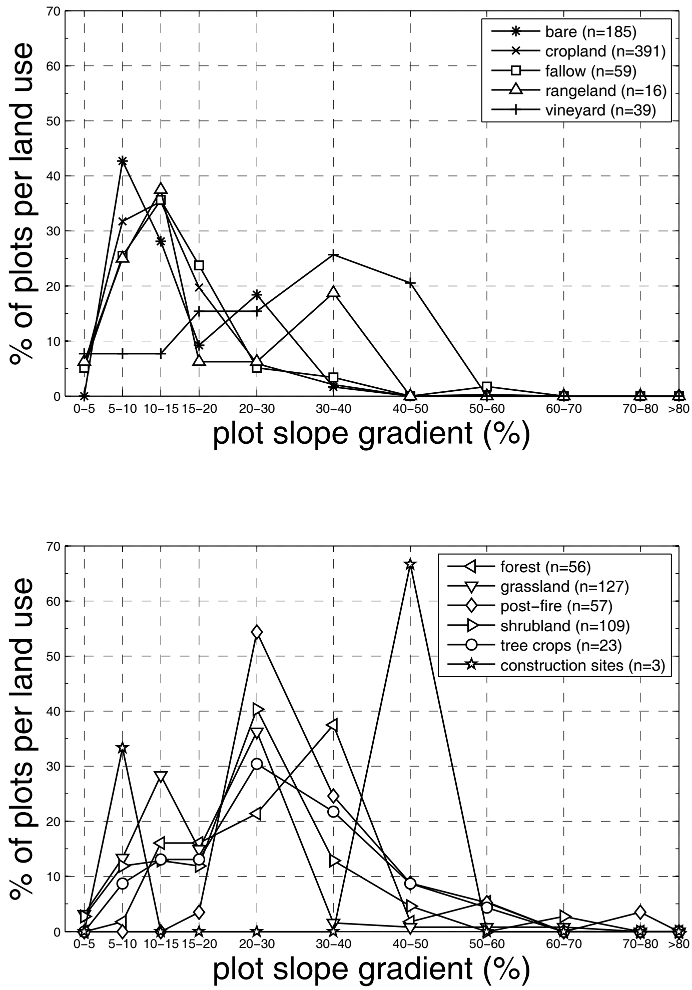


Figure 2.6: Frequency distribution of the percentage of plots for the different land use types in Europe and the Mediterranean according to plot slope gradient class. n= total number of plots for a given land use type.

While the use of runoff and soil loss plots allows for a relatively easy assessment of R_a, runoff has received considerably less attention than soil loss in the literature as can be noted in the literature review (Table 1.1) and the number of plots and plot-years (Table 2.5). One of the problems underlying this discrepancy is the complex relationship between R_a and environmental variables (Wischmeier, 1966) and as a result also the relation between R_a and SL_a is less studied.

Nevertheless, the assessment of R_a is an important part of many erosion models (Merritt, 2003) and hence a better understanding of runoff generation and runoff-soil loss relationships would contribute towards better erosion models (e.g. Kinnell and Risse, 1998). Runoff generation is also an important problem in itself, both on-site (e.g. loss of plant available water: Wallace, 2000) and off-site (e.g. flash floods). For instance, water is a key resource in the Mediterranean (Araus, 2004; Vanmaercke et al., 2011b) and RC_a are often higher than RC_a for comparable land uses in temperate regions, while SL_a rates tend to be lower in the pan-Mediterranean climatic zone than in the temperate climatic zone (Table 3.2, Fig. 3.3), hence excessive runoff may be of more concern than soil loss.

2.4 Reliability of runoff and soil loss plot data.

Several authors indicate that differences in experimental methodology can substantially affect the observed measurements (e.g. Evans, 1995; Bagarello and Ferro, 1998; Blanco-Canqui et al., 2004b; Boix-Fayos et al., 2007; Hudson, 1993; Zöbisch et al., 1996; Stroosnijder, 2005). While some publications specifically address technical and methodological aspects of plot runoff and soil loss measurements (e.g. Cammeraat, 1993; Hudson, 1993), there is no strictly defined universal protocol for the design of runoff and soil loss plots and many details of the experimental set-up are often determined by logistical constraints during the study. Basic aspects of good plot design such as plot borders and flow splitters between subsequent collection tanks, and measurement methodology such as thoroughly stirring the runoff water in collector tanks for a representative sediment sampling are generally well observed and reported by researchers. When clear measurement failures occurred such as collector tank overtopping or plot border failure, the reported runoff and soil loss data were not incorporated in the database. However, the magnitude and impact of other measurement uncertainties is hard to assess, as even replicate (i.e. identical adjacent plots on the same site) show considerable variability in measured R_a and SL_a rates. Nearing et al. (1999) measured coefficients of variation between 14% and 150% for event SL from replicated plots over a wide range of plot measuring stations and plot conditions. Nearing et al. (1999) also observed that the coefficient of variation in measured data is larger for smaller SL_a rates. Wendt et al. (1985) found coefficients of variation of about 20% for both event R and SL for replicated plots at the same study site. As both studies (Nearing et al., 1999; Wendt et al., 1985) considered replicate plots, the observed variability is largely due to the inherent natural variability of interrill and rill erosion processes under field conditions .

Few assessments of the reliability of runoff and soil loss plot data with respect to experimental methodology exist. For instance, Bollinne (1982) performed an error assessment of measured R and SL data due to experimental procedures such as sample weighing and cleaning of the collection gutters. Errors for measurements of soil loss, runoff and sediment concentration were found to be at maximum 14%, 20% and 5%, respectively. In the same study, the variability of R, SL and sediment concentration of individual plots in a set of three replicates was found to be generally larger than the observed measuring uncertainties. The deviations of individual plots from the average of three replicates were found to range between -18.6% and +81,5% for SL, between -9.6% and +107,1% for R and between -58.9% and +82,9% for sediment concentration over 4 observed events. While these assessments of measuring uncertainties are certainly useful, they are very rare in the international literature.

With respect to the R_a and SL_a data included in this study, care was taken to include only data from reliable scientific sources (i.e. peer-reviewed publications, project reports, edited books and PhD. theses) that meet the standards for data quality described in section 2.1.1. Nevertheless, there are still sources for uncertainty in the compiled data. Firstly, whenever volumetric rill measurements of soil loss ($m^3 \cdot ha^{-1} \cdot yr^{-1}$; Govers and Poesen, 1988; Feiza et al., 2007; Jankauskas and Fullen, 2002; Jankauskas and Jankauskiene, 2003a; Jankauskas et al., 2004) are converted to masses ($Mg \cdot ha^{-1} \cdot yr^{-1}$), there are additional uncertainties on the measured dry bulk density and the 25% added to the measured SL_a to account for interrill erosion (based on Govers and Poesen (1988)). Outside the Boreal climatic zone, plots with volumetric SL_a measurements constitute a minor part of the database (number of plots= 5, number of plot-years= 117) and the uncertainties involved will not affect overall results substantially. However, when SL_a in the Boreal climatic zone is considered separately, as a substantial part of the plot data from this climatic zone is based on the volumetric measurements of rills on plots at the Lithuanian Research Centre of Agriculture and Forestry, Kaltinenai, Lithuania (number of plots= 103, number of plot-years= 792; Feiza et al., 2007; Jankauskas and Fullen, 2002; Jankauskas and Jankauskiene, 2003a; Jankauskas et al., 2004), and comparison of SL_a rates from the Boreal climatic zone with SL_a rates from other climatic zones should be done with caution. Secondly, also the extrapolation procedures for the plots where at least two thirds of P_a was measured and rainfall was measured (see section 2.1.2) introduce a degree of uncertainty. Nevertheless, the extrapolated plot data represent less than 2.4% of the total number of plots and less than 0.6% of the total number of plot-years.

On the whole, the uncertainties due to the data compilation procedures are of minor importance, both compared to the natural variability often reported in plot runoff and soil loss studies (e.g. Nearing et al., 1999; Wendt et al., 1985) and as percentage of the total number of plots and number of plot-years in the database. Hence, it is reasonable to expect that the uncertainties and variability experienced in the field assessment of R_a and SL_a are a good reflection of the variability in R_a and SL_a actually occurring in the field. A large part of the challenge in erosion research today is dealing with this uncertainty and complexity. Hence, rather than dismissing plot-measured data as incorrect and uninformative on the basis of the observed variability (e.g. Hudson, 1993), future research should recognise this as an essential aspect of R and SL assessment and further develop methods to incorporate these uncertainties in the analysis.

2.5 Conclusions

The plot database compiled for this study comprises 227 plot-measuring sites in Europe and the Mediterranean, with SL_a data for 1056 plots representing 7024 plot-years and R_a data for 804 plots representing 5327 plot-years. This study is the largest currently available database of plot-measured R_a and SL_a data, covering the whole of Europe and the Mediterranean and is contains a substantially larger amount of data in comparison to previously compiled databases (Table 1.1). It is also the first compilation of plot runoff and soil loss data to explicitly consider both R_a and SL_a data and the relations between R_a and SL_a. The large number of data and inclusion of individual annual data, rather than only per-plot averages, allows to better assess mean values, as well as frequency distributions of R_a and SL_a data. Furthermore, the inclusion of several key environmental factors such as annual precipitation, plot length, plot gradient and soil texture allows to assess the effects of these factors on both R_a and SL_a, as well as on the relation between both.

The large number of plots and plot-years also means that exceptional values can be better identified. Also bias in the database, such as the predominance of plots with standard USLE dimensions (plot length of 22.13 m on a slope gradient of 9%), and differences in slope frequency distributions between different land use types can also be better identified, allowing a better interpretation of the results. R_a rates have received considerably less attention in the literature than SL_a, and are only considered to a limited extent in review studies. Compared to other land use types such as cropland and shrubland, some erosion-prone land use types such as vineyards (with respect to R_a) and tree crops (with respect to both R_a and SL_a) have attracted relatively little research attention by means of runoff and soil loss plot measurements under natural rainfall. This plot data compilation also allows to direct future research towards specific land use conditions which have been under-researched.

A decrease in the number of studies using runoff and soil loss plots under natural rainfall is observed since the mid-1990s in Europe and the Mediterranean. This is attributed to the labour-intensive and costly nature of runoff and soil loss plot studies under natural rainfall and a replacement of field plot studies by modelling studies. Nevertheless, there is an ongoing interest in plot-measured data and compilation of these data. As more data likely exist but are not easily accessible, further expansion of this database with other data in the 'grey literature' is useful, especially as collecting these field data is labour- and time-consuming and hence costly. Many of the original source data of these measurements are at risk of being lost as original reference copies of the publications disappear as well as researchers who originally collected the data retire.

Chapter 3

Effects of land use on annual runoff and soil loss in Europe and the Mediterranean: A meta-analysis of plot data

This chapter is based on: Maetens, W., Vanmaercke, M., Poesen, J., Jankauskas, B., Jankauskiene, G. and Ionita, I., 2012. Effects of land use on annual runoff and soil loss in Europe and the Mediterranean: A meta-analysis of plot data. Progress in Physical Geography 36(5): 597 - 651. doi:10.1177/0309133312451303

3.1 Introduction

Runoff and soil loss due to interrill and rill erosion are important processes of soil degradation that cause significant on-site and off-site problems (e.g. Boardman and Poesen, 2006; Montgomery, 2007b; Poesen and Hooke, 1997). An integrated approach to these problems at sub-continental scale requires runoff and soil loss to be assessed for a wide range of representative environmental conditions. An extensive assessment and mapping approach for large areas (e.g. Evans, 2002; Le Gouée et al., 2010; Oldeman et al., 1991) may provide an overview of the scale of the problem and locate erosion hotspots, but to gain insight in these processes and to develop strategies to mitigate their impacts, more detailed field-measured experimental data that accurately quantify soil loss are needed. Recently, such quantitative assessment of annual soil loss (SL_a, $Mg \cdot ha^{-1} \cdot yr^{-1}$) rates at a pan-continental scale for Europe has seen several applications like risk assessment mapping through modelling (Kirkby et al., 2004), exploration of spatial variability and controlling factors of SL_a rates (Cerdan et al., 2006, 2010), assessment of scale effects on sediment production (Vanmaercke et al., 2012a) and the development of indicator systems to identify and monitor problem areas (e.g. Gobin et al., 2004). All these continental-wide applications either directly make use of available field-measured soil loss data or conclude they would benefit from a validation with such data.

While aforementioned studies place strong emphasis on the assessment of SL_a rates, runoff plays an important role as a causal factor of SL_a and the relations between annual runoff (R_a, $mm \cdot yr^{-1}$), annual runoff coefficients (RC_a, %) and SL_a are not yet fully understood quantitatively. Nevertheless, process-based erosion models use runoff in order to estimate SL_a rates (Merritt, 2003) and good knowledge on these relations is an important part of soil loss modelling (e.g. Jetten and Favis-Mortlock, 2006; Kinnell and Risse, 1998). Furthermore, R_a and RC_a also directly relate to on-site problems like agricultural productivity (Rockström et al., 2010) and off-site problems like export of nutrients and pesticides (Rossi Pisa et al., 1999), flash floods (e.g. García-Ruiz et al., 2010; Poesen and Hooke, 1997) and the potential activation of other sediment sources (such as river banks and gullies) further downstream (Vanmaercke et al., 2011a). Hence, it is important to assess both R_a and SL_a rates in conjunction, as well as the effect of key controlling factors on R_a and SL_a rates.

Over the last 60 years, numerous quantitative experimental studies on R_a and SL_a have been conducted throughout Europe and the Mediterranean region, using different experimental methods (e.g. runoff plots, rainfall simulations, rill volume measurements, tracer methods). From these, bounded runoff plot studies under natural rainfall conditions can be considered as the most used and standardised experimental method (e.g. Cammeraat, 1993; Cerdan et al., 2006; Hudson, 1993; Evans, 1995; Boix-Fayos et al., 2006). Studies on runoff and soil loss plots have been extensively used in large-scale coordinated research projects in the United States, leading to the development of the (R)USLE(2) equation (Wischmeier and Smith, 1960, 1978; Renard et al., 1997). However, projects of such an extent have not taken place in Europe where many individual runoff and soil loss plot studies have been reported. As a result, the findings of runoff plot studies in Europe and the Mediterranean region are dispersed over numerous scientific papers, reports and theses. They are mostly designed to analyse effects of erosion controlling factors on R_a and SL_a in a particular area. These individual studies have provided better insights in runoff and soil loss processes at local scales, but the diversity and natural variability of runoff and soil loss plot studies limit the potential to extrapolate these findings to other environmental conditions. Boardman (1998) showed that published soil erosion rates for large areas, based on a small number of observed data should be interpreted with care or may not be relevant at all. Furthermore, an overall assessment of runoff and soil loss rates is also hampered by a high temporal variability (e.g. Bagarello et al., 2011; Martínez-Casasnovas et al., 2002; Ollesch and Vacca, 2002).

From this lack of an overview and difficulties to extrapolate local R_a and SL_a data to larger areas arises the need for a pan-European compilation of all available R_a and SL_a data. As a response to this need, national-scale datasets on soil erosion have recently been assembled for most countries in Europe, although the methodology used and the erosion processes considered differ (Baade and Rekolainen, 2006; Boardman and Poesen, 2006). With respect to R_a and SL_a, several recent studies have compiled field-measured plot runoff and/or soil loss data at the regional, national or sub-continental scale (Boardman and Poesen, 2006, Table 1.1).

From these compilations, a better insight in some key factors determining rates and variability of R_a and SL_a at (sub-)continental and regional scales has been gained. The dominant control of land use type on SL_a was illustrated by Cerdan et al. (2006, 2010), where also soil type, plot length and slope gradient were used to account for further variability. Kosmas et al. (1997) found that the relation between annual precipitation (P_a, $mm \cdot yr^{-1}$) and R_a and SL_a at plot-scale at eight different sites in the Northern Mediterranean is mainly influenced by land use and hence temporal and spatial patterns of vegetation cover (Fig. 1.4). For shrubland, these authors also observed a vegetation feedback mechanism whereby with increasing P_a (up to P_a=200-300 $mm \cdot yr^{-1}$) SL_a first increases and then decreases with increasing P_a. This effect is similar to the one described by Langbein and Schumm (1958), who demonstrated for catchments in the United States that the relation between P_a and catchment sediment yield does not only reflect the increasing erosion potential of higher P_a but also includes feedback effects from a larger vegetation biomass with increasing P_a, effectively reducing sediment yield above an P_a threshold of about 254 to 381 $mm \cdot yr^{-1}$ (10-15 $inch \cdot yr^{-1}$). For the other land uses studied by Kosmas et al. (1997), R_a and SL_a were all positively related to P_a.

However, several key elements to obtain a comprehensive understanding of R_a, RC_a and SL_a rates and controls in Europe and the Mediterranean region are not fully considered in these studies. First and foremost, the existing overviews mainly consider SL_a, while R_a and RC_a are studied to a lesser extent (Table 2.1). While Europe-wide assessments of plot-scale SL_a exist (Cerdan et al., 2010; Vanmaercke et al., 2012a), this is not the case for R_a. In addition, most of these studies assess the effects of one or more controlling factors on R_a and SL_a, but recognise that they lack information on other important controlling factors. As such, Cerdan et al. (2006, 2010) acknowledge the importance of P_a but did not include this in their analysis. Similarly, Kosmas et al. (1997) do not assess the effect of plot length or slope gradient but nevertheless cite its importance. Which controlling factors are included in the analysis, and how they are assessed, may have a significant impact on the discussion of R_a and SL_a rates. For instance, Fleskens and Stroosnijder (2007) argued that average SL_a rates in olive groves are unlikely to exceed 10 $Mg \cdot ha^{-1} \cdot yr^{-1}$, which was contested by Gómez et al. (2008) on the basis that several scale and environmental factors like plot length were not sufficiently taken into account.

Hence, there are still unresolved questions with respect to the relationships between precipitation, runoff (coefficients) and soil loss, and the effect of different land uses on these relations. While several local-scale studies address part of these questions, it is not known whether the findings of these studies also apply to other regions. At a continental-wide scale, there are no studies that comprehensively and quantitatively explore all of these relations. Nevertheless, such analysis is of great use to support model assumptions and contribute to an integrated approach towards soil degradation. Therefore, this study aims i) to provide an overview of both R_a and SL_a rates by interrill and rill erosion, measured at the plot scale for Europe and the Mediterranean region; ii) to assess the variability in both R_a and SL_a rates for different land uses and different climatic regions in the study area and iii) to analyse the relationship between observed $R_a$and SL_a-rates, and their relationship to annual precipitation (P_a, $mm \cdot yr^{-1}$).

3.2 Analysis of the runoff and soil loss plot database

Plot length, slope gradient and soil characteristics need to be taken into account to explain variability in R_a, RC_a and SL_a. Previous studies indicated that the relationship between plot length or slope gradient and R_a or SL_a is non-linear (Cerdan et al., 2010; Nearing, 1997; Poesen and Bryan, 1989; Wischmeier, 1966; Wischmeier and Smith, 1978). The effect of plot length and slope gradient on R_a and SL_a was assessed by calculating the non-parametric Spearman's rank correlation coefficients (r_s). For SL_a, also the correlations with the RUSLE length factor (L_{PL}, Eq. 3.1; Renard et al., 1997; Renard1997), a slope factor (S_{PL}, Eq. 3.3; Nearing1997) and the product of the L_{PL} and S_{PL} (LS-factor, Eq. 3.3) were calculated. For all plots with a land use type for which SL_a was significantly correlated to the LS-factor, the annual unit plot soil loss (SL_u) was calculated using Eq. 3.4; Bagarello et al., 2010b (Bagarello et al., 2010b).

$$L_{PL} = \left(\frac{\lambda}{22.13}\right)^{0.5} \quad \text{(Eq. 3.1; Renard et al., 1997)}$$

$$S_{PL} = -1.5 + \frac{17}{1 + \exp\left(2.3 - 6.1 \times \sin\theta\right)} \quad \text{(Eq. 3.2; Nearing, 1997)}$$

$$LS_{PL} = L_{PL} \times S_{PL} \quad \text{(Eq. 3.3)}$$

$$SL_u = \frac{measured\ plot\ SL_a}{LS_{PL}} \quad \text{(Eq. 3.4; Bagarello et al., 2010b)}$$

Where: L_{PL}= plot length factor, λ= plot length (m), S_{PL} = plot slope gradient factor, θ = plot slope angle (°), LS_{PL}= plot LS-factor, SL_u = annual unit plot soil loss ($Mg \cdot ha^{-1} \cdot yr^{-1}$).

As no standard procedure for the correction of R_a and RC_a values to a unit plot exists, no correction factor could be calculated for R_a and RC_a. Regardless of other factors, it can be expected that the reliability of average R_a and SL_a measurements increases with an increasing number of plot-years, as they capture more of the occurring natural variability. According to the central limit theorem, the standard error of a sample is inversely related to the square root of the number of observations (Tijms, 2004). Therefore, average R_a, RC_a and SL_a values for the different land use types were calculated by weighting the reported mean value of each individual plot by the square root of the number of plot-years for that plot. While this is a very basic approach and does not take into account the complex temporal variation in SL_a and often non-normal distributions of SL_a time series (Maetens et al., 2011), it can be expected that this weighting results in more reliable estimates of average R_a, RC_a and SL_a rates (Cerdan et al., 2010; Vanmaercke et al., 2011b). A k-sample Kolmogorov-Smirnov test (KS-test) was used to test for significant differences in the distribution of R_a, RC_a, SL_a and SL_u data between each combination of two land use types, whereby the significance level of the test was adapted using Bonferroni's Inequality to account for family-wise error in multiple corrections (Brittain, 1987). The same procedure was applied to test for significant differences between the climatic zones for each land use type. To further explore the runoff-soil loss relationship, regression equations of the form:

$$Y = aX^b \quad \text{(Eq. 3.5)}$$

were calculated, by log-transforming both the dependent and independent variable and performing a linear regression. The dependent variable Y was either SL_a or SL_u, the independent variable X was either R_a or RC_a and a and b are the empirical regression constants. Both unweighted and regressions weighted according to the square root of the number of plot-years of each plot were calculated.

Precipitation has been studied intensively as the causal factor of runoff and soil loss. The rainfall erosivity index (EI30) has been proposed as a good measure for relating precipitation to (potential) SL_a (Wischmeier and Smith, 1958). While P_a was reported in most of the studies and is generally widely available with detailed spatial and temporal coverage, EI30 values were not consistently reported for the plots included in the database and are not available from other data sources (Gabriëls, 2006). As an alternative, the Modified Fournier Index (Arnoldus, 1980, Eq. 3.6) has been used as a measure of climatic erosivity (Gabriëls, 2006):

$$MFI = \sum_{1}^{12} \frac{p_m^2}{P_a} \qquad \text{(Eq. 3.6)}$$

Where: MFI= Modified Fournier Index ($mm^2 \cdot mm^{-1}$), p_m= average monthly precipitation ($mm \cdot month^{-1}$), P_a=average annual precipitation ($mm \cdot yr^{-1}$). MFI was calculated for each of the plot measurement sites, using monthly precipitation data obtained from the CRU CL 2.0 dataset (New et al., 2002).

3.3 Results

3.3.1 Effects of plot length and slope gradient on annual runoff, runoff coefficient and soil loss

The range of R_a and SL_a values recorded in the plot database varies over almost 4 orders of magnitude (Fig. 3.1). To allow a comparison of the results obtained on plots with different plot lengths and slope gradients, the importance of these topographic variables was examined using the correlation coefficient between plot length and slope gradient and R_a and RC_a (Table 3.1). R_a was found to be positively correlated with plot length for plots having bare soil, cropland, grassland and tree crops. A significant negative relationship between R_a and plot length was found for plots under forest and plots recently affected by fire. With respect to the relation between slope gradient and R_a only a significant positive correlation was found for post-fire conditions. For grassland and shrubland the relation between R_a and plot slope gradient was found to be significantly negative. With the exception of vineyards, where a significant negative correlation between slope gradient and SL_a was observed, the same trends, albeit with slightly different r_s values were found for SL_a.

For SL_a, a significant positive correlation with the LS-factor (Eq. 3.3) was found for bare plots, cropland, fallow, shrubland and tree crops (Table 3.1). For cropland, the correlation was significant with both the L_{PL} and S_{PL}, while for bare and tree crops only a significant correlation with L_{PL} was found.

Table 3.1: Spearman's rank correlation coefficient (r_s) and p-values for the correlation between annual runoff (R_a), annual runoff coefficients (RC_a) and plot length and slope gradient on the one hand and annual soil loss (SL_a) and slope length factor (L_{PL},, eq. 1), slope gradient factor (S_{PL}, e.q. 2) and LS-factor (LS_{PL}, eq. 3) on the other hand for different land use types in Europe and the Mediterranean. Values in bold indicate significance at α=0.05. NA: not available

	R_a				RC_a				SL_a						
land use type	pl. length		slope grad.		pl. length		slope grad.		Lpl		Spl		LSpl		
	r_s	p	r_s	p	r_s	p	r_s	p	r_s	p	r_s	p	r_s	p	
bare	0.21	**0.02**	0.07	0.45	0.23	**0.03**	0.09	0.41	0.16	**0.04**	0.15	0.06	0.22	**<0.01**	
cropland	0.56	**<0.01**	-0.03	0.65	0.56	**<0.01**	-0.07	0.30	0.31	**<0.01**	0.14	**<0.01**	0.28	**<0.01**	
fallow	0.16	0.29	-0.13	0.40	0.21	0.16	-0.21	0.17	0.21	0.11	0.25	0.06	0.39	**<0.01**	
forest	-0.55	**<0.01**	-0.02	0.90	-0.33	**0.03**	0.08	0.62	-0.13	0.39	-0.08	0.58	-0.04	0.79	
grassland	0.30	**0.02**	-0.56	**<0.01**	0.32	**0.03**	-0.59	**<0.01**	0.05	0.61	-0.04	0.66	0.08	0.44	
post-fire	-0.58	**<0.01**	0.47	**0.001**	-0.71	**<0.01**	0.38	**0.01**	-0.06	0.69	0.11	0.44	0.09	0.52	
rangeland	-0.35	0.24	0.01	0.98	-0.43	0.14	-0.22	0.47	-0.31	0.24	0.10	0.71	0.07	0.79	
shrubland	-0.03	0.79	-0.26	**0.03**	0.07	0.57	-0.26	**0.04**	0.13	0.20	0.30	**<0.01**	0.32	**<0.01**	
tree crops	0.82	**<0.01**	0.15	0.63	0.80	**<0.01**	0.25	0.41	0.71	**<0.01**	0.24	0.29	0.55	**0.01**	
construction sites	NA	NA	0.87	0.67	NA	NA	0.87	0.67	NA	NA	0.87	0.67	0.87	0.67	
vineyard	0.20	0.34	-0.23	0.26	-0.16	0.44	-0.60	**<0.01**	0.61	**<0.01**	-0.24	0.15	0.04	0.82	

Nevertheless, the correlation between SL_a and S_{PL} is also relatively strong. For shrubland, only S_{PL} was significantly correlated with SL_a while for vineyards there was only a significant correlation with L_{PL}. Therefore, unit plot SL_a (SL_u, Eq. 3.4; Bagarello et al., 2010b) was calculated for plots where the land use was bare, cropland, fallow, shrubland or tree crops. Although the correlation between SL_a and LS_{PL} was not significant for construction sites, this is likely due to the limited number of plots for this land use type. Nevertheless, SL_u values were also calculated for plots on construction sites since these plots all have a bare soil surface.

3.3.2 Characteristics of the frequency distributions of annual runoff and soil loss for various land uses

Weighted mean values and box-plots indicating the range of R_a, RC_a, SL_a and SL_u per land use type are shown in Fig. 3.2. Weighted mean values for the different land use types are always higher than the median value for that land use type for R_a, RC_a and SL_a, which indicates that individual plot mean R_a, RC_a and SL_a have a positively skewed distribution, such as the log-normal distribution (e.g. Bagarello et al., 2010a). Construction sites have consistently the highest R_a, RC_a and SL_a. After correction for the plot length and slope gradient, SL_u values for construction sites are smaller, which is attributed to the steep slopes associated with the construction site plots (Fig. 2.6).

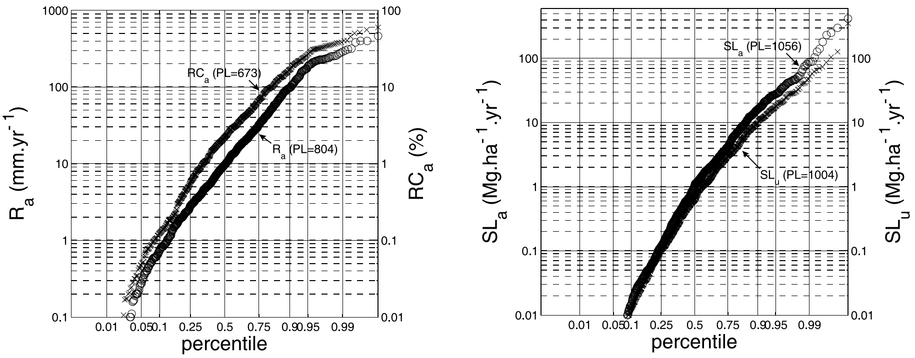


Figure 3.1: Frequency distribution of annual runoff (R_a), annual runoff coefficients (RC_a), annual soil loss (SL_a) and annual unit plot soil loss (SL_u, Eq. 3.4; Bagarello et al., 2010b) rates in the plot database for Europe and the Mediterranean. PL= total number of plots

Nevertheless, SL_u for construction sites remains considerably higher than for other land use types. However, the number of plots on construction sites is low, so the corresponding mean values for R_a, RC_a and SL_a rates should be interpreted with caution, although they do indicate a high vulnerability to erosion of this land use type. This is attributed to a presence of bare, disturbed soil with a low structural stability on relatively steep slopes causing very high soil loss rates (Borselli et al., 2006). The remainder of the land use types can be divided into two groups: a first group consisting of bare plots or plots with some type of crop cultivation (i.e. cropland, fallow, tree crops and vineyards) show weighted mean R_a rates between 30 and 60 mm·yr^{-1}, SL_a rates between 5 and 15 % and SL_a rates between 1 and 20 Mg·ha^{-1}·yr^{-1}. The second group of land use types consists of plots with a (semi-)natural vegetation cover (i.e. forest, post-fire, shrubland, rangeland and grassland), with R_a, RC_a and SL_a rates less than 30 mm·yr^{-1}, 5% and 1 Mg·ha^{-1}·yr^{-1}, respectively (Table 3.2). While bare plots and plots with crop cultivation have consistently higher mean R_a, RC_a and SL_a rates than plots with (semi-)natural vegetation, the ranking of the weighted mean values for the different land use types within these two groups varies for R_a, RC_a and SL_a (Fig. 3.2). The difference between these two groups was confirmed by the application of the Kolmogorov-Smirnov test, whereby significant differences between land use types were mostly observed between land use types where crop cultivation was applied and land use types with (semi-)natural vegetation. Furthermore, for 25 out of 55 pairwise combinations of all the 11 land use types, the distribution of SL_a data was found to be significantly different, while for R_a and RC_a only 12 and 8 combinations were found to be significantly different, respectively.

Table 3.2: Weighted mean, median and coefficient of variation (CV) for annual runoff (R_a), annual runoff coefficient (RC_a), annual soil loss (SL_a) and annual unit plot soil loss (SL_u) per land use type for all data and grouped per climatic zone in Europe and the Mediterranean. Weighting is according to the square root of the number of plot-years. For number of plots and plot-years corresponding to each land use type, see Table 5. NC: not calculated because the relation between LS-factor and annual soil loss for this land use type was not significant (see Table 3.1). NA: no data available.

land use class	climatic zone	R_a mean (mm)	R_a median (mm)	R_a CV (-)	RC_a mean (%)	RC_a median (%)	RC_a CV (-)	SL_a mean ($Mg \cdot ha^{-1} \cdot yr^{-1}$)	SL_a median ($Mg \cdot ha^{-1} \cdot yr^{-1}$)	SL_a CV (-)	SL_u mean ($Mg \cdot ha^{-1} \cdot yr^{-1}$)	SL_u median ($Mg \cdot ha^{-1} \cdot yr^{-1}$)	SL_u CV (-)
bare	all data	35.0	15.8	1.8	5.0	2.5	1.5	12.8	5.7	1.9	9.2	3.9	1.7
	pan-Mediterranean	36.0	20.5	1.9	4.9	3.8	1.4	9.1	3.6	1.5	7.4	3.1	1.5
	temperate	33.1	13.5	1.7	5.2	2.4	1.6	18.9	8.7	1.8	12.4	5.6	1.7
	cold	44.5	45.5	0.3	3.0	3.1	0.3	14.8	15.5	0.6	4.7	5.0	0.7
cropland	all data	50.3	11.9	1.7	8.0	3.2	1.4	6.5	1.3	2.8	3.3	0.9	2.2
	pan-Mediterranean	51.0	17.6	1.7	8.6	4.2	1.2	2.9	1.2	1.8	2.3	0.5	2.2
	temperate	19.3	5.8	1.8	3.3	1.3	1.5	9.6	0.9	2.7	4.1	1.1	2.1
	cold	148.9	199.7	0.6	19.5	28.9	0.6	9.0	3.0	1.3	4.1	1.7	1.3
fallow	all data	50.4	8.3	1.6	7.3	1.9	1.5	5.8	1.7	2.1	2.7	1.1	1.7
	pan-Mediterranean	51.1	25.8	1.1	7.7	3.9	1.0	1.5	0.7	1.5	0.7	0.4	1.4
	temperate	17.2	4.5	2.1	3.6	0.8	2.5	7.1	2.9	1.5	4.1	2.1	1.2
	cold	213.3	221.5	0.2	31.3	31.8	0.2	20.0	15.1	1.3	7.4	3.9	1.4
forest	all data	13.9	4.3	2.5	2.9	1.6	1.5	0.7	0.1	5.2	NC	NC	NC
	pan-Mediterranean	9.6	4.6	1.9	2.8	1.6	1.1	0.4	0.1	2.2	NC	NC	NC
	temperate	28.7	2.0	1.9	3.3	0.9	1.7	1.3	0.0	3.6	NC	NC	NC
	cold	3.9	3.9	NA	NA	NA	NA	0.0	0.0	NA	NC	NC	NC
grassland	all data	15.4	3.5	2.3	4.0	0.6	2.1	0.7	0.1	2.0	NC	NC	NC
	pan-Mediterranean	11.7	3.8	1.3	6.2	1.8	1.9	0.6	0.2	1.6	NC	NC	NC
	temperate	7.7	2.2	1.9	1.1	0.3	2.0	1.0	0.1	2.0	NC	NC	NC
	cold	133.9	131.0	0.6	27.5	26.0	0.4	0.6	0.1	2.1	NC	NC	NC
post-fire	all data	28.3	4.6	1.6	3.5	0.9	1.6	1.5	0.2	2.1	NC	NC	NC
	pan-Mediterranean	26.3	4.0	1.7	3.2	0.7	1.7	1.6	0.2	2.1	NC	NC	NC
	temperate	53.0	41.5	0.8	8.0	8.6	0.3	1.1	1.2	0.9	NC	NC	NC
	cold	41.7	41.7	0.1	NA	NA	NA	0.8	0.8	0.3	NC	NC	NC
rangeland	all data	18.8	6.0	1.8	2.7	1.7	0.9	0.7	0.2	2.2	NC	NC	NC
	pan-Mediterranean	19.6	5.5	1.8	2.8	1.8	0.9	0.8	0.2	2.2	NC	NC	NC
	temperate	6.6	6.6	NA	0.8	0.8	NA	0.3	0.3	1.4	NC	NC	NC
shrubland	all data	18.3	5.1	1.6	4.4	1.1	1.8	0.6	0.1	2.0	0.3	0.1	1.6
	pan-Mediterranean	18.5	5.1	1.6	4.4	1.1	1.8	0.6	0.2	1.9	0.3	0.1	1.6
	temperate	14.3	0.0	1.4	3.4	0.0	1.6	0.5	0.0	2.6	0.2	0.0	2.0
tree crops	all data	46.2	36.9	1.2	8.2	5.0	1.1	13.4	9.0	1.4	3.3	2.0	1.7
	pan-Mediterranean	52.7	34.2	1.2	10.2	5.5	1.0	11.6	7.4	1.5	3.0	1.8	1.8
	temperate	34.7	36.9	0.7	4.7	5.0	0.7	18.4	13.8	1.0	4.2	3.9	0.8
construction sites	all data	227.7	230.0	0.2	57.0	59.7	0.3	325.1	299.0	0.4	115.8	73.4	1.0
	pan-Mediterranean	241.0	241.0	0.1	61.9	61.9	0.1	357.5	357.5	0.2	63.1	63.1	0.2
	temperate	168.0	168.0	NA	34.8	34.8	NA	180.0	180.0	NA	351.3	351.3	NA
vineyard	all data	38.8	21.0	1.5	7.4	4.4	1.2	17.9	0.9	4.1	NC	NC	NC
	pan-Mediterranean	47.6	21.0	1.5	9.8	6.9	1.0	1.8	0.3	1.8	NC	NC	NC
	temperate	25.2	23.1	1.0	3.7	4.3	0.8	32.1	12.4	3.1	NC	NC	NC

3.3.3 Relationships between annual runoff coefficients and soil loss for various land uses

The median and distributions of all R_a and SL_a data for different land use types are shown in Fig. 3.2. This figure indicates that there is a general trend towards higher median RC_a rates with higher median SL_a. For land uses under crop cultivation, tree crops have the highest median RC_a and SL_a. Vineyards and tree crops have comparable median RC_a and 75th percentile SL_a values. However, median SL_a is markedly lower for vineyards, indicating that high SL_a rates in the database are more rare for vineyards than for tree crops. The interquartile ranges for cropland and fallow plots are similar for RC_a and SL_a. Median RC_a is lower for fallow plots however, indicating that low RC_a are more prevalent on fallow plots. Although RC_a rates on bare and cropland plots are similar, bare plots show higher SL_a rates than cropland plots. Plots with (semi-)natural vegetation are characterised by consistently low median SL_a (0.08 - 0.3 $Mg \cdot ha^{-1} \cdot yr^{-1}$), but show somewhat more variation in median RC_a rates (0.5 - 1.1%).

Using all plot data for which respectively pairs of R_a-SL_a, RC_a-SL_a, R-SL_u and RC_a-SL_u were available, the best regression correlations (Eq. 3.5) are generally observed between R_a and SL_a (Table 3.3). Only for plots under grassland and post-fire better correlations between RC_a and SL_a were found. As a lack of data on P_a, plot length or plot slope gradient data does not allow the calculation of SL_a or SL_u, respectively, a smaller number of plots was available to calculate SL_a-SL_a, R_a-SL_u and RC_a-SL_u regressions. Nevertheless, the results of the regression analyses did not change using a subdataset of 611 plots where all four variables (R_a, RC_a, SL_a and SL_u) are available. In general, regression with unweighted R_a, RC_a, SL_a and SL_u data yields slightly better correlation than weighted regression, but weighted regressions were considered to be more appropriate as the number of plot-years is taken into account (cfr. section 3.2).

Regressions between R_a and SL_a were found not to be significant for fallow, vineyards and construction sites and hence these land use types were not included in Fig. 3.4a. The large exponents for tree crops, forests and post-fire plots may be attributed to the increase in kinetic energy of falling water drops caused by canopy dripping, hence causing more soil detachment for each mm of runoff. While no specific research on this process was found for Europe and the Mediterranean, Brandt (1988) found that the kinetic energy of rainfall in single-canopy rainforests was increased 1.86 times by canopy dripping. Fig. 3.4b gives a comparison of the regressions for the different land use types in Fig. 3.4a. The difference in regression slopes (i.e. the exponents in the power laws) between the different land use types was not always significant however (Table 3.4). As a large number of plots is available for cropland and crop type is expected to

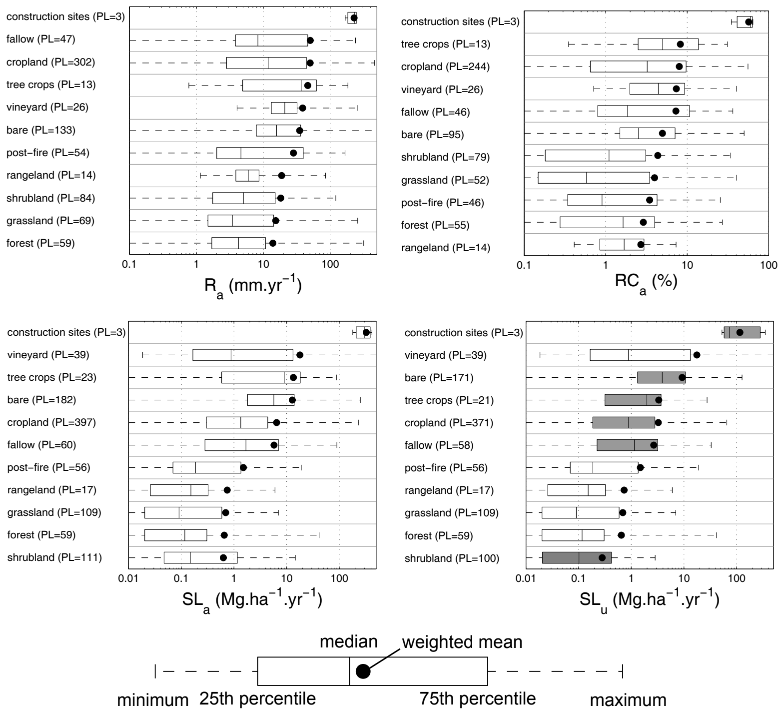


Figure 3.2: Frequency distribution of mean annual runoff (R_a), mean annual runoff coefficient (RC_a), mean annual soil loss (SL_a), mean annual unit plot soil loss (SL_u, eq. 4) of all plots in Europe and the Mediterranean, grouped per land use type. Box-plots in grey indicate mean annual soil losses scaled to a unit plot (SL_u= plot length= 22.1m, plot slope gradient= 9%). For the other land use types, the original, measured SL_a was used as no significant relation between SL_a and the LS-factor was found for these land use types (α=0.05). Black dots indicate the mean R, RC_a, SL_a and SL_u, weighted by number of plot years (PY). Box plots are ordered according to descending weighted mean.

have an important effect on SL_a rates in this land use type (e.g. Auerswald 2009; Gabriëls et al. 2003), a further subdivision according to crop type was made (Table 2.2). The regression between R_a and SL_a was only significant for cereals (Fig. 3.5), most likely due to a lack of sufficient data for the other crop types. Nevertheless, for maize, sugar beet and potatoes the available data show high R_a and SL_a rates on nearly all of the plots where these crops were planted (Fig. 3.5).

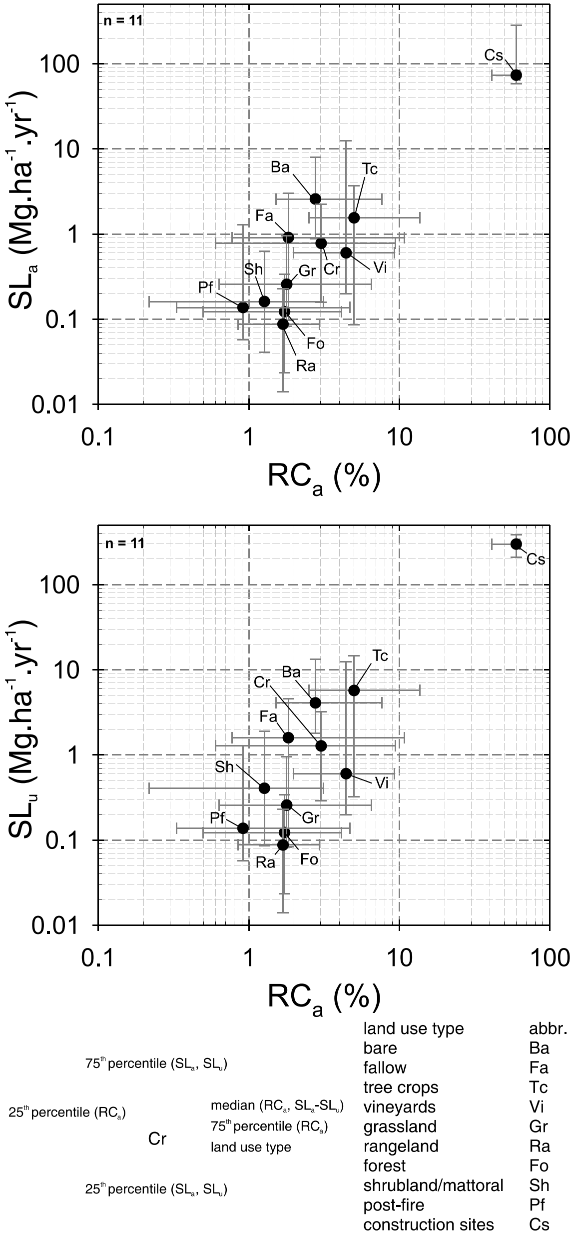


Figure 3.3: Relation between median annual runoff coefficient (RC_a), median annual soil loss (SL_a) and median annual unit plot soil loss (SL_u) of all plots in Europe and the Mediterranean per land use with indication of the 25th and 75th percentile for RC_a and SL_a, based on the plot database (for details see Table 2.5 and Table 3.2). n= number of land use types

Table 3.3: Coefficients of determination (r^2) and number of plots (PL) for the unweighted (unwg.) and weighted (wg.) annual runoff - annual soil loss (R_a vs. SL_a), annual runoff coefficient - annual soil loss (RC_a vs. SL_a), annual runoff – annual unit plot soil loss (R_a vs. SL_u) and annual runoff coefficient – annual unit plot soil loss (RC_a vs. SL_u) regressions. Weighting was done using the square root of the number of plot years. For each land use, the regression with the highest r^2 value is indicated in bold. n.s.: regression is not significant at α=0.05, NA: no regression was calculated as no SL_u values were calculated for these land use types.

land use type	crop	R_a vs. SL_a			RC_a vs. SL_a			R_a vs. SL_u			RC_a vs. SL_u		
		PL	r^2 unwg.	r^2 wg.	PL	r^2 unwg.	r^2 wg.	PL	r^2 unwg.	r^2 wg.	PL	r^2 unwg.	r^2 wg.
bare		126	**0.47**	0.45	89	0.37	0.35	118	0.40	0.37	84	0.34	0.30
cropland	all data	281	**0.19**	0.19	228	0.13	0.12	259	0.16	0.15	217	0.10	0.09
	cereals	132	0.29	0.28	105	**0.29**	0.26	116	0.23	0.22	99	0.22	0.18
	maize	15	n.s.	n.s.	15	n.s.	n.s.	15	n.s.	n.s.	15	**0.27**	n.s.
	sunflower	8	0.54	n.s.	8	0.64	0.56	8	0.54	n.s.	8	**0.64**	0.59
	leguminous	23	n.s.	n.s.	21	n.s.	n.s.	23	n.s.	n.s.	21	n.s.	n.s.
	potatoes + sugar beet	6	n.s.	n.s.	4	n.s.	n.s.	5	n.s.	n.s.	4	n.s.	n.s.
	rotation	77	**0.05**	n.s.	55	n.s.	n.s.	72	n.s.	n.s.	50	n.s.	n.s.
	other	20	**0.38**	0.36	20	0.27	0.24	20	0.25	0.21	20	n.s.	n.s.
fallow		40	n.s.	n.s.	39	n.s.	n.s.	38	n.s.	n.s.	38	n.s.	n.s.
forest		50	0.48	0.47	48	**0.58**	0.58	NA	NA	NA	NA	NA	NA
grassland		47	0.26	0.25	30	**0.28**	0.27	NA	NA	NA	NA	NA	NA
post-fire		53	**0.44**	0.44	45	0.40	0.39	NA	NA	NA	NA	NA	NA
rangeland		14	**0.31**	n.s.	14	0.30	n.s.	NA	NA	NA	NA	NA	NA
shrubland		81	**0.14**	0.08	76	0.12	0.11	71	0.08	n.s.	66	0.08	n.s.
tree crops		13	**0.83**	0.83	13	0.66	0.65	13	0.81	0.80	13	0.62	0.61
vineyard		26	n.s.	n.s.	26	n.s.	n.s.	NA	NA	NA	NA	NA	NA
construction sites		3	n.s.	n.s.	3	n.s.	n.s.	3	n.s.	n.s.	3	n.s.	n.s.

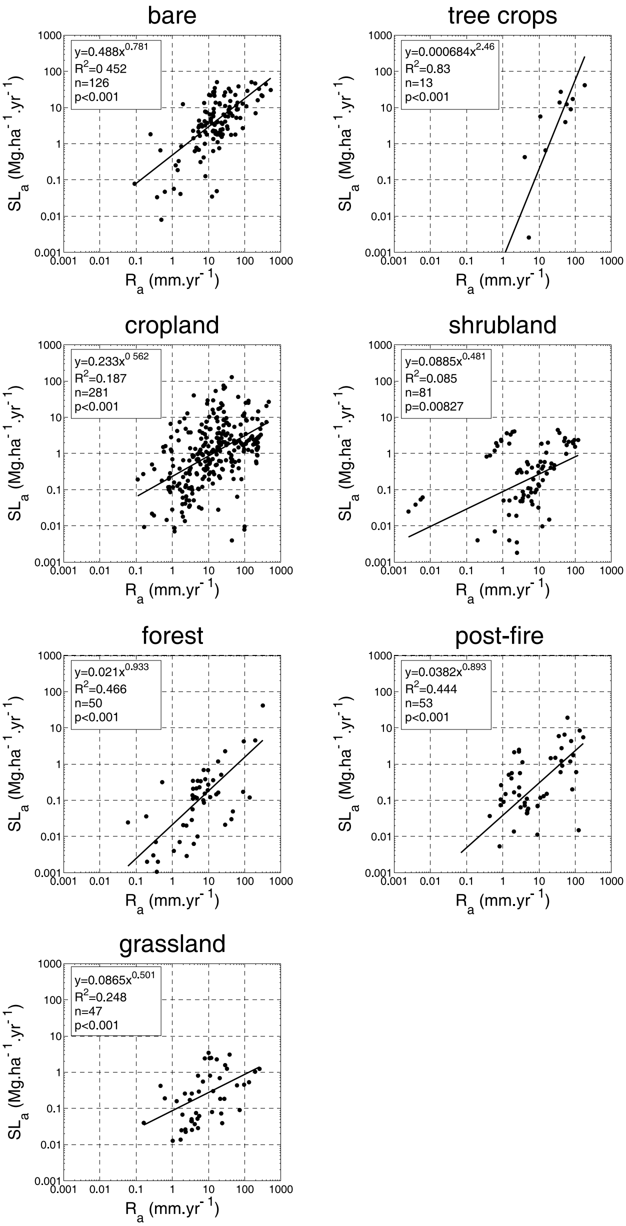


Figure 3.4a: Significant (α=0.05) relationships between annual runoff (R_a) and annual soil loss (SL_a) for different land use types in Europe and the Mediterranean. Each point corresponds to the measuring period mean R_a and SL_a for a single plot or replicate plots for which the joint mean R_a or SL_a was reported. Regressions are weighted according to the square root of number of plot years for each plot. n= number of plots.

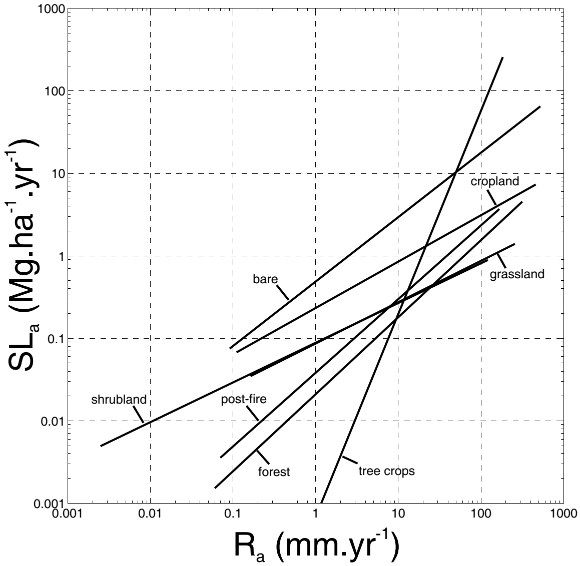


Figure 3.4b: Comparison of the significant (α=0.05) weighted regressions (SL_a=a$R_a{}^b$) between annual runoff (R_a) and annual soil loss (SL_a) for different land use types in Europe and the Mediterranean. Regressions are weighted according to the square root of the number of plot years for each plot. For regression parameters, see Fig. 3.1

Table 3.4: P-values for the differences between the slopes of the regressions for the different land use types in Fig. 3.4b. Land use types for which the regression slopes are significantly different at α=0.05 are indicated with a p-value in bold font.

land use type	bare	cropland	forest	grassland	post-fire	shrubland
cropland	**0.04**					
forest	0.42	**0.04**				
grassland	0.13	0.71	0.06			
post-fire	0.48	**0.02**	0.85	0.06		
shrubland	**0.04**	0.50	**0.02**	0.92	**0.01**	
tree crops	**<0.01**	**<0.01**	**<0.01**	**<0.01**	**<0.01**	**<0.01**

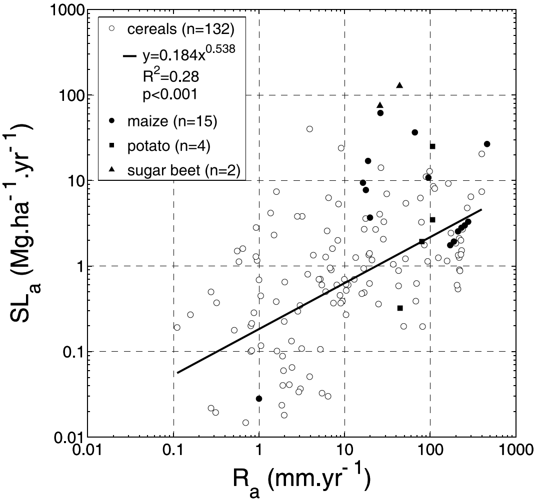


Figure 3.5: Relationship between annual runoff (R_a) and annual soil loss (SL_a) for cereal, maize, potato and sugar beet in Europe and the Mediterranean. Only the regression for plots with cereals was found to be significant (α=0.05) and is given here. n= number of plots, p= p-value.

3.3.4 Effects of land use on runoff coefficient and soil loss for different climatic zones

In order to investigate possible impacts of the climatic zones on the relation between RC_a and SL_a for different land use types, plot data were grouped according to climatic zone. After initial data analysis, the division of all plot data into 7 different climatic zones, some of which contain only a few plots, was found to be too detailed. Therefore, the Mediterranean and Anatolian climatic zone were grouped in a new climatic zone hereafter referred to as "pan-Mediterranean" (M). Likewise, the Atlantic, Continental and Steppic climatic zones were grouped in a "temperate" climatic zone (T) and the Boreal and Alpine climatic zones in a "cold" climatic zone (C). Weigthed mean values, median values and coefficients of variation for R_a, RC_a, SL_a and SL_u for each land use and climatic zone are given in Table 3.2. Shrubland, rangeland and post-fire occur mainly in the pan-Mediterranean zone and the few plots in the temperate zone are located relatively close to the pan-Mediterranean zone. Therefore, these land use types were not further considered in this analysis. Also construction sites were disregarded due to insufficient plot data. For the other land use types the differences in median value and the distribution of R_a and SL_a data per climatic zone are indicated in Fig. 3.6.

Mean SL_a values are smaller in the pan-Mediterranean than in the other two climatic zones for all land use types (Table 3.2). With respect to the median, median SL_a rates for bare, fallow, tree crops and vineyards are significantly lower in the pan-Mediterranean zone than in the other climatic zones, except the differences between the pan-Mediterranean and cold climatic zone for bare plots, and the difference between pan-Mediterranean and temperate climatic zone for tree crops. These non-significant differences in SL_a are likely due to insufficient data for bare plots in the cold climatic zone and tree crops in the temperate climatic zones. Median SL_a in the pan-Mediterranean zone is not smaller for cropland, grassland and forests. However, mean SL_a for all these land use types is smaller in the pan-Mediterranean zone compared to the temperate zone, indicating that the highest SL_a on cropland occur more frequently in the temperate zone than in the pan-Mediterranean (Fig. 3.6).

Only differences between the pan-Mediterranean and temperate climatic zones for cropland and between pan-Mediterranean and cold climatic zones for grassland were found to be significant. With respect to RC_a, median RC_a is significantly higher in the cold climatic zones than in the other climatic zones for cropland, fallow and grassland. Also, mean RC_a is the highest in the cold climatic zone for all land use types. For grassland, there is no significant difference in median SL_a between the different climatic zones, but differences in median RC_a between different climatic zones are significantly different, indicating while grassland has consistently low SL_a rates throughout the study area, RC_a rates differ depending on climatic zone.

3.3.5 Effects of annual precipitation on annual runoff and soil loss for different land uses

Figure 3.7 displays the relation between P_a and the weighted mean R_a, RC_a, SL_a and SL_u for the different land use types. As all plots on construction sites fall within the 250-500 mm·yr^{-1} P_a-class, they were not included in this figure. Separate graphs displaying the R_a, RC_a and SL_a per precipitation class for the different land use types are presented in Fig. 3.8a, Fig. 3.8b and Fig. 3.8c. It should be noted that for some land use types, only a few plots are available and hence some of the observed trends are uncertain. Nevertheless, clear trends can be observed. For all land use types, R_a generally increases with increasing P_a (Fig. 3.8a). For plots under cropland, there is already a substantial increase in R_a for 500-750 mm·yr^{-1} P_a, while for plots with a (semi-)natural vegetation cover, the most substantial increases in R_a occur in the 750-1000 mm·yr^{-1} and >1000 mm·yr^{-1} P_a classes. Application of the Kolmogorov-Smirnov test between subsequent precipitation classes indicated that the distribution of R_a data was only significantly different between the 250-

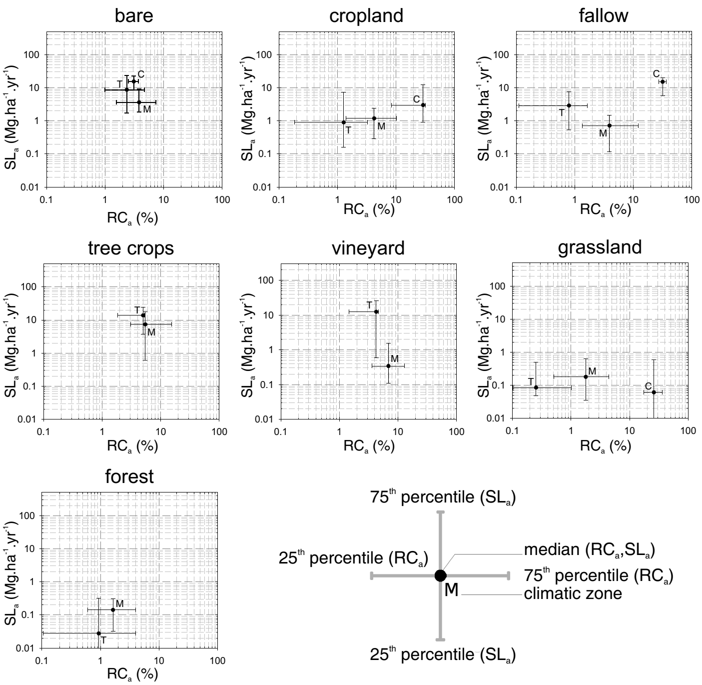


Figure 3.6: Relation between median annual runoff coefficient (RC_a) and median annual soil loss (SL_a) for all plots in Europe and the Mediterranean per land use type and climatic zone, with indication of the 25th and 75th percentile for R_a and SL_a. M= pan-Mediterranean, T= temperate, C= cold. For the number of plots and number of plot years, refer to Table 2.5, for the division in climatic zones, see Fig. 2.1.

500 mm·yr^{-1} P_a classes for cropland, fallow, post-fire forest and shrubland plots. For forest, also the difference between the 0-250 and 250-500 mm·yr^{-1} classes was significant. For SL_a, however, no clear general trend with P_a over all land use types can be noted (Fig. 3.8b).

With respect to SL_a (Fig. 3.8c), there is a general trend towards higher SL_a with higher P_a in plots with crop cultivation (vineyard, tree crops, bare, cropland and fallow). Mean SL_a under bare conditions increases between 250-500 mm·yr^{-1} and 500-750mm, but levels afterwards and even decreases between the 750-1000 mm·yr^{-1} and >1000 mm·yr^{-1} P_a classes. Similarly, SL_a in cropland generally increases with increasing P_a, but mean SL_a for the >1000 mm·yr^{-1} P_a class is higher than for the 750m-1000 mm·yr^{-1} class. Annual SL_a increases gradually with increasing P_a for post-fire, rangeland and forest plots. For grassland, the highest SL_a are observed in the 500-750 mm·yr^{-1} class.

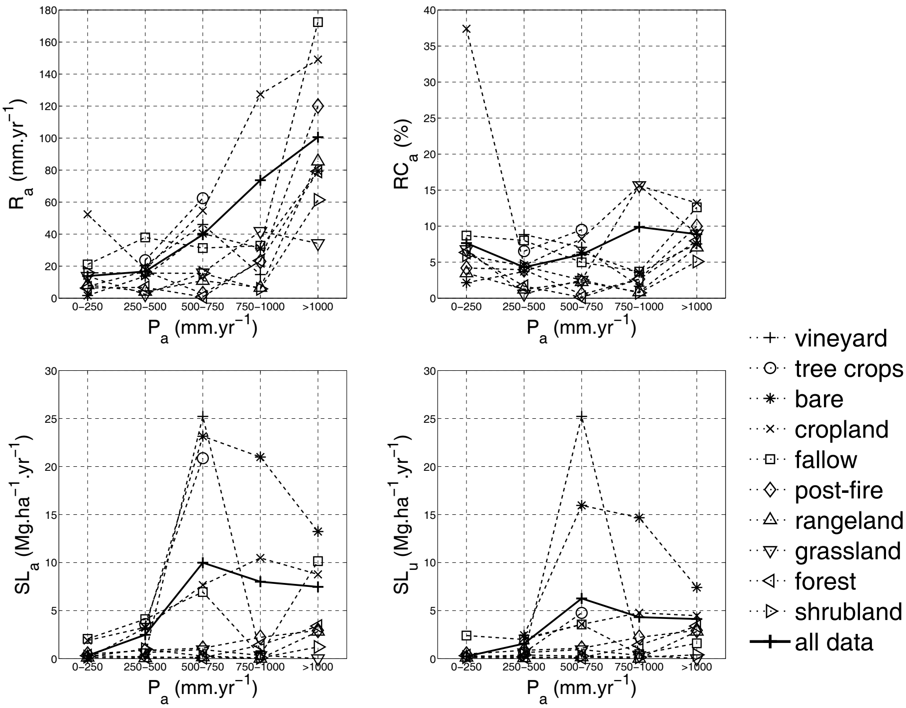


Figure 3.7: Weighted mean annual runoff (R_a), annual runoff coefficient (RC_a), annual soil loss (SL_a) and annual unit plot soil loss (SL_u) for each land use type in Europe and the Mediterranean as a function of the annual precipitation (P_a) interval.

For shrubland, SL_a first increases to a maximum in the 250-500 mm·yr^{-1} class, and then declines over the 500-750 mm·yr^{-1} and 750-1000 mm·yr^{-1} classes. Mean SL_a increases again between the 750-1000 and >1000 mm·yr^{-1} classes, but the latter class corresponds to only 3 plots so results are not conclusive. Significant differences in SL_a between subsequent P_a classes were found between the 0-250 mm·yr^{-1} and 250-500 mm·yr^{-1} classes for shrubland and between the 250-500 mm·yr^{-1} and 500-750 mm·yr^{-1} classes for bare, cropland and shrubland plots. For cropland, also the 500-750 mm·yr^{-1} and 750-1000 mm·yr^{-1} classes were found to have significantly different distributions. Corrections for plot length and slope gradient resulted generally in SL_u rates that are lower than the original SL_a rates. Nevertheless, the trends observed in the graph depicting the relation between P_a and SL_a (Fig. 3.8c) were found to persist. Correlating the Modified Fournier Index with R_a, RC_a and SL_a did not yield clearer trends than using P_a measured on the plots (Fig. 3.9).

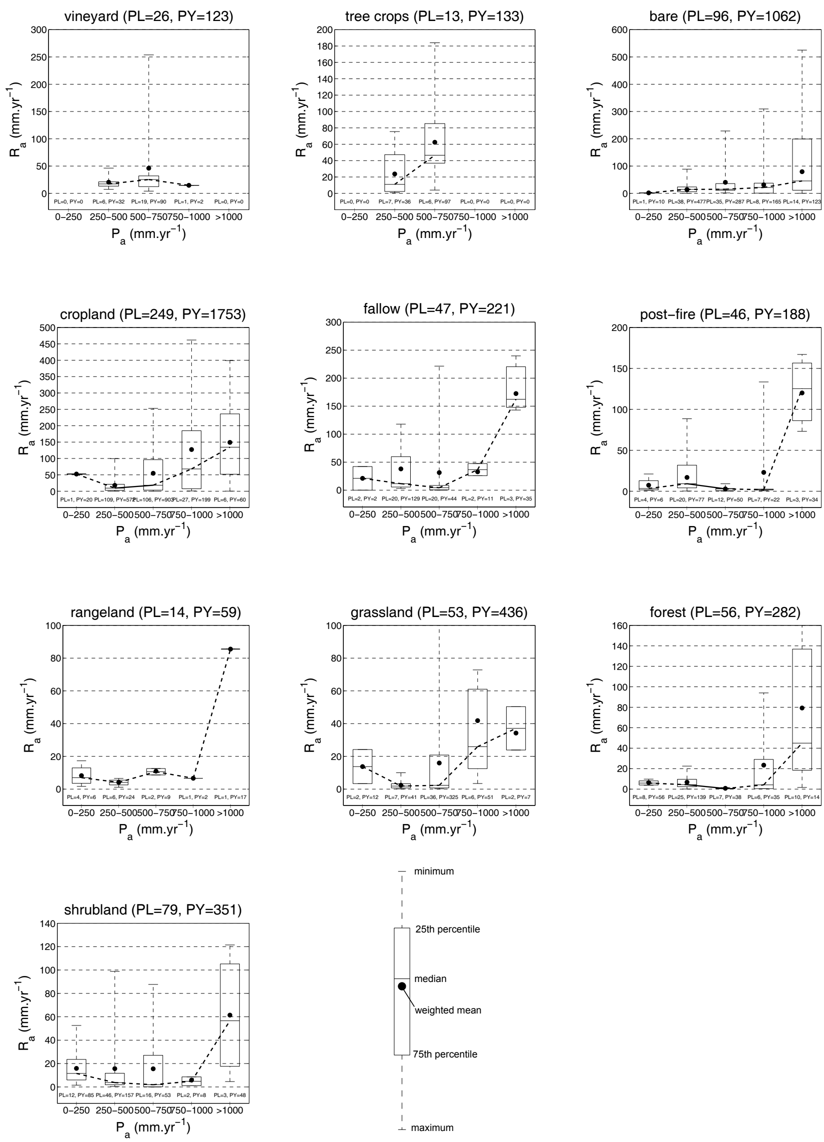


Figure 3.8a: Median trend, distribution and weighted mean of annual runoff (R) per annual precipitation (P_a) interval and land use type in Europe and the Mediterranean. Significant differences (α=0.05) in R_a between subsequent P_a classes are indicated by full lines, while insignificant differences are indicated by dashed lines. PL= number of plots, PY= number of plot-years (next to land use type: total for that land use type, under boxplots: total for that specific boxplot).

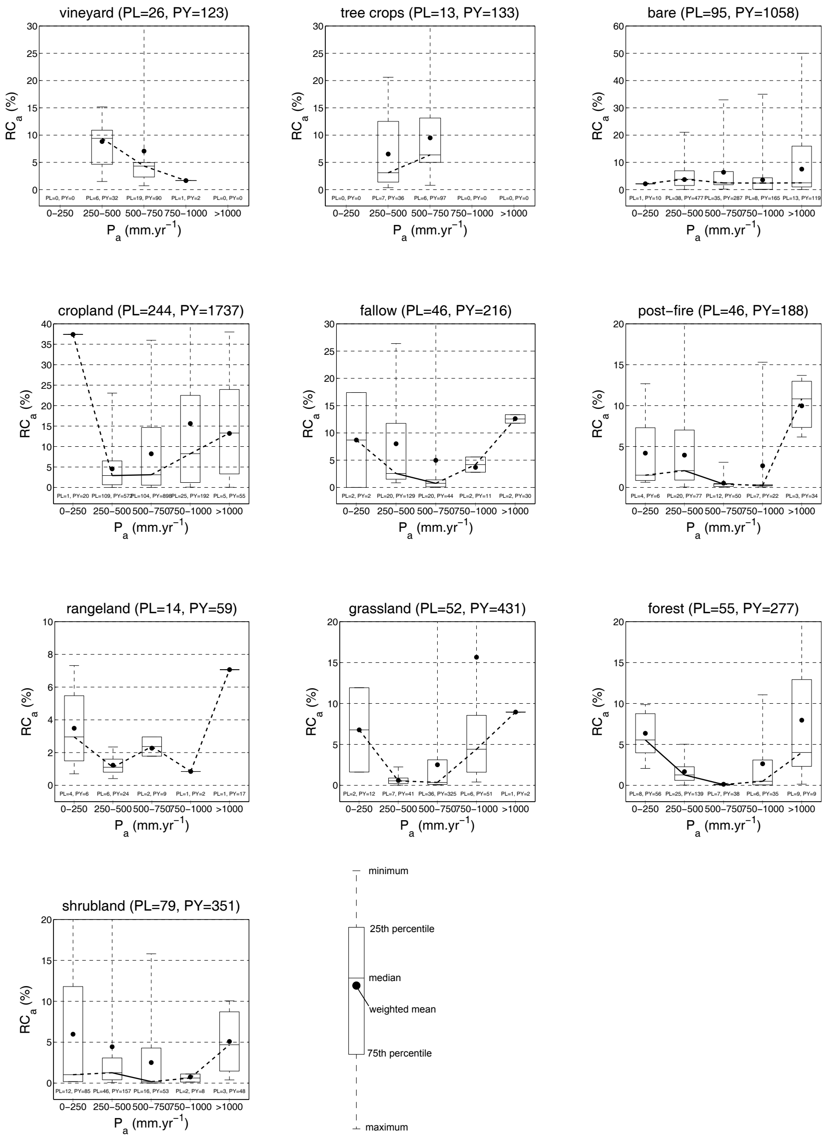


Figure 3.8b: Median trend, distribution and weighted mean of annual runoff coefficient (RC$_a$) per annual precipitation (P$_a$) interval and land use type in Europe and the Mediterranean. Significant differences (α=0.05) in RC$_a$ between subsequent P$_a$ classes are indicated by full lines, while insignificant differences are indicated by dashed lines. PL= number of plots, PY= number of plot-years (next to land use type: total for that land use type, under boxplots: total for that specific boxplot).

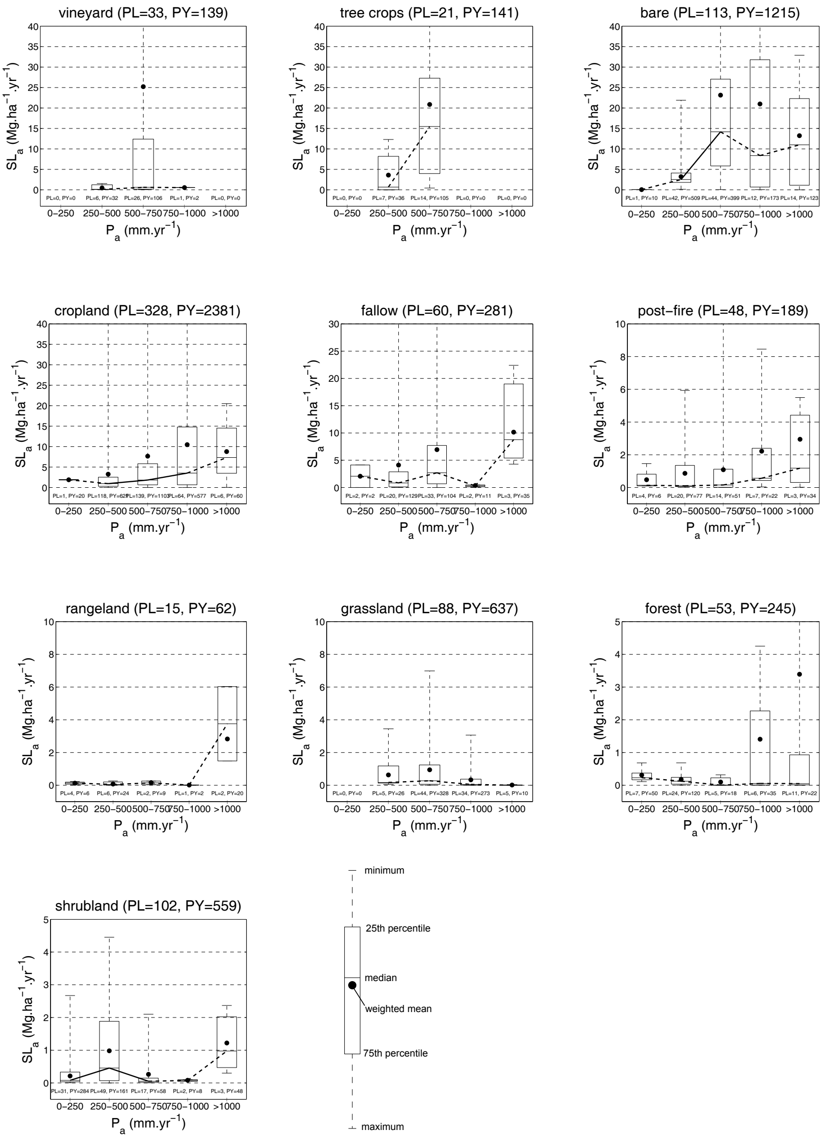


Figure 3.8c: Median trend, distribution and weighted mean of annual soil loss (SL_a) per annual precipitation (P_a) interval and land use type in Europe and the Mediterranean. Significant differences (α=0.05) in R_a between subsequent P_a classes are indicated by full lines, while insignificant differences are indicated by dashed lines. PL= number of plots, PY= number of plot-years (next to land use type: total for that land use type, under boxplots: total for that specific boxplot).

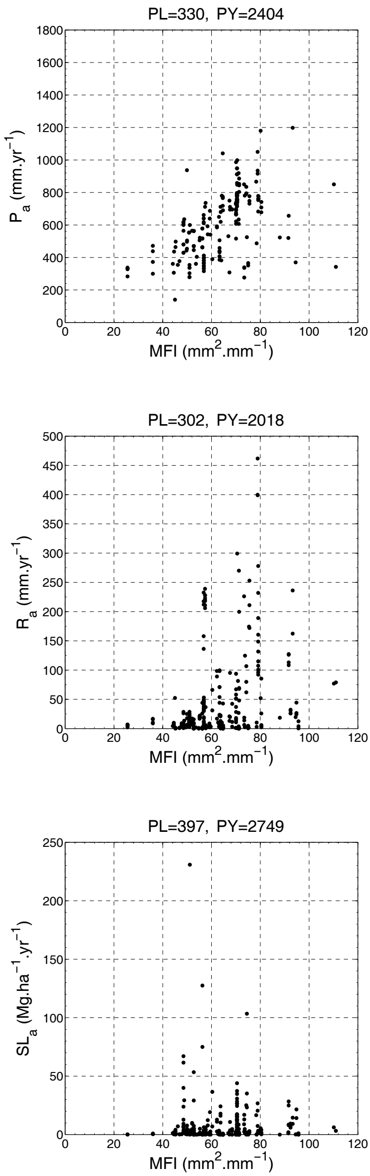


Figure 3.9: Relations between Modified Fournier Index (MFI) and annual precipitation (P_a), annual runoff (R_a) and annual soil loss (SL_a) for cropland plots in Europe and the Mediterranean. PL= number of plots, PY= number of plot-years.

3.4 Discussion

3.4.1 Effects of plot length and slope gradient on annual runoff, runoff coefficient and soil loss

Annual SL_a on bare, cropland and tree crop plots are positively correlated to plot L-factor (Table 3.1). This finding concurs with results found by Cerdan et al. (2010). For these land use types, also R_a and RC_a are positively correlated with plot length (Table 3.1). This can be attributed to the high connectivity along flow paths under these land use types, allowing runoff to accumulate along the flow path over longer plot lengths and resulting in higher SL_a due to the increased detachment and transport capacity of the overland flow (Govers and Poesen, 1988; Prosser and Rustomji, 2000). A significant effect of plot S-factor on SL_a was only found for cropland and shrubland, although the correlations between the plot S-factor and SL_a are also relatively strong for bare plots and fallow plots (Table 3.1, p=0.06).

With respect to climatic zone, Cerdan et al. (2010) found that for arable and bare plots, the relation between LS-factor (Eq. 3.4; Bagarello et al., 2010b) and SL_a was only significant for plot data collected outside the Mediterranean climatic zone and not for plots in the Mediterranean climatic zone, which was attributed to the higher surface rock fragment cover in the Mediterranean, especially on steeper slopes (Poesen et al., 1998). In this study, similar results are found for bare plots in the temperate climatic zone where the relation between $SL_a$and the L-factor (r_s=0.62, p<0.01), S-factor (r_s=0.25, p=0.03) and LS-factor (r_s=0.44, p<0.01) was significant, while for plots in the pan-Mediterranean climatic zone none of these relations were significant. However, for cropland in both the temperate and pan-Mediterranean climatic zones significant relations between L-factor and SL_a (r_s=0.36, p<0.01 and r_s=0.20, p=0.01 respectively) and between LS-factor and SL_a (r_s=0.33, p<0.01 and r_s=0.17, p=0.03 respectively) were found. This could be due to the inclusion of more pan-Mediterranean cropland plots in the database as compared to the database used by (Cerdan et al., 2010). By including more plots, a better overall representation of environmental conditions in the Mediterranean is obtained which contributes to a better assessment of topographic effects on SL_ain this climatic zone. Nevertheless the correlation coefficient between LS-factor and SL_a remains smaller for cropland plots in the pan-Mediterranean (r_s=0.17, p=0.03) than for cropland plots in the temperate zone (r_s=0.33, p<0.01).

Similar to results obtained by Cerdan et al. (2010), no significant correlations were found between the L-, S- or LS-factor and $SL_a$for grassland. This can be explained by the high root density of grasses, which reduce soil erodibility (De Baets et al., 2006). The correlation between plot length and R_a was significantly positive however (Table 3.1), which may be explained by the fact that runoff is more likely to converge on longer slopes, and the effect of grass cover on R_a retention is often less pronounced and more variable than the effect on SL_a, as is shown by studies on the effectiveness of grass buffer strips (e.g. Blanco-Canqui et al., 2004a) and grassed waterways (e.g. Fiener and Auerswald, 2003). Contra-intuitively, a significant negative relation was noted between slope gradient and both R_a and RC_a for grassland. This is probably due to the fact that grasslands in the Northern regions often lie on gentle slopes but are very wet due to a clayey subsoil. These soils often need drainage (e.g. Øygarden, 1996; Øygarden et al., 1997; Turtola and Paajanen, 1995; Turtola et al., 2007; Warsta et al., 2009). For plots under forest and post-fire a significant negative correlation between plot length and R_a and SL_a was found (Table 3.1). This is probably related to the heterogeneity of soil cover and macropore distribution in these land use types. As plot length increases, runoff is more likely to flow through patches with increased roughness or infiltration capacity. For shrubland, a significant negative correlation between slope gradient and R_a and RC_a is observed, while the correlation between the S-factor and SL_a is positive (Table 3.1). This can be explained by the reduced tendency in surface sealing of bare patches with steeper slopes (Poesen, 1984, 1986b), reducing runoff but increasing splash erosion (Bradford et al., 1987). Nevertheless, plots under shrubland are characterised by a high spatial variability at plot scale and hence the establishment of relations between plot length and slope gradient and R_a, RC_a and SL_a is difficult (Cammeraat, 2002).

No significant effect of plot length on R_a or RC_a was found for vineyards, but the L-factor was significantly correlated with SL_a (Table 3.1). This is mainly due to the fact that high R_a and RC_a already occur on short plots, resulting in high runoff rates independent of plot length. For tree crops, which are also considered as a land use type with permanent cultivation, a positive relation between plot length and R_a and RC_a was found. This difference with vineyards may be due to differences in soil types or the generally higher vegetation cover associated with tree crops, though no data were available to check these hypotheses. As was already indicated by Cerdan et al. (2010), our results illustrate that the relations between plot length and/or slope gradient and SL_a depend on the considered land use. In addition, our analyses show that also relations of plot length and slope gradient with R_a and RC_a depend on land use type. Plot length and slope gradient may affect R_a and RC_a differently than they affect SL_a. Furthermore, a further subdivision of the major land use types analysed by Cerdan et al. (2010) (bare, arable, permanent cultivation and permanent vegetation) reveals that also within these groups, relations between topographic factors and SL_a may vary. For instance, in the permanent vegetation group, no significant relation between LS-factor and SL_a was found for forest, grassland and rangeland plots, but there was a significant relationship between the LS-factor and SL_a for shrubland (Table 3.1).

3.4.2 Frequency distributions and relationships between annual runoff and soil loss for various land uses

Previous studies (e.g. Cerdan et al., 2010; Kosmas et al., 1997) showed the importance of vegetation cover as an important determinant of SL_a on a (sub-) continental scale. This study further confirms this, as significant differences between the distributions of SL_a data were mostly found between land use types with cultivation and land use types with semi-natural vegetation. Construction sites have the highest observed SL_a rates (Fig. 3.2, Table 3.2), although the limited amount of available data (Table 2.5) does not allow clear conclusions on the frequency distribution of the observed values. Bare plots and plots with crop cultivation (cropland, vineyards and tree crops) form a second cluster with high R_a, RC_a and SL_a rates (Fig. 3.2, Fig. 3.3). All these land use types are characterised by severe anthropogenic disturbance (construction sites and bare plots) or intensive tillage of the soil for agriculture (cropland, tree crops and vineyards). On these plots, SL_a regularly exceeds tolerable soil loss rates (T-values) of 5 to 12 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ (Montgomery, 2007b). The mean SL_a rate of 11.6 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ for tree crops in the pan-Mediterranean zone (Table 3.2) was found to be much higher than reported in previous studies such as Cerdan et al. (2010) who found an average SL_a rate for orchards in the Mediterranean of 1.67 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ and Fleskens and Stroosnijder (2007) who concluded that SL_a in olive orchards is unlikely to exceed 10 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$. Annual SL_a in olive orchards is often the result of infrequent high-intensity rain events and depends strongly on spatial scale and land management (Gómez et al., 2008). Depending on which data are included in the analysis and the way the available data are analysed they may be deemed low (Fleskens and Stroosnijder, 2007) or frequently in excess of tolerable rates (Gómez et al., 2008). The use of a larger database (20 plots, corresponding to 80 plot-years) in this study indicates that SL_a in tree crops in the pan-Mediterranean zone can frequently exceed tolerable levels. Hence, care should also be taken when interpreting continental-wide assessments of SL_a (e.g. Cerdan et al., 2010; Maetens et al., 2012b) for land use types that are based on a limited amount of data. While these are the best estimates currently available, they remain highly uncertain with respect to accuracy of the estimated mean and variability of of SL_a.

Vineyards show a high coefficient of variation for SL_a in addition to a high average SL_a value (Table 3.2). This is attributed to the high erosion susceptibility of this land use type after vineyard establishment. Wicherek (1991) reports a sharp decline in soil losses over the first three years after vineyard establishment in the Aisne region, Northern France (57.3, 28.7 and 1.4 $Mg \cdot ha^{-1} \cdot yr^{-1}$, respectively). Tropeano (1984) also reports the highest soil loss (8.3 $Mg \cdot ha^{-1} \cdot yr^{-1}$) to occur in the first year after vineyard establishment in the Piemonte region, North-West Italy, which was reduced to 1.2 $Mg \cdot ha^{-1} \cdot yr^{-1}$ in the following year. This is also illustrated by Engels (2009) who notes that soil loss from an old undisturbed vineyard in the Moselle region, Western Germany is considerably lower (0.5 $Mg \cdot ha^{-1} \cdot yr^{-1}$) than that from an adjacent vineyard where vines were removed and roots destroyed (4,4 $Mg \cdot ha^{-1} \cdot yr^{-1}$). Nevertheless, no clear trends in SL_a between 8 and 17 years after vineyard establishment were found for vineyards in the Douro region, Northern Portugal (de Figueiredo and Gonçalves Ferreira, 1993; de Figueiredo and Poesen, 1998, de Figueiredo, personal communication). Brenot et al. (2006) measured soil loss by vine stock unearthing (i.e. over the complete period since vineyard establishment) for vineyards of different ages (10-54 years) in Burgundy, France, but no trend in SL_a with respect to vineyard age is observed. In contrast to the high SL_a observed for several vineyards, some vineyards established on very steep slopes (ca. 45 %) with very stony soils show comparatively very low mean SL_a values in the Douro region, Portugal (0.2 - 0.5 $Mg \cdot ha^{-1} \cdot yr^{-1}$: de Figueiredo and Gonçalves Ferreira, 1993; de Figueiredo and Poesen, 1998) or the Moselle region (0.02-0.5 $Mg \cdot ha^{-1} \cdot yr^{-1}$: Richter, 1980). Although the number of available plots on which SL_a data were collected directly after vineyard (re)planting (n=1) or removal of the vines (n=1) is limited, these results indicate that vineyards are in general very vulnerable to soil loss after establishment or periodical re-planting of the vines due to the often intense soil disturbances (Borselli et al., 2006). After a number of years, SL_a in vineyards often decreases to relatively low rates as is shown by the median SL_a of 0.9 $Mg \cdot ha^{-1} \cdot yr^{-1}$ for the whole of Europe and the Mediterranean (n=39, Table 3.2). This is attributed to the stony soils often associated with vineyards which develop an erosion pavement with a high rock fragment cover which drastically reduces SL_a (Poesen et al., 1994; Poesen and Lavee, 1994).

Land use types with a (semi-)natural vegetation generally have mean SL_a values well below the T-value, illustrating the dominant control of vegetation cover over SL_a (Montgomery, 2007b). However, SL_a rates up to 7 $Mg \cdot ha^{-1} \cdot yr^{-1}$ can be observed on all land use types and differences between land use types are mainly situated in the frequency of occurrence for high SL_a values (i.e. above 10 $Mg \cdot ha^{-1} \cdot yr^{-1}$) (Fig. 3.2). Scaling of SL_a values to unit plot soil loss rates results generally in somewhat lower SL_u rates (Fig. 3.1), but does not change the differences between different land use types observed above (Fig. 3.2, Table 3.2).

Hence, while plot length and slope gradient may have an effect on SL_a values, they cannot fully account for the differences between different land use types.

The distribution of R_a and RC_a values follows the same pattern as SL_a, with land use types with a (semi-) natural vegetation cover having generally lower R_a and RC_a rates (Fig. 3.2). However, less significant differences are observed between different land use types combinations than for SL_a. There is a larger overlap between the R_a and RC_a data distributions (Fig. 3.2) which indicates that the effect land use types have on SL_a is more pronounced than they are on R_a and RC_a. No standard exists for tolerable R_a and RC_a levels, but excessive runoff also has several negative off-site effects such as gully formation (Poesen et al., 2006) and flooding (García-Ruiz et al., 2010).

With respect to the relations between R_a/RC_a and SL_a/SL_u, the small differences between r^2 values obtained for weighted and unweighted regressions (Table 3.3) can be explained by the distribution of the number of plot-years since a large part of the number of plots corresponds to a low number of plot-years and hence have similar weights (Fig. 2.3). The generally better correlations between R_a and SL_a than between RC_a and SL_a (Table 3.3) can be explained by the fact that mainly the total runoff volume, rather than the runoff coefficient determines the erosive power and transport capacity of the overland flow. Nevertheless, the inclusion of differences in P_a makes it easier to compare RC_a rates between different plots. Hence, both R_a and RC_a are important variables to consider. The general trend in the relation between R_a and SL_a is that (semi-)natural land uses show SL_a rates of up to one order of magnitude less than SL_a values for land use types with crop cultivation for the same R_a rate (Fig. 3.4b). This is not the case for tree crops however, where the increase in SL_a with increasing R_a is much stronger than for other land use types (Fig. 3.4a). However, this trend should be interpreted with caution as it is based on only 13 plots. For shrubland, clusters of data that correspond to individual plot measuring stations show a good correlation between R_a and SL_a (Fig. 3.4a), while the global regression is affected by more scatter in the data, which is most likely determined by local environmental factors that differ from plot site to site. Summer crops like maize, potatoes and sugar beet were found to result in R_a and SL_a rates which are among the highest for cropland (Fig. 3.5).

3.4.3 Effects of land use on runoff coefficient and soil loss for different climatic zones

The higher RC_a observed in the cold zone compared to the other climatic zones may be attributed to the combination of snowmelt and a frozen soil in spring (e.g. Alström and Bergman, 1990; Wade and Kirkbride, 1998) and the generally lower annual evapotranspiration rate at high latitudes (Weiß and Menzel, 2008). Furthermore, R_a and RC_a rates in the pan-Mediterranean zone are generally higher than in the temperate climatic zone (Table 3.2, Fig. 3.6). This is likely due to the combination of soil properties, the often lower and more discontinuous natural vegetation cover in the pan-Mediterranean region, and the seasonality of the rainfall with a large fraction of the P_a concentrated in a few important events during a short winter season (Altava-Ortiz et al., 2011; Mehta and Yang, 2008).

Nevertheless, these higher R_a and RC_a rates in the pan-Mediterranean do not result in high SL_a (Fig. 3.6, Table 3.2). Smaller SL_a rates in the Mediterranean were also observed by Cerdan et al. (2010) and can be explained by the generally higher rock fragment cover in Mediterranean soils which is known to reduce soil loss rates (Poesen et al., 1994; Poesen and Lavee, 1994). Especially for vineyards, which are often located on stony soils, SL_a rates in the pan-Mediterranean climatic zone are much smaller than those in the temperate climatic zone (Fig. 3.6). Nevertheless, Sanchis et al. (2008) observed that also for soils with a low rock fragment content (<10% content by mass) soil erodibility was lower in the Mediterranean. This is attributed to a dominance of clay rich soils in arable land in the Mediterranean and the associated low soil erodibility of these soils (Torri et al., 1997). Furthermore, very erodible soil types like loess-derived soils occur almost exclusively in the temperate climatic zone. Hence, it should be noted that differences between climatic zones include more than just a climatic effect as climatic zone is also a proxy for particular soil properties which affect R_a and SL_a like soil susceptibility to cracking. Apart from differences between precipitation and soil characteristics between the temperate and pan-Mediterranean climatic zones, also differences in conventional land management like tillage frequency and depth in cropland may account for part of the observed variability. Furthermore, for grassland and forest, there is no significant difference between median SL_a rates for plots in the pan-Mediterranean zone and plots in the temperate climatic zone, which again indicates the important effect of vegetation in controlling SL_a.

3.4.4 Effects of annual precipitation on annual runoff and soil loss for different land use types

For all land uses, there is a consistent trend towards higher R_a with increasing P_a which is more pronounced for land use types with crop cultivation than land use types with (semi-) natural vegetation (Fig. 3.8a). For all land use types with crop cultivation, except for vineyards, there is also a trend towards increasing RC_a with increasing precipitation, indicating that as rainfall increases, there is a larger percentage of excess rainfall (Fig. 3.8b). This may be related to distribution of rainfall patterns throughout the year, with areas with high P_a generally having a more uniform precipitation distribution throughout the year. This causes seasonal saturation of the soil and faster runoff formation (Ponce and Hawkins, 1996). A more detailed analysis and discussion of the rainfall-runoff relationship is given in chapter 5.

For SL_a, there is also an increase in SL_a rates with increasing P_a for most land use types (Fig. 3.8c), but unlike R_a, the increase is more gradual. For bare and cropland plots, there is even a decrease in SL_a in the highest P_a classes (>750 mm·yr^{-1} for bare and >1000 mm·yr^{-1} for cropland). This may be attributed to variations in seasonality of the rainfall, as the relatively uniform rainfall distribution in regions with high P_a in Europe and the Mediterranean makes these regions less prone to unfrequent extreme rainfall events in periods of the year when the soil is vulnerable to erosion (Edwards and Owens, 1991; Larson et al., 1997).

Shrubland is generally limited to the pan-Mediterranean region and a maximum in SL_a is observed in the 250-500 mm·yr^{-1} P_a class (Fig. 3.8c), and both the differences between the lower and higher precipitation class are significant. A similar trend was noted by Kosmas et al. (1997) who attributed this to an initial increase in SL_a with increasing P_a as also the erosion potential of the rain increases, combined with insufficient vegetation cover in drylands. However, as P_a futher increases also the vegetation cover increases, effectively reducing SL_a for higher P_a. This is similar to the mechanism and trend for sediment yield proposed by Langbein and Schumm (1958). However, this trend is not observed for other land use types, as vegetation cover in shrubland can be expected to be the most sensitive to changes in P_a. Contrary to the study by Kosmas et al. (1997), this maximum is not noticeable for R_a, which may indicate that at a Mediterranean-wide scale, runoff volume is more determined by other environmental characteristics than vegetation cover, such as surface sealing and rock fragment cover.

In general, significant differences in the frequency distribution of both R_a and SL_a were mostly found between the 250-500 mm·yr^{-1} and 500-750 mm·yr^{-1} P_a classes (Fig. 3.8a,Fig. 3.8c). This may indicate that significant changes in the rainfall-runoff and rainfall-soil loss relations occur at around these P_a values which are likely related to changes in rainfall regime and distribution throughout the year. Hence, comparison of R_a and SL_a rates with measures that take rainfall distribution throughout the year into account could improve the analysis. Nevertheless, the use of MFI as an indicator of climatic effect on R_a, RC_a and SL_a did not yield better correlations than the use of P_a (Fig. 3.9). The MFI values used in this analysis represent long-term average (i.e. climatological) values which may not be representative for the specific years of plot measurements. This explains why low R_a and SL_a rates occur regularly in zones with high MFI values. Furthermore, the relation between MFI and P_a (Fig. 3.9) and MFI and EI30 values (Torri et al., 2006) is not straightforward.

3.5 Conclusions

Recently, several studies have assessed the erosion problem by reviewing existing soil loss data from plots and by using these data in empirical studies or erosion model validations. Many of these studies investigated soil loss rates in relation to land use, topography and soil properties. However, plot-scale runoff has been largely neglected in these studies and the relation between runoff and soil loss has not been reviewed on a scale covering Europe and the Mediterranean. Nevertheless, runoff estimation and its relation to soil loss are important parts of many erosion models. On a (sub-)continental scale also climatological differences are often not taken into account in overview studies. To address these issues, the largest dataset of runoff and soil loss plot data for Europe and the Mediterranean region was compiled in this study, which includes for the first time both annual runoff and annual runoff coefficients at the continental scale.

In general, soil loss studies using runoff plots mainly focused on bare plots and cropland and less data are available for construction sites, tree crops and vineyards, in spite of the high soil losses that may be associated with these land use types. Variation in annual runoff and soil loss rates observed on the plots range over several orders of magnitude. While land use types with crop cultivation (cropland, fallow, tree crops and vineyards) have higher mean soil loss rates than land use types under (semi-)natural vegetation (grassland, rangeland, shrubland, forest and post-fire), there are still large variations within each of these land use types, which can only be partly accounted for by topographic (i.e. plot length and slope gradient) differences. Annual runoff rates follow the same pattern as annual soil loss rates, but differences between land uses are less clear. The generally good relations between annual runoff and annual soil loss illustrate the key importance of the relation between runoff and soil loss for a good assessment of soil loss rates. Further quantitative analysis of this relation may also contribute significantly to improve predictions erosion models. Furthermore, the relation between annual runoff and soil loss also depends on climate, with comparatively high runoff coefficients in cold climates and lower soil losses in the pan-Mediterranean region. This indicates that runoff-soil loss relations may show important regional variations. Apart from the importance of runoff as a causal factor of soil loss, runoff in itself is associated to several problems such as flooding and plant-available water. Techniques specifically aimed at reducing runoff and runoff coefficients can contribute to a more efficient use of rainwater on-site to increase food production, especially in drier regions like the Mediterranean.

As expected, annual rainfall was found to be related to annual runoff and soil loss, and the vegetation feedback effect for shrubland proposed by Kosmas et al. (1997) was also observed on a Mediterranean-wide scale for soil loss, but not for runoff. Nevertheless, a large part of the variation in runoff and soil loss rates remains unexplained and better relations between annual precipitation and runoff and soil loss can likely be obtained by accounting for rainfall erosivity, but use of a Modified Fournier Index did not yield better results than the use of the annual precipitation measured on the plots. Hence there are still possibilities to expand the research to better account for rainfall erosivity. Further research may also focus on the effects of several important soil characteristics (e.g. texture, organic matter content) and a more detailed analysis of several relations for which general trends were established in this study, e.g. the relation between annual runoff and plot length and plot slope gradient, the relation between annual runoff and soil loss. In conclusion, this meta-analysis of field plot data for Europe and the Mediterranean allows a quick assessment of the impact of land use change scenarios on annual runoff, runoff coefficient and soil loss on a continental scale.

Chapter 4

Inter-annual variability of plot scale annual runoff and soil loss

4.1 Introduction

As discussed in chapter 2, section 2.3 and chapter 3, measuring periods (MP) for plots are relatively short (mean: 6 yrs., median: 4 yrs.). However, many models for the prediction of annual soil loss (SL_a) rates (e.g. (R)USLE; Wischmeier and Smith, 1978; Renard et al., 1997, the PESERA map; Kirkby et al., 2004 and the Soil Erosion Map of Europe (SEM): Cerdan et al., 2010) indicate that the predicted values should be considered *"long-term average"* or *"stable average"* SL_a rates. Wischmeier and Smith (1965) indicate USLE-predicted values should be considered mean SL_a rates over measuring periods of 20 years, but relatively few information of the exact meaning of *"long-term"* (i.e. the exact measuring period that is considered long-term) is available in plot-scale erosion research. The relatively short measuring periods for most runoff and soil loss plots in chapter 2 have several consequences with respect to the reliability and representativeness of mean annual runoff (R_a), annual runoff coefficient (RC_a), and annual soil loss (SL_a) rates. These short measuring periods also complicate a global assessment of inter-annual variability in R_a and SL_a rates that is valid beyond the measuring period of the single study.

A considerable year-to-year variation in annual R_a and SL_a data is observed in many runoff and soil loss plot measuring studies (e.g. Kosmas et al., 1997; Øygarden, 1996; Stroosnijder, 2005). This effect is also replicated in modelling studies (e.g. Renschler et al., 1999), and is mostly explained by the inter-annual variability in rainfall erosivity. While most multi-annual runoff and soil loss plot studies provide a quantification of the inter-annual variability of measured R_a and SL_a data in the form of mean R_a and SL_a along with standard deviations, they provide little information on how representative the measured values are for the long term. Nevertheless, the measured mean R_a and SL_a are often compared between different studies without a detailed analysis of the representativeness of the mean R_a and SL_a data (e.g. Auerswald et al., 2009; Fleskens and Stroosnijder, 2007).

Some runoff and soil loss plot studies include a study of the cumulative R_a and SL_a (e.g. Bagarello and Ferro, 2004; de Figueiredo and Poesen, 1998; Francia Martínez et al., 2006; Hudek and Rey, 2009; Martin, 1999; Martínez-Murillo and Ruiz-Sinoga, 2007; Withers et al., 2006), but no detailed studies on the inter-annual variation of R_a and SL_a exist for Europe and the Mediterranean. Long-term studies (>10 yrs.) of plot-measured R_a and SL_a rates are limited to a relatively small number of plots and plot measuring stations. Hence there is very little information on how long R_a and SL_a should be measured to obtain reliable mean R_a and SL_a rates. Several authors have stressed the importance of low frequency-high magnitude erosion events in total measured R_a and SL_a (e.g. de Figueiredo et al., 1998; Edwards and Owens, 1991; Larson et al., 1997; Poesen et al., 1996) in this respect. Recent review studies incorporate this effect by weighting the mean plot R_a and SL_a data with the square root of the number of plot-years (PY) (Vanmaercke et al., 2011b,chapter 2,chapter 3) or number of plot-months (Cerdan et al., 2006, 2010) in the calculation of global mean R_a and SL_a values (e.g. per land use type). Thereby, more weight is given to mean R_a and SL_a rates that represent a longer measuring period in accordance with the central limit theorem (Tijms, 2004). However, the validity of this weighting procedure for global mean plot R_a and SL_a data remains unproven and it assumes a uniform convergence to a long-term mean value for all plots and plot measuring stations (i.e. independent of environmental factors that may control temporal variation in plot-measured R_a and SL_a rates).

It can be assumed that temporal variability in R_a and SL_a may depend on a number of environmental variables. A major source of temporal variability in runoff and soil loss rates is temporal variability in precipitation intensity and annual precipitation depth (P_a). Measures for annual rainfall erosivity such as the (R)USLE R-factor (Renard et al., 1997) and Modified Fournier Index (Gabriëls, 2006) are known to vary strongly from year to year (e.g. Angulo-Martínez and Beguería, 2009; Ferro et al., 1999; Kosmas et al., 1997;

Maetens et al., 2012b; Renschler et al., 1999; Renard and Freimund, 1994; Verstraeten et al., 2006b). Based on detailed long-term precipitation records, Verstraeten et al. (2006b) showed that annual rainfall erosivity had a coefficient of variance of 31% in central Belgium and observed that temporal patterns in the critical season for erosion (May-June in central Belgium) may be different than the annual patterns of rainfall erosivity. Kirkby and Cox (1995) also indicate the importance of monthly variations in soil erosion potential due to rainfall seasonality and also the vegetation response to variations in rainfall depth and air temperature. In addition, variability in annual precipitation (P_a) and monthly or even event precipitation distribution has also indirect effects on R_a and SL_a through feedback effects from changes in vegetation cover in response to rainfall depth and patterns (Kawabata et al., 2001; Richard and Poccard, 1998), which were suggested to affect R_a and SL_a (Kosmas et al., 1997; Maetens et al., 2012b) and catchment sediment yield (Langbein and Schumm, 1958).

Apart from climatic factors, several other environmental or experimental factors such as plot length, slope gradient, soil texture and land use type can have an effect on the observed inter-annual variability of R_a and SL_a rates. While Vanmaercke et al. (2012b) found only weak relations between environmental or experimental factors and the inter-annual variation in catchment sediment yield, these factors may still have a significant effect on the inter-annual variability of R_a and SL_a rates at the plot scale and should be considered as controlling factors of inter-annual variability in R_a and SL_a.

Furthermore, for many applications such as flood-related hazard assessment, hydrologic engineering, and the planning of soil and water conservation techniques, it is necessary to estimate the range and especially extreme values of R_a and SL_a rates that may occur in specific conditions and for a specific return period. Hence, apart from (long-term) mean R_a and SL_a rates, there is also a practical need for the assessment of the variability with respect to the (long-term) mean R_a and SL_a. Wendt et al. (1985) assess the frequency distribution of event runoff and soil loss observed on 40 uniform (i.e. replicated) experimental plots. While the study by Wendt et al. (1985) is mainly directed at the estimation of spatial variability an measurement error between replicated plots, also the correlations between the 25 different events in the study are asessed. Nearing et al. (1999) and Nearing (2000) also assess the variability observed in a large dataset of soil loss data from pairs of replicated plots, containing data from 2061 events, corresponding to 797 annual soil loss measurements and 53 multi-year soil loss totals from 13 plot measuring stations in the U.S.A. They showed a negative correlation between the variability in soil loss observed on replicated plots and the absolute value of the measured soil loss, and that this relation is independent of whether event, annual or multi-annual soil loss data are used, which is important form the point of view of temporal variation. With respect to models, a more stochastic approach to the RUSLE model was also developed by Hession et al. (1996), Snyder and Thomas (1987) and Thomas et al. (1988), which allows for temporal uncertainty in SL_a to be addressed.

However, a detailed assessment of temporal variability in plot-measured R_a and SL_a requires the estimation of the statistical distribution (i.e. probability density functions) of R_a and SL_a time series. No such studies exist for time series of annual plot-measured runoff and soil loss, but a limited number of studies exists for plot-measured event runoff and soil loss. Istok and Boersma (1986) estimated joint frequency distributions of two events leading to large soil losses and Mills et al. (1986) directly estimated soil loss probabilities. However, both these studies are based on modelled soil loss responses to climatic variables due to a lack of plot-measured data to assess large erosion events. Baffaut et al. (1998) fitted log-Pearson type III distributions (LP III, Bobée and Ashkar, 1991) to plot measured soil loss data from six sites, measured over 6 to 10 years, and compared them to time series of model predicted soil loss generated by the WEPP model. Both plot-measured and model-predicted frequency curves were found to fall within the 95% confidence interval of the LP III distributions. Bagarello et al. (2010a) on the other hand used the log-normal distribution to compare time series of event soil loss and concluded that after normalisation of event soil loss over different replicates, the probability distribution of the normalised soil losses was independent of both the temporal scale (i.e. event or annual soil loss) and the plot length. Bagarello et al. (2011) then explored possibilities for using these frequency distributions for conservation planning. Finally, at the catchment scale Vanmaercke et al. (2012b) used the Weibull distribution to determine the uncertainty of mean measured sediment yield values with respect to the long term average sediment yield.

While the use of frequency distributions is relatively limited in erosion research, their use for the assessment of extreme events is far more common in hydrology and flood modelling. While the log-Pearson type III distribution (LP III) has been adopted by the U.S. federal agencies as the standard distribution for flood frequency analysis (Griffis and Stedinger, 2007), several other distributions from the log-normal family and the extreme value family (i.e. the Gumbel, Fréchet and Weibull distributions) have been used in hydrological modelling (e.g. Al-Mashidani et al., 1978; Canfield et al., 1980; Ferdows and Hossain, 2005; Griffis and Stedinger, 2007; Heo et al., 2001; Karim and Chowdhury, 1995; Loaiciga and Leipnik, 1999; Millington et al., 2011). While these statistical techniques are also applicable to plot-measured annual runoff and soil loss time series, frequency distributions of R_a and SL_a have not been addressed extensively in the literature due to a lack of plot-measured data with a sufficiently long measuring periods.

The above-mentioned questions about the general reliability of measured mean R_a, RC_a and SL_a measured throughout Europe and the Mediterranean, the factors affecting temporal variability of these measurements and underlying frequency distributions of R_a and SL_a can not be answered by studies restricted to a single or a limited number of plots and plot measuring stations. The large number of annual plot-measured R_a and SL_a data compiled in this study (chapter 2) allows for the first time to address these questions. Therefore, the objectives of this study are (1) to quantify the inter-annual variation of R_a and SL_a measured on a wide range of runoff and soil loss plots throughout Europe and the Mediterranean, (2) to explore which factors control the inter-annual variation of R_a and SL_a, (3) explore the underlying statistical distributions in plot-measured R_a and SL_a time series and the uncertainty on the long-term mean RC_a and SL_a in function of the measuring period, and (4) to discuss the implications of inter-annual variation in R_a and SL_a rates for further erosion research and its practical applications.

4.2 Material and methods

4.2.1 Annual runoff and soil loss data selection and description

Time series for which R_a and/or SL_a data for the individual years were available and with a minimum period of five years of R_a and/or SL_a measurements were selected from the runoff and soil loss plot database for Europe and the Mediterranean described in chapter 2 (Fig. 4.1, Table 4.1). With respect to runoff and soil loss plot studies, there is no literature to support the arbitrary choice of a five-year minimum measuring period. Vanmaercke et al. (2012b) set a mimimum measuring period of seven years for the assessment of inter-annual variability of catchment sediment yield. However, measuring periods for runoff and soil loss plot studies (Maetens et al., 2012b) are generally shorter than catchment sediment yield studies (Vanmaercke et al., 2011b). Using a minimum measuring period of seven years for runoff and soil loss plot studies would substantially lower the number of available time series (Fig. 4.2) and hence also the range of environmental conditions (e.g. climate, plot length and slope gradient, land use types) over which R_a and SL_a data were measured. The arbitrary minimum measuring period of five years was chosen to provide a balance between data availability and representativeness for Europe and the Mediterranean on the one hand and a measuring period that is long enough to capture the temporal variability on the other hand.

This resulted in a dataset of 234 R_a and 307 SL_a time series, representing 2523 and 3120 plot-years, respectively (Table 4.1), with plot data from 62 plot measuring sites throughout Europe and the Mediterranean (Fig. 4.1). For 217 R_a time series, representing 2359 plot-years, also P_a matching the R_a data were available, allowing the calculation of annual runoff coefficients (RC_a). For 206 time series, representing 2250 plot-years, both R_a and SL_a for all individual years of the time series were available, enabling a comparison between inter-annual variation of R_a and SL_a.

Table 4.1: Overview of number of plots (PL), number of plot-years (PY), and plot data sources for annual runoff (R_a) and annual soil loss (SL_a) time series with at least five years of consecutive measurements data per country in Europe and the Mediterranean. Numbers between brackets refer to the plot measuring stations indicated in Fig. 4.1.

Country	R_a		SL_a		Source
	PL	PY	PL	PY	
Albania	6	30	6	30	Grazhdani et al., 1996 [38-39]; Grazhdani, personal communication [38-39]
Algeria	4	60	4	60	Mazour et al., 2008 [37]
Austria	5	30	5	30	Klik, 2003 [15-17]; Klik, 2010 [15-17]; Klik, personal communication [15-17]
Croatia	-	-	1	5	Basic et al., 2001 [21]; Basic et al., 2004 [21]
Finland	2	18	1	8	Turtola et al., 2007 [7]; Uusi-Kämppä, 2005 [7]
France	12	96	-	-	Wicherek, 1986 [10]
Germany	10	71	10	71	Jung and Brechtel, 1980 [11-14]
Israel	9	72	9	72	Lavee et al., 1998 [58-60]; Lavee, personal communication [58-60]
Italy	28	181	34	219	Bagarello and Ferro, 2010 [24]; Bagarello et al., 2010a [24]; Bagarello et al., 2010b [24]; De Franchi and Linsalata, 1983 [23]; Postiglione et al., 1990 [23]; Vacca et al., 2000 [25]; Ollesch and Vacca, 2002 [25]; Vacca, personal communication [25]; Zanchi, 1983 [22]; Zanchi, 1988b [22]; Zanchi, 1988a [22]
Lithuania	-	-	52	429	Jankauskas and Jankauskiene, 2003a [8]; Jankauskas and Jankauskiene, 2003b [8]; Jankauskas et al., 2004 [8]; Jankauskas et al., 2007 [8]; Jankauskas, personal communication [8]
Macedonia	-	-	3	15	Jovanovski et al., 1999 [40-42]
Morocco	13	65	17	85	Chaker et al., 2001 [36]; Laouina et al., 2003 [33]; Yassin et al., 2009 [34-35]; Yassin, personal communication [34-35]
Norway	8	73	9	79	Børresen, personal communication [1-5]; Grønsten and Lundekvam, 2006 [2,5]; Lundekvam, 2007 [1,3-4]; Øygarden, 1996 [6]
Poland	4	53	4	53	Gil, 1986 [18]; Gil, 1999 [18]
Portugal	18	191	18	191	de Figueiredo and Gonçalves Ferreira, 1993 [31]; de Figueiredo et al., 1998 [31]; de Figueiredo, personal communication [31]; Roxo et al., 1996 [32]
Romania	4	95	4	95	Ionita, 2000 [20]; Nistor and Ionita, 2002 [19]
Spain	20	227	38	423	Andreu et al., 1998a [27]; Andreu et al., 1998b [27]; Castillo et al., 1997 [28]; Cerdà and Lasanta, 2005 [26]; García-Ruiz et al., 1995 [26]; Guerra et al., 2004 [61-62]; Lasanta et al., 2006 [26]; Martinez-Mena et al., 1999 [29]; Nadal Romero and Lasanta, personal communication [26]; Puigdefábregas et al., 1996 [30]; Rodríguez Rodríguez et al., 2002 [61-62]; Solé, personal communication [30]
Turkey	81	1172	81	1178	Köse et al., 1996 [45]; Köse and Taysun, 2002 [45]; Oguz et al., 2006 [43-44,46-57]; Oguz, personal communication [43-44,46-57]
United Kingdom	11	89	11	77	Fullen, 1998 [9]; Fullen and Booth, 2006 [9]; Fullen et al., 2006 [9]
Total	234	2523	307	3120	

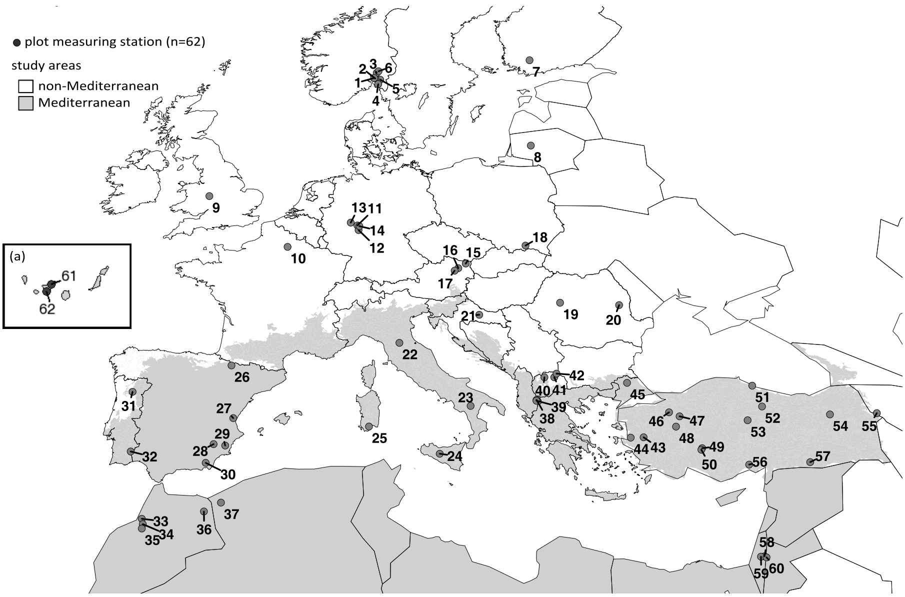


Figure 4.1: Geographical distribution of runoff and soil loss plot measuring stations from which data for annual runoff and annual soil loss time series in Europe and the Mediterranean are used in this study. The division between Mediterranean and Non-Mediterranean was derived from the LANMAP2 database (Mücher et al., 2010; Metzger et al., 2005). Plot measuring site numbers refer to the date sources indicated in Table 4.1. n= number of plot measuring stations.

An overview of the frequency distribution of measuring periods for R_a and SL_a time series is given in Fig. 4.2. The measuring period for the majority of R_a and SL_a time series is relatively short (R_a time series: mean= 10.8 yrs., median=9 yrs., mode= 5 yrs., SL_a time series: mean= 10.2 yrs., median=8 yrs., mode= 6 yrs.). The longest R_a and SL_a time series were measured in Perieni, Romania over 30 years (Ionita, 2000). While longer time series of runoff and soil loss plot measurements exist at Vale Formoso, Portugal (i.e. 32 yrs.: Roxo et al., 1996), a land use rotation between cropland and fallow was applied and the plots are not considered as continuous measurements for the purposes of this study. Other notable exceptions to the relatively short measuring periods are the long-term runoff and soil loss plot studies at various sites in Turkey (20-27 yrs.: Oguz et al., 2006), at Szymbark, Poland (19 yrs.: Gil, 1986) and rill volume measurements in Kaltinenai, Lithuania (18 yrs.: Jankauskas and Jankauskiene, 2003a,b; Jankauskas et al., 2004, 2007).

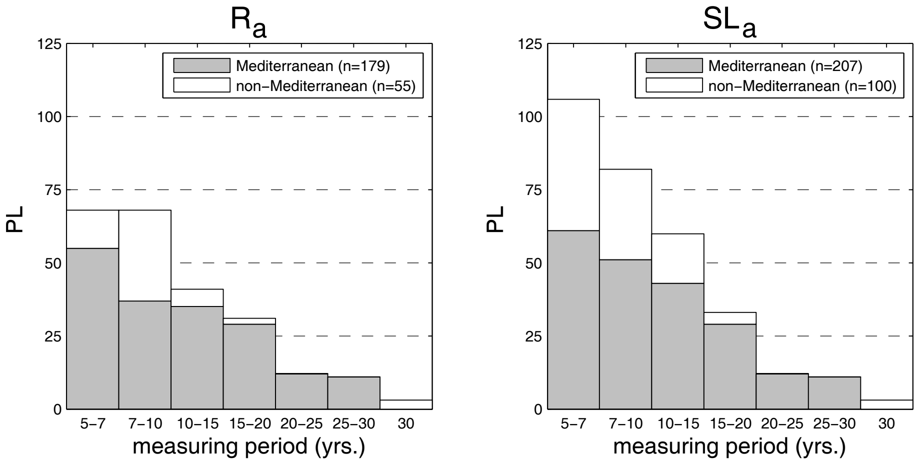


Figure 4.2: Frequency distribution of the measuring period (MP) for the annual runoff (R_a) and annual soil loss (SL_a) time series. The division between Mediterranean and non-Mediterranean was derived from the LANMAP2 database (Mücher et al., 2010; Metzger et al., 2005). PL= number of plot-years, n= number of time series.

One of the major causes of temporal variability in $R_a$and $SL_a$is the distribution of daily precipitation and rainfall erosivity (e.g Renschler et al., 1999). However, very few studies report time series of daily precipitation and/or rainfall erosivity. To explore the effect of the frequency distribution of daily precipitation (P_d) over the time series measuring periods, time series of P_d matching the measuring period of each time series where extracted from the E-OBS climatic database (Haylock et al., 2008), which contains spatially distributed daily precipitation records from 1950 to 2006, interpolated at a 25km resolution. For each of these P_d time series where a full record of $P_d$values (i.e. no missing values) could be obtained from E-OBS, the Precipitation Concentration Index (PCI) as defined by Martin-Vide (2004) was calculated. This Precipitation Concentration Index is defined as a Gini-coefficient for daily precipitation distributions and is calculated as:

$$PCI = \frac{2 \cdot S'}{10\ 000} \qquad \text{(Eq. 4.1b)}$$

$$S' = 5000 - \int_0^{100} ax \cdot \exp^{(bx)} \qquad \text{(Eq. 4.1b)}$$

Where PCI is the Precipitation Concentration Index, a and b are parameters obtained by fitting the Lorentz curve to the relative cumulative distribution of daily precipitation, and S' is the area enclosed by the equidistribution line (i.e. 1:1 line) and the Lorentz curve for the observed daily precipitation distribution (Fig. 4.3).

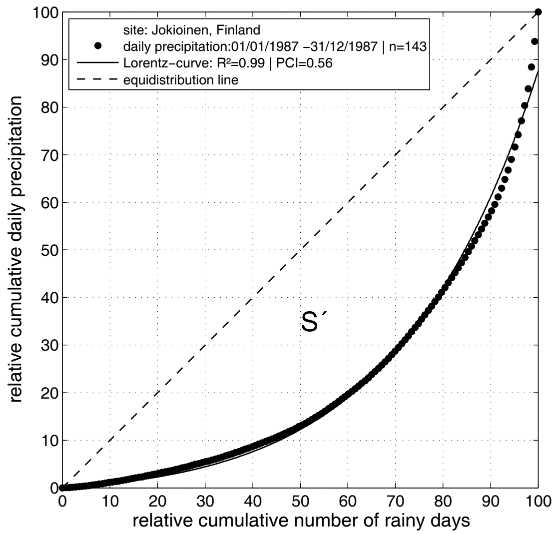


Figure 4.3: Example for the calculation of the daily Precipitation Concentration Index (PCI) for the daily precipitation time series at Jokioinen, Finland for the measuring period between 1 January 1987 and 31 December 1987. S' is the area enclosed between the equidistribution line and the Lorentz-curve. n= total number of rainy days in the measuring period. R^2: R-squared for the fit of the Lorentz-curve to the daily precipitation data.

A graphical example of the calculation of the Gini coefficient is given in Fig. 4.3. For the calculation of the PCI corresponding to each RC_a and SL_a time series, the distribution of P_d during the full measuring period is used.

4.2.2 Analysis of inter-annual variability of runoff and soil loss time series

To characterise the time series of R_a, RC_a and SL_a, a number of descriptive statistical measures were calculated for each time series such as time series minimum to maximum ratio, time series skewness and time series coefficient of variation (CV):

$$\mathrm{CV} = \frac{\sigma}{\mu} \qquad \text{(Eq. 4.2: Hazewinkel, 2001b)}$$

$$\text{skewness} = \frac{\mu_3}{\sigma^3} = \frac{\frac{1}{n}\sum_{i=1}^{n}(x_i-\mu)^3}{\left(\frac{1}{n}\sum_{i=1}^{n}(x_i-\mu)^2\right)^{3/2}} \qquad \text{(Eq. 4.3: Hazewinkel, 2001a)}$$

Where CV= time series coefficient of variation, σ= time series standard deviation, μ= time series mean, μ_3= time series third moment about the mean.

The Liliefors test was used to test the normality of time series (Lilliefors, 1967). Correlations between the time series characteristics and continuous variables such as slope length, plot gradient and P_a were tested by calculating the Pearson correlation coefficient (Rodgers and Nicewander, 1988) and Spearman's rank correlation coefficient (Zar, 1972). Differences between discrete groups of data (i.e. climatic zones and land use types) were tested using Kolmogorov-Smirnov tests (KS-test) (Massey, 1951), with application of a Bonferroni correction by dividing the confidence level by the number of possible comparisons (Brittain, 1987) to correct for family-wise error caused by multiple comparisons within the same dataset.

To make an assessment of the measuring period that is required to obtain reliable long-term average RC_a and SL_a rates, a Monte Carlo simulation was applied. For all RC_a and SL_a time series with a measuring period equal to or longer than 10 years (RC_a: n= 98, SL_a: n=116), several continuous distributions previously used in the assessment of temporal variability in runoff and soil loss (i.e. the log-normal, (log-)Pearson type III, Gumbel, Weibull and Fréchet distributions) were evaluated using the "fidistrplus" package in the R software (Delignette-Muller et al., 2012). The Weibull distribution (Weibull, 1951) was found to best describe the RC_a and SL_a time series, although differences between goodness of fit statistics between the Weibull and other distributions are relatively small and differ between individual time series. Hence, there may not be a universally best distribution to fit RC_a and SL_a time series, and as described for hydrological studies in section 4.1, different distributions may provide better fits depending on the study. However, for consistency only the Weibull distribution is used in the remainder of this chapter.

As the Weibull distribution does not include zero values and zeros are observed in both RC_a and SL_a time series (i.e. no runoff and/or soil loss measured), the distribution fit was optimised with the inclusion of a location parameter (θ; Eq. 4.4). This parameter allows zeros to be included in the Weibull distribution fit. Fitting was done using maximum likelihood estimation. For the Weibull distribution, other distribution fitting techniques exist for when zeros are present in the observed data (e.g. the L-moments method, Hosking, 1990), but these do not allow the prediction of zeros by the fitted distribution.

$$f(x;\theta,\lambda,k) = \frac{k}{\lambda}\left(\frac{x+\theta}{\lambda}\right)^{k-1} \cdot e^{-\left(\frac{x+\theta}{\lambda}\right)^{k}} \qquad \text{(Eq. 4.4)}$$

Where k>0 is the shape parameter, λ>0 is the scale parameter, and θ is a location parameter. θ>0 when zeros are observed in the RC_a or SL_a time series and θ=0 when no zeros are observed.

For 51 RC_a time series and 68 SL_a time series, the Cramér - von Mises criterion (Anderson, 1962; Laio, 2004) indicated a significant fit for the Weibull distribution at α=0.1. The lack of a significant fit for about half of the RC_a and SL_a time series with a measuring period larger than or equal to 10 years is attributed to the fact that 10 to 30 years is still a relatively short measuring period for the estimation of the underlying distributions, which means that the estimation of the probability of large RC_a and SL_a occurring (i.e. the right tail of the frequency distribution) is subject to large uncertainty. For all time series where the Cramér - von Mises criterion indicated a significant fit at α=0.1, the long-term mean RC_a and SL_a was calculated using a numerical approximation of the expected value (E(x)) of the fitted Weibull distributions, which are left-censored at θ:

$$E(x) = \frac{1}{10000}\sum_{i=1}^{10000}(x_i^*) \quad where \quad \begin{array}{ll} x_i^* = x_i - \theta & if\ x_i \geq \theta \\ x_i^* = 0 & if\ x_i < \theta \end{array} \qquad \text{(Eq. 4.5)}$$

Where E(x) is the expected long-term mean RC_a/SL_a value, and x_i values are randomly drawn from the fitted Weibull distributions (Eq. 4.4).

Subsequently, 50 observations for RC_a and SL_a were randomly drawn from the fitted distributions, corrected for the location parameter θ and left-censored at 0. The estimated relative error on the long term mean for plot RC_a and SL_a for measuring periods ranging between 1 and 50 years was then calculated as:

$$Rel.\ Err.\ on\ mean\ RC_{a,i} = \frac{\left(\overline{RC_{a,i}} - E(RC_a)\right)}{E(RC_a)} \qquad \text{(Eq. 4.6)}$$

$$Rel\ Err.\ on\ mean\ SL_{a,i} = \frac{\left(\overline{SL_{a,i}} - E(SL_a)\right)}{E(SL_a)} \qquad \text{(Eq. 4.7)}$$

Where Rel. Err. on mean $RC_{a,i}$ and Rel. Err. on mean $SL_{a,i}$ are the relative errors of the sample mean for a measurement period (MP) of i years (1≤MP≤50), $\overline{RC_{a,i}}$ and ($\overline{SL_{a,i}}$) are the means of the first i values of the 50 randomly drawn observeations of RC_a and SL_a, respectively. $E(RC_a)$ and $E(SL_a)$ are the expected long-term mean RC_a and SL_a, respectively (Eq. 4.5).

This procedure was then repeated 1000 times for each fitted distribution that was significant at α=0.1 (i.e. 51 RC_a time series and 68 SL_a time series). This results in 1000 estimates of the relative error on the long-term mean RC_a and SL_a for different measuring periods between 1 and 50 years. The 2.5^{th} and 97.5^{th} percentiles of each set of 1000 relative errors were then calculated as an assessment of the maximum errors in comparison to the long-term average RC_a and SL_athat are expected for a given measuring period. This procedure is then repeated for each of the significant (α=0.1) RC_a and SL_a time series. Note that this approach does not distinguish between plots in different land use types or climatic zones. While both land use type and climatic zone may have an effect on the temporal variability of RC_a and SL_a, an analysis of their effects is beyond the scope of this research.

4.3 Results and discussion

4.3.1 Characteristics of inter-annnual variability in R_a and SL_a

Visual representation of the RC_a and SL_a time series (Fig. 4.4) shows that time series median values are generally smaller than mean values, indicating right-tailed distributions for both RC_a and SL_a data. There is also no clear relation between mean values and the range of RC_a and SL_a rates in the time series.

A Lilliefors normality test on the individual time series showed that 93 R_a time series (39.7%), 63 RC_a time series (29.0%) and 175 SL_a time series (57.0%) do not come from a normally distributed populations (α=0.05). Hence, a larger proportion of SL_a data than R_a data is not normally distributed. The lower percentage of not normally distributed RC_a time series (29%) compared to the percentage of not normally distributed R_a time series (39.7%) is attributed to the fact that variability in $P_a$rates is also taken into account in the calculation of the RC_avalues. Furthermore, Fig. 4.5 shows that with increasing measuring period, a larger proportion of time series tend to come from a non-normal distribution. In addition, CV and skewness increase with increasing measuring period, while the minimum/maximum ration decreases with increasing measuring period for both RC_a and SL_a time series.

As time series frequency distributions tend to be not-normal and right-tailed, data descriptors that do not assume a normal distribution (such as skewness) and non-parametric statistics should be used (e.g. Bonett and Seier, 2006; Bonett, 2006; Efron, 1981) for the time series analysis. However, while for non-normal distributions the CV can not be used for Gaussian statistics such as e.g. the estimation of confidence intervals, the coefficient of variation remains a useful measure of data variability, even for non-normal distributions (e.g. Hastings, 1965; Vanmaercke et al., 2012b). Both CV and skewness were used in the analysis of the time series, but only the results for CV are given and discussed when the analysis of skewness did not yield additional insights. The minimum to maximum ratio was not further considered in this research as it was found to be highly sensitive to the minimum value in the time series and low R_a and SL_a values are known to have a high natural variability (Nearing et al., 1999) and are hence not very indicative of the inter-annual variability in R_a and SL_a data.

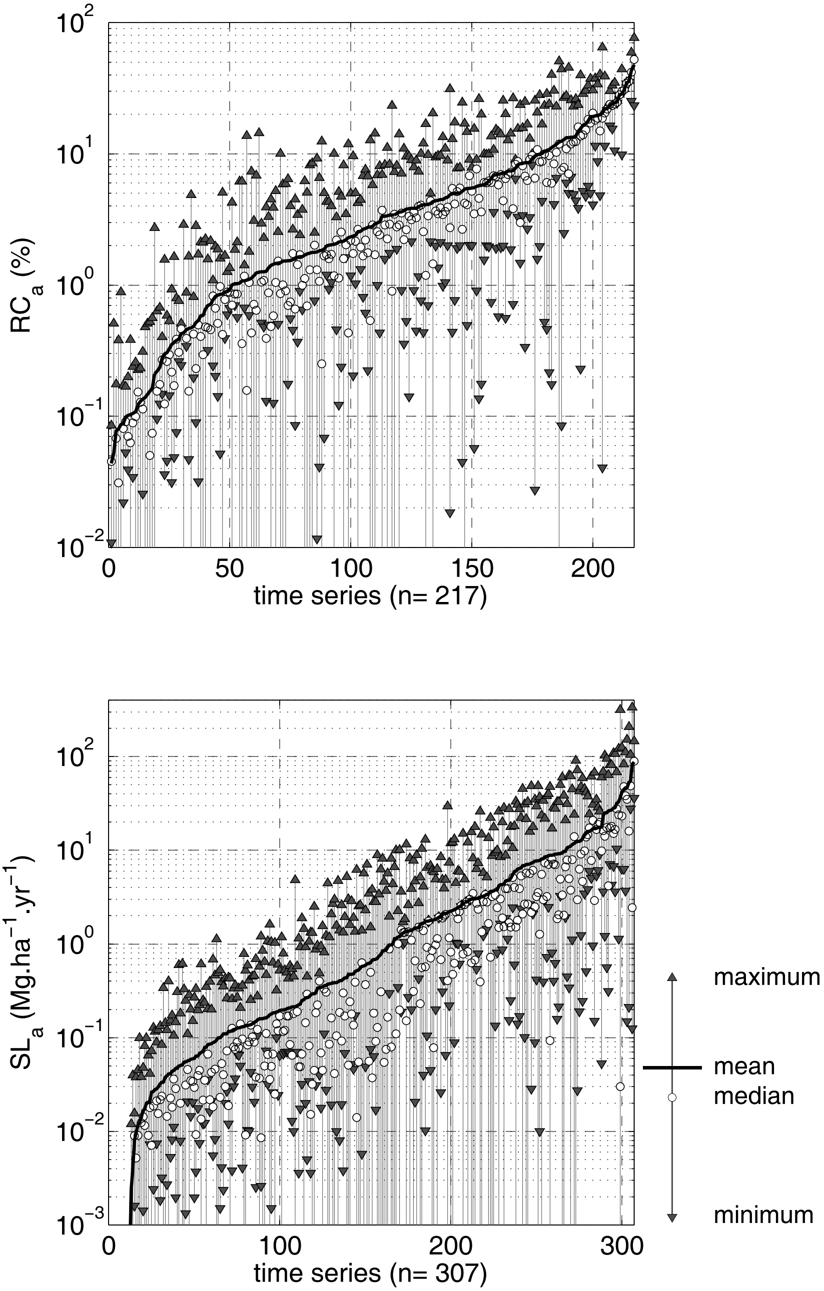


Figure 4.4: Maximum, median, mean and minimum recorded annual values for all time series for RC_a and SL_a, sorted in order of ascending mean value. n= number of time series.

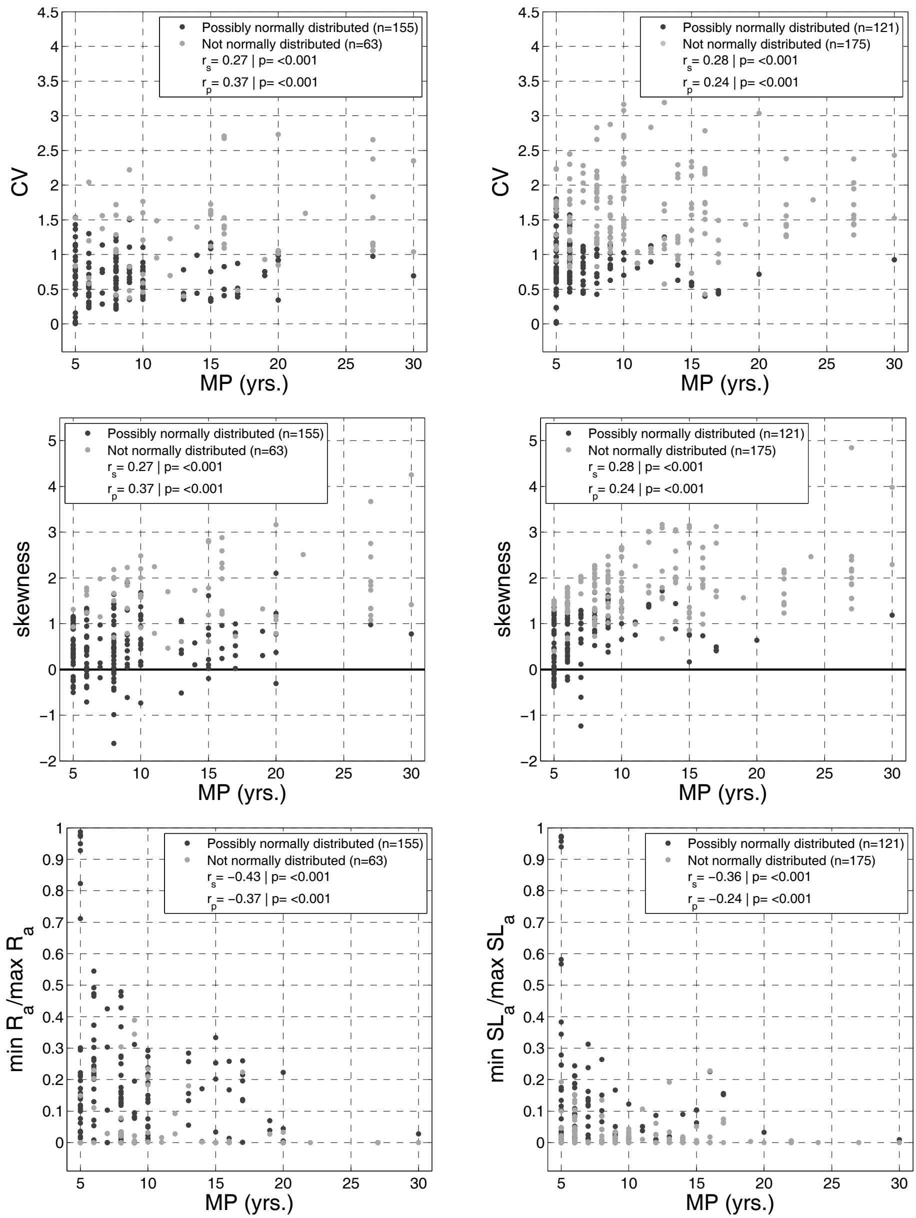


Figure 4.5: Relation between time series measuring period (MP) and the time series coefficient of variation (CV), skewness, and minimum to maximum ratio for annual runoff coefficients (RC_a) and annual soil loss (SL_a). n= number of time series. r_s= Spearman's rank correlation coefficient, r_p= Pearson correlation coefficient.

4.3.2 Factors controlling inter-annual variability of R_a and SL_a

Magnitude of time series mean annual runoff and soil loss

Fig. 4.6a shows a significant negative correlation (R^2=0.12, p<0.001) between the base-10 logarithm of the time series mean R_a and the time series R_a CV. Similarly, there is a significant negative correlation (R^2=0.05, p<0.001) between the base-10 logarithm of the time series mean SL_a and the time series SL_a CV, but it is subject to more scatter than the correlation for R_a. Because of the dependence of the coefficient of variation and the mean (Eq. 4.2: Hazewinkel, 2001b), care should be taken when interpreting these correlations. The dependence of CV values on the mean indicates that the time series R_a and SL_a standard deviations do not increase proportionally with the time series R_a and SL_a, i.e larger mean R_a and SL_acorrespond to a proportionally smaller differences between the individual annual measurements. A similar relation between mean SL_a and the CV of SL_a measured on replicate plots was noted by Nearing et al. (1999), indicating that the observed variability between replicated plots has some similar characteristics compared to the variability observed between different years of measurements on the same plot, indicating randomness in both types of variability. Conversely, Vanmaercke et al. (2012b) observed a positive correlation between catchment time series mean sediment yield and the sediment yield CV (Fig. 4.6b). This difference indicates fundamental differences in the controlling factors of inter-annual variability between plot-measured SL_a and catchment sediment yield. With respect to inter-annual variability in plot-measured SL_a, the CV on plots with a small mean SL_a is mostly determined by whether or not a highly erosive event occurs within the year, resulting in a high inter-annual CV while for plots with large mean SL_a rates, such erosive events will occur more frequently (i.e. at least annually), causing SL_a rates to be more uniform throughout and hence causing inter-annual variability to be lower. For catchment sediment yield on the other hand, there is a relatively reliable baseflow and minimum sediment yield (i.e. consistent lower limit to individual annual sediment yield values; Vanmaercke et al., 2012b). Hence, when there are no additional disturbances in the catchment that result in higher mean annual sediment yield, catchments with a small mean annual sediment yield generally also have a small inter-annual variability of catchment sediment yield (Vanmaercke et al., 2012b). This outlines some important differences between the inter-annual variability of $SL_a$and inter-annual variability of sediment yield.

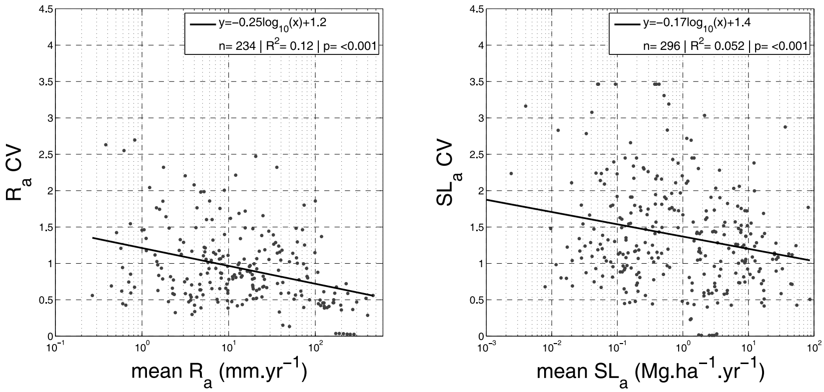


Figure 4.6a: Comparison between time series median annual runoff (R_a) and annual runoff coefficient of variation (R_a CV) and between time series median annual soil loss (SL_a) and annual soil loss coefficient of variation (SL_a CV). n= number of time series.

Climatic zone and land use type

The previous section outlines the importance of measuring period as a factor controlling the observed inter-annual variability in RC_a and SL_a time series. measuring period is mainly an experimental factor however, and several environmental factors may also have an effect on the inter-annual variability of runoff and soil loss plot measurements. A comparison of RC_a and SL_a time series CV and skewness between Mediterranean and non-Mediterranean plots (see Fig. 4.1), and between different land use types (as defined in Table 2.2) is given in Fig. 4.7a. A Kolmogorov-Smirnov (KS)-test (α=0.05) indicates no significant difference between Mediterranean and non-Mediterranean plots in the

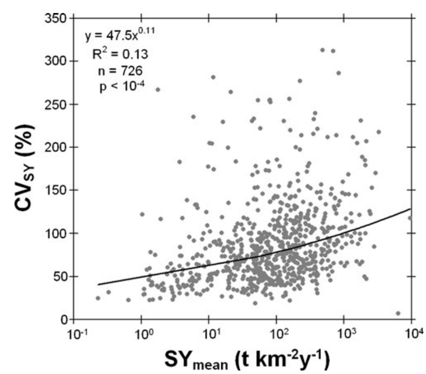


Figure 4.6b: Mean annual sediment yield (SY_{mean}) versus the coefficient of variation of the annual sediment yield values (CV_{SY}) (source: Vanmaercke et al., 2012b).

distribution of CV for both RC_a and SL_a time series. With repect to skweness, a significant difference between Mediterranean and non-Mediterranean plots was found for RC_a (p=0.015) but not for SL_a time series. The skewness of RC_a time series tends to be lower for non-Mediterranean plots compared to Mediterranean plots (Fig. 4.7a). This difference is mainly attributed to the relatively larger variation in P_a as well as a more seasonal rainfall pattern in the Mediterranean. Other differences between the Mediterranean and non-Mediterranean regions such as a more variable vegetation cover or higher soil rock fragment cover in the Mediterranean (Poesen and Lavee, 1994; Poesen et al., 1994; Yaalon, 1997) may also contribute to this effect however.

Pairwise KS-tests between each possible combination of different land use types showed a number of significant differences (α=0.0014 after Bonferroni correction) in the distributions of CV and to a lesser extent also skewness for both RC_a and SL_a time series (Fig. 4.7a). For most land use type combinations where a significant difference in CV or skewness was noted, there was also a significant difference in measuring periods for the same land use type combination however. As shown in Fig. 4.5, this difference in measuring periods can be the main cause for the observed difference in CV or skewness. This was not the case for the difference in RC_a CV between bare soil and vineyard plots (p=0.0008), which is attributed to the limited number of time series (n=5) for vineyards. There was also a significant difference in SL_a CV between cropland and grassland plots (p=0.0012), while the frequency distributions of the respective measuring periods were not significantly different. Grassland plots show high SL_a CV values (Fig. 4.7a), but generally low mean SL_a rates (Maetens et al., 2012b). This is consistent with the general trend observed in Fig. 4.6a. Nevertheless, no strong evidence for an effect of land use type on the CV and skewness of RC_a and SL_a time series was found.

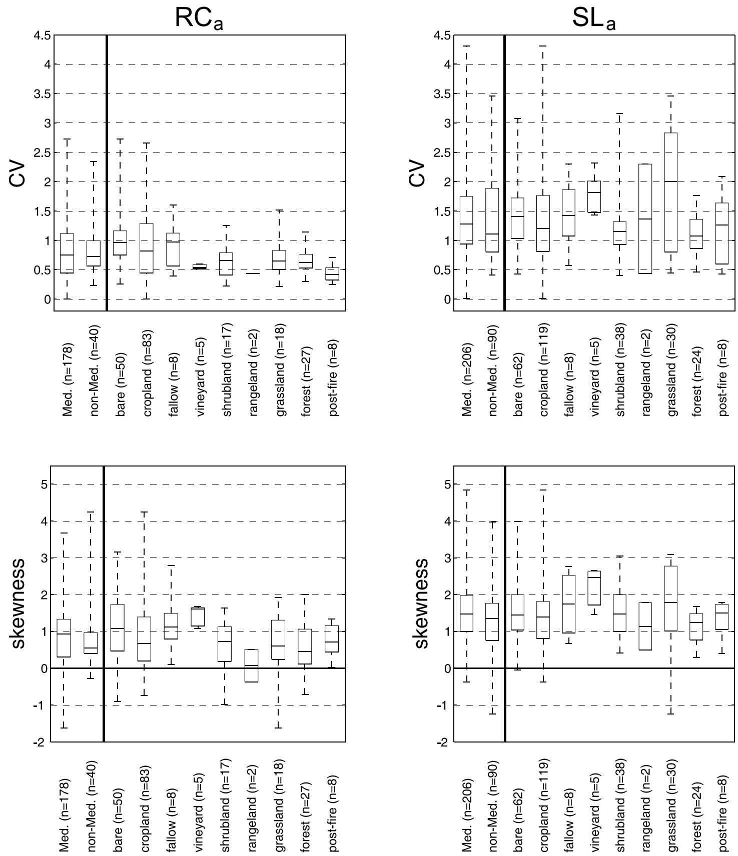


Figure 4.7a: Comparison between annual runoff coefficient (RC$_a$) and soil loss (SL$_a$) time series coefficients of variation (CV) and skewness for Mediterranean and non-Mediterranean plots, and between time series CV and skewness for different land use types (Table 2.2). The legend for the box plots in this graph is given in Fig. 4.7b. n= number of time series.

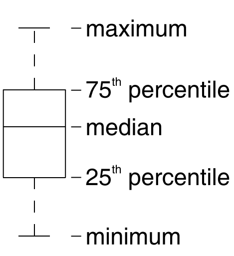


Figure 4.7b: Legend for the box plots used throughout this chapter

Plot length and slope gradient

The relation between plot length and slope gradient on the one hand and RC_a and SL_a time series CV on the other hand is given in Fig. 4.8. Most plots have slope lengths smaller than 25m and slope gradients between 5% and 10%, which complicates the assessment of the effect of slope length and plot slope gradient on the inter-annual variability of RC_a and SL_a. Nevertheless, there is a significant negative correlation between RC_a time series CV and both slope length and gradient, but the relation is subject to substantial scatter and the frequency distribution of plot length and slope gradients is biased towards shorter plots with smaller slope gradients (Fig. 4.8).

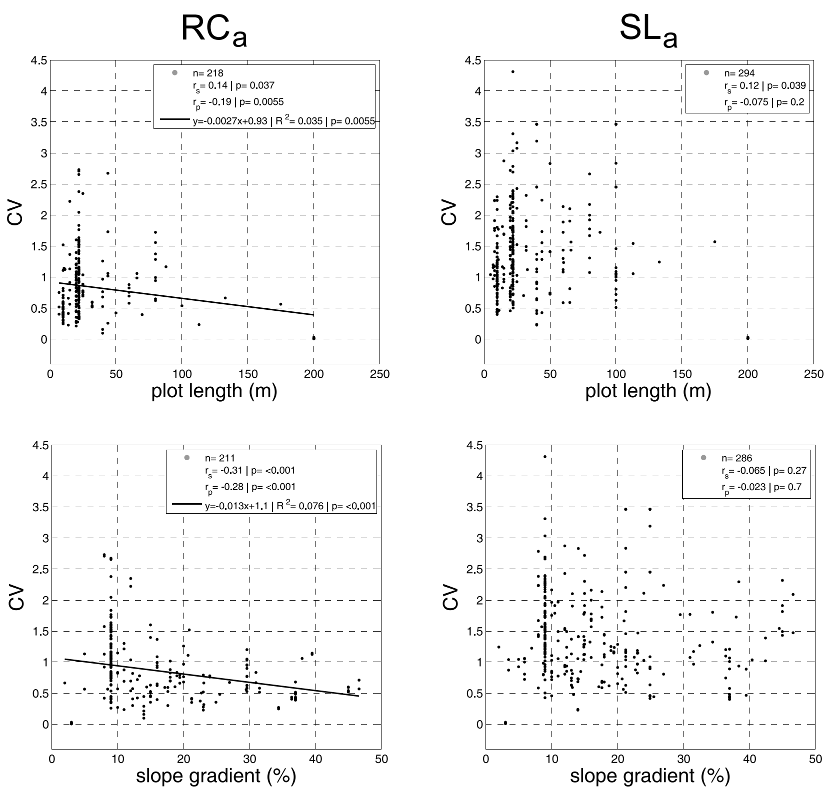


Figure 4.8: Relation between plot length and slope gradient and the time series coefficient of variation (CV) for annual runoff coefficient (RC_a) and annual soil loss (SL_a) time series. n= number of time series. r_s= Spearman's rank correlation coefficient, r_p= Pearson correlation coefficient.

The general effect of plot length and slope gradient on field-measured runoff is not well understood (Maetens et al., 2012b; Poesen and Bryan, 1989; Wischmeier, 1966), but a faster runoff response on steeper slopes and greater likelihood of flow convergence on longer slopes could cause a more consistent runoff response than on shorter slope where other factors like soil permeability and presence of cracks may play a more important role in runoff generation and cause the observed scatter in RC_a time series CV. With respect to SL_a time series CV, no consistent relations with plot length and slope gradient are observed. This is in line with findings by Bagarello et al. (2010a) who found no significant effect of plot length on the distribution of normalised event soil loss for plots with slope lengths of 11, 22, 33 and 44m. As no significant correlations between plot length or slope gradient and SL_a CV were found, no trendlines for these correlations are given in Fig. 4.8.

Annual precipitation and precipitation distribution

Significant negative correlations (α=0.05) between median time series P_a and both time series R_a CV and SL_a CV were found Fig. 4.9. However, visual interpretation of the data shows that R_a CV and SL_a CV first increase with increasing median P_a up to ca. 300mm·yr^{-1} median P_a, and then decrease with increasing median P_a. The (semi-)arid regions with median P_a <300mm·yr^{-1} are characterised by erratic rainfall, where R_a and SL_a are generally low in most years and hence have a small CV. As P_a increases to 400-500mm·yr^{-1} median P_a, $R_a$and SL_aalso occur more frequently and show a higher CV. The highest R_a CV and SL_a are observed in regions with median P_a values typical for a Mediterranean climate, which is characterised by a high inter-annual variability in P_a and a seasonal rainfall distribution within the year, and this variability persist in time series R_a CV and SL_a CV. For regions with higher median P_a (P_a>600mm·yr^{-1}), inter-annual variability in P_a tends to decrease and the distribution of rainfall within each years is more homogeneous.

Direct comparison of the time series P_a CV with time series R_a CV and with time series SL_a CV (Fig. 4.10), does not reveal significant correlations. This lack of correlations can be explained by the fact that SL_a values for individual years (and hence also time series SL_a CV) are strongly affected by the occurrence of exceptional events (i.e. large storms) within a year (de Figueiredo et al., 1998; Edwards and Owens, 1991; Larson et al., 1997; Poesen et al., 1996), which contribute proportionally more to total SL_a than to the total P_a for that year. At the catchment scale, Vanmaercke et al. (2012b) found better relations between inter-annual variation of P_a and catchment sediment yield.

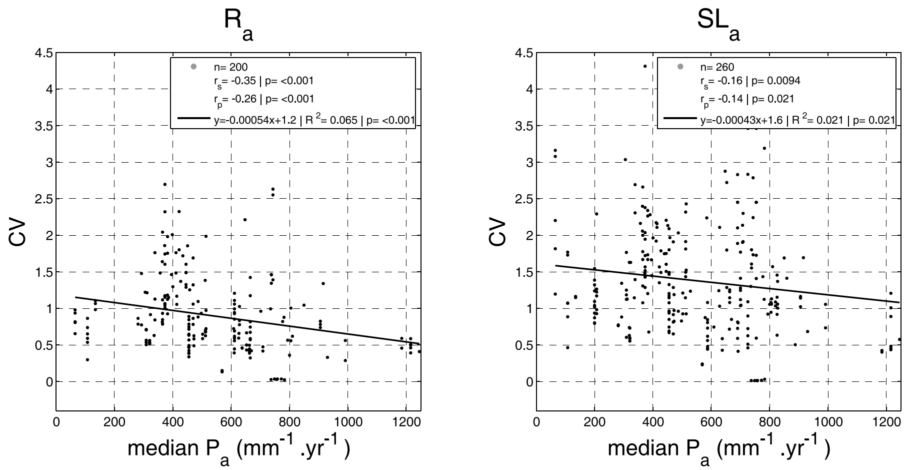


Figure 4.9: Relation between time series median annual precipitation (P_a) and the time series coefficient of variation (CV) for annual runoff and annual soil loss. n= number of time series. r_s= Spearman's rank correlation coefficient, r_p= Pearson correlation coefficient.

However, no clear effects of the distribution of event rainfall within the measuring period, as expressed by the PCI were found (Fig. 4.10). The relation between time series PCI and SL_a CV was found to be significant (p=0.001), but this can be attributed relatively high SL_a CV values for PCI<0.45, while the correlation as a whole is weak. The lack of relations between time series PCI on the one hand and R_a CV and SL_a CV on the other can be explained by (1) the fact that PCI calculations do not consider whether events are consecutive or occur during a period that the soil is vulnerable to runoff and soil loss (e.g. after tillage and before crop germination), (2) bad matches between time series P_d data extracted from the E-OBS database and the real time series P_d as occuring in the field (see also section 5.2.3), and (3) to the fact that extreme events can be highly localised and may not be well represented in the E-OBS database.

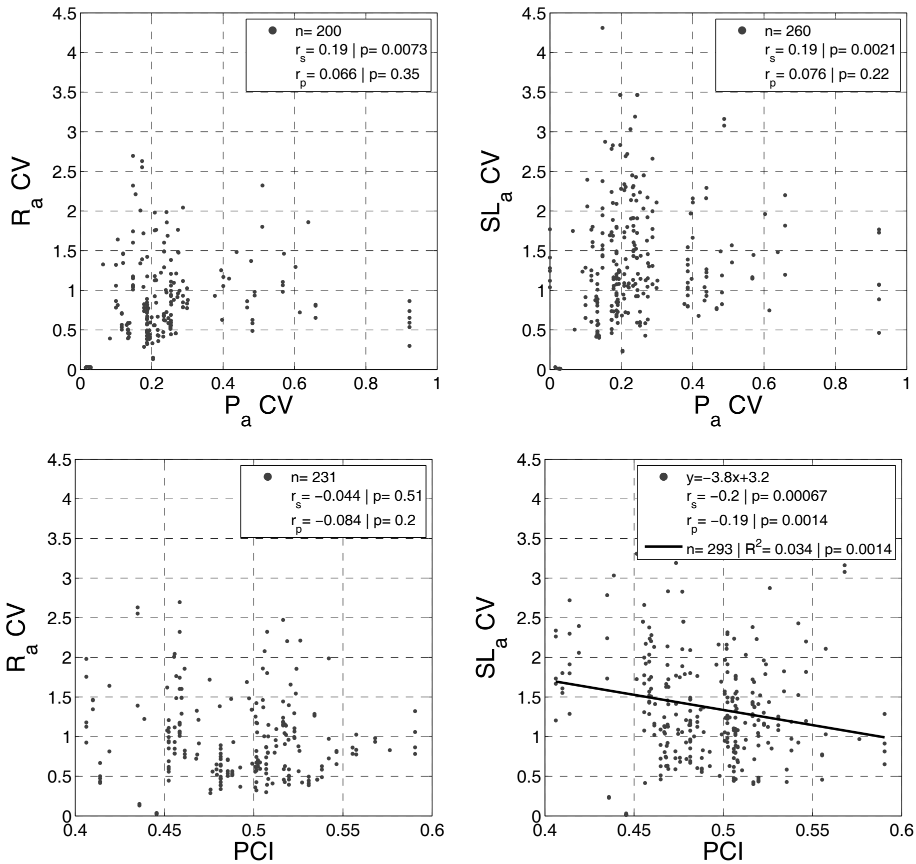


Figure 4.10: Upper: Relation between the coefficient of variation for time series of annual precipitation (P_a CV) and the coefficient of variation for time series of annual runoff (R_a CV) and annual soil loss (SL_a CV). **Lower:** Relation between the time series Precipitation Concentration Index (PCI; Martin-Vide, 2004)and R_a CV and SL_a CV. n= number of time series. r_s= Spearman's rank correlation coefficient, r_p= Pearson correlation coefficient.

Relation between inter-annual variability in runoff and soil loss

Significant correlations between time series R_a CV and SL_a CV (r_p= 0.68 , $p<$ 0.001), and between time series R_a skewness and SL_a skewness (r_p=0.53 , $p<$ 0.001) were found. Orthogonal regression analysis shows that 85% of observed variance in the coefficient of variation and 76% of observed variance in skewness can be explained by the orthogonal regression(Fig. 4.11). Time series SL_a CV are generally larger than the corresponding R_a CV, but this is less the case for skewness. Vanmaercke et al. (2012b) reported similar relations between inter-annual variability in catchment runoff and catchment sediment yield, indicating that inter-annual variability in runoff is an important erosion-controlling factor across different scales.

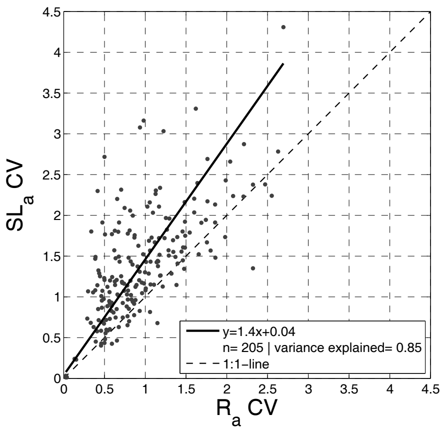


Figure 4.11: Left: relation between the coefficient of variation for time series of annual runoff (R_a CV) and the coefficient of variation for time series of annual soil loss (SL_a CV). **Right:** relation between the skewness for time series of annual runoff (R_a skewness) and the skewness for time series of annual runoff (SL_a skeweness). The thick black lines indicate the orthogonal regression equation. n= number of time series.

4.3.3 Uncertainty of mean plot-measured R_a and SL_a rates

As discussed in section 4.1, knowledge on the uncertainty of mean plot-measured R_a and SL_a rates and how well they represent long-term averages could greatly aid comparison of measured values between different studies and the validation of erosion models.

Results of the Monte Carlo analysis (Eq. 4.6, Eq. 4.7) performed on the Weibull distributions (Eq. 4.4) where a significant distribution fit (α=0.1) for RC_a (n=51) and/or SL_a (n=68) time series with a minimum measuring period of 10 years was found are given in Fig. 4.12. Each of the boxplots displays the 2.5 and 97.5 percentiles of the relative error on the mean for 1000 iterations of the Monte Carlo procedure described in section 4.2.2 for the 51 RC_a and 68 SL_a time series. For any given measuring period, the relative error on the mean tends to be larger for SL_a time series than for RC_a time series. This indicates that as precipitation causes runoff, and precipitation together with runoff cause soil loss, progressively more inter-annual variability is introduced in RC_a and SL_atime series, respectively. Furthermore, relative errors on the long-term mean RC_a converge faster to the long therm mean than relative errors on the long-term mean SL_a. Nevertheless, the convergence justifies the use of weighted mean values based on the square root of the measuring period when global mean R_a and SL_a are calculated (section 3.2, Maetens et al., 2012b,a).

Due to the right-tailed nature of the RC_a and SL_a frequency distributions, the measuring period mean will be more likely to underestimate the long-term mean RC_a and SL_a. However, over-estimations can be large (e.g. for a measuring period of one year: larger than ca. 230% for $RC_a$and larger than ca. 620% for $SL_a$for 25% of the time series, Fig. 4.12). The distribution of the 2.5 and 97.5 percentiles of the expected relative errors for a specific measuring period also differ strongly between different time series. Hence, a generally valid estimate of how long measuring periods should be for plot-measured averages to be reliable estimates of the long-term mean within a given relative error is difficult. For time series of RC_a, a measuring period of 5 years is enough for 75% of time series averages to fall within a 100% deviation of the estimated long-term average (with a confidence interval of 95%). For SL_a, a measuring period of 25 to 30 years is needed to achieve a relative error on the mean within 150% for 75% of the time series considered (with a confidence interval of 95%). The largest reduction in uncertainty on the long-term mean SL_aoccurs within the first 10 years however, after which longer measuring periods become progressively less cost-efficient for more accurate assessments of the long-term average SL_a. While these estimations of temporal variability in plot RC_a and SL_a are conservative, they do give an idea of the maximum deviation from the long-term mean RC_a and SL_a that can be expected for a given measuring period.

For time series of SL_a, relative errors on the mean that can be expected for any measuring period are larger than those observed for catchment sediment yield by Vanmaercke et al. (2012b), who found that for the 202 studied time series of catchment sediment yield, median 2.5 and 97.5 percentiles for relative errors on long-term mean catchment sediment yield after a measuring period of 10 years range between ca. -50% and +50%, respectively. For the 68 SL_a time series studied here, median 2.5 and 97.5 percentile for relative errors on long-term mean SL_arange between ca. -50% and +100%, respectively (Fig. 4.12).

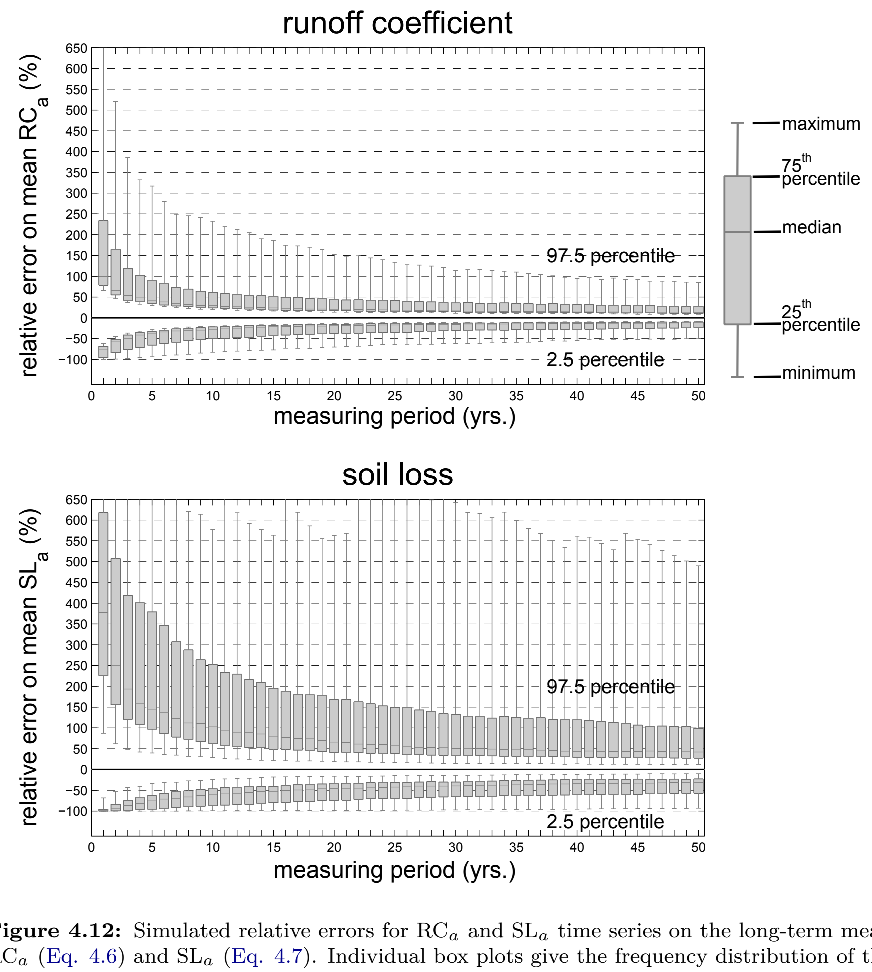


Figure 4.12: Simulated relative errors for RC_a and SL_a time series on the long-term mean RC_a (Eq. 4.6) and SL_a (Eq. 4.7). Individual box plots give the frequency distribution of the estimated 2.5 and 97.5 percentiles of the relative error for each RC_a (n= 51) and SL_a (n= 68) time series for the measuring period indicated.

Fig. 4.12 can be used to make an assessment of the likely deviation of the long-term mean RC_a and SL_a values from the plot-measured mean RC_a and SL_a values. However, some important caveats should be noted. The relative error ranges are only valid insofar the plot-measured RC_a and SL_a time series used in the Monte Carlo analysis are representative for the long-term climatic conditions on the plot-measuring stations, which may not always be the case given the limited measuring periods. In addition the random sampling procedure assumes that subsequent years of RC_a and SL_a measurements are not correlated to each other, which may not be the case. For instance, Wendt et al. (1985) showed that event soil loss magnitudes show the strongest correlations with events that are close in time to the event considered, although this effect is less likely to apply to annual plot data. Furthermore, the limited measuring periods also mean that there are considerable uncertainties with respect to the tails of the fitted distributions and caution should be taken when estimating extreme RC_a and SL_a values from these distributions.

4.4 Conclusions and recommendations

Assessment of the inter-annual variability of plot-scale R_a and SL_a values is an important and hitherto under-researched topic, especially with respect to flood prevention and soil and water conservation planning. In this study a quantification of the inter-annual variability of 234 R_a (representing 2523 plot-years) and 307 SL_a (representing 2359 plot-years) time series measured on runoff and soil loss plots at 62 plot measuring stations throughout Europe and the Mediterranean was made. The majority of measuring periods for the R_a and SL_a time series is shorter than 10 years, with a maximum of 30 years. Overall, inter-annual variability in both R_a and SL_a is often not-normal and has a right-tailed distribution, with coefficients of variation reaching up to 2.70 and 4.31, respectively. Time series R_a and SL_a CV, as well as R_a and SL_a skewness increase with increasing measuring period, illustrating the importance of a sufficiently long measuring period to make a correct assessment of R and SL variability.

The environmental factors evaluated in this study can only account for a limited fraction of the observed inter-annual variability in R_a and SL_a. Differences in R_a CV and SL_a between Mediterranean and non-Mediterranean plots and between different land use types are limited. Also the effect of plot length and slope gradient on R_a and SL_a CV is limited. While a negative correlation between median time series P_a on the one hand and annual R_a CV and SL_a CV was found, it could not be explained by inter-annual variability of P_a (as expressed by P_a CV), or daily precipitation concentration (PCI). This indicates that nor P_a CV nor PCI represent the important climatic factors that may explain inter-annual variability in R_a and SL_a well. While the occurrence of low frequency - high intensity events (i.e. large storms) plays a very important role in the temporal variability of R_a and SL_a (e.g. de Figueiredo et al., 1998; Edwards and Owens, 1991; Larson et al., 1997; Poesen et al., 1996), it was not possible to demonstrate this with the available data and individual storm data are likely needed.

Fundamental differences between inter-annual variability of plot-measured SL_a and the inter-annual variability of catchment sediment yield were noted. In this study, a negative relation between R_a CV, SL_a CV and mean time series R_a and SL_a was found, indicating an important random component in time series R_a CV and SL_a CV. Conversely, Vanmaercke et al. (2012b) found that relation between catchment sediment yield CV and mean value was positive. Runoff variability is an important factor explaining inter-annual variability in both plot-measured SL_a and catchment sediment yield, but the relation of P_a CV with plot-measured SL_a CV is weaker than the relation with catchment sediment yield CV. The latter is likely due to the fact that annual rainfall erosivity, rather than volume is an important controlling factor of SL_a.

Convergence of the relative error on the mean with increasing measuring period (Fig. 4.12) is faster for time series of R_a than for time series of SL_a and supports the use of the weighted mean calculation procedure for global mean R_a and SL_a values outlined in section 3.2. Ideally, the measuring period duration should be chosen based on an analysis of the likelihood of a representative number of extreme events occurring. Such analysis would require detailed data on the relation between the return periods of single storm rainfall depth and corresponding event soil loss rates (e.g Baffaut et al., 1998). However, insufficient data is available for such study on a continental scale. Alternatively, the measuring period on the plots could be extended until a sufficient number of erosive events is measured to obtain reliable mean R_a and SL_a values, but this is rarely the case due to logistic and financial constraints.

A Weibull distribution was found to provide the best fit for the time series of RC_a and SL_a, but differences with other evaluated distributions are small and different distributions may provide better fits in local studies. Results of the Monte Carlo analysis indicate that for any measuring period, there are large uncertainties on how well the the sample mean reflects the expected long-term average RC_a and SL_a, but relative errors are smaller for RC_a time series compared to SL_a time series (Fig. 4.12).

Chapter 5

Application of the runoff curve number method to predict annual runoff

5.1 Introduction

Compared to direct estimates of soil loss (SL) rates, the assessment and prediction of plot runoff rates have received considerably less attention in erosion studies (chapter 3). Yet, the prediction of runoff is an important component of many process-based erosion models (Merritt, 2003) and better prediction of runoff will lead to significant improvements of erosion models (e.g. Kinnell and Risse, 1998). Furthermore, accurate runoff estimation can contribute to a better prediction of the soil water balance and to a more effective local water management in agriculture (e.g. Rockström et al., 2010). Early applications of precipitation - runoff (P-R) modelling based on the rational method were mostly directed at engineering problems such as urban sewer design, drainage system design and reservoir spillway design (Todini, 1988). Further developments in P-R modelling focussed mainly on the prediction of catchment runoff, with special emphasis on the prediction of flood peaks, giving rise to several runoff models (e.g. Beven, 2012).

Two main approaches for modelling P-R relations have been followed (e.g. King et al., 1999; Wilcox et al., 1990), i.e.: (1) the use of the Green-Ampt infiltration model, in which R is estimated as precipitation (P) that can not infiltrate and (2) the SCS-Curve Number (CN) equation, which relates daily precipitation (P_d; which often serves as a proxy for the unknown event precipitation) directly to daily runoff (R_d) through an empirical model (Hawkins et al., 2009). Advantages of the CN method are that it is conceptually simple, the required model parameters (i.e. the Curve Number value: CN_D) can easily be selected from tables, and the only variable needed (i.e. P_d) is easily measurable or available from climatic records. Furthermore, it has been shown that it is possible for simple runoff models with few parameters to provide results that are as least as good as more complex models (e.g. Jakeman and Hornberger, 1993; Loague and Freeze, 1985).

Nevertheless, a disadvantage of empirical models such as the SCS-CN method is that the validity and accuracy of these models outside the region for which they were calibrated is unknown (Govers, 2010), and consistent validations of the SCS-CN method or published CN_D values that are applicable for large areas are rare outside the United States, and non-existent for Europe and the Mediterranean. Furthermore, scale effects in runoff processes have been demonstrated by several authors (Langhans et al., 2010; Leys et al., 2010; Shentsis et al., 1999) and as a general rule, runoff models should only be applied at the spatial scale for which they have been developed (e.g. Koren et al., 1999). The Curve Number method was initially developed for small catchments in the United States (NEH4, 2004), but the CN model has since been successfully fitted at the plot, field and small catchment scale for several land use types in different parts of the world, and CN values for those specific environments have been published (e.g. Auerswald and Haider, 1996; Boughton, 1989; Descheemaeker et al., 2008; Simanton et al., 1996; Wilcox et al., 1990). Simanton et al. (1996) performed an analysis of the effect of drainage area on the estimated CN values for 18 plots and semiarid catchments in Walnut Gulch, Arizona with a drainage area ranging between 0.00069 ha and 785.3 ha, and found that the estimated CN value was inversely related to catchment area. This negative relation was attributed to spatial variability in rainfall and infiltration losses in the larger catchments and shows that CN values are indeed scale dependent between the plot and catchment scales. However, these examples also show that the CN method approach is valid at smaller spatial scales than the catchment scale (i.e. the field and plot scale), but CN_D values may need adjustment at these scales.

Detailed plot-measured daily precipitation and runoff data representative for at least a full year are relatively rare for Europe and the Mediterranean, but a substantial number of annual precipitation (P_a) and annual runoff (R_a) plot data to assess the P-R relationship is available (Maetens et al., 2012b). In contrast to many widely applied erosion models that successfully predict annual SL (SL_a) (e.g. (R)USLE: Renard et al., 1997), no models have been developed to predict R_a, probably because the applications of hydrological models focus on the event scale (e.g. modelling of peak flows). Nevertheless, models predicting R_a could significantly contribute to the improvement of erosion models, as estimation of R is an important part of several erosion models (Merritt, 2003). Furthermore, as annual precipitation (P_a) and R_a data measured at the plot scale cover a wide range of conditions throughout Europe and the Mediterranean, an assessment of the important controlling factors controlling the annual rainfall-runoff (P_a-R_a) relation at a continental scale is possible, which is a prerequisite for a model that aims to be widely applicable. Assuming that factors controlling the P_a-R_a relationship will also be important controls of the daily rainfall-runoff (P_d-R_d) relationship, analysis of the P_a-R_a relationship at a continental scale can also yield important insights for the application of event-based models in areas where detailed daily or event runoff data for validation of the models is not available.

In this chapter, a comparison is made between the performance of two simple models to predict the P_a-R_a relation observed on runoff plots throughout Europe and the Mediterranean: linear regression and an adapted SCS-CN method. Furthermore, several factors that may have an important effect on the P_a-R_a relation are evaluated (i.e. concentration and distribution of daily rainfall within the year, hydrologic soil group of the plot, plot length, plot slope gradient and spatial variability), and the importance of these controlling factors on a continental scale is discussed.

5.2 Material and methods

5.2.1 Description of the annual rainfall and runoff dataset

Pairs of P_a and R_a observations were selected from the runoff plot database described in chapter 2. To obtain a good representation of the P_a-R_a relationship, only pairs of data corresponding to values for a measuring period (MP) of one single year were selected. P_a and R_a data averaged over multiple years were omitted as preliminary examination of the annual P_a-R_a relation in chapter 3 (Fig. 3.7, Fig. 3.8a) shows that the annual rainfall-runoff relation is possibly not linear. Averaging several P_a and R_a values observed on a plot where there is a non-linear annual rainfall-runoff relationship would result in pairs of averaged P_a and R_a that are not a true representation of the actual annual rainfall-runoff relationship. Hence, every pair of P_a and R_a data in this analysis corresponds to 1 plot-year (PY). A per-country overview of the literature sources and number of plot data is given in Table 5.1, the location of the plot measuring stations from which P_a and R_a were used is given in Fig. 5.1.

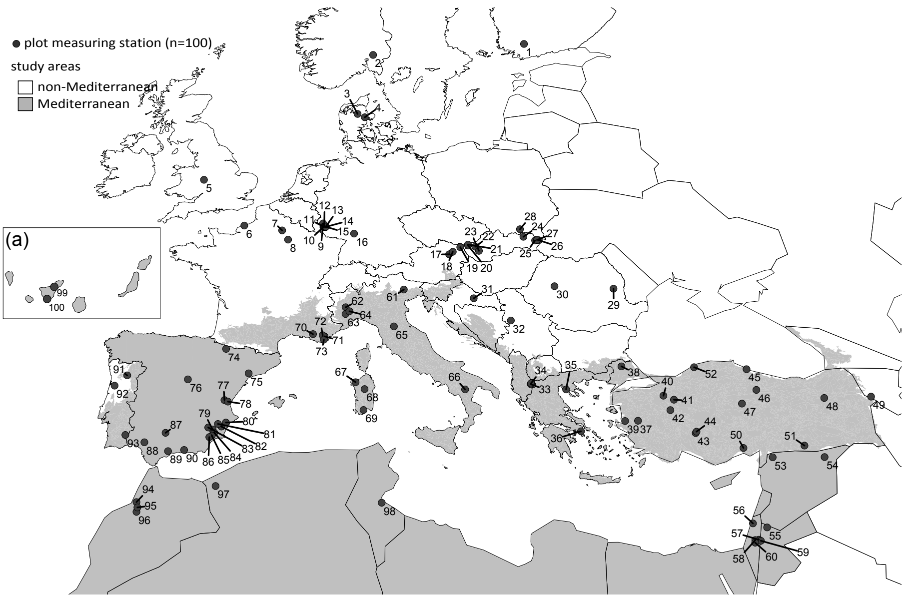


Figure 5.1: Geographical distribution of runoff and soil loss plot measuring stations (n=100) in Europe and the Mediterranean included in the analysis of relations between annual rainfall and annual runoff. Plot measuring station numbers correspond to the literature sources indicated in Table 5.1. The division between the climatic zones was derived from the LANMAP2 database (Metzger et al., 2005; Mücher et al., 2010). (a) Canary Islands.

Table 5.1: Overview of number of plots (#PL), number of plot-years (#PY), and plot data sources for pairs of annual precipitation and annual runoff data per country in Europe and the Mediterranean. Numbers between brackets refer to the plot measuring stations indicated in Fig. 5.1.

Country	#PL	#PY	Source
Albania	6	30	Grazhdani et al., 1996 [33-34]; Grazhdani, personal communication [33-34]
Algeria	24	80	Mazour, 1992 [97]; Morsli et al., 2004 [97]; Mazour et al., 2008 [97]
Austria	6	33	Klik, 2003 [17-19]; Klik, 2010 [17-19]; Klik, personal communication [17-19]
Croatia	4	8	Basic et al., 2001 [17-19]; Basic et al., 2004 [31]
Denmark	10	29	Schjønning et al., 1995 [3-4]
Finland	12	40	Turtola and Paajanen, 1995 [1]; Turtola et al., 2007 [1]; Uusi-Kämppä, 2005 [1]
France	26	139	Ballif, 1989 [8]; Clauzon and Vaudour, 1969 [70]; Clauzon and Vaudour, 1971 [70]; Le Bissonnais et al., 2004 [6]; Martin, 1990 [73]; Martin et al., 1997 [73]; Viguier, 1993 [71-72]; Wicherek, 1986 [7]; Wicherek, 1991 [7]
Germany	18	42	Dikau, 1983 [16]; Dikau, 1986 [16]; Richter, 1980 [15]; Richter, 1985 [9-14]
Greece	30	42	Diamantopoulos et al., 1996 [35]; Kosmas et al., 1996 [36]
Israel	25	128	Inbar et al., 1997 [56]; Inbar et al., 1998 [56]; Lavee et al., 1998[57-59]; Lavee, personal communication [57-59]
Italy	56	235	Basso et al., 1983a [66]; Basso et al., 1983b [66]; Bini et al., 2006 [61]; Caredda et al., 1997 [67]; De Franchi and Linsalata, 1983 [66]; Ollesch and Vacca, 2002 [69]; Porqueddu and Roggero, 1994 [67]; Postiglione et al., 1990 [66]; Rivoira et al., 1989 [68]; Tropeano, 1984 [62-64]; Vacca et al., 2000 [69]; Vacca, personal communication [69]; Zanchi, 1983 [65]; Zanchi, 1988a [65]; Zanchi, 1988b [65]
Jordan	2	2	Abu-Zreig, 2006 [55]; Abu-Zreig et al., 2011 [55]
Morocco	16	75	Laouina et al., 2003 [94]; Yassin et al., 2009 [95-96]; Yassin, personal communication [95-96]
Norway	5	25	Øygarden, 1996 [2]
Palestinian territories	7	17	Al-Seekh and Mohammad, 2009 [60]; Mohammad and Adam, 2010 [60]
Poland	4	53	Gil, 1986 [28]; Gil, 1999 [28]
Portugal	28	208	de Figueiredo and Gonçalves Ferreira, 1993 [91]; de Figueiredo and Poesen, 1998 [91]; de Figueiredo, personal communication [91]; Roxo et al., 1996 [93]; Shakesby et al., 1994 [92]
Romania	5	99	Ionita, 2000 [29]; Nistor and Ionita, 2002 [30]
Serbia	2	6	Sekularac and Stojiljkovic, 2007 [32]
Slovakia	68	76	Chomanicová, 1988 [25-26]; Fulajtár and Janský, 2001 [20-24]; Suchanic, 1987 [27]
Spain	78	374	Andreu et al., 1998a [78]; Andreu et al., 1998b [78]; Campo et al., 2006 [77]; Castillo et al., 1997 [83]; Castillo et al., 2000 [81]; Cerdà and Lasanta, 2005 [74]; Chirino et al., 2006 [80]; Durán Zuazo et al., 2004 [90]; Durán Zuazo et al., 2008 [90]; Francia Martínez et al., 2006 [90]; García-Ruiz et al., 1995 [74]; Gimeno-Garcia et al., 2007 [77]; Gómez et al., 2004 [87]; Gómez et al., 2009 [88]; Gómez Plaza, 2000 [81]; Guerra et al., 2004 [99-100]; Lasanta et al., 2006 [74]; Lopez-Bermudez et al., 1991 [79]; Martínez-Murillo and Ruiz-Sinoga, 2007 [89]; Nadal Romero and Lasanta, personal communication [74]; Rodríguez Rodríguez et al., 2002 [99-100]; Romero-Díaz et al., 1999 [79]; Romero-Díaz and Belmonte Serrato, 2008 [84-86]; Rubio et al., 1997 [78]; Sanchez et al., 1994 [82]; Soler et al., 1994 [75]; Williams et al., 1995 [76]
Syrian Arab Republic	5	12	Bruggeman et al., 2005 [53]; Shinjo et al., 2000 [54]; Masri et al., 2005 [53]
Tunisia	4	4	Bourges et al., 1973 [98]; Bourges et al., 1975 [98]
Turkey	96	1187	Kara et al., 2010 [52]; Köse and Taysun, 2002 [37]; Köse et al., 1996 [37]; Oguz et al., 2006 [38-51]; Oguz, personal communication [38-51]
United Kingdom	20	109	Fullen, 1992 [5]; Fullen and Booth, 2006 [5]; Fullen and Reed, 1986 [5]; Fullen et al., 2006 [5]
Total	**557**	**3053**	

In order to be able to better compare the results of this study for the different land uses and crop types with the different Hydrologic Soil-Cover Complexes used in the SCS-CN method (NEH4, 2004, Table 9-1), the land use types as defined in chapter 2 were reclassified as indicated in Table 5.2. Cropland plots with crops not fitting in one of the crop categories defined in the SCS-CN method (NEH4, 2004, Table 9-1) are not included in the analysis (Table 5.2). Due to the limited number of plots at construction sites, these were not included in this analysis. While the 'Woods-grass combination (orchard or tree farm)' in NEH4 (2004), Table 9-1 is similar to the 'tree crops' land use type (Table 5.2), no comparison between the tree crops land use type and the 'Woods-grass combination' Hydrologic Soil-Cover Complex is made as none of the plots considered in the tree crops land use type has any grass cover.

Table 5.2: Reclassification of land use types described in chapter 2 to match Hydrologic Soil-Cover Complexes (NEH4, 2004).

chapter 2 land use type	**chapter 3 crop**	**This chapter land use type**	**Hydrologic Soil-Cover Complex cover type**
bare		bare	Fallow (bare soil)
fallow		fallow	-
cropland	cereals	cereals	Small grain (straight row)
	maize	row crops	Row crops (straight row)
	sunflower	row crops	Row crops (straight row)
	sugar beet	row crops	Row crops (straight row)
	potato	row crops	Row crops (straight row)
	leguminous	leguminous	Close-seeded or broadcast legumes or rotation meadow (straight row)
	rotation	rotation	-
	other	*not included*	-
tree crops		tree crops	Woods-grass combination (orchard or tree farm)
vineyards		vineyards	-
grassland		grassland	Meadow-continuous grass, protected from grazing and generally mowed for hay
rangeland		rangeland	Pasture, grassland or range- continuous forage for grazing
forest		forest	Woods
shrubland/matorral		shrubland	Brush-brush-forbs-grass mixture with brush the major element
post-fire		post-fire	-
construction sites		*not included*	-

5.2.2 Factors controlling the annual precipitation - annual runoff relation

Several environmental factors are known to control the precipitation-runoff relation. For instance, apart from the effect of land use, also the effect of soil texture is included in the SCS-CN method through the hydrologic soil group (HSG: Hawkins et al., 2009). Furthermore, also rainfall distribution throughout the year (e.g. Kirkby et al., 2005), plot length (e.g. Langhans et al., 2010; Leys et al., 2010; Poesen and Bryan, 1989; Wischmeier, 1966), and plot slope gradient (e.g. Poesen, 1984; Wischmeier, 1966) have been shown to control the precipitation-runoff relation. Therefore, data on these factors was collected to analyse their effect on the P_a-R_a relation.

Determination of Hydrologic Soil Groups (HSG)

The HSG concept is used to account for the effects of differences in soil texture on the P_d-R_d relation. Four different HSG groups are defined based on the USDA soil texture class (Table 5.3) and CN-values for each of these HSG groups are given for each cover type (NEH4, 2004, Table 9-1). Thus each 'Hydrologic Soil – Cover Complex' is associated with a CN-value describing the P_d-R_d relation.

Table 5.3: Definition of the Hydrologic Soil Groups (HSG) based on soil texture (Source: Hawkins et al., 2009).

USDA soil texture	HSG
Sand, loamy sand, sandy loam	A
Silt loam or loam	B
Sandy clay loam	C
Clay loam, silty clay loam, sandy clay, silty clay or clay	D

The HSG for each of the individual pairs of P_a and R_a data was determined by using (in order of preference according to data quality); (1) published soil texture data (% clay, % silt, % sand), (2) the soil texture class, soil type or other soil information provided by the author(s) of the publication, (3) from other runoff plots at the same plot measuring station, or (4) soil texture at the plot measuring station derived from the European Soil Database (ESDB, 2004; Panagos, 2006; Panagos et al., 2012). Soil texture data (% clay, % silt, % sand) were available for 2 364 (77.15%) of the pairs of P_a and R_a data. For these data pairs, the USDA soil texture class was determined (Fig. 5.2) and

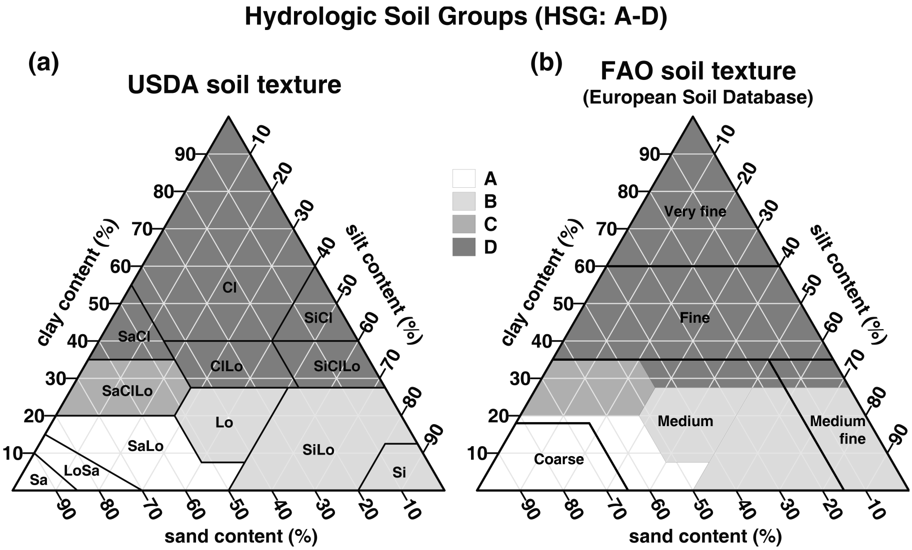


Figure 5.2: Soil texture classification according to (a) the USDA system and (b) the FAO system used in the European Soil Database (ESDB, 2004; Panagos, 2006; Panagos et al., 2012), with indication of the Hydrological Soil Groups (A through D). Cl: clay, SiCl: silty clay, SaCl: sandy clay, ClLo: clay loam, SiClLo: silty clay loam, SaClLo: sandy clay loam, Lo: loam, Sa: sand, LoSa: loamy sand, SaLo: sandy loam, SiLo:silty loam, Si: silt.

the corresponding HSG was assigned (Table 5.3). For another 227 (7.41%) P_a-R_a pairs, no detailed soil texture data was reported, but enough details (e.g. soil texture class or soil type and description) was provided by the author to determine the HSG. For 375 (12.24%) of the remaining P_a-R_a pairs, HSG was determined by using the same HSG as that for other plots or soil outside the plots at the same plot measuring station. For 63 (2.06%) of the remaining P_a-R_a pairs, HSG could be derived unambiguously from the European Soil Database texture at the plot location (i.e. the texture was either Very fine or Fine and HSG D was assigned, or Coarse and HSG A was assigned). For 35 (1.14%) P_a-R_a pairs, no HSG could reliably be determined, and these data were omitted from the analyses of the effects of HSG on P_a-R_a relations.

Rainfall distribution throughout the year

Few publications report complete series of daily rainfall depth throughout the years of measurement, or report only the rainfall events causing runoff. Therefore, time series of daily precipitation were extracted from the E-OBS gridded dataset (Haylock et al., 2008), which contains daily precipitation from 1950 to 2006, interpolated at a 25 km resolution. Only time series for which there were no missing data during the full MP for each pair of P_a and R_a data were selected. The correspondence between the individual P_a data observed on each plot and the sum of all daily precipitation values for the same MP extracted from the E-OBS database at the location of that plot is given in Fig. 5.3.

There is a clear correspondence between the precipitation time series obtained from the E-OBS database and the precipitation measured at the plots (Fig. 5.3). Nevertheless, the substantial deviations from the 1:1 line observed for the P_a data in Fig. 5.3 indicate that also the daily precipitation data obtained from the E-OBS time series can not be used as accurate approximations of the actual daily precipitation depth at the plot measuring stations. However, the daily

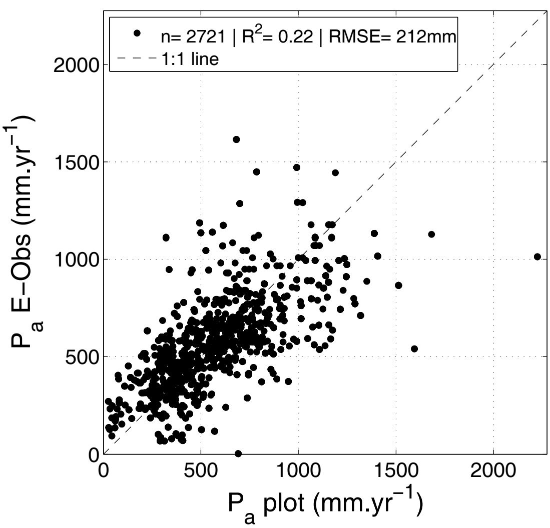


Figure 5.3: Correspondence between the individual P_a data observed on each plot (P_a plot) and the sum of all daily precipitation values for the same MP extracted from the E-OBS database at the location of that plot (P_a E-OBS). n= number of pairs of P_a-R_a data for which a complete time series of daily precipitation could be extracted from the E-OBS database for the entire MP. R^2 and RMSE: R-squared and Root Mean Square Error with respect to the 1:1-line.

precipitation distribution, rather than the daily precipitation depth, obtained from the E-OBS database may still be used as a description of climatic conditions at the plot measuring stations during the measuring period. Therefore, the daily Precipitation Concentration Index (PCI) derived from the Gini concentration index was calculated for each of the selected time series of daily precipitation, following the procedure described by Martin-Vide (2004). For more details on the calculation of this index, see section 4.2.1.

Originally, the Antecedent Moisture Content (AMC), and later, the Antecedent Runoff Condition (ARC), was also considered as a factor in the SCS-CN method, but its use has been the subject of debate (Grabau et al., 2006; Hawkins et al., 2009; NEH4, 2004). Furthermore, AMC and ARC rely on the concept that rainfall and soil conditions in the days before the event have an influence on the P_d-R_d relation. While it has been shown that initial soil moisture content at the start of an event affects the runoff response (Govers et al., 1990), this concept is probably less useful at an annual scale, as specific rainfall and soil conditions in the previous year or years will have little influence on the P_a-R_a relation in the year that runoff measurements are taken. For these reasons, the AMC and ARC concepts were not further explored in this research.

Other factors

Data on plot length, plot slope gradient and soil organic matter content (SOM) was collected from the reviewed literature and the effect of these environmental and experimental factors on the P_a-R_a relation was assessed. While several other factors such as vegetation cover (Descheemaeker et al., 2008, e.g.) or rock fragment cover (e.g. Poesen et al., 1990; Smets et al., 2011a) have been shown to control the P_a-R_a relation, not enough data on these factors was available to assess their effect. Nevertheless, these environmental conditions can be assumed to be relatively homogeneous at the individual plot measuring stations. To account for the variability in the P_a-R_a relation that can be attributed to these differences between plot measuring stations, the plot measuring station was designated as a random effect factor in the mixed models that were used to assess the P_a-R_a relation (cfr. section 5.2.3).

Furthermore, a distinction was made between Mediterranean (i.e. the Mediterranean and Anatolian climatic zones) and non-Mediterranean plots (i.e. the Atlantic, Boreal, Continental, Steppic and Alpine climatic zones) based on plot measuring station location (Fig. 5.1) to evaluate whether spatial differences in plot locations have an effect on the P_a-R_a relation.

5.2.3 Model equations

Two different models were used to describe the P_a-R_a relationship observed on the runoff plots (Fig. 5.1, Table 5.1):

Linear mixed effects model

For each combination of land use type Table 5.2 and HSG Table 5.3 for which more than 5 observations were available, a linear mixed model (McLean et al., 1991) with P_a designated as a fixed effect and the plot measuring station designated as a random effect on the intercept was fitted using the "lme4" package in the R software (Bates et al., 2012). The effect of the different HSG was accounted for by including a dummy variable:

$$R_a = \delta_A (m_A P_a + b_A) + \delta_B (m_B P_a + b_C) + \delta_C (m_C P_a + b_C)$$
$$+\delta_D (m_D P_a + b_D) + Zu + \epsilon \quad \text{for} \quad P_a > P_{T,HSG} \qquad \text{(Eq. 5.1a)}$$

$$R_a = 0 \quad \text{for} \quad P_a \leq P_{T,HSG} \qquad \text{(Eq. 5.1b)}$$

$$P_{T,HSG} = \frac{-b_{HSG}}{m_{HSG}} \qquad \text{(Eq. 5.1c)}$$

Where m_{HSG} and b_{HSG} are respectively the regression slope and intercept for the different HSG. $P_{T,HSG}$ is the threshold annual precipitation for the different HSG (i.e. the P_a depth below which no R_a is expected, Eq. 5.1c.), δ_{HSG} are dummy variables for the different HSG which take on a value of 1 for the HSG corresponding to the observation and 0 otherwise. u is the random effects (i.e. plot measuring station) vector with regressor matrix Z, and ϵ is the error term.

To be physically consistent, the concept of P_T is included in Eq. 5.1a and Eq. 5.1b. Pairs of $P_a - R_a$ data where R_a= 0 mm·yr^{-1} can be considered left-censored. However, the censoring applies at the level of the individual storm and not at the annual level. Applying a censored linear mixed effects on the annual data resulted in regression fits with implausible intercepts. Hence, pairs of $P_a - R_a$ data where R_a= 0 mm·yr^{-1} were omitted from the dataset as their inclusion (without censoring) was found to have little effect on the fitted regression slope values.

As the calculation and interpretation of an R^2 statistic is not straightforward for a model with mixed effects (Edwards et al., 2008), a different approach to obtain a goodness-of-fit statistic was used. Furthermore, the residuals of the regression fits were found to be heteroskedastic, which does not affect the estimation of regression parameters but precludes inference based on variance estimates (Wooldridge, 2013). Therefore, a Markov Chain Monte

Carlo bootstrap procedure (R software package “languageR”; Baayen, 2011) was applied to evaluate the significance of the individual regression parameters. Only regression slopes for the HSG that were found to be significant at α=0.05 were further analysed. To check whether slopes for the different HSG are significantly different from each other ($\alpha = 0.05$), a pair-wise Mann-Whitney U test was then applied on the bootstrap slope samples for all combinations of regression slopes for different HSG. To correct for family-wise error in the multiple comparisons between different HSG, a Bonferroni correction (Abdi, 2007) was applied by dividing the overall significance level ($\alpha = 0.05$) by the total number of paired comparisons.

Quantile regression (e.g. Cade and Noon, 2003) was applied to further assess the variability in regression slopes for the different land use types. In quantile regression, different quantiles of the response variable are fitted in stead of the mean. Therefore, a linear quantile mixed model with the same model specifications as the linear mixed model in Eq. 5.1a was fitted using the R software package “lqmm” (Geracin, 2012).

modified SCS-Curve Number method

Applying the SCS-CN method to annual data would require summation of the individual daily SCS-CN expressions (Hawkins et al., 2009; NEH4, 2004) for all rainy days in a full year (Eq. 5.2):

$$R_a = \sum_{i=1}^{n} R_{d,i} = \sum_{i=1}^{n} \frac{(P_i - \lambda S_D)^2}{P_i + (1-\lambda)\, S_D} \times (P_i > \lambda S_D) \qquad \text{(Eq. 5.2)}$$

Where n is the number of days with precipitation in the full year, $R_{d,i}$ is the total daily runoff and $P_{d,i}$ is the total daily precipitation for the i^{th} rainy day, S_D is the dimensionless S-value derived from CN_D values as described in (Eq. 5.3), λ is a dimensionless parameter, and $(P_i > \lambda S_D) = 1$ if true and 0 if false.

For an easy interpretation and representation in tables, the daily S-values used in the SCS-CN method (S_D) are converted to daily CN-values (CN_D) according to the following transformation (Hawkins et al., 2009):

$$CN_D = \frac{25400}{254 + S_D} \qquad \text{(Eq. 5.3)}$$

Where CN_D and S_D are the daily CN and S numbers, respectively.

CN_D values are restricted to values ranging between 30 and 100. Hence, CN_D and S_D values are inversely related.

Subsequently, Eq. 5.2. would need to be solved for $\sum_{i=1}^{n} P_{d,i} = P_a$ so as to allow the use P_a data to predict R_a. However, Eq. 5.2 can not be solved without P_d data and hence, the SCS-CN method can not be applied when only P_a data are available.

Nevertheless, while preliminary analysis of the P_a-R_a relation in chapter 3 (Fig. 3.8a) shows substantial scatter precluding the identification of the precise form of the P_a-R_a relation, the data show a curvilinear trend of stronger increase in R_a with increasing P_a for most land use types (Fig. 3.8a). This type of curvilinear relation is similar to that observed for the P_d-R_d relation (e.g. Descheemaeker et al., 2008), which is described by the SCS-CN method. This suggests that, after re-parametrisation and re-interpretation of the model concepts, the equation structure of the SCS-CN method may be more useful to describe the P_a-R_a relation than using a linear model.

Therefore, a non-linear mixed effects model was fitted to assess the P_a-R_a relation for all combinations of land use type and HSG for which more than 10 observations are available. Designating the plot measuring station as a random effect factor, the following model was fitted using the R software package “nlme” (Pinheiro et al., 2013):

$$R_a = \frac{(P_a - \lambda(S_A + S_r))^2}{P_a + (1-\lambda)(S_A + S_r)} \quad \text{for} \quad P_a > \lambda(S_A + S_r) \qquad \text{(Eq. 5.4a)}$$

$$R_a = 0 \quad \text{for} \quad P_a \leq \lambda(S_A + S_r) \qquad \text{(Eq. 5.4b)}$$

Where λ is a dimensionless parameter, S_A is the annual S-value, and S_r is the random effect parameter of the plot measuring station.

Eq. 5.4a and Eq. 5.4b are equal to the SCS-CN equation (Hawkins et al., 2009; NEH4, 2004), but use P_a and R_a as variables and S_A as the fitted regression parameter. For some land use types, the joint fitting of all HSG using dummy variables did not lead to convergence on the parameter estimations and hence the combinations of land use type and HSG were fitted individually.

The λ-value in the SCS-CN method relates the S-value to the initial abstraction (I_a) through:

$$I_a = \lambda \times S_D \qquad \text{(Eq. 5.5)}$$

Where the initial abstraction (I_a) is the precipitation depth that is required for the initiation of runoff, i.e. a precipitation threshold for each event.

Applied to annual data, the concept of an I_a value needs to be reconsidered. In addition, λ is set to a fixed 0.2 value in the SCS-CN method, but several sources report better results for other values of λ (e.g. Descheemaeker et al., 2008; Woodward et al., 2003). Therefore, in a first exploration, the effect of different λ values was evaluated by fitting Eq. 5.4a to the P_a and R_a data for each land use type, without accounting for the random effect of the study site, but with optimization for both λ and S_A, and with λ as a fixed parameter with values of 0.2, 0.05 and 0. S_A values were then converted to CN_A values using the same transformation as in the SCS-CN method (Eq. 5.2). For each of these model fits, the model efficiency (ME) was calculated as:

$$ME = 1 - \frac{\sum_{i=1}^{n} \left(R_{o,i} - R_{m,i}\right)^2}{\sum_{i=1}^{n} \left(R_{o,i} - \overline{R_o}\right)^2} \qquad \text{(Eq. 5.6, Nash and Sutcliffe, 1970)}$$

Where ME is the model efficiency, n is the total number of observations (i.e. P_a-R_a pairs), $R_{o,i}$ is the i^{th} observation of R_a, $R_{m,i}$ is the modelled R_a for the i^{th} observation, and $\overline{R_o}$ is the mean value of all n observed R_a. Negative ME values indicate the model performs worse than taking the mean of all observations as predicted value.

It should be clearly stated that the standard SCS-CN method (Hawkins et al., 2009; NEH4, 2004) should only be used to predict R_d values from P_d data. However, the use of a curvilinear CN-type of equation (Eq. 5.4a and Eq. 5.4b to examine the P_a-R_a relationship has the advantage that a substantial amount of literature is available on the strengths, weaknesses and interpretation of the method. Also, by using the CN-equation, the P_a-R_a relationship is described by a single parameter (i.e. the S_A- or CN_A- value), provided that λ is kept constant. Hence a CN-type model is less complex compared to the two-parameter (i.e. slope and intercept) linear model. Less complex models can have significant advantages in hydrologic modelling (e.g. Jakeman and Hornberger, 1993).

5.2.4 Relation between annual and daily rainfall-runoff relations

Further exploration of the relation between annual and daily rainfall-runoff relations requires detailed records on daily rainfall and runoff (as a proxy for event rainfall and runoff), including both runoff generating and non-runoff generating events, very few of which were reported in the literature. As remarked in section 5.2.2, the correspondence between measured P_a and annual rainfall data from the E-OBS database is too small to reliably use the E-OBS precipitation series for comparison with measured data. To obtain an assessment of the variability in the P_a-R_a relation that can be explained by the precipitation distribution during the year, pairs of P_a-R_a data were simulated for all plot measuring stations in the database (see chapter 2) in the area covered by the E-OBS database.

For each daily rainfall value in the time series, sets of corresponding daily runoff were calculated. A range of CN_D values between 30 and 100 was selected, and the corresponding S_D-value for λ=0.05 (S_D) (Hawkins et al., 2009) were calculated. Using the curve number method (NEH4, 2004), these S_D were then used to generate daily R values for the time series of daily precipitation values extracted from the E-OBS database. The resulting daily precipitation and runoff values for different CN_D were then summed to annual values and the annual P_a-R_a relation was modelled using a modified CN model (Eq. 5.7):

$$R_a = \frac{(P_a - k_1 S_D)^2}{P_a + k_2 S_D} \qquad \text{(Eq. 5.7)}$$

Where S_D is the S-value obtained from the CN_D as defined by the SCS-CN method (NEH4, 2004) and k_1 is a regression parameter relating the S_D value to the initial abstraction (i.e. amount of P_a that does not cause runoff) and k_2 is a regression parameter relating the daily S-value (S_D) to the annual S-value (S_A).

Subsequently, the relation between the P_d-R_d relation (i.e. CN_D values) and the P_a-R_a relation (i.e. CN_A values) is examined.

5.3 Results and discussion

5.3.1 Linear mixed effects model

Fitted regression parameters for the linear mixed model to the different combinations of land use types and HSG (Eq. 5.1a) are given in Table 5.4. A graphical representation of the fitted equations, along with the results of the linear mixed quantiles regression is given in Fig. 5.4. Combinations of land use type and HSG for which the slope parameter was not significant at (α=0.05) according to the the Markov Chain Monte Carlo bootstrap procedure (section 5.2.3) are not given in Table 5.4 and Fig. 5.4, as no meaningful relation between P_a and R_a can be obtained. For Tree crops, none of the fitted HSG slope parameters was significant at α=0.05. Note that the model specification in Eq. 5.1a implies a random effect on the intercept only, i.e. the differences between different plot measuring stations are explained by differences in the intercept only, and regression slopes are identical for all plot measuring stations with the same combination of land use type and HSG. This model specification was chosen so as to assess a single P_a-R_a relation for each combination of land use type and HSG on all plot measuring sites.

The effect of HSG on the P_a-R_a relation is complex Fig. 5.4. For bare plots, regressions for HSG A, HSG B, and HSG C show the expected relative positions of stronger runoff response for HSG A through C, while the runoff response (i.e. regression line) for HSG D is lower than all the other HSG. For the other land use types, regression slopes for HSG B and C show different trends relative to each other depending on the land use. Apart from uncertainty in the determination if the correct HSG based on the procedure outlined in section 5.2.2, this may also indicate that the distinction between these 4 HSG (Fig. 5.2) may not be optimal for soil and climate conditions in Europe and the Mediterranean. While few of the regression slopes for HSG A were significant, runoff response for HSG A is generally small, but high runoff coefficients on sandy soils do occur (e.g. in vineyards). These are most likely attributed to other factors such as steep slopes and high connectivity in vineyards. A Mann-Whitney U test (Gibbons, 1985) performed on the Markov Chain Monte Carlo bootstrapped slope samples for each land use type shows that for all pair-wise combinations of two HSG with a significant slope fit ($\alpha = 0.05$), median slopes were also significantly different from the other HSG in the same land use type at α=0.05 (after Bonferroni-correction by dividing α by the number of pair-wise combinations in each land use type). These results were relatively robust against changes in the number of Markov Chain Monte Carlo Bootstrap simulations, with only the difference between HSG A and HSG C for leguminous plots becoming not significant (p= 0.61) when the number of bootstrap samples was reduced to 100.

A remarkable observation for bare plots and the land use types with crop cultivation (i.e. cereals, row crops, leguminous, and rotation) is the low runoff response for HSG D with respect to the other HSG. This is attributed to two very different behaviours of the fine-textured soils in the HSG D (Fig. 5.2). Clayey soils generally have a low hydraulic conductivity (e.g. Saxton and Rawls, 2006) and are therefore prone to high runoff coefficients which would result in high regression slopes consistent with the original HSG concept (NEH4, 2004). Visual interpretation of the P_a-R_a pairs in Fig. 5.4 shows that some of the highest R_a values for several land use types are indeed observed on plots in HSG D. However, many fine-textured soils are also prone to cracking upon drying, resulting in macropores that are very effective in increasing infiltration rates and reduce runoff (Arnold et al., 2005; Johnson, 1962). This different hydrologic behaviour within HSG D calls for the separate treatment of cracking and non-cracking fine-textured soils in runoff modelling. Inference of soil cracking and soil hydrological behaviour is hard to determine from soil texture data alone however (Wagner et al., 2001; Wösten et al., 2001). Insufficient information was available to separate cracking from non-cracking soils in the dataset and hence, the importance of cracking in the P_a-R_a relation could not be further assessed. Nevertheless, soil cracking occurs mostly in seasonally dry environments such as the Mediterranean and hence a difference between Mediterranean and non-Mediterranean plots is expected. This is adressed in section 5.3.4.

Furthermore, Fig. 5.4 shows a large residual variability on the fitted regressions for most combinations of land use type and HSG. The random effect of the plot measuring station explains between 16.9% (shrubland) and 93.5% (vineyards) of the total observed variance in the residuals (Table 5.4). Hence, the observed P_a-R_a relations can be strongly affected by the characteristics of individual plot measuring stations, and the results of this study are only valid for the range of environmental conditions represented in the runoff plot database. Especially for combinations of land use types and HSG where the number of plot measuring sites is small, care should be taken when applying the results in Table 5.4 and Fig. 5.4 to other plot measuring sites.

The relative positions of the quantile curves for the slope of the P_a-R_a relations show an effect of HSG, largely corresponding to results of the linear mixed model (Table 5.4), but also reflect the large residual variability in the results of the linear mixed quantile regressions Fig. 5.4. Quantile curves are not smooth which indicates important differences between the plot measuring stations, and hence other environmental characteristics than HSG (e.g. vegetation cover or rainfall distribution within the year) also play an important role in the P_a-R_a relations. Furthermore, R_a data for individual plot measuring stations tend to be clustered in the frequency distributions of the predictor variable due to a specific climatic regime, which is not ideal for the application of mixed models.

Table 5.4: Results of the linear mixed effects model fitting for different combinations of land use types and Hydrologic Soil Groups. var. explained= residual variance explained by the random effect of plot measuring station. n= number of P_a-R_a pairs, PL= number of plots, MS= number of plot measuring stations, b= regression intercept, m= regression slope, p= regression slope p-value, n.c.= not calculated as n<5, n.s.= regression slope not significant.

land use type	var. expl. (%)	HSG A						HSG B					
		n	PL	MS	b	m	p	n	PL	MS	b	m	p
bare	85.1	92	26	12	-23.03	0.142	<0.001	209	17	7	-38.96	0.176	<0.001
cereals	76.1	40	12	8	n.s.	n.s.	0.229	99	46	16	-46.19	0.169	<0.001
row crops	81.3	72	19	4	n.s.	n.s.	0.498	63	22	10	-82.51	0.215	<0.001
leguminous	63.5	7	3	3	-3.37	0.079	0.014	8	2	2	-167.93	0.275	<0.001
rotation	39.1	1	1	1	n.c.	n.c.	n.c.	73	9	2	-62.35	0.311	<0.001
fallow	51.6	14	5	3	n.s.	n.s.	0.954	77	18	6	-137.68	0.294	<0.001
tree crops	59.9	4	1	1	n.c.	n.c.	n.c.	5	2	2	n.s.	n.s.	0.9056
vineyard	93.5	59	8	4	47.85	0.076	0.013	4	3	3	n.c.	n.c.	n.c.
shrubland	16.9	9	3	3	n.s.	n.s.	0.756	104	29	12	n.s.	n.s.	0.3778
rangeland	19.8	3	1	1	n.c.	n.c.	n.c.	1	1	1	n.c.	n.c.	n.c.
grassland	80.4	146	22	5	n.s.	n.s.	0.86	30	6	4	-37.34	0.129	<0.001
forest	72.2	78	17	5	3.17	0.011	0.007	47	10	5	n.s.	n.s.	0.0626
post-fire	48.2	30	13	7	-36.5	0.118	0.01	16	6	3	n.s.	n.s.	0.9114

Table 5.4: *Continued.* Results of the linear mixed effects model fitting for different combinations of land use types and Hydrologic Soil Groups. n= number of P_a-R_a pairs, PL= number of plots, MS= number of plot measuring stations, b= regression intercept, m= regression slope, p= regression slope p-value, n.c.= not calculated as n<5, n.s.= regression slope not significant.

land use type	**HSG C**						**HSG D**					
	n	**PL**	**MS**	**b**	**m**	**p**	**n**	**PL**	**MS**	**b**	**m**	**p**
bare	101	8	4	-10.27	0.241	<0.001	348	40	18	-23.11	0.111	<0.001
cereals	54	5	4	-43.64	0.118	0.005	214	42	20	-27.94	0.118	<0.001
row crops	52	3	2	17.22	0.097	0.013	59	14	7	-13.06	0.087	<0.001
leguminous	23	2	2	38.61	0.041	0.019	136	22	7	-9.21	0.049	<0.001
rotation	18	2	1	n.s.	n.s.	0.657	59	6	3	-108.25	0.269	<0.001
fallow	1	1	1	n.c.	n.c.	n.c.	70	13	8	-39.07	0.171	<0.001
tree crops	1	1	1	n.c.	n.c.	n.c.	13	4	3	n.s.	n.s.	0.2764
vineyard	1	1	1	n.c.	n.c.	n.c.	11	5	2	n.s.	n.s.	0.117
shrubland	1	1	1	n.c.	n.c.	n.c.	89	22	9	-11.1	0.072	<0.001
rangeland	1	1	1	n.c.	n.c.	n.c.	35	7	4	-31.58	0.092	<0.001
grassland	1	1	1	n.c.	n.c.	n.c.	30	10	3	n.s.	n.s.	0.6158
forest	17	3	2	-16.83	0.07	0.012	34	8	3	7.55	0.027	<0.001
post-fire	4	1	1	n.c.	n.c.	n.c.	88	18	2	60.31	0.044	0.017

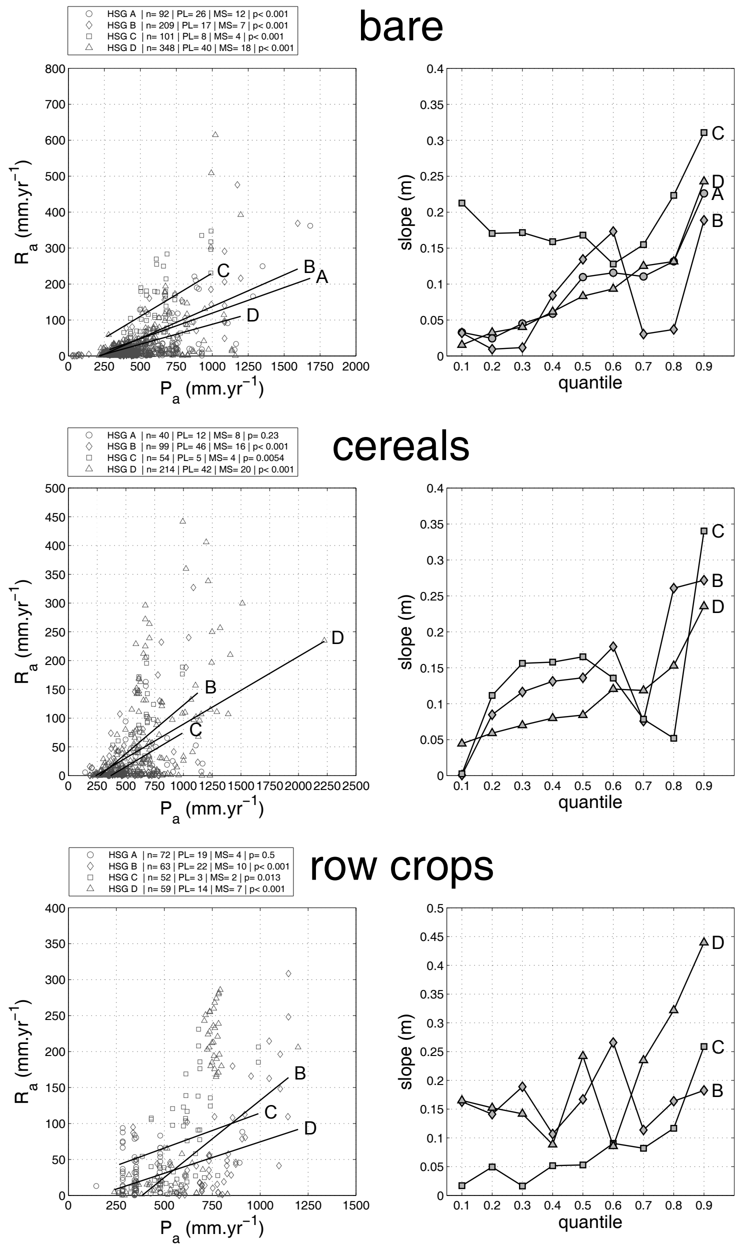


Figure 5.4: Results of the fitting of **(left)** a linear mixed effects model (Eq. 5.1a) and **(right)** a linear quantile mixed model to the P_a-R_a data for different land use types.

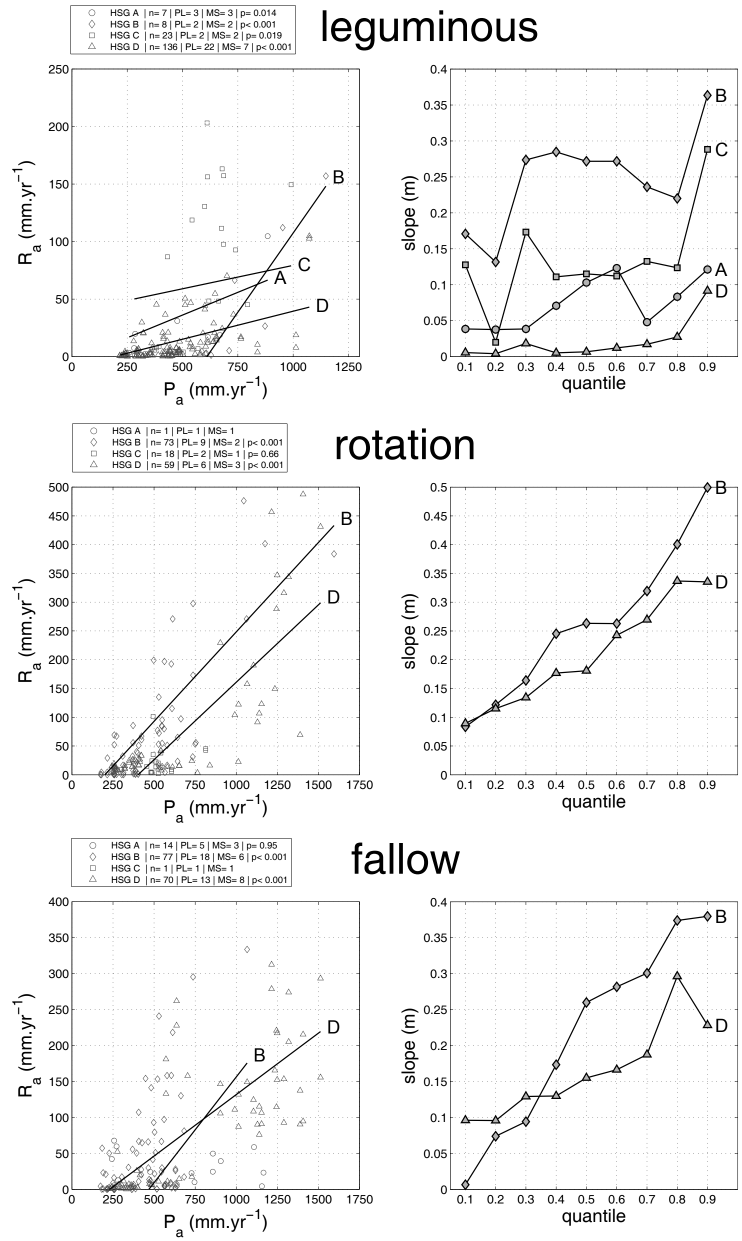


Figure 5.4: *Continued.* Results of the fitting of **(left)** a linear mixed effects model (Eq. 5.1a) and **(right)** a linear quantile mixed model to the P_a-R_a data for different land use types.

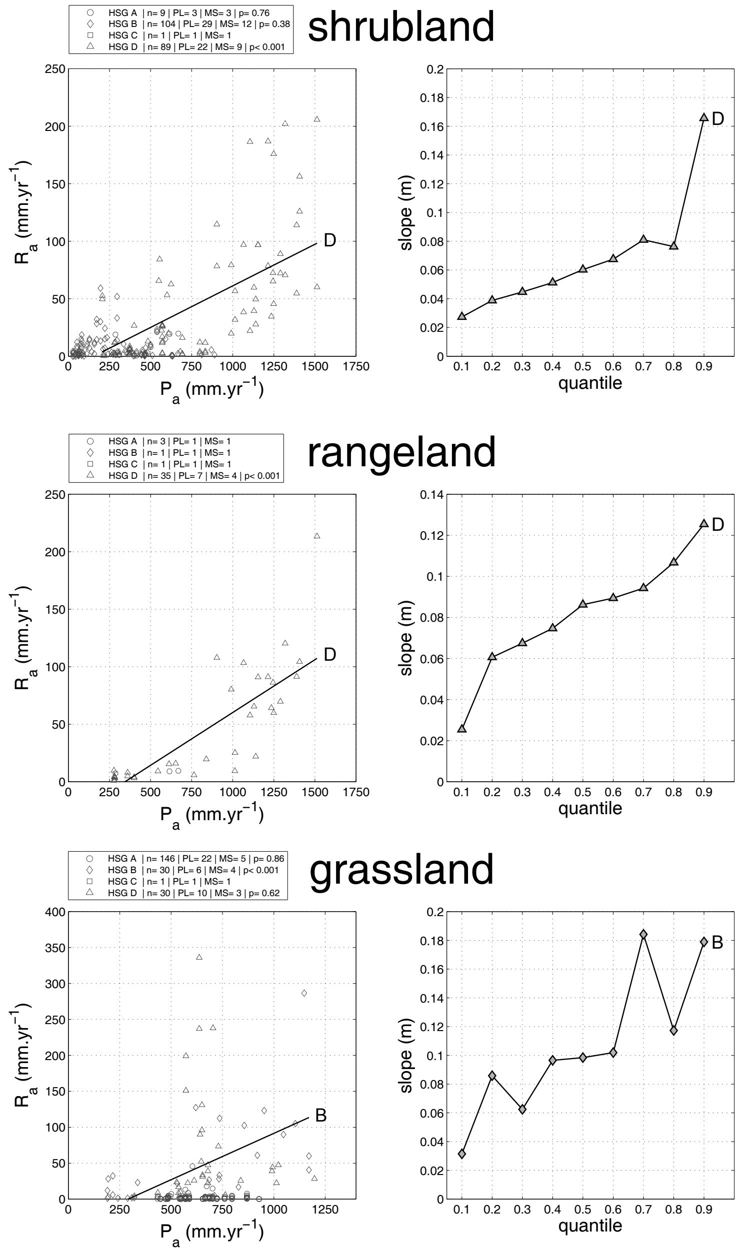


Figure 5.4: *Continued.* Results of the fitting of **(left)** a linear mixed effects model (Eq. 5.1a) and **(right)** a linear quantile mixed model to the P_a-R_a data for different land use types.

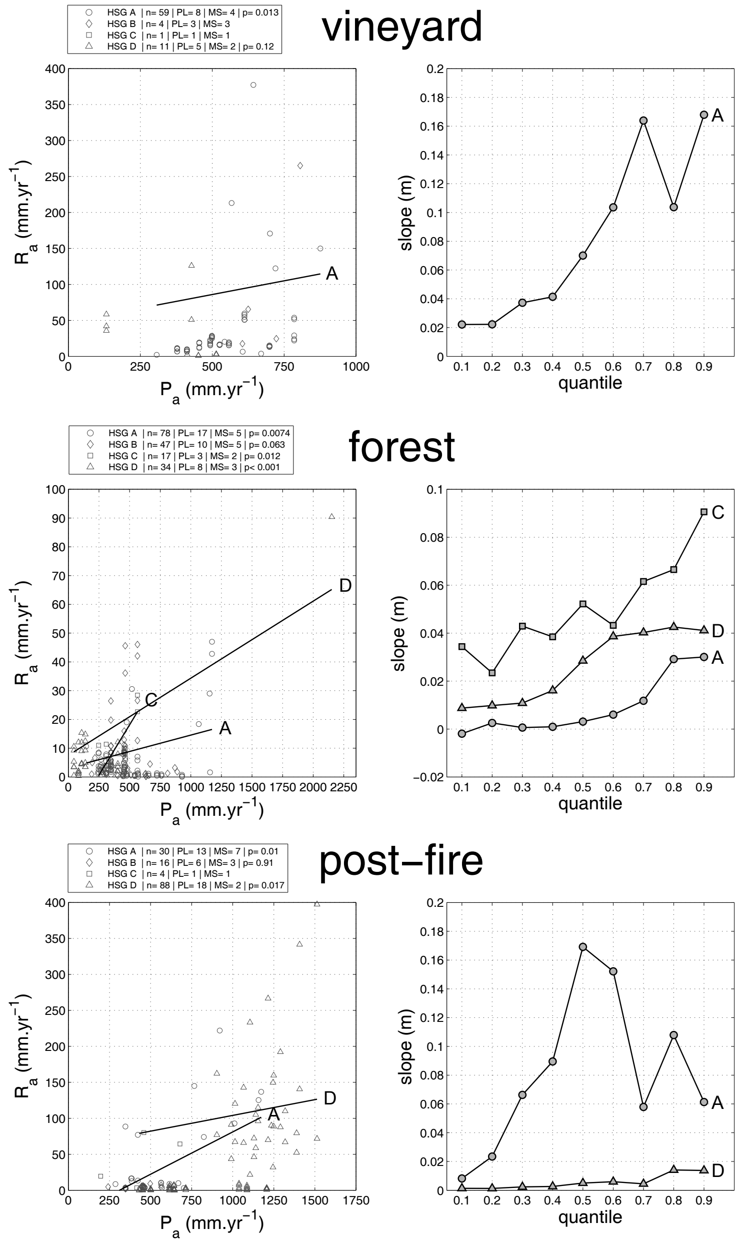


Figure 5.4: *Continued.* Results of the fitting of **(left)** a linear mixed effects model (Eq. 5.1a) and **(right)** a linear quantile mixed model to the P_a-R_a data for different land use types.

A comparison of the fitted slope parameter values for the linear mixed model (Table 5.4) with published CN_D values (Hawkins et al., 2009), and of the fitted annual P_a for any R_a to be generated (i.e., the annual precipitation threshold, P_T) with Ia values for the SCS-Curve Number method (Eq. 5.5) is given in Fig. 5.5. Fig. 5.5 is interpreted only visually as the number of points for each individual HSG is too small for meaningful correlation coefficients to be calculated. Overall, correlation between fitted slope values (m) and published CN_D values is weak, but within each separate HSG, there is a positive correlation between the fitted slopes and published CN_D values. Slopes for HSG D are relatively small compared to the high published CN_D values for this HSG, which is in accordance with results observed in Table 5.4 and Fig. 5.4.

Also between P_T and I_a, a positive relation is observed, although many of the fitted intercepts (Eq. 5.1a) were found to be not significant at (α= 0.05), indicating that there is a lot of variability associated with the P_T values. Apart from illustrating the usefulness of the Curve Number concept for the analysis of the P_a-R_a relation (see section 5.3.2), these relations also indicate links between the daily rainfall-runoff (P_d-R_d) relation and the annual rainfall-runoff (P_a-R_a relation). This is further explored in section 5.3.5.

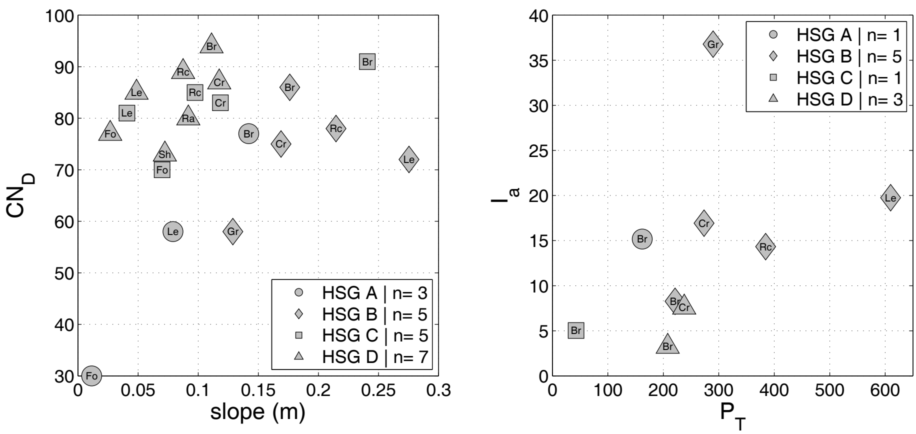


Figure 5.5: (left) Relation between the fitted slope values for the linear mixed effects model (Table 5.4) and CN_D values (Hawkins et al., 2009). **(Right)** Relation between the fitted precipitation threshold values for the linear mixed effects model (Eq. 5.1c) and initial abstraction values (I_a, Eq. 5.5). CN_D= daily CN-values, I_a= Initial abstraction, P_T= P_a threshold for R_a observation. Br: bare, Ce: cereals, Rc: row crops, Le: leguminous, Ro: rotation, Fa: fallow, Tc: tree crops, Vi: vineyard, Sh: shrubland, Ra: rangeland, Gr: grassland, Fo: forest, Pf: post-fire. n= number of different combinations of land use type and HSG with significant regressions parameters.

5.3.2 Modified SCS-Curve Number method

The results of fitting the Modified Curve Number Method Eq. 5.4a and Eq. 5.4b without a random effect to the P_a-R_a data for each land use type and different λ values are given in Table 5.5. As is expected, the best model efficiencies (ME) are obtained when optimizing for both λ and CN_A. However, λ values are generally small, and allowing λ to vary between different land use types precludes an easy interpretation of the CN_A value as a direct indication of the susceptibility of a specific land use type to runoff generation. When considering only the regressions with a fixed λ value, the best results (i.e. largest ME values) are generally obtained for λ=0. For bare plots, tree crops, vineyards, rangeland and post-fire, somewhat higher ME values are obtained for λ=0.05 and/or λ=0.2, but differences are small. For forest plots, the Modified Curve Number with a λ value of 0.2 performs worse than just taking the mean of all observations, hence the negative ME.

Table 5.5: Results of the modified modified CN model fit (Eq. 5.4a,Eq. 5.4b) for different λ-values for the different land use types. n= number of P_a-R_a pairs, λ= Lambda-value, CN_A= fitted CN-value, ME= Model Efficiency (Eq. 5.6, Nash and Sutcliffe, 1970).

Land use type	n	Optimal λ value			$\lambda = 0$		$\lambda = 0.05$		$\lambda = 0.2$	
		λ	CN_A	ME	CN_A	ME	CN_A	ME	CN_A	ME
bare	908	0.03	4.65	0.33	3.11	0.32	5.79	0.33	11.24	0.27
cereals	485	0.00	2.66	0.29	2.66	0.29	4.97	0.24	9.61	0.07
row crops	258	0.00	4.69	0.29	4.69	0.29	7.49	0.27	13.63	0.20
leguminous	186	0.01	2.98	0.33	2.20	0.32	4.96	0.30	10.43	0.17
rotation	152	0.02	5.89	0.62	5.08	0.62	7.32	0.61	12.25	0.56
fallow	168	0.00	2.95	0.38	2.95	0.38	4.92	0.30	9.02	0.18
tree crops	22	0.29	20.50	0.34	6.31	0.29	9.65	0.32	17.04	0.34
vineyard	74	0.05	6.66	0.13	3.40	0.12	6.61	0.13	13.48	0.11
shrubland	203	0.01	2.10	0.57	1.50	0.56	3.30	0.56	7.18	0.51
rangeland	38	0.05	3.19	0.75	1.51	0.72	3.24	0.75	7.13	0.71
grassland	209	0.01	1.79	0.06	1.09	0.06	3.44	0.06	8.11	0.01
forest	176	0.00	0.50	0.27	0.50	0.27	1.60	0.03	3.68	*-0.05*
post-fire	139	0.06	3.90	0.38	1.65	0.35	3.49	0.38	7.53	0.37

Therefore, regression fitting with a fixed λ=0 value was adopted in the remainder of the analysis. CN_A values for the P_a-R_a (i.e. annual) relations are considerably lower than CN_D values for the P_d-R_d (i.e. event) relation, which is due to the much larger fitted S-values for the P_a-R_a relation. It should be noted that Eq. 5.3. was introduced only to scale S-values used in the Curve Number equation (Hawkins et al., 2009; NEH4, 2004) to an easily interpretable range of CN_D values between 30 and 100 (Hawkins et al., 2009). Hence, it would be possible to redefine the transformation of S_A-values to CN_A values to result in CN_A values in the 30 to 100 range. This is not done in this research to avoid confusion between CN_D and CN_A values.

Results for the fitting of the Modified Curve Number model (Eq. 5.4a, Eq. 5.4b) for the different combinations of land use types and HSG (λ=0) are given in Table 5.6 and Fig. 5.6). For row crop plots in HSG D, the fitting algorithm did not converge and hence no CN_A value was obtained. However, Fig. 5.6 shows a consistent relation between P_a and R_a that would result in a small CN_A value, if not for a single cluster of P_a-R_a data pairs with R_a>150 mm·yr^{-1}corresponding to 7 plots at 3 measuring sites; i.e. Mugello (Italy; Zanchi, 1988a), Lumalas and Drithas (Croatia; Grazhdani et al., 1996), the latter of which acted as control plots in a study to determine the effectiveness of drainage. Fig. 5.6 confirms findings for the fitting of the linear mixed model (section 5.3.1).

For bare plots, CN_A for HSG A < CN_A for HSG B < CN_A for HSG C as expected, but CN_A for HSG D is smaller than the fitted CN_A values for the other three HSG. Also for the other land use types, fitted CN_A values for HSG D are smaller than fitted CN_A values for at least one of the other HSG for all land use types except cereals and shrubland. As mentioned in section 5.3.1, this is attributed to the cracking behaviour of some clayey soils. As already mentioned, the relative order of the different HSG is not always as expected (i.e. HSG A < CN_A for HSG B < CN_A for HSG C < CN_A for HSG D). Fitting of the Modified Curve Number Model (Eq. 5.4a, Eq. 5.4b, λ=0) without taking into account the random effect of the plot measuring station resulted in fitted CN_A values that correspond better to the expected relative order for the different HSG (Fig. 5.7, Table 5.7), although not for every land use type and CN_A values for HSG D remain low in comparison to the other HSD. The fact that not considering the mixed effect results in a clearer distinction between the different HSG indicates that while HSG does not cause a clear effect on every single plot measuring station due to other environmental factors that control the P_a-R_a relation, it is nevertheless an important overall controlling factor. This is supported by the results for the bare plots, where both including and excluding the random effect of plot measuring station gives similar results as the important controlling effect of differences is vegetation cover is not present in this land use type.

Table 5.6: Results of the modified CN model fit (λ=0) for the different combinations of land use type and Hydrological Soil Groups with mixed effects for the plot measuring stations. n= number of P_a-R_a data pairs, PL= number of plots, MS= number of plot measuring stations, CN_A= fitted Curve Number, p= p-value. n.c.= not calculated. Results for CN_A that are not significant at α=0.05 are given in italic and small print as some of them are marginally significant.

land use type	HSG A					HSG B				
	n	PL	MS	CN_A	p	n	PL	MS	CN_A	p
bare	94	26	12	1.96	<0.001	279	17	7	2.93	0.00
cereals	40	12	8	1.25	0.01	147	54	16	0.65	<0.001
row crops	72	19	4	3.49	<0.001	63	22	10	1.83	<0.001
leguminous	7	3	3	n.c.	n.c.	8	2	2	n.c.	n.c.
rotation	1	1	1	n.c.	n.c.	73	9	2	7.26	<0.001
fallow	16	6	3	0.73	0.02	82	22	6	1.46	*0.06*
tree crops	4	1	1	n.c.	n.c.	5	2	2	n.c.	n.c.
vineyard	59	8	4	3.05	0.02	4	3	3	n.c.	n.c.
shrubland	9	3	3	n.c.	n.c.	104	29	12	0.67	0.01
rangeland	3	1	1	n.c.	n.c.	1	1	1	n.c.	n.c.
grassland	149	22	5	*0.15*	0.05	30	6	4	2.05	0.01
forest	78	17	5	0.49	<0.001	47	10	5	*0.39*	0.11
post-fire	30	13	7	3.01	<0.001	16	6	3	0.69	<0.001

land use type	HSG C					HSG D				
	n	PL	MS	CN_A	p	n	PL	MS	CN_A	p
bare	121	8	4	4.08	<0.001	414	40	18	1.53	<0.001
cereals	72	5	4	1.26	0.10	226	42	20	1.65	<0.001
row crops	56	3	2	3.04	*0.07*	1	1	1	n.c.	n.c.
leguminous	29	2	2	5.56	<0.001	142	22	7	1.20	<0.001
rotation	19	2	1	1.82	<0.001	60	7	4	4.54	<0.001
fallow	1	1	1	n.c.	n.c.	70	13	8	2.95	0.00
tree crops	1	1	1	n.c.	n.c.	13	4	3	6.52	0.03
vineyard	1	1	1	n.c.	n.c.	11	5	2	*2.55*	0.24
shrubland	1	1	1	n.c.	n.c.	89	22	9	1.79	0.00
rangeland	1	1	1	n.c.	n.c.	35	7	4	1.52	<0.001
grassland	1	1	1	n.c.	n.c.	30	10	3	*2.12*	0.05
forest	17	3	2	1.49	<0.001	34	8	3	0.52	<0.001
post-fire	4	1	1	n.c.	n.c.	89	18	2	2.24	<0.001

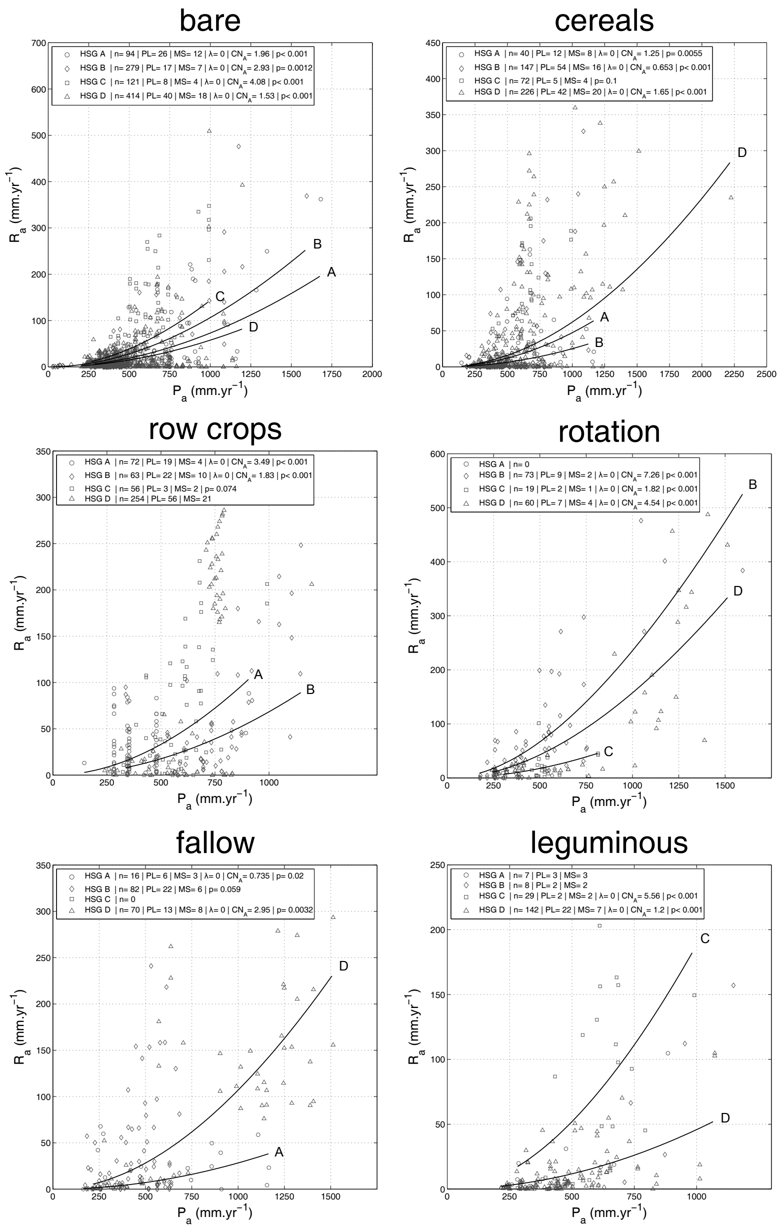


Figure 5.6: Results of the fitting of the Modified SCS-Curve Number model (Eq. 5.4a, λ=0) to the P_a-R_a data for different land use types. n= number of P_a-R_a pairs, PL= number of plots, MS= number of plot measuring stations, CN_A= annual CN-value, p= p-value.

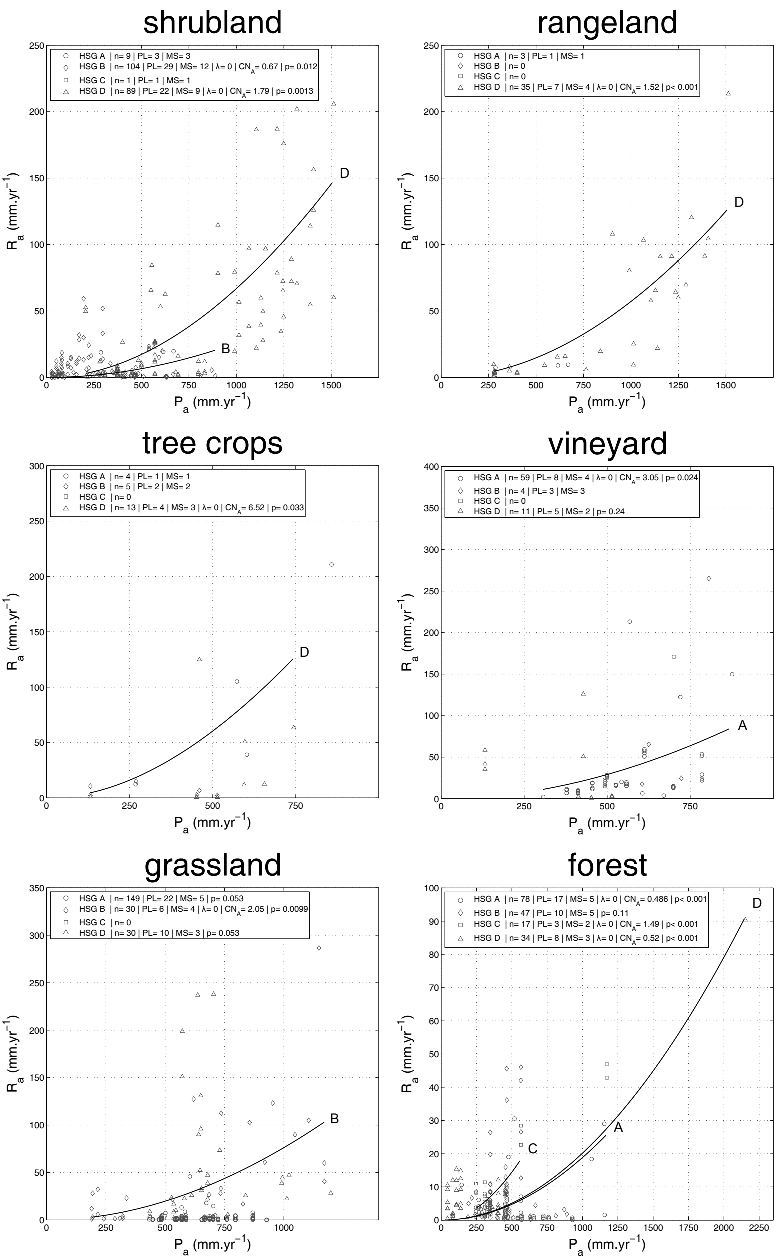


Figure 5.6: *Continued.* Results of the fitting of the Modified SCS-Curve Number model (Eq. 5.4a, λ=0) to the P_a-R_a data for different land use types. n= number of P_a-R_a pairs, PL= number of plots, MS= number of plot measuring stations, CN_A= annual CN-value, p= p-value.

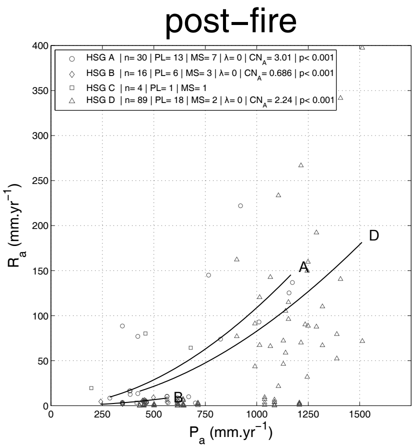


Figure 5.6: *Continued.* Results of the fitting of the Modified SCS-Curve Number model (Eq. 5.4a, λ=0) to the P_a-R_a data for different land use types. n= number of P_a-R_a pairs, PL= number of plots, MS= number of plot measuring stations, CN_A= annual CN-value, p= p-value.

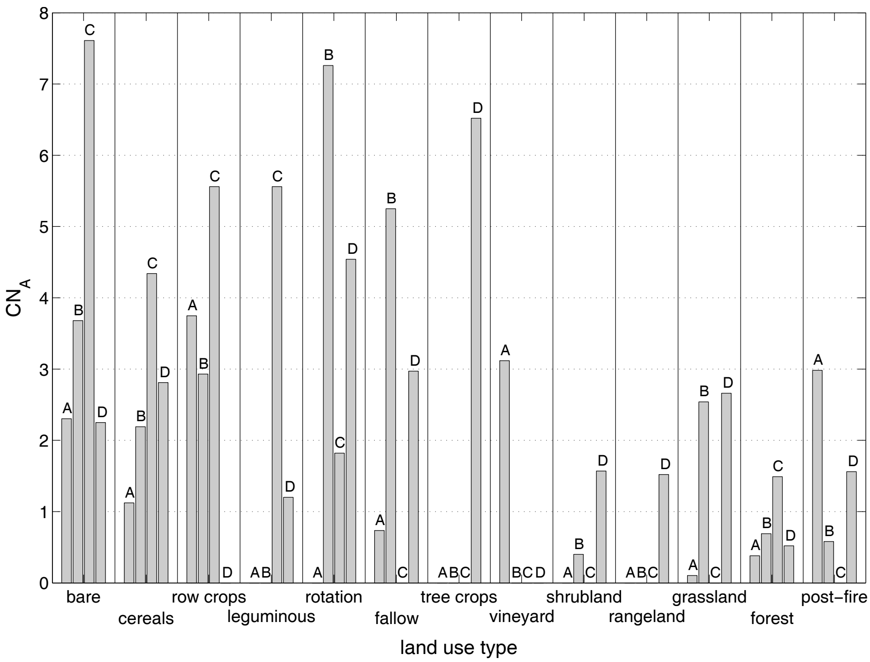


Figure 5.7: Results of the fitting of the Modified SCS-Curve Number model (Eq. 5.4a, λ=0) to the P_a-R_a data for different land use types without random effects. For number of P_a-R_a pairs and regression p-values, see Table 5.7.

Table 5.7: Results of the modified CN model fit (λ=0) for the different combinations of land use type and Hydrological Soil Groups without mixed effects. n= number of P_a-R_a data pairs, CN_A= fitted Curve Number, p= p-value. n.c.= not calculated. n.s.= not significant at α= 0.05.

land use type	**HSG A**			**HSG B**			**HSG C**			**HSG D**		
	n	**CN_A**	**p**	**n**	**CN_A**	**p**	**n**	**CN_A**	**p**	**n**	**CN_A**	**p**
bare	94	2.30	<0.001	279	3.68	<0.001	121	7.61	<0.001	414	2.25	<0.001
cereals	40	1.12	0.001	147	2.19	<0.001	72	4.34	<0.001	226	2.81	<0.001
row crops	72	3.75	<0.001	63	2.93	<0.001	56	5.56	<0.001	1	n.c.	n.c.
leguminous	7	n.c.	n.c.	8	n.c.	n.c.	29	5.56	<0.001	142	1.20	<0.001
rotation	1	n.c.	n.c.	73	7.26	<0.001	19	1.82	0.001	60	4.54	<0.001
fallow	16	0.73	0.019	82	5.25	<0.001	1	n.c.	n.c.	70	2.97	<0.001
tree crops	4	n.c.	n.c.	5	n.c.	n.c.	1	n.c.	n.c.	13	6.52	0.029
vineyard	59	3.12	<0.001	4	n.c.	n.c.	1	n.c.	n.c.	11	n.s.	0.234
shrubland	9	n.c.	n.c.	104	0.40	0.010	1	n.c.	n.c.	89	1.57	<0.001
rangeland	3	n.c.	n.c.	1	n.c.	n.c.	1	n.c.	n.c.	35	1.52	<0.001
grassland	149	0.10	<0.001	30	2.54	<0.001	1	n.c.	n.c.	30	2.66	0.004
forest	78	0.38	<0.001	47	0.69	<0.001	17	1.49	<0.001	34	0.52	<0.001
post-fire	30	2.98	<0.001	16	0.58	<0.001	4	n.c.	n.c.	89	1.56	<0.001

5.3.3 Comparison between the linear and modified SCS-Curve Number models

Linear mixed model fits with a positive intercept (e.g Vineyards, HSG A: Fig. 5.4) only have local meaning and cannot be extrapolated beyond the observed P_a. In this respect, the modified CN method is more consistent with the runoff generation process, as P_a approaches 0 mm·yr^{-1} , so does the predicted R_a. Therefore, the modified CN Model can be extrapolated more easily without producing physical inconsistencies (i.e. predicting R_a when P_a= 0 mm·yr^{-1}). On the other hand, for combinations of land use types and HSG where the linear mixed model fit has a positive intercept (e.g vineyard, HSG A: Fig. 5.4), or a steep slope (e.g forest, HSG C: Fig. 5.4) no significant fit can be obtained with the modified CN method. Nevertheless, there is a relatively good correlation between the fitted slope values (m) for the linear mixed model and the fitted CN_A values for the modified CN Model with random effects for the plot measuring station (Fig. 5.8).

A comparison of the residuals for both the linear mixed model and modified CN model with mixed effects (Fig. 5.9 gives an example for bare plots, other land use types show similar results) shows considerable unexplained scatter in both models. The linear mixed tends to overpredict R_a values more than the modified CN model. On the other hand, the the modified CN model tends to underpredict R_a values more in the 500-1000 mm·yr^{-1} P_a range, an effect that becomes smaller for P_a>1000 mm·yr^{-1}. These differences are attributed to the curvilinear form of the modified CN model which is a better approximation of the R_a-P_a relation (Fig. 5.4, Equation Eq. 5.6, Nash and Sutcliffe, 1970).

Examination of the P_T values (Fig. 5.4) shows that a clear demarcation of P_T values or P_a thresholds for runoff generation is not possible and the P_T concept (Eq. 5.1b and Eq. 5.1c) is a poor predictor of whether R_a will occur or not. This is likely due to the fact that the effects of the initial abstraction and precipitation threshold work at the event scale, and can not easily be derived from annual data. Conversely, also the annual effective precipitation (i.e. the annual sum of P_d causing runoff) for different land use types can not accurately be determined using P_a-R_a data. Many small and evenly distributed rainfall events can result in large P_a values which nevertheless do not yield any R_a, while a few intense rain events can cause relatively high R_a for limited P_a. Hence, the usefulness of incorporating a P_T or initial abstraction (I_a, Eq. 5.5) concept to obtain a better fit is limited, justifying the choice a fixed λ=0. On the other hand, the elimination of I_a also means that any value of P_a, however small, will result in a certain amount of R_a, which is not physically consistent with concepts of surface storage, infiltration and evaporation (e.g. Gayle and Skaggs, 1978; Kamphorst et al., 2000; Le Bissonnais et al., 2005a).

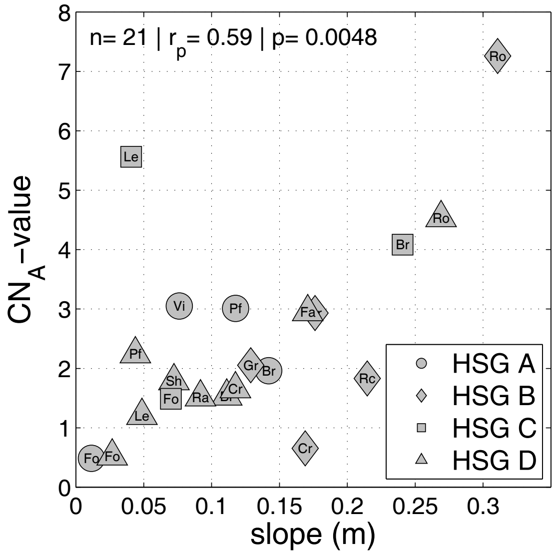


Figure 5.8: Relation between linear mixed effects model slopes (Eq. 5.1a) and SCS-Curve Number model CN_A values (Eq. 5.4a) for different combinations of land use type and Hydrologic Soil Groups for which both the linear mixed model slopes and the CN_A values are significant at α=0.05. Br: bare, Ce: cereals, Rc: row crops, Le: leguminous, Ro: rotation, Fa: fallow, Tc: tree crops, Vi: vineyard, Sh: shrubland, Ra: rangeland, Gr: grassland, Fo: forest, Pf: post-fire. n= number of different combinations of land use type and HSG with significant regressions for both models. r_p= Pearson correlation coefficient, p= p-value.

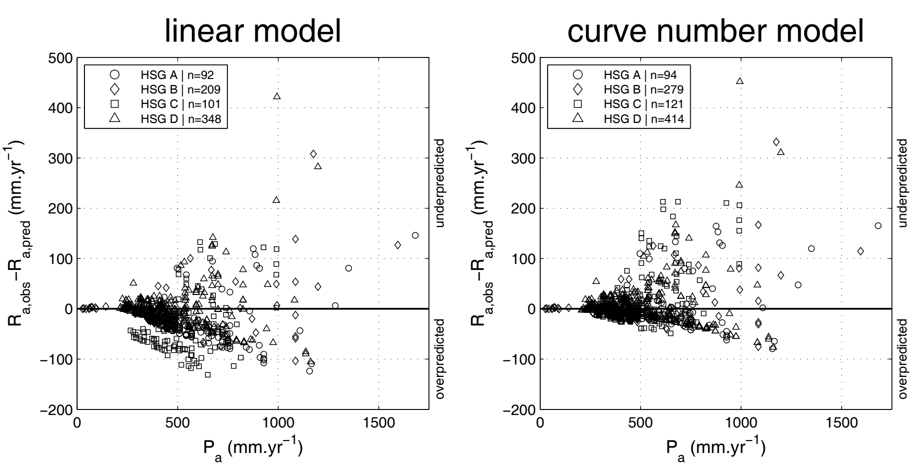


Figure 5.9: Plot of the residuals for the linear mixed model (Eq. 5.1a) and modified CN mixed model (Eq. 5.4a) fitted to data for bare plots. n: number of P_a-R_a pairs.

5.3.4 Other factors controlling the P_a-R_a relationship

Variation between Mediterranean and non-Mediterranean plots

Comparison of the modified CN model fits without random effects (λ=0) for Mediterranean and non-Mediterranean plots (Fig. 5.10) shows that fitted CN_A values are generally higher for the Mediterranean plots than for the non-Mediterranean plots, although the number of combinations of land use type and HSG for which the CN_A fit is significant in both the Mediterranean and non-Mediterranean. For HSG D, there are no pairs of CN_A values available for which the Modified Curve Number Model gave a significant fit for both the Mediterranean and non-Mediterranean plots, and hence the hypothesis that the CN_A value for HSG D is lower in the Mediterranean than in the non-Mediterranean plots could not be verified.

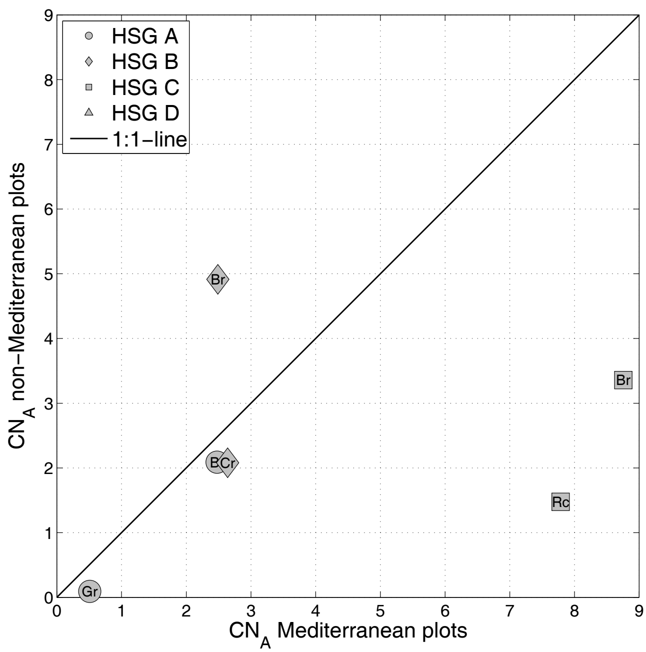


Figure 5.10: Relation between fitted CN_A values for Mediterranean and non-Mediterranean plots for which both CN_A values fitted by the modified CN model without random effects are significant at α=0.05. Br: bare, Cr: cereals, Rc: row crops.

The general trend for higher CN_A for Mediterranean plots is attributed mainly to climatic differences. Precipitation in the Mediterranean is often concentrated in a few events during a short rainy season (e.g. Altava-Ortiz et al., 2011; Mehta and Yang, 2008), therefore causing larger R_a. Nevertheless, various other factors are known to affect runoff response and may also be a factor to explain the observed differences between the Mediterranean and non-Mediterranean runoff plots. Soil stone content and surface cover by rock fragments is generally higher in the Mediterranean than in the non-Mediterranean (e.g. Cerdan et al., 2010), which can have different effects on runoff response, depending on the placement and depth of burial of the stones (Poesen et al., 1990; Smets et al., 2011a). Also other factors such as differences in slope gradient distribution between the Mediterranean and non-Mediterranean may play a role.

Other environmental and experimental factors

Residuals plots of the modified CN model fits (λ=0) for the different HSG as a function of plot length (Fig. 5.11a), slope gradient (Fig. 5.11b) and SOM (Fig. 5.11c) indicate that they explain little added variability in the P_a-R_a relationship. While there may be weak trends in the data presented here (e.g. for plot length and slope gradient on bare plots), with possibilities for further model improvement, no generally valid effect can be found and at continental scale, the environmental effects explored here are subordinate to other, unexplained effects such as precipitation distribution throughout the year.

Residuals analysis of the modified CN model fits for the different HSGs shows that also the precipitation concentration index (PCI) does not explain any substantial added variability (Fig. 5.11d). While the distribution of precipitation events throughout the year may be an important source of scatter in the P_a-R_a relationship, this effect is difficult to capture in a single parameter that is easy to quantify such as the PCI. For instance, PCI does not consider whether events are consecutive or occur during times of the year when the runoff response will be more rapid (e.g. when there is little vegetation cover or surface roughness is low). In addition, severe storms causing a significant proportion of R_a can be highly localised and may not be well represented in the E-OBS database. Further analysis of the correspondence between the P_a-R_a relationships and distribution of rainfall events during the measuring period is a key factor to make further use of the annual dataset described in this research, but also of other seasonal and monthly P-R data for a range of applications such as reservoir design or soil available water management.

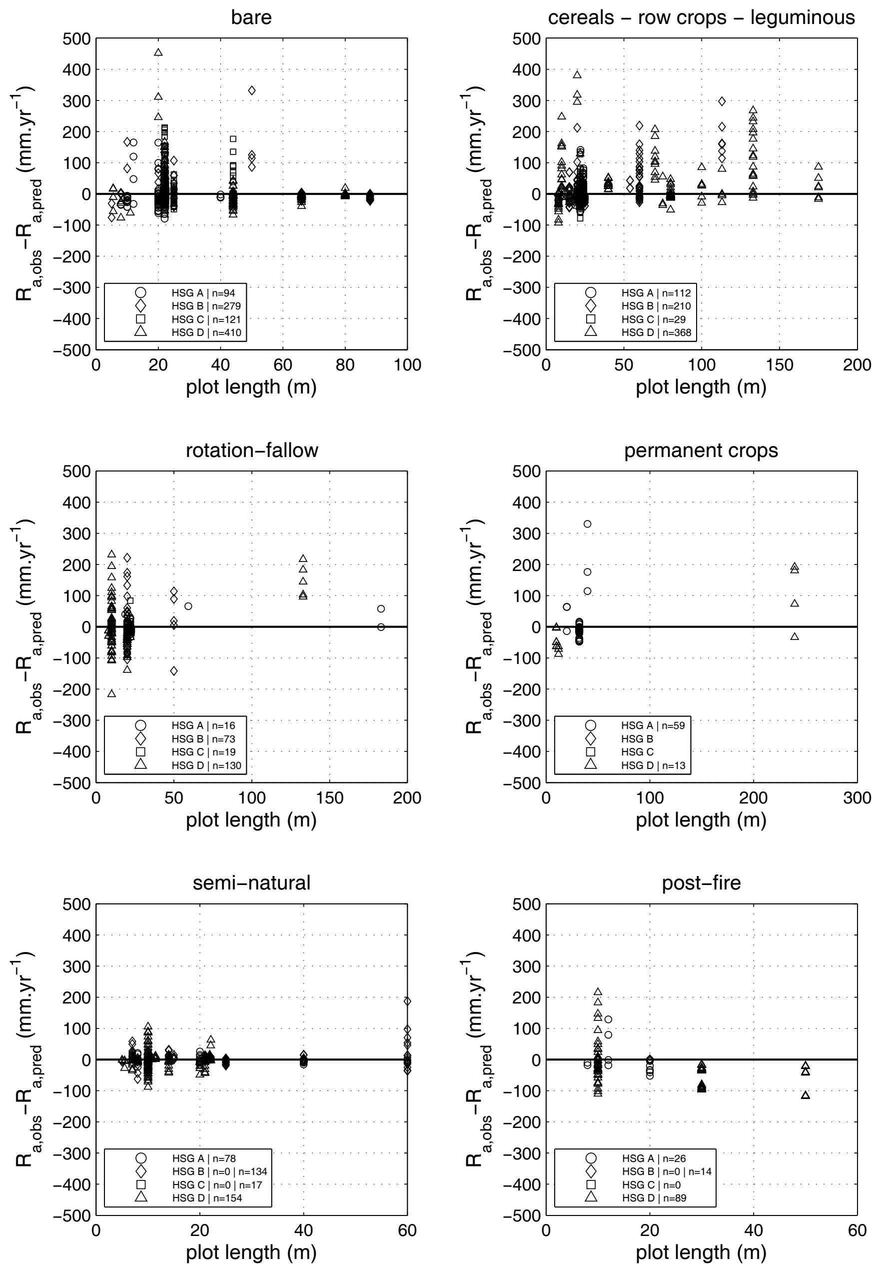


Figure 5.11a: Residuals plot for the modified CN model fits (λ=0) for the different Hydrologic Soil Groups in function of the plot length. n= number of P_a-R_a data pairs.

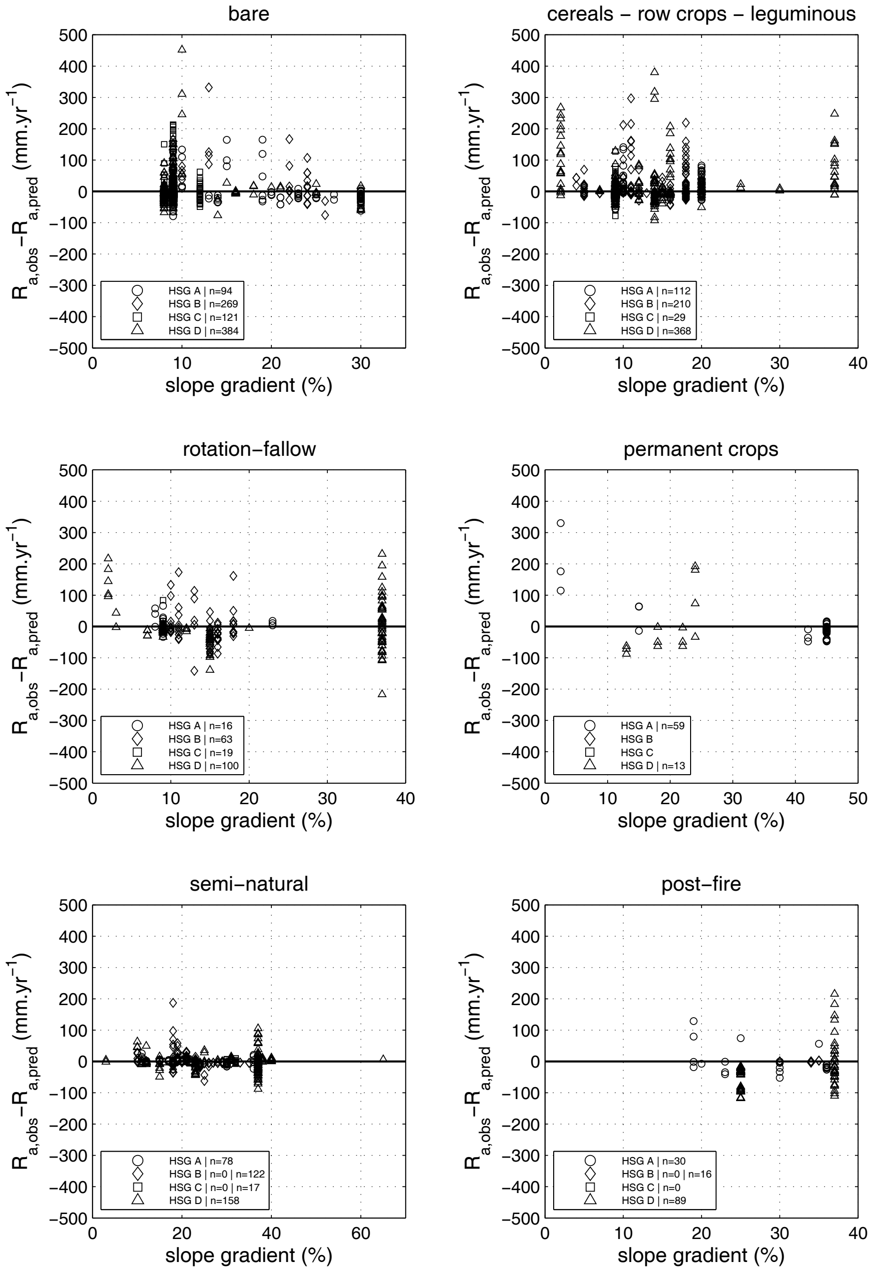


Figure 5.11b: Residuals plot for the modified CN model fits (λ=0) for the different Hydrologic Soil Groups in function of the slope gradient. n= number of P_a-R_a data pairs.

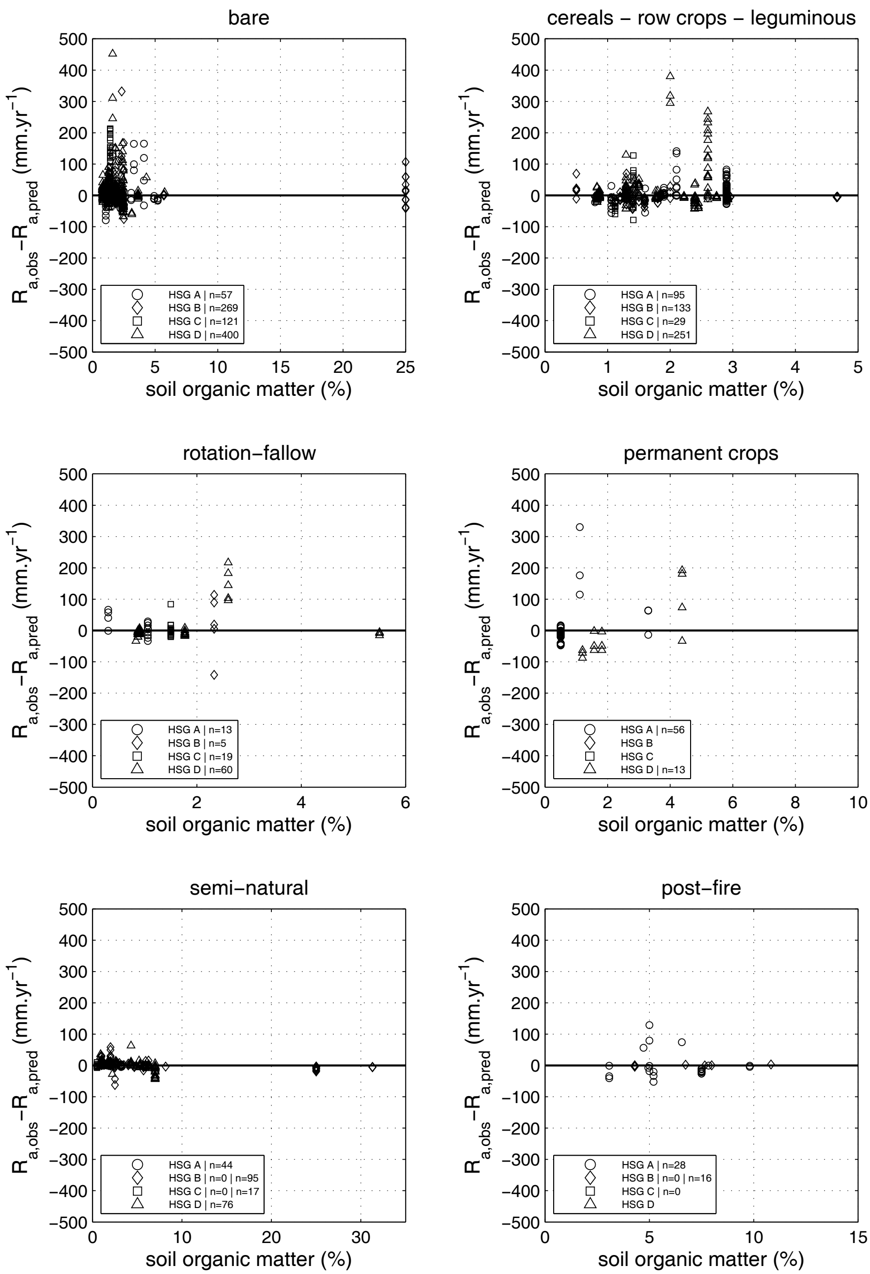


Figure 5.11c: Residuals plot for the modified CN model fits (λ=0) for the different Hydrologic Soil Groups in function of the soil organic matter content. n= number of P_a-R_a data pairs.

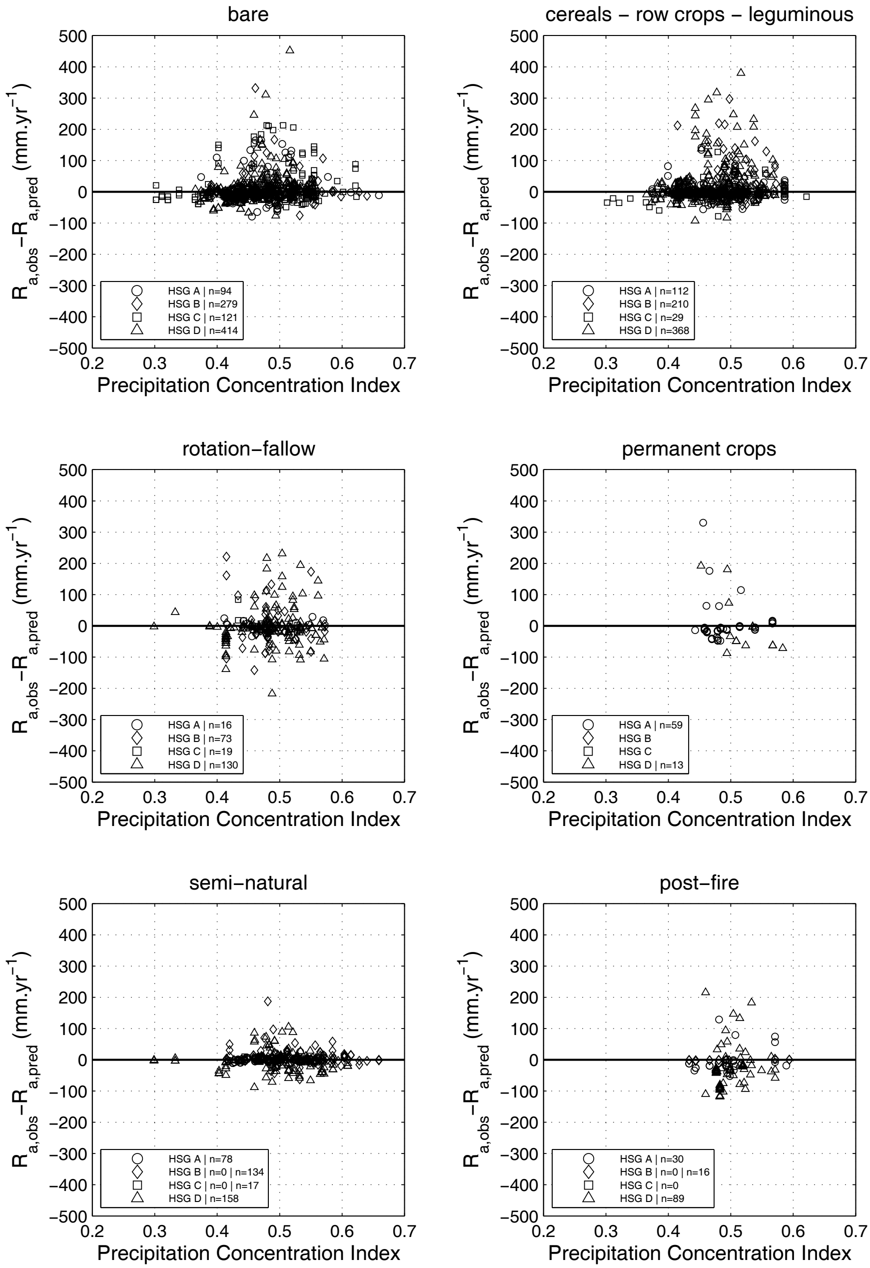


Figure 5.11d: Residuals plot for the modified CN model fits (λ=0) for the different Hydrologic Soil Groups in function of the Precipitation Concentration Index (Eq. 5.8; Martin-Vide, 2004). n= number of P_a-R_a data pairs.

5.3.5 Relation between annual and daily rainfall-runoff relations

Fitting of the modified curve number model (Eq. 5.7) showed that k_1 values were very small or zero. Hence, a simplified model with a fixed k_1-value equal to zero was adopted and the P_a-R_a relation is fully described by the k_2 parameter. These results further indicate that there is no precipitation threshold or initial abstract concept at the annual scale, which is in concurrence with the observations for plot-measured P_a-R_a data.

For small CN_D values, little runoff is generated in most years and the P_a-R_a relation can not be reliably evaluated. Therefore, only P_a-R_a data pairs with $R_a > 10$ mm·yr^{-1}and where there were at least seven years with $R_a > 10$ mm·yr^{-1}at the location were used in the analysis. The relationship between the parameters describing the P_d-R_d relationship (CN_D) and the parameter describing the P_a-R_a relationship (k_2) is given in Fig. 5.12. As expected, there is a negative relation between CN_D and k_2, indicating that the effects of land use type (as expressed in a different CN_D value) at the event scale persist at the annual scale.

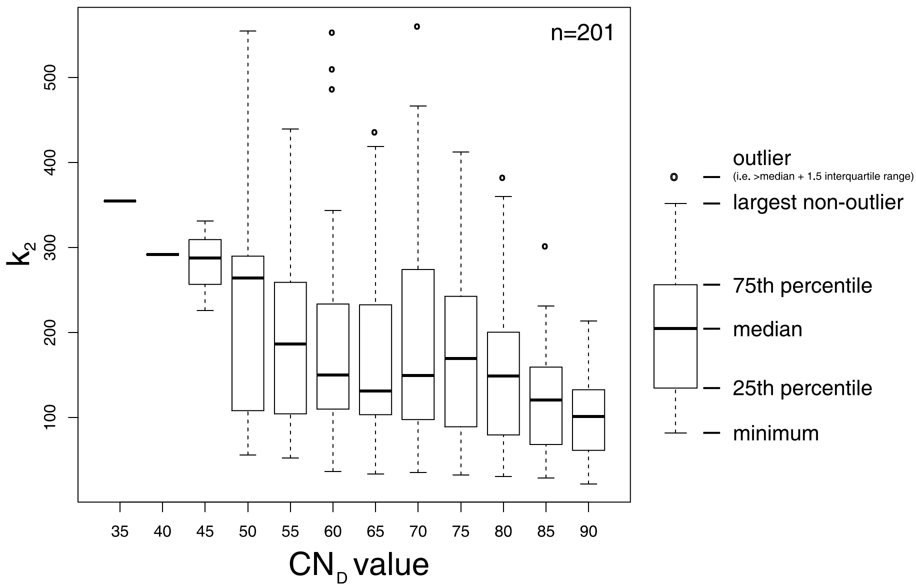


Figure 5.12: Relation between the CN_D-values and the fitted k_2 parameter for the annual curve number model (Eq. 5.7) for all plot measuring stations where E-OBS precipitation time series could be obtained. n= number of plot measuring stations.

Fig. 5.12 also shows a large variation in the CN_D-k_2 relation, which can only be explained by 1) variation in the precipitation distribution between different years at the same plot measuring station (i.e. inter-annual or temporal variability), which affects the fit of the k_2 value, and 2) variation in precipitation distributions between different plot measuring stations (i.e. spatial variability). Hence, even when there is a perfect correlation between rainfall and runoff at the event scale described by the SCS-CN model, spatio-temporal variation in precipitation distribution can account for a large part of the observed variation in the P_a-R_a relationship. In reality, there will not be a perfect relation between P_d-R_d (e.g. Descheemaeker et al., 2008) and CN_D values will vary during the year due to vegetation or crop growth, making even larger variability in the P_a-R_a relation observed in plot-measured data likely.

It should also be noted that neither the runoff simulations (Fig. 5.12), nor the PCI consider the sequence of rainfall events as reflected in the AMC/ARC concept. The use of AMC is now discouraged (Hawkins et al., 2009; NEH4, 2004) and has been replaced by an ARC concept in which ARC I represents the CN_D with 90% exceedance probability, ARC II the 50% exceedance probability CN_D value and ARC I the 10% exceedance probability value (Grabau et al., 2006), but precise guidelines for the actual determination of the corresponding CN_D values are lacking.

Hence, to better understand the P_a-R_a relation, it is crucial to understand the way in which this relation is affected by spatio-temporal variability in daily precipitation distribution. Therefore, the relations between the CN_D and k_2 values are further explored for the different LANMAP2 climatic zones (Fig. 5.13) and plot measuring stations (Fig. 5.14).

Fig. 5.13 shows that the relation between CN_D and k_2 is indeed climate-dependent. Another observation is that variability in the relation decreases for increasing CN_D values. These results indicate that further subdivision into climatic zones may explain additional variability the P_a-R_a relation observed in plot-measured data. Assuming a linear relation between CN_D (and thus S_D) and k_2, a relation of the form:

$$k_{2,cl} = a_{cl} \left(S_D + S_{cl} \right) \qquad \text{(Eq. 5.8)}$$

Where $k_{2,cl}$ is a climate-specific parameter describing the P_a-R_a relation and acl and Scl are climate-specific parameters that can be derived from the relations as shown in Fig. 5.13.

Substituting Eq. 5.8 in Eq. 5.7 gives:

$$R_a = \frac{\left(P_a - k_1 S_D \right)^2}{P_a + a_{cl} \left(S_D + S_{cl} \right) S_D} \qquad \text{(Eq. 5.9)}$$

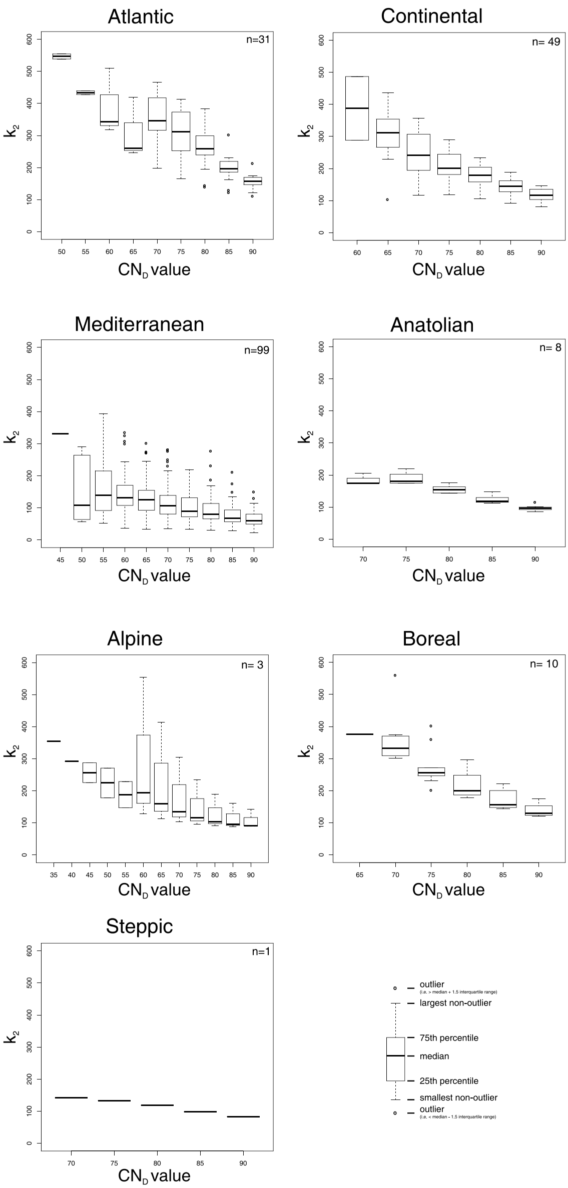


Figure 5.13: Relation between CN_D-values and the fitted k_2 parameter for the modified CN model (Eq. 5.9) for the different LANMAP climatic zones (Metzger et al., 2005; Mücher et al., 2010). n= number of plot measuring stations.

This model accounts for both land use type and HSG effects (through S_D) and climatic effects (through a_{cl} and S_{cl}). However, the limited number of data for non-Mediterranean plots (Fig. 5.10) makes it difficult to examine the climatic effect in actual plot-measured data and validate Eq. 5.9.

For individual plot measuring stations, relatively clear relations between CN_D and k_2 can be observed (Fig. 5.14). Both form and slope of the relation between CN_D and k_2 were found to vary between study sites, reflecting differences in daily precipitation distribution. This shows that using a subdivision in LANMAP2 climatic zones and a linear relation between CN_D and k_2 is only an approximation. Deriving a more accurate general form of the relation between CN_D and k_2 that is valid for the whole of Europe and the Mediterranean and thus linking R_d to P_d at a continental scale is difficult given the many factors affecting the local daily precipitation distribution (e.g. Cosma et al., 2002; Kutiel and Paz, 1998; Lana et al., 1995). However, there is scope to improve the climatic classification specifically aimed at runoff predication in future research.

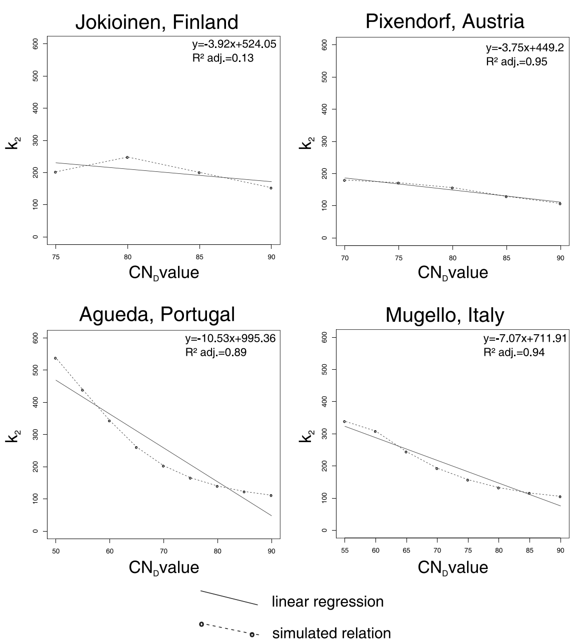


Figure 5.14: Relation between CN_D-values and the fitted k_2 parameter for the modified CN model (Eq. 5.7) for four individual plot measuring stations. Stations have been chosen to represent different types of observed relations.

5.4 Conclusions

A better understanding of annual plot-scale rainfall-runoff relations and controlling variables for the whole of Europe and the Mediterranean has several applications related to soil loss prediction, prediction of soil water balance, soil and water conservation and management of streams and drainage systems. Several models for (plot-scale) runoff prediction have been developed, but many of these models lack validation or are only validated at specific study sites. Hence, the applicability of these models to other areas and environmental conditions controlling the annual rainfall-runoff relation at a continental scale are largely unknown. Plot-measured runoff data throughout Europe and the Mediterranean are generally reported as annual values and few detailed time series of event or daily runoff data exist. Therefore, the annual rainfall (P_a) - annual runoff (R_a) relationship was examined to determine a general form for the relationship and to assess the importance of several factors that control this relationship throughout Europe and the Mediterranean.

The SCS-Curve Number (CN) method, which was developed for the prediction of daily runoff (R_d) from daily precipitation (P_d) was modified to model the P_a-R_a relation (Eq. 5.4a, Eq. 5.4b). It should be clearly stated that the standard SCS-CN method (Hawkins et al., 2009; NEH4, 2004) should only be used to predict R_d values from P_d data. Although predicting power for both the modified Curve Number model and the linear mixed effects model is low, the curvilinear nature of the modified CN model is a better approximation of the actual P_a-R_a relation and Eq. 5.4b) has the advantage that the P_a-R_a relationship is described by a single parameter (i.e. the S_A or CN_A value), provided that λ is kept constant. Hence a CN-type model is less complex compared to the two-parameter (i.e. slope and intercept) linear model, which can be an advantage (e.g. Jakeman and Hornberger, 1993). Furthermore, it was shown that the best overall fit for the modified CN method with a fixed λ value is obtained for λ=0.

The plot data allowed to quantify the different effects of different land use type on the P_a-R_a relation. Notable effects of the Hydrologic Soil Group (HSG) were also found, but a large residual variability after accounting for land use type and HSG remains. Nevertheless, the other environmental factors that were examined (i.e. plot length, slope gradient, soil organic matter content and precipitation concentration index) did not explain much variability in the observed relations. This is in accordance with the original SCS-model where the CN_D value that describes the P_d-R_d relation is dependent on land use type and HSG. Plots in the HSG D (i.e. fine-textured soils) were not associated with the highest CN_A values for a specific land use type however, which is probably attributable to the cracking behaviour of many fine-textured soils, which enhances infiltration and reduces runoff.

A considerable part of the unexplained variability was shown to be explained by the plot measuring station, indicating an important effect of measuring station-specific factors, implying that care should be taken when applying runoff models to other land use types or to soils with different textures or cracking properties than the ones for which they are developed or parametrised. However, apart from HSG, no such factors could be clearly identified, albeit that data on some important possible controlling factors such as vegetation cover and storm intensity are not available. As shown in the analysis of the precipitation concentration index, relating climatic effects to plot-measured R_a data is not straightforward. Hence, future research should further explore the climatic effect on R_a to increase our understanding of runoff generation in relation to precipitation characteristics.

The substantial amount of unexplained variability in the fitted relations nevertheless precludes the use of the modified CN model presented in this paper for an accurate prediction of R_a in specific field conditions. However, the modified CN model does allow the estimation of the likely R_a response to changes in land use type or vegetation type, which can be a useful tool in scenario analyses of the effects of climatic changes or land use changes on R_a.

Chapter 6

Confrontation of measured soil loss plot data with model predictions

6.1 Introduction

Recently, several model-based assessments of annual soil loss (SL_a) at a continental-wide or even global scale have been made (e.g. Kaplan and Vanwalleghem, 2012; Van Oost et al., 2007; Yang et al., 2003). Specifically for Europe, two model-based continental scale-assessments of annual soil loss (SL_a) have been published recently; the Pan-European Soil Erosion Risk Assessment (PESERA) map (Kirkby et al., 2004, 2008) and Soil Erosion Model (SEM) map (Cerdan et al., 2010).

The SEM map

The SEM map (Cerdan et al., 2010) is based on an empirical model that predicts annual soil loss (SL_a) due to sheet and rill erosion. Based on a literature review weighted mean plot-measured SL_a for different land use types and corresponding to a 100 m slope length are calculated. Per grid cell, the land use type is determined from the CORINE land cover map European Environment Agency, 1999, 2002 and weighted mean SL_a for that land use type are then corrected for the slope gradient of that grid cell derived from the SRTM (CIAT, 2004) and GTOPO30 (USGS, 2012) databases, and for soil texture and surface rock fragment characteristics derive from the European Soil Database (Panagos, 2006; Panagos et al., 2012). The resulting map has a 100x100m resolution (Fig. 6.1). While the plot-measured SL_a data compiled by Cerdan et al. (2010) correspond to a large part of the plot-measured SL_a data in this research for the same cover area, Cerdan et al. (2010) do not use the exact geographical location of the plot data for prediction of SL_a in the model, as the weighted mean of all plots for each land use type is used as a base value for prediction of SL_a in each grid cell.

The PESERA map

PESERA is a process-based model for regional soil erosion risk assessment and predicts SL_a by several processes at the hillslope scale (i.e. splash, interrill and rill erosion, and to some extent also gullying). It does not include channel delivery processes or flow routing in channels (Kirkby et al., 2008). The PESERA map (Kirkby et al., 2004) is based on a simplified version of PESERA model. The model uses climatological (MARS database; SAI-JRC, 1996), land use (CORINE database; European Environment Agency, 1999, 2002), soil characteristics (European Soil Database; Panagos, 2006; Panagos et al., 2012) and topography (EROS DEM; USGS, 2011) data available for the whole of Europe. The PESERA map model converts distributions of daily rainfall for each month to overland runoff via a bucket storage model and then uses a power law sediment model to predict SL_a delivered at the base of each individual map cell (Fig. 6.1, resolution: 1 km^2). Due to data limitations, validation of the PESERA model is restricted to internal validation of model equations and a limited external validation by comparing PESERA-predicted SL_a with plot-measured SL_a and small catchment sediment yield (Cerdan, 2003; Licciardello et al., 2009; Tsara et al., 2005; Van Rompaey et al., 2003)

Few independent validations of the PESERA and SEM maps have been made. In a comparison of the frequency distribution of plot-measured SL_a and catchment sediment yield with the frequency distribution of SL_a values predicted by the PESERA and SEM maps, Vanmaercke et al. (2012a) found that both the PESERA and SEM maps tended to under-predict plot-measured SL_a and catchment sediment yield. This was attributed to a research bias in plot-measured SL_a towards more erosion-prone situations and the non-inclusion of several important erosion and transport processes at the catchment scale, respectively (Vanmaercke et al., 2011b).

Using plot-measured data for validation of erosion models may lead to several misapplications and misconceptions (Govers, 2010). Measuring periods of runoff and soil loss plots are generally shorter (chapter 4) than the long-term mean SL_a predicted by PESERA and SEM, and runoff and soil loss plots are considerably smaller (chapter 2, Maetens et al., 2012b) than the 1 km^2 and 1 ha resolutions of the PESERA and SEM models. As it is directly based on plot-measured data, the SEM model is expected to correspond better to plot-measured SL_a data than the process-based PESERA model.

Much can be learned from a confrontation of plot-measured SL_a and model-predicted PESERA and SEM map SL_a. (1) Such confrontation gives an idea the representativeness of the PESERA and SEM maps for plot-measured SL_a rates, which aids interpretation of these maps for practical purposes such as soil conservation planning. (2) Such confrontation can indicate whether either a process-based model (PESERA) or an empirical model (SEM) performs better at a continental scale. (3) While a real validation of these models is not possible, exploration of several possible causes of differences between model output and plot measured SL_a such as land use type, slope length, slope gradient and soil characteristics can indicate areas where these models could be improved.

6.2 Material and methods

6.2.1 Annual plot soil loss database

Plot-measured SL_a data from all plot measuring stations for which the corresponding PESERA and/or SEM SL_a values could be determined (i.e. within the map cover areas; Fig. 6.1) were selected from the database described in chapter 2 (Table 6.1). Only SL_a data from plots with a land use type that is comparable to the plot measuring station land use cover on the respective versions of the CORINE map was used, as the CORINE land cover (CLC) map (European Environment Agency, 1999) is an essential part of both the PESERA map (European Environment Agency, 2002) and the SEM map (European Environment Agency, 2012). For reference purposes, bare plots were also retained, irrespective of the corresponding CORINE land use cover. This resulted in a database of 622 plots (corresponding to 3837 plot-years) from 128 plot measuring stations for the PESERA map and 687 plots (corresponding to 4197 plot-years) from 148 plot measuring stations for the SEM map (Table 6.1)

Table 6.1: Overview of number of plots (PL), number of plot-years (PY), and plot data sources for plot-measured annual soil loss data that were used in the confrontation with the Pan-European Soil Erosion Risk Assessment (PESERA; Kirkby et al., 2004) map and the Soil Erosion Model (SEM; Cerdan et al., 2010) map data.

Country	PESERA		SEM		Source
	PL	PY	PL	PY	
Albania	-	-	14	66	Grazhdani et al., 1996; Grazhdani et al., 1999; Grazhdani, personal communication
Austria	3	33	3	33	Klik, 2003; Klik, 2010; Klik, personal communication
Belgium	2	17	2	17	Bollinne, 1982; Govers and Poesen, 1988
Bulgaria	34	243	34	243	Kroumov and Malinov, 1989; Rousseva et al., 2006
Croatia	2	10	2	10	Basic et al., 2001; Basic et al., 2004
Denmark	10	41	10	41	Schjønning et al., 1995
Finland	-	-	19	82	Puustinen et al., 2005; Puustinen et al., 2007; Turtola and Paajanen, 1995; Turtola et al., 2007
France	20	166	18	164	AREDVI, 2003; Ballif, 1989; Brenot et al., 2006; Clauzon and Vaudour, 1969; Clauzon and Vaudour, 1971; Le Bissonnais et al., 2004; Martin, 1990; Martin et al., 1997; Messer, 1980; Viguier, 1993; Wicherek, 1986; Wicherek, 1988; Wicherek, 1991

Table 6.1: *Continued*

Country	PESERA		SEM		Source
	PL	PY	PL	PY	
Germany	80	251	80	251	Ammer et al., 1995; Auerswald et al., 2009; Barkusky, 1990; Botschek, 1991; Deumlich and Frielinghaus, 1994; Deumlich and Gödicke, 1989; Dikau, 1983; Dikau, 1986; Dubber, 1968; Emde, 1992; Emde et al., 2005; Engels, 2009; Felix and Johannes, 1993; Fleige and Horn, 2000; Frielinghaus, 1998; Jung and Brechtel, 1980; Richter, 1980; Richter, 1985; Saupe, 1990; Saupe, 1992; Voss, 1978
Greece	20	48	20	48	Arhonditsis et al., 2000; Diamantopoulos et al., 1996; Dimitrakopoulos and Seilopoulos, 2002; Kosmas et al., 1996
Hungary	10	52	10	52	Kertész and Huszár-Gergely, 2004; Kertesz et al., 2007; Pinczés, 1982; Richter and Kertesz, 1987; Richter, 1987
Italy	76	585	76	585	Bagarello et al., 2010a,b; Bagarello and Ferro, 2010; Basso et al., 1983a; Basso et al., 1983b; Basso et al., 2002; Caredda et al., 1997; Caroni and Tropeano, 1981; Chisci, 1989; Chisci and Zanchi, 1981; De Franchi and Linsalata, 1983; Ollesch and Vacca, 2002; Porqueddu and Roggero, 1994; Postiglione et al., 1990; Rivoira et al., 1989; Tropeano, 1984; Vacca et al., 2000; Vacca, personal communication; Zanchi, 1983; Zanchi, 1988a; Zanchi, 1988b
Lithuania	103	792	103	792	Jankauskas and Fullen, 2002, 2006; Jankauskas and Jankauskiene, 2003a,b; Jankauskas et al., 2004, 2007, 2008; Jankauskas, personal communication
Macedonia	-	-	3	15	Jovanovski et al., 1999
Poland	10	79	10	79	Gil, 1986; Gil, 1999; Rejman et al., 1998; Skrodzki, 1972; Stasik and Szafranski, 2001; Szpikowski, 1998
Portugal	46	262	46	262	de Figueiredo and Gonçalves Ferreira, 1993; de Figueiredo and Poesen, 1998; de Figueiredo, personal communication; Lopes et al., 2002; Nunes and Coelho, 2007; Roxo et al., 1996; Shakesby et al., 1994
Romania	19	503	19	503	Bucur et al., 2007; Ene, 1987; Ionita, 2000; Motoc et al., 1998; Nistor and Ionita, 2002; Teodorescu and Badescu, 1988
Slovakia	59	96	59	96	Chomanicová, 1988; Fulajtár and Janský, 2001; Gajdová et al., 1999; Stankoviansky et al., 2006
Slovenia	4	19	4	19	Horvat and Zemljic, 1998; Hrvatin et al., 2006
Spain	121	621	119	619	Andreu et al., 1998a; Andreu et al., 1998b; Andreu et al., 2001; Aspizua, 2003; Bautista et al., 1996; Bautista et al., 2007; Bienes et al., 2006; Campo et al., 2006; Cerdà and Lasanta, 2005; Chirino et al., 2006; Durán Zuazo et al., 2004; Durán Zuazo et al., 2008; Francia Martínez et al., 2006; García-Ruiz et al., 1995; Gimeno-Garcia et al., 2007; Gómez et al., 2009; González-Pelayo et al., 2010; Ingelmo et al., 1998; Lopez-Bermudez et al., 1991; Martínez-Murillo and Ruiz-Sinoga, 2007; Martinez Raya et al., 2006; Nadal-Romero and Lasanta, personal communication; Puigdefábregas et al., 1996; Romero-Díaz and Belmonte Serrato, 2008; Romero-Díaz et al., 1999; Rubio et al., 1997; Sanchez et al., 1994; Schnabel et al., 2001; Sole, personal communication; Soler et al., 1994; Soto and Díaz-Fierros, 1998; Williams et al., 1995
Sweden	-	-	4	36	Ulén, 1997
The Netherlands					Kwaad et al., 1998; Kwaad, 1991; Kwaad, 1994; Kwaad, personal communication
United Kingdom	-	-	29	164	Bhattacharyya et al., 2008; Fullen and Brandsma, 1995; Fullen, 1998; Mitchell et al., 2003; Morgan and Duzant, 2008; Quinton and Catt, 2004
Total	**622**	**3837**	**687**	**4197**	

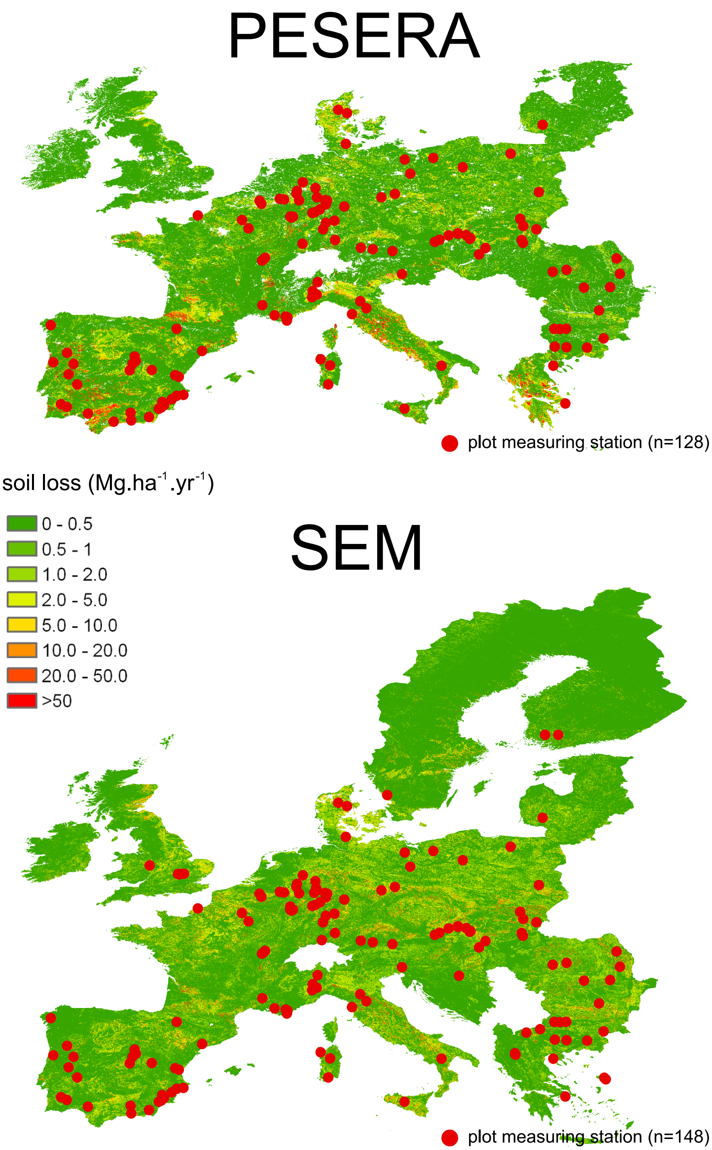


Figure 6.1: Geographical distribution of soil loss plot measuring stations from which annual soil loss data were compared with the Pan-European Soil Erosion Risk Assessment (PESERA; Kirkby et al., 2004) map and the Soil Erosion Model (SEM; Cerdan et al., 2010) map. n= number of plot measuring stations.

6.2.2 Comparing plot-measured and model-predicted annual soil loss rates

The frequency distribution of the plot measuring sites over the different soil loss classes matches the frequency distribution of those classes over the entire cover areas of the PESERA and SEM maps relatively well (Fig. 6.2). Nevertheless, the lowest soil loss class (0-5 Mg·ha^{-1}·yr^{-1}) is under-represented in both the PESERA and SEM maps, while soil loss plots in the medium erosion classes (0.5 - 5 Mg·ha^{-1}·yr^{-1}) and soil loss plots in the higher erosion classes (>2 Mg·ha^{-1}·yr^{-1}) are somewhat overrepresented in the PESERA and SEM maps, respectively. This indicates a (limited) research bias towards the measurement of plot soil loss in areas with higher model-predicted SL$_a$. On the whole, the database of plot-measured runoff and soil loss plots is a representative sample of the PESERA and SEM maps however. Nevertheless, Vanmaercke et al. (2012a) noted that there are substantial differences between frequency distributions of the plot land use type in the plot database and the fraction of area covered by those land use types over the whole of Europe. Furthermore, Cerdan et al. (2010) shows that some land use types such as vineyards are under-represented in soil loss plot measurements, while others such as shrubland are over-represented with respect to the total amount of erosion on those land uses.

Different measures of correspondence between plot-measured SL$_a$ and model-predicted SL$_a$ were evaluated:

$$R_{diff} = \frac{model\ SL_a\ -\ plot\ SL_a}{model\ SL_a\ +\ plot\ SL_a} \qquad \text{(Eq. 6.1; Nearing et al., 1999)}$$

$$\gamma = log_{10}\left(\frac{model\ SL_a}{plot\ SL_a}\right) \qquad \text{(Eq. 6.2)}$$

Where: R$_{diff}$= relative difference, γ= log$_{10}$-ratio, model SL$_a$= model-predicted (PESERA or SEM map) annual soil loss (Mg·ha^{-1}·yr^{-1}), SL$_a$= annual plot soil loss (Mg·ha^{-1}·yr^{-1}).

Many publications do not report the exact location of the plot measuring station by means of coordinates, but rather indicate the location on a map or just mention the municipality where the plots were located. Hence, there is a substantial uncertainty involved in the determination of the plot measuring station locations. To assess the impact of this uncertainty, log$_{10}$-ratios as defined in Eq. 6.2 were also calculated using the mean SL$_a$ for square areas around the plot measuring stations with a 1km, 3km, 5km, 10km and 15km resolution for the PESERA and SEM maps.

Both the relative difference (R$_{diff}$) and log$_{10}$-ratio (γ) result in positive values when the model over-predicts the plot-measured SL$_a$ and negative values when the model under-predicts the plot-measured SL$_a$, while zero represents a perfect match. However, R_{diff} is restricted between -1 and 1, while γ is open-ended. As was also noted by Vanmaercke et al. (2012a), differences between model-predicted and plot-measured SL_a were found to be frequently in the order of magnitudes and values for R_{diff} tend towards -1 and 1, while γ-values are better clustered around zero. Therefore, γ (Eq. 6.2) was used in the remainder of the analysis. Note that γ is not defined for plots where either the model-predicted SL_a or plot-measured SL_a is zero, so those data were not included in the analysis.

Differences between model-predicted and plot-measured SL_a were statistically tested using a one-sample, two-sided Wilcoxon signed-rank test (Wilcoxon, 1945), indicating whether the median γ value differs significantly from zero (i.e. over- or under-prediction by the models). This non-parametric test was chosen as a Lilliefors normality test (Lilliefors, 1967, α=0.05) indicated that the frequency distribution of the γ-values for the PESERA map did not come from a normal distribution (p=0.001). In addition, γ-values for several data subsets for the individual land use types and climatic zones were not normally distributed (α=0.05). Relations between γ and continuous variables such as plot length, slope gradient, and annual precipitation (P_a) were tested by using the Pearson correlation coefficient (r_p: Rodgers and Nicewander, 1988) and the Spearman's rank correlation coefficient (r_s: Zar, 1972). When significant correlations were found, linear regression was used to better identify the trend.

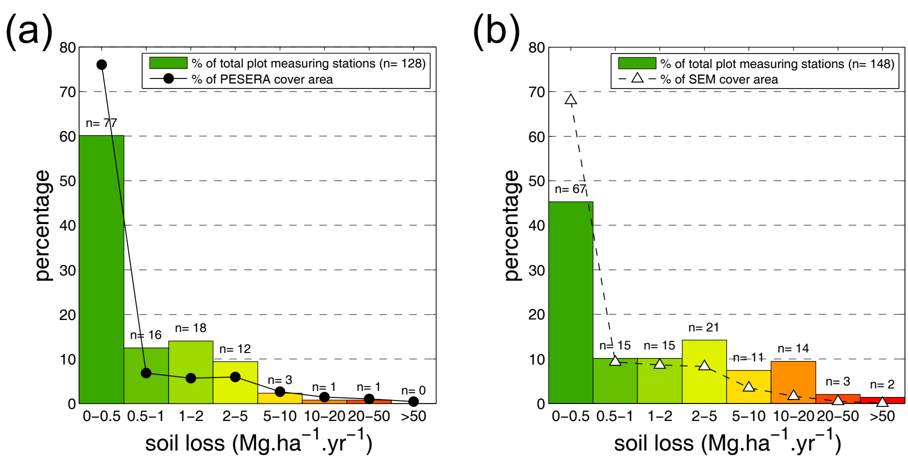


Figure 6.2: Frequency distribution of the percentage of plot measuring stations over different annual soil loss classes compared to the frequency distribution of those classes for the entire cover area for **(a)** the Pan-European Soil Erosion Risk Assessment (PESERA; Kirkby et al., 2004) and **(b)** the Soil Erosion Model (SEM; Cerdan et al., 2010). n= number of plot measuring stations.

6.3 Results and discussion

6.3.1 General correspondence between plot-measured and predicted annual soil loss rates

Fig. 6.3a shows that model-predicted SL_a values for the original map resolution (i.e. 1km for PESERA and 500m for SEM) have a tendency to under-predict plot-measured SL_a values. Not counting bare plots, the PESERA and SEM maps under-predict respectively 64.5% and 58.5% of the plot-measured SL_a, while over-predicting 34.5% and 41.6%, respectively. Similarly, Vanmaercke et al. (2012a) noted that the cumulative frequency distribution of plot-measured SL_a rates shows higher SL_a rates than the cumulative frequency distributions of both the PESERA and SEM models, which was attributed to a soil loss plot research bias towards more erosion-prone situations. However, the tendency for under-prediction is also present at the individual plot (Fig. 6.3a). Nearing (1998) has shown that erosion models have a strong tendency to over-predict small soil losses and under-predict large soil losses, due to the significant random component or unexplained variability in soil loss measurements. Due to the often right-tailed distributions of both plot-measured and model-predicted SL_a data (Fig. 6.2), under-prediction is generally more likely than over-prediction. Using the average single-storm soil loss of 40 plots (data source: Wendt et al., 1985) and the single-storm soil loss measured on a replicated plot (data source: Nearing, 1998) as predictors of individual plot soil loss, Nearing (1998) placed the transition between over-prediction and under-prediction of the majority of the plots at 6.3 $Mg{\cdot}ha^{-1}$ and 2.2 $Mg{\cdot}ha^{-1}$, respectively. This also explains the tendency for under-prediction observed in Fig. 6.3a. Also uncertainty in the determination of the location of plot measuring stations plays a role in the tendency for the PESERA and SEM maps to under-predict plot-measured SL_a. Deviations in the location determination of plot measuring stations are more likely to assign them to a cell with a smaller model-predicted SL_a rate than to a cell with a larger model-predicted SL_a due to the frequency distribution of SL_a rates in both the SEM and PESERA maps (Fig. 6.2).

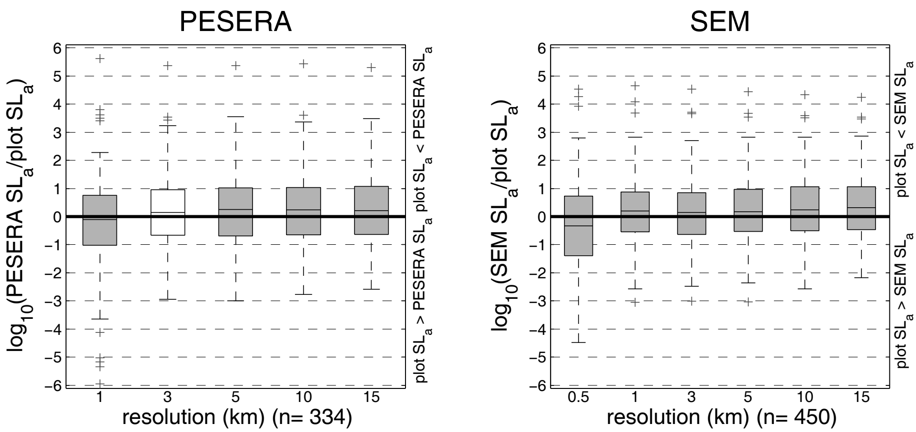


Figure 6.3a: Effect of spatial aggregation of annual soil loss rates (SL_a) on the correspondence between plot-measured and model-predicted SL_a. Bare plots are not included. Boxplots in grey are significantly different from zero according to a Wilcoxson signed-rank test (α=0.05). The legend for the box plots in this graph is given in Fig. 6.3b. n= number of plots.

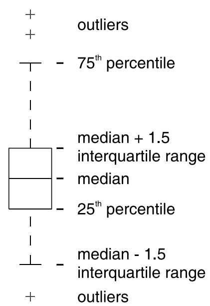


Figure 6.3b: Legend for the box plots used throughout this chapter

Using the mean of the model-predicted SL_a over square areas around the plot measuring stations eliminates the tendency for under-prediction, and averaging for even small areas introduced a tendency to over-predict measured SL_a values (Fig. 6.3a), although averaging has little effect on the variability of the γ-values. Elimination of the under-prediction bias by spatial averaging is attributed to the effects of spatial aggregation on model results (Van Rompaey et al., 1999). For the SEM map, even the smalles aggregation possible area (1 km resolution) results in median γ values that are significantly larger than zero (α=0.05), despite the median γ values still being close to zero in comparison to the total range of γ-values. This is attributed to the fact that as the area for which the mean model-predicted SL_a value is calculated increases, so does the chance of incorporating large SL_a values. These spatially aggregated SL_a rates are not direct model outcomes however. As the purpose of this analysis is to confront model-predicted SL_a values with plot-measured SL_a and explore systematic deviations between the models and the field plots rather than improve the accuracy of the predictions, the SL_a for the original model resolution was used. Furthermore, this avoids the inclusion of multiple land use types in the model-predicted SL_a values.

Fig. 6.4a gives a direct comparison between mean SL_a for all plots at each individual site Fig. 6.4. This figure shows that for any given plot measuring station, there is a considerable range of plot-measured SL_a on individual plots. As was shown by Nearing et al. (1999), considerable variability in plot-measured SL_a rates is found even between replicated plots. Thus deviations between plot-measured and model-predicted SL_a rates are caused by natural variability of plot SL_a as well as model errors. As not much is known about the factors causing natural variability in plot SL_a rates (e.g. Maetens et al., 2012b; Nearing et al., 1999), this complicates the identification of systematic model errors from the variability that is partly caused by natural variability in plot-measured SL_a, especially at large (e.g. continental) scales with very heterogeneous environmental conditions. Furthermore, even when the SL_a values represented by the PESERA and SEM maps are accurate long-term average SL_a rates, many practical applications like conservation planning would benefit from an assessment of likely deviations from the predicted SL_a values. Furthermore, Fig. 6.4a also illustrates the tendency for over-prediction of small plot-measured SL_a (i.e. $SL_a < 1$ Mg·ha^{-1}·yr^{-1}) and under-prediction of large SL_a (i.e. $SL_a >$ 1 Mg·ha^{-1}·yr^{-1}).

In Fig. 6.4b, estimated 95% confidence intervals around the mean of plot-measured mean SL_a for each individual plot are given. The 95% confidence interval for each plot was determined as the mean of the 68 estimated 95% confidence intervals on the relative error on the mean that are expected for plots with that measuring period, using the simulation of relative errors on the mean as performed in chapter 4, section 4.3.3. This figure shows clearly that while more than half of the plots-measured SL_a are under-predicted for both models, over-predictions of mean SL_a are more likely to be substantial (i.e. fall outside the 95% confidence interval around the plot-measured mean SL_a).

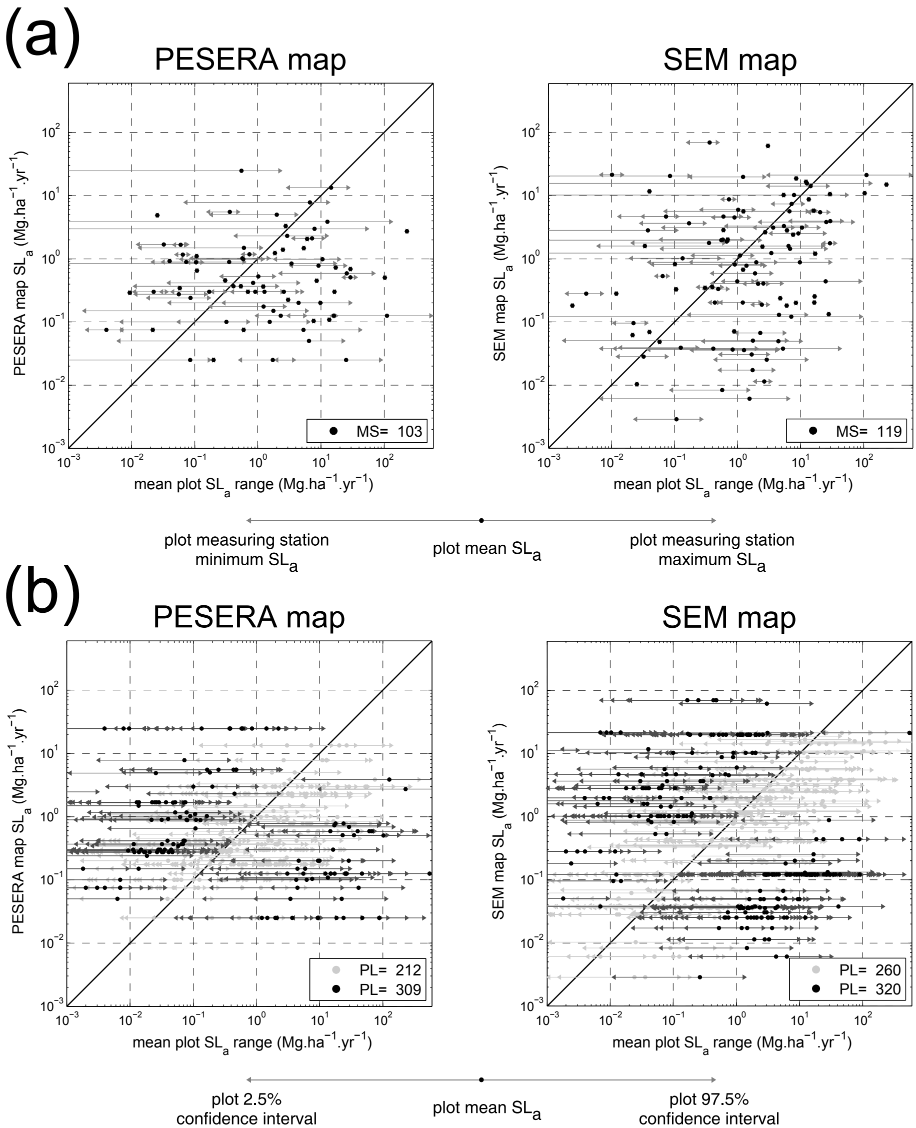


Figure 6.4: **(a)** Comparison of the range and mean plot-measured annual soil loss rates (SL_a) and model-predicted SL_a at the different plot measuring stations. MS= number of plot measuring stations. **(b)** Comparison of mean plot-measured SL_a and 95% confidence interval around the mean against model-predicted SL_a rates for individual plots. Plots where the 95% confidence interval around the plot-measured mean SL_a does not include the model-predicted value are drawn in dark grey, other plots in light grey. PL = number of plots. Bare plots are not included.

6.3.2 Effect of climatic region and land use type

Differences between model-predicted SL_a values and mean plot-measured SL_a rates for different climatic zones and land use types is given in Fig. 6.5. Due to the limited number of plot measuring stations for the Boreal (n=4), Alpine (n=2) and Steppic (n=1) climatic zones, the data are more likely to represent the specific conditions of these plot measuring stations rather than constitute a representative sample of the climatic zone and hence they were excluded from this analysis.

As expected, SL_a on bare plots are under-predicted by both the PESERA and SEM models for all climatic zones. While the erosion-sensitive bare soil in these plots does not occur as a realistic land use type, this result does show that it is possible to detect systematic deviations between plot-measured SL_a and model-predicted SL_a. The effect of climatic zone and land use type on the deviations between plot-measured and model-predicted SL_a is complex (Fig. 6.5) however. For both the PESERA and SEM maps, all significant differences between median plot-measured SL_a values in the Continental zone are under-predictions. This is attributed to climatic effects that are not included in the models such as the effect of freeze-thaw cycles (Ferrick and Gatto, 2005; Gatto, 2000). Plot-measured SL_a rates in vineyards are generally underpredicted in the Atlantic and Continental climatic regions, while they are over-predicted in the Mediterranean (Fig. 6.5). Over-prediction in the Mediterranean can be attributed to the erosion-limiting effect of surface rock fragments (Poesen et al., 1994; Poesen and Lavee, 1994).,This effect is not accounted for in the PESERA map, and while the SEM model does account for the effect of rock fragments in the soil, it does so rather arbitrarily by reducing SL_a rates by 30% when the soil contains >30% rock fragments. Under-prediction of SL_a in the Atlantic and Continental Climatic Zones is more likely attributed to an underestimation of the often steep slopes of vineyards in the Atlantic and Mediterranean (e.g. Ballif, 1989; Emde, 1992; Engels, 2009; Messer, 1980; Wicherek, 1991) when Digital Elevation Models such as SRTM (CIAT, 2004) are used. The PESERA model under-predicts SL_a on rangeland plots and over-predicts SL_a on shrubland plots in the Mediterranean, whereas this is not the case for the SEM model (Fig. 6.5). This difference may indicate that the process-based PESERA model is less capable of accurately predicting SL_a on shrubland and rangeland plots due to the complex vegetation patterns and heterogeneity in these land use types, while this problem is smaller for the SEM model as it relies on measured mean SL_a values, rather than having to estimate them.

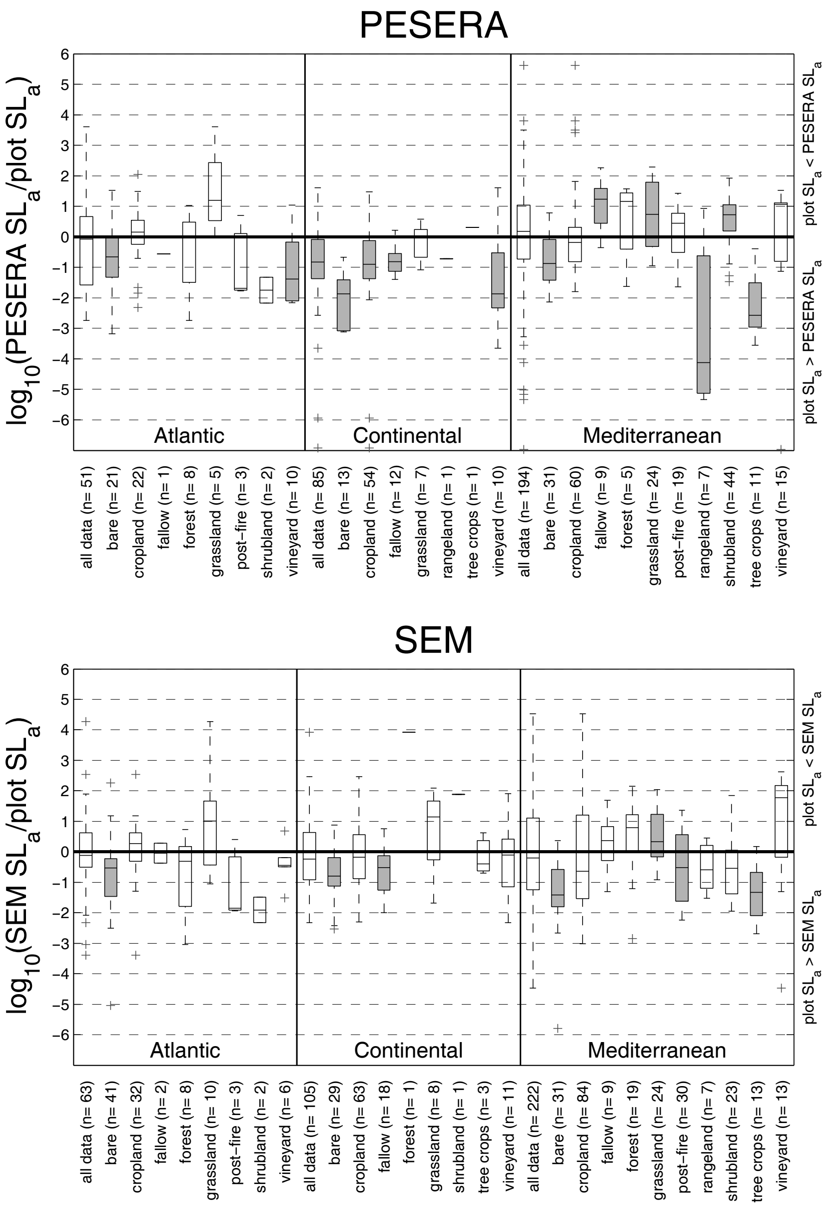


Figure 6.5: Correspondence between mean annual soil loss (SL_a) rates per plot and model-predicted SL_a rates for different land uses in the Atlantic, Continental and Mediterranean climatic zones. Boxplots in grey are significantly different from zero according to a Wilcoxson signed-rank test (α=0.05). Bare plots are not included in the box plots with all data for the different climatic zones. The legend for the box plots in this graph is given in Fig. 6.3b. n= number of plots.

Patterns of under- and over-prediction for different land use types are similar for the PESERA and the SEM model, with the exception of grassland plots in the Continental, and post-fire and shrubland plots in the Mediterranean. This is attributed to the important role of the CORINE land cover map (European Environment Agency, 1999) in both the PESERA model map and the SEM model map in the prediction of SL_a rates. The CORINE land cover map is not specifically designed for use in erosion models. As illustrated in Table 6.2, CORINE land cover for the different plot measuring stations do not always accurately reflect the land use types on the plot. For instance, the definition of the CORINE land cover *"land principally occupied by agriculture, with significant areas of natural vegetation"* (Table 6.2) clearly encompasses shrubland, grassland and forests, but the PESERA and SEM models would likely use reference values for arable land in determining SL_a for this CORINE land cover. Similarly, the CORINE land cover *"discontinuous urban fabric"* (Table 6.2) is difficult to interpret in terms of erosion sensitivity. This ambiguity in the land cover types in the CORINE map can for instance account for the over-prediction of grassland by both the PESERA and SEM maps (Fig. 6.5), as the majority of the plot measuring sites with grassland plots correspond to the *"land principally occupied by agriculture, with significant areas of natural vegetation"* and *"non-irrigated arable land"* CORINE classes, rather than to the "pastures" class. These inconsistencies between land use type on the plots and the corresponding CORINE land cover limits the interpretation of the effect of land use types on the difference between plot-measured and model-predicted SL_a. Verstraeten et al. (2003) also found that it was difficult to predict catchment sediment yield in Spain based on the CORINE database as the land cover data were not a good representation of soil cover. Nevertheless, applying a stricter interpretation of the CORINE land cover types would greatly restrict the number of available plot data and make the distribution of γ-values more likely to reflect plot measuring site characteristics other than the land use type such as soil characteristics.

Analysis of the γ-values for the individual annual plot-measured SL_a data (measuring period= 1 yr.), rather than the mean plot-measured SL_a value for the entire measuring period (Fig. 6.6), shows that both the PESERA and SEM maps tend to over-predict individual annual SL_a more than the mean SL_a values for the entire measuring period. This is a consequence of the right-tailed frequency distribution of time series of SL_a data (chapter 4). The Mediterranean climatic region is more sensitive to this effect than the Atlantic and Continental climatic regions (Fig. 6.6) due to the larger inter-annual variability in that climate. Hence, both the PESERA and SEM maps should be considered estimates of long-term annual SL_a. These regional differences between the model-predicted SL_a values and individual annual SL_a values also imply that both maps are less suitable as a prediction of likely maximum annual SL_a rates in specific regions, which is nevertheless an important consideration for conservation planning.

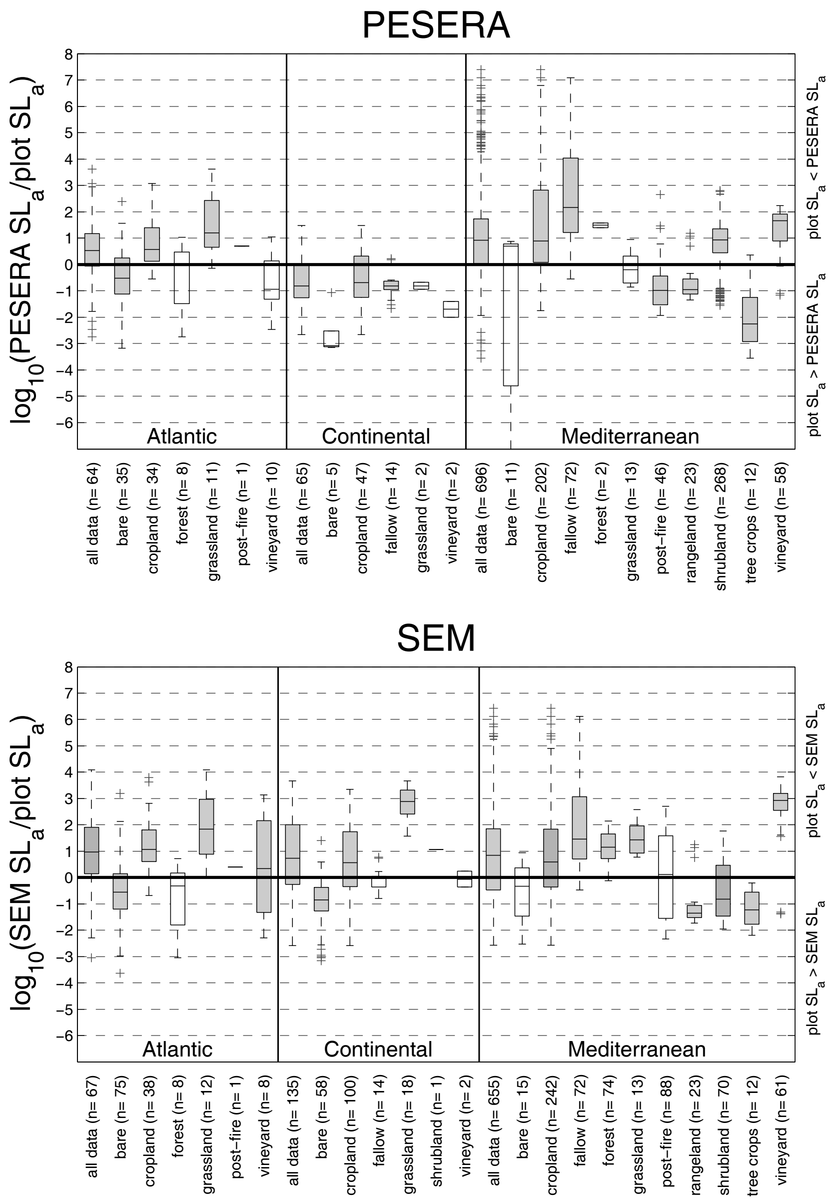


Figure 6.6: Correspondence between individual annual soil loss (SL_a) rates and model-predicted SL_a rates for different land uses in the Atlantic, Continental and Mediterranean climatic zones. Boxplots in grey are significantly different from zero according to a Wilcoxson signed-rank test (α=0.05). Bare plots are not included in the box plots with all data for the different climatic zones. The legend for the box plots in this graph is given in Fig. 6.3b. n= number of plot-years.

Table 6.2: Comparison between land use types on the soil loss plots and the corresponding CORINE land cover types (European Environment Agency, 1999). PL= number of plots.

climatic zone	land use type	CORINE land cover	PESERA PL	SEM PL
Atlantic	bare	Complex cultivation patterns	1	1
		Discontinuous urban fabric	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	2	2
		Natural grasslands	1	1
		Non-irrigated arable land	18	31
		Pastures	7	7
	cropland	Complex cultivation patterns	1	1
		Discontinuous urban fabric	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	3	3
		Non-irrigated arable land	18	28
	fallow	Non-irrigated arable land	1	2
	forest	Broad-leaved forest	8	8
	grassland	Discontinuous urban fabric	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	1	1
		Non-irrigated arable land	4	9
	post-fire	Broad-leaved forest	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	2	2
	shrubland	Land principally occupied by agriculture, with sig. areas of nat. veg.	2	2
	vineyard	Non-irrigated arable land	3	1
		Vineyards	7	7
Continental	bare	Broad-leaved forest	6	6
		Complex cultivation patterns	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	2	2
		Mineral extraction sites	1	1
		Non-irrigated arable land	12	12
		Pastures	8	8
		Vineyards	1	1
	cropland	Complex cultivation patterns	4	4
		Discontinuous urban fabric	-	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	11	11
		Non-irrigated arable land	67	67
	fallow	Natural grasslands	5	5
		Non-irrigated arable land	19	19
	forest	Mixed forest	1	1
	grassland	Complex cultivation patterns	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	4	4
		Non-irrigated arable land	5	5
		Pastures	1	1
	rangeland	Land principally occupied by agriculture, with sig. areas of nat. veg.	1	1
		Discontinuous urban fabric	-	1
		Transitional woodland-shrub	-	1
	tree crops	Discontinuous urban fabric	2	2
		Non-irrigated arable land	1	1
	vineyard	Non-irrigated arable land	4	4
		Vineyards	7	7
Mediterranean	bare	Agro-forestry areas	1	1
		Airports	1	1
		Complex cultivation patterns	8	8
		Coniferous forest	1	1
		Non-irrigated arable land	13	13
		Olive groves	1	1
		Permanently irrigated land	1	1
		Sclerophyllous vegetation	3	3
		Vineyards	3	3
	cropland	Agro-forestry areas	1	1
		Complex cultivation patterns	33	36
		Land principally occupied by agriculture, with sig. areas of nat. veg.	1	9
		Non-irrigated arable land	37	40
	fallow	Complex cultivation patterns	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	1	1
		Non-irrigated arable land	4	4
		Sclerophyllous vegetation	3	3
	forest	Broad-leaved forest	12	12
		Coniferous forest	4	4
		Land principally occupied by agriculture, with sig. areas of nat. veg.	3	3
	grassland	Complex cultivation patterns	8	8
		Land principally occupied by agriculture, with sig. areas of nat. veg.	1	1
		Non-irrigated arable land	6	6
		Sclerophyllous vegetation	9	9
	post-fire	Agro-forestry areas	1	1
		Broad-leaved forest	6	6
		Coniferous forest	14	11
		Sclerophyllous vegetation	9	9
		Transitional woodland-shrub	-	3
	rangeland	Agro-forestry areas	5	5
		Coniferous forest	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	1	1

Table 6.2: *Continued*

climatic zone	land use type	CORINE land cover	PESERA PL	SEM PL
Mediterranean	shrubland	Agro-forestry areas	1	1
		Coniferous forest	9	9
		Land principally occupied by agriculture, with sig. areas of nat. veg.	4	2
		Sclerophyllous vegetation	11	11
		Sparsely vegetated areas	19	19
	tree crops	Complex cultivation patterns	2	2
		Olive groves	3	3
		Permanently irrigated land	8	8
	vineyard	Complex cultivation patterns	1	1
		Land principally occupied by agriculture, with sig. areas of nat. veg.	1	1
		Non-irrigated arable land	3	3
		Vineyards	11	11

6.3.3 Effect of other plot characteristics

Both the PESERA and SEM model maps apply topographic information in the prediction of SL_a values. The PESERA map applies a local relief factor based on the standard deviation of elevation in a radius of 3km around each cell (Kirkby et al., 2004). The SEM map on the other hand accounts for slope gradient by applying the slope factor proposed by Nearing (1997) and subsequently correcting calculated SL_a data to 100m long slopes based on the (R)USLE slope length factor for land use types where a significant relation between plot length and plot-measured SL_a was found (Cerdan et al., 2010). Fig. 6.7 shows that the PESERA map tends to underpredict SL_a rates on longer slopes. There are no significant over- or under-predictions with respect to the slope gradient however, and relations are generally weak and explain very little additional variability. Hence, nor plot length and slope gradient are important causes of the observed variability between plot-measured and model-predicted SL_a (Fig. 6.7). The PESERA map accounts for the effect of precipitation based on the MARS climatic database (Kirkby et al., 2004), but the SEM map does not account for precipitation (Cerdan et al., 2010). Nevertheless, the relation between annual precipitation measured on the plots and γ values is weak for both the PESERA and SEM map (Fig. 6.7).

The effect of the topsoil erodibility factor, crusting factor and the resulting texture correction factor (Cerdan et al., 2010) on the relation between plot-measured model predicted SL_a is given in Fig. 6.8. Especially the cells with a topsoil correction factor of 3 are over-predicted by the SEM map, which is contrary to the overall trend for the other topsoil correction factors (Fig. 6.8). Hence, improvements to the SEM model with respect to the effects of topsoil texture can be made. For the PESERA map, not enough details on the soil properties used in the model were available to allow a full analysis.

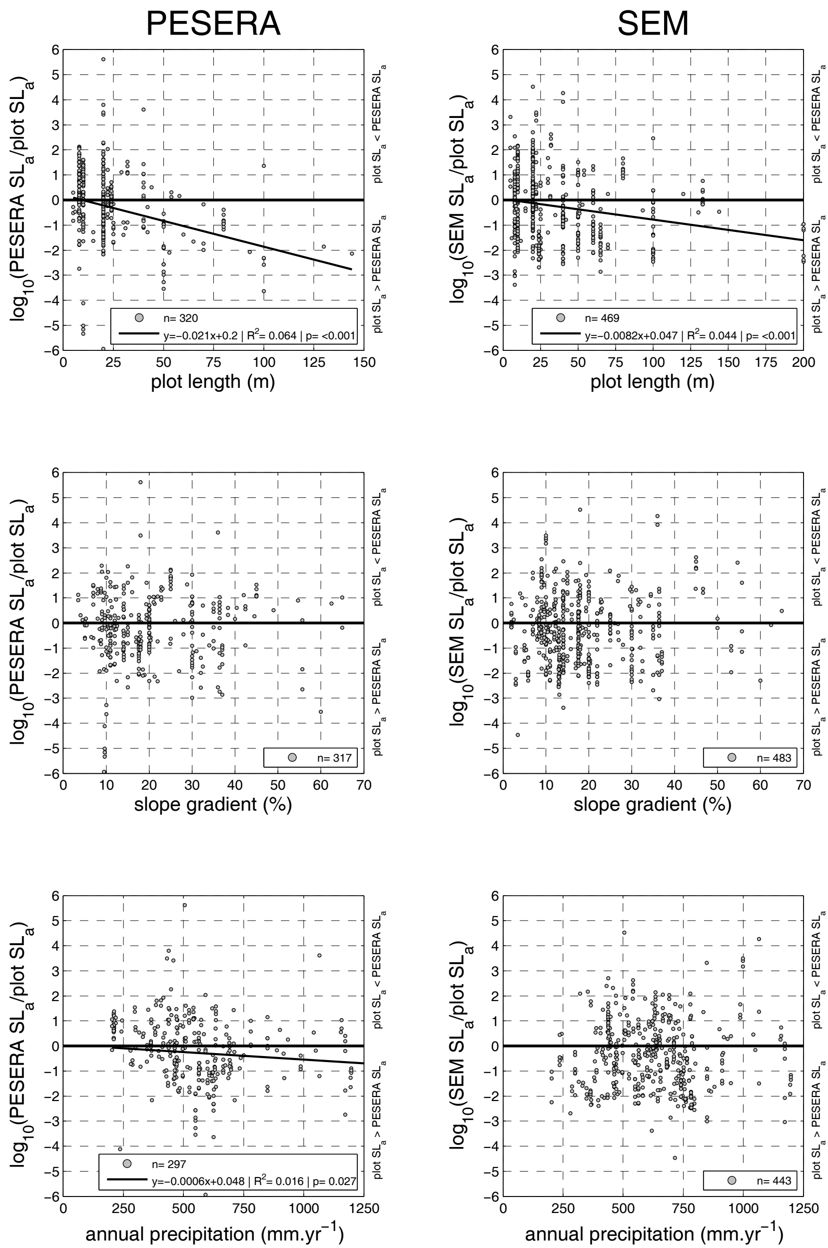


Figure 6.7: Effect of plot length, slope gradient and annual precipitation on the correspondence between plot-measured and model-predicted annual soil loss (SL_a) rates. Bare plots are not included. n= number of plots.

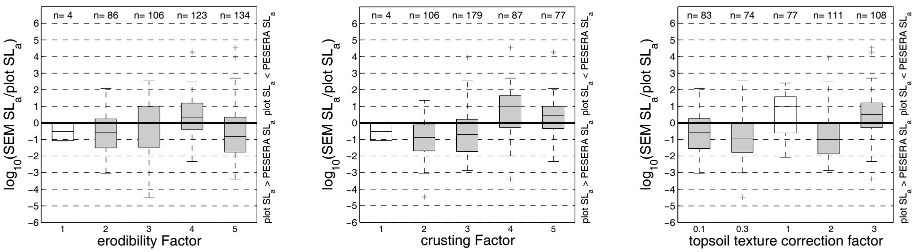


Figure 6.8: Effect of the erodibility factor and crusting factor (European Soil Database; ESDB, 2004), and topsoil texture correction factor (Le Bissonnais et al., 2005b; Cerdan et al., 2010) on the correspondence between plot-measured and model-predicted annual soil loss (SL_a) rates. Boxplots in grey are significantly different from zero according to a Wilcoxson signed-rank test (α=0.05). Bare plots are not included. The legend for the box plots in this graph is given in Fig. 6.3b. n= number of plots.

6.4 Conclusions and recommendations

This study is by no means a real validation of the PESERA and SEM maps. Nevertheless, it offers some important insights for the interpretation of the continental-wide assessments of SL_a in the PESERA and SEM maps. Confrontation of the continental-wide assessments of SL_a rates in the PESERA and SEM maps with plot-measured SL_a data showed that there is considerable variability of plot-measured SL_a rates, both when individual plots are considered as well as when only the average SL_a of all plots at the individual plot measuring station are considered (Fig. 6.4). On the whole, differences between the process-based PESERA model and the empirical SEM model are limited. Both models are more likely to under-predict the mean plot SL_a (i.e. more than 50% of plot mean SL_a are under-predicted), but over-predictions of plot mean SL_a are more likely to fall outside the 95% confidence interval around the plot-measured mean (Fig. 6.4a). The over-prediction of small plot-measured SL_a and under-prediction of large plot-measured SL_a is attributed to a significant part of unexplained random variability in SL_a measurements and was also observed by (Nearing, 1998). Both model maps should be considered to represent estimates of long-term average SL_a rates, but there is no data available to evaluate their accuracy in that respect and the large variability between model-predicted and plot-measured data has consequences for the practical applications of these maps. While they may serve to identify erosion hotspots in Europe, other techniques that are currently not available are also needed for the quantification of actual plot-scale SL_a rates on a continental scale that can be used for e.g. conservation design purposes for which the probability of large SL_a rates occurring, rather than long-term mean SL_a are needed.

With respect to the SEM model, the use of the extra plots included in this research will not affect the model predictions considerably as the weighted mean SL_a for the different land use types in this research are not considerably different from the weighted mean values obtained in this research (Table 3.2), except for some land use types such as vineyards and tree crops where the SL_a is highly variable and even after inclusion of more plots in this research, the accuracy of the weighted mean SL_a remains uncertain. However, an evaluation of the use of the erodibility factor, crusting factor and topsoil texture factor may contribute to better predictions by the SEM model. Significant added value to these maps can alsoe be created if apart from long-term mean SL_a, also the likely variability of plot-scale SL_a rates with respect to the model-predicted SL_a rates (i.e. the model uncertainty with respect to both individual annual SL_a and spatial variability within each model cell) can be assessed. The calculation of this temporal variability is possible for a process-based model using frequency distributions of daily rainfall like the PESERA map (Kirkby et al., 2004), but no literature or data on this topic are currently available. However, erosion models are susceptible to model equifinality (Govers, 2010) and correctness of the range of individual annual SL_a rates can not be inferred from a correct mean SL_a rate. Therefore it would be interesting to compare the range of SL_a rates generated by the PESERA map model with the range of plot-measured SL_a rates at specific locations.

Similarly for an empirical model such as the SEM map, quantification of uncertainty on the model-predicted SL_a rates may contribute to practical applications of this continental-scale model. This requires more detailed knowledge of the causes for uncertainty however. Confrontation between model-predicted SL_a and plot-measured SL_a in this chapter showed that SL_a values for the Continental climatic zone tend to be under-predicted (Fig. 6.5), probably due to the fact that many climate-associated controlling factors of SL_a such as freeze-thaw cycles and snowmelt erosion are not included in the PESERA and SEM map models. Furthermore, the use of the CORINE land cover map (European Environment Agency, 1999) is an important controlling factor in both the PESERA and SEM maps (Table 6.2). As CORINE is not specifically designed for the assessment of erosion, it is difficult to infer erosion sensitivity for many of the land cover types, causing large uncertainties on the model-predicted SL_a. The above-mentioned causes of variability between model-predicted and plot-measured SL_a rates are much more important at a continental scale than the effects of plot length and slope gradient (Fig. 6.7). Hence, priorities for model improvement should focus on better determination of erosion sensitivity based on land use type, identification of important climatic controls in the Continental region and assessment of the expected variability of SL_a rates at a continental scale.

Chapter 7

How effective are soil conservation techniques in reducing plot runoff and soil loss in Europe and the Mediterranean?

This chapter is based on: Maetens, W., Poesen, J. and Vanmaercke, M., 2012. How effective are soil conservation techniques in reducing plot runoff and soil loss in Europe and the Mediterranean? Earth-Science Reviews 115(1-2): 21-36. doi:10.1016/j.earscirev.2012.08.003

7.1 Introduction

Soil and Water Conservation Techniques (SWCTs) have long existed as a means to combat the detrimental effects of soil loss through interrill and rill erosion (Cerdà et al., 2009; Montgomery, 2007a; Morgan, 2005). The aim of SWCTs is to reduce both on-site runoff (R) and soil loss (SL) as well as the off-site consequences of erosion such as sedimentation of reservoirs, deterioration of water quality and flooding (e.g. Owens et al., 2005; Vanmaercke et al., 2011a; Verstraeten and Poesen, 1999). Recent research also focuses on the role of

SWCTs in the conservation of various ecosystem functions of the soil and its role in bio-geochemical cycles, including carbon sequestration (e.g. Conley, 2000). Whereas the role of SWCTs in reducing soil loss is well recognised (e.g. Boardman and Poesen, 2006; Morgan, 2005), there is still a need to integrate SWCTs effectively into good agricultural and sustainable land management practices. This need is exemplified by the goals of recent policy developments such as the European Commission's Thematic Strategy for Soil Protection (European Commission, 2012). Furthermore, several international scientific projects focus on both quantifying the effectiveness of different SWCTs in reducing R and SL as well as on their successful implementation (e.g. DESIRE, 2007; Karlen, 2008; Louwagie et al., 2009b; Römkens, 2010).

Successful SWCT application schemes are sufficiently effective in reducing R and SL to sustainable levels, while not being overdimensioned so that they are economically feasible. Implementing successful schemes therefore requires both qualitative assessments of the effects and applicability of SWCTs (e.g. Schwilch et al., 2011) as well as reliable quantitative data on of the R and SL reduction by the SWCT. The most widely used measure to quantify the effectiveness of SWCTs in reducing SL is the soil loss ratio (SLR), i.e. the ratio of SL from a plot with SWCT application and SL from a reference plot with the same characteristics but without SWCT application (e.g. Castillo et al., 1997; Cogo et al., 1984; Gilley and Risse, 2000; Smets et al., 2008a). SLR values are similar to the widely used (R)USLE cover management (C) and support practice (P) factor (Renard et al., 1997). However, the calculation of C- of P-factors for specific soil conservation techniques is not straightforward and the validity of the empirical relations for C- and P-factors given by Renard et al. (1997) outside the Midwestern U.S.A. is uncertain. Quantification of SWCT effectiveness for other regions requires local measurements of SLR (e.g. Hessel and Tenge, 2008).

Furthermore, C- and P-factors apply only to SL and not to R. While runoff ratios (RR), the equivalent of SLR, have been used in some studies (e.g. Gilley and Risse, 2000), quantification of SWCT effectiveness remains mainly oriented at SL. Nevertheless, the term "soil *and water* conservation techniques" implies that also an effect on runoff is expected or desired. Despite the limited attention, runoff reduction remains an important concern. On-site, conservation of plant-available water is an important issue for agricultural production (Rockström et al., 2010; Wallace, 2000) and may be a more important concern than soil loss, e.g. in areas where water is a key resource. Furthermore, sediment yield at the catchment scale is in many cases strongly controlled by the occurence and magnitude of a few flood events (e.g. Gonzalez-Hidalgo et al., 2010). Hence, runoff reduction is a crucial part of integrated catchment management (Nyssen et al., 2010; Vanmaercke et al., 2010; Verstraeten and Poesen, 1999). In addition, runoff generation and soil loss for various land use types are closely related

(Maetens et al., 2012b) and information on the effectiveness of SWCTs in reducing R can also improve insights in their effectiveness in reducing SL.

There are also strong indications that the effectiveness of SWCT depends on environmental factors such as land use, saturated conductivity and storm size (Hessel and Tenge, 2008) or plot slope length (e.g. Gilley and Risse, 2000; Smets et al., 2008b,a) and plot slope gradient (e.g. Renard et al., 1997; Syversen, 2005). Nevertheless, very few quantitative assessments of the effects of these environmental factors on SWCT effectiveness in reducing R and SL have been made. Limited understanding of environmental effects on SWCT effectiveness in reducing R and SL also limits the incorporation of SWCT application in erosion models. (e.g. Hessel and Tenge, 2008).

Finally, a comprehensive assessment of the effectiveness of SWCTs also needs to consider temporal aspects of SWCT application: the temporal variability in SWCT effectiveness and how this effectiveness evolves over the years since the initial application. The latter has been studied for the build-up of soil organic carbon (Hao et al., 2002), soil biochemical properties (Madejón et al., 2009) and crop yield (Rusinamhodzi et al., 2011; Van den Putte et al., 2010). However, no such study exists with respect to the long-term effects of SWCTs on R or SL.

An overview and meta-analysis of available field-measured data on the effectiveness of various SWCTs in reducing both R and SL can provide important additional insights and can improve our ability to model the effects of SWCT in reducing R and SL under various conditions. However, relatively few comprehensive overviews are currently available. A global assessment of SWCT effectiveness in reducing SL was made by Montgomery (2007b), but this analysis does not include R nor does it allow a quantification of the effectiveness of specific techniques. For Europe and the Mediterranean, available overviews of SWCTs effectiveness are very limited (Table 1.1) Several of the available overviews of erosion rates and their controlling factors do not consider SWCTs explicitly (Table 1.1). Furthermore, the reviews that consider SWCTs often do not include the effectiveness of SWCTs in reducing R. Studies that do consider R are limited to a few specific techniques (Table 1.1).

Therefore, the objectives of this paper are (1) to provide an overview of field plot data on effectiveness of SWCT in reducing annual runoff (R_a, mm·yr^{-1}) and annual soil loss (SL_a, Mg·ha^{-1}·yr^{-1}) in Europe and the Mediterranean, (2) to quantify the effectiveness of different SWCT types in reducing both R_a and SL_a and to explore the effect of SWCTs on the relations between R_a and SL_a and (3) to explore the relations of SWCT effectiveness with some important variables that were reported in the experimental studies (i.e. magnitude of R_a and SL_a, plot length, plot slope gradient, annual precipitation (P_a, mm·yr^{-1}), and the number of consecutive years of SWCT application).

7.2 Data collection

Annual runoff and soil loss data, measured on bounded plots where a SWCT was applied, for Europe and the Mediterranean were collected from research papers, books, project reports and PhD. theses (Fig. 7.1). Each plot represents a combination of a soil type, a plot length, a slope gradient, and a land use type and is associated with one type of SWCT (Table 2.1). SWCTs were classified into three groups according to Morgan (2005) (Table 2.1). Only runoff and soil loss measurements from bounded runoff plots under natural rainfall, with a minimum length of 5m were retained. Only annual data are considered: either plot data were collected during at least a full year, or the reported data could be extrapolated to represent a full year with a sufficient degree of reliability, i.e. when measurements were conducted for at least 80% of the year and rainfall was uniformly distributed throughout the year (Maetens et al., 2012b), or when authors indicated that the measurements were representative for a full year. For each plot, the corresponding number of plot-years was determined, whereby 1 plot-year corresponds to a measuring period of 1 year on a single runoff plot. Most plots were equipped with tanks for collecting runoff and soil loss. However, for a small number of plots (n=27), soil loss was determined by measuring rill volumes (Feiza et al., 2007; Jankauskas and Jankauskiene, 2003b; Jankauskas et al., 2007). These volumetric measurements were converted to SL_a data (in $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$) by assuming a soil bulk density of 1.5 $g{\cdot}cm^{-3}$ and by adding 25% to account for interrill erosion. This value was based on a literature review by Govers and Poesen (1988). If available, also the plot length, plot gradient, annual precipitation and measuring period were included in the data compilation.

7.3 Characteristics of the collected plot data

Fig. 7.1 shows the SWCTs that have been tested in Europe and the Mediterranean. Relatively few plot data are available for Northern latitudes and for Eastern European countries compared to other regions. This may be partly explained by the fact that results from most plot studies in Eastern Europe are only reported in the local language and have not been published internationally. Nevertheless, this study represents the largest compilation of SWCT effectiveness measurements in Europe and the Mediterranean. An overview of the collected data per country is given in Table 7.1. Analyses of these data revealed no clear regional differences in the type of SWCTs evaluated (Table 7.1).

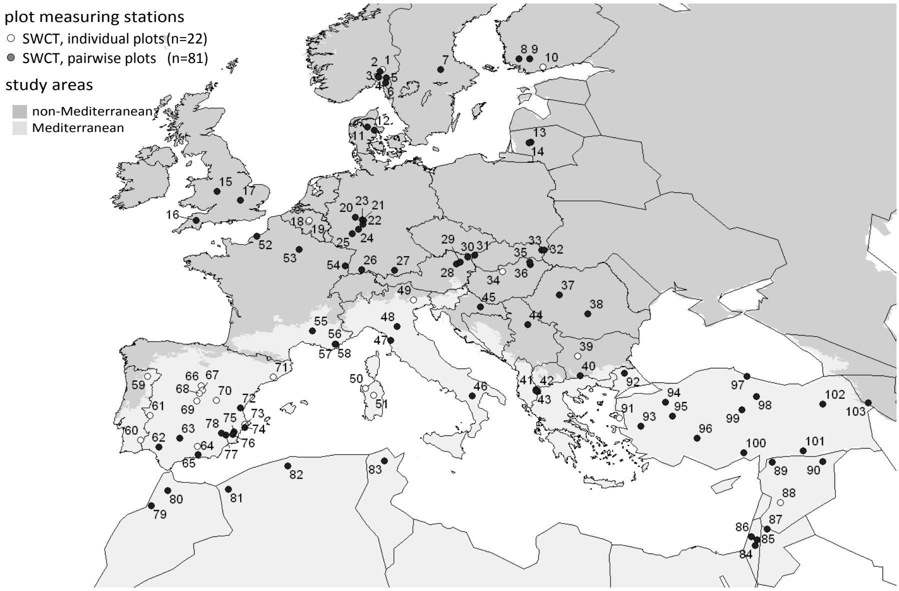


Figure 7.1: Geographical distribution of runoff and soil loss plot measuring stations in Europe and the Mediterranean for the individual SWCT plot database (i.e. plots where SWCTs were applied without a control plot with conventional practice) and pairwise SWCT plot database (i.e. SWCT plots were also a control plot with conventional practice was applied). Black circles represent stations included in the paired plot database, while open circles represent stations which are only included in the individual plot database. The division between Mediterranean and Non-Mediterranean was derived from the LANMAP2 database (Mücher et al., 2010; Metzger et al., 2005). n= number of plot measuring stations.

Table 7.1: Overview of the different soil and water conservation techniques (SWCTs) included in the global plot database and publications from which annual runoff and soil loss data were extracted in each country in Europe and the Mediterranean. References followed by * report data for plots where only SWCTs were applied, other references contain plots included in the paired plot database. References followed by ^{TS} were used for the analysis of plot data time-series (consecutive years) of SWCT application. Numbers between brackets refer to the plot measuring stations indicated in Fig. 7.1.

Country	SWCT	Source
Albania	no-tillage	Grazhdani et al., 1999 [42]
	reduced tillage	Grazhdani et al., 1999 [42]
	drainage	Grazhdani et al., 1996 [41-43]; Grazhdani, pers. comm. [41-43]
Algeria	cover crops	Arabi and Roose, 1993 [82]
	buffer strips	Arabi and Roose, 1993 [82]
	exclosure	Mazour, 1992 [81]; Morsli et al., 2004 [81]
	no-tillage	Arabi and Roose, 1993 [82]
	contour tillage	Mazour, 1992 [81]; Mazour et al., 2008^{TS} [81]
Austria	cover crops	Klik, 2003 [28-30]; Klik, 2010; Klik, pers. comm.
Belgium	cover crops	Laloy and Bielders, 2008*;Laloy and Bielders, 2010* [18,19]
	no-tillage	Laloy and Bielders, 2008*;Laloy and Bielders, 2010* [18,19]
	reduced tillage	Laloy and Bielders, 2008*;Laloy and Bielders, 2010* [18,19]
Bulgaria	buffer strips	Biolchev, 1975* [39]; Malinov, 1999* [39], cited by Rousseva et al., 2006
	soil amendment	Kroumov and Malinov, 1989 [40]
Croatia	no-tillage	Basic et al., 2001^{TS},Basic et al., 2004^{TS} [45]
	contour tillage	Basic et al., 2001, 2004^{TS} [45]
Denmark	cover crops	Schjønning et al., 1995* [11,12]
	reduced tillage	Schjønning et al., 1995* [11,12]
Finland	cover crops	Turtola et al., 2007* [9]
	buffer strips	Puustinen et al., 2005, 2007 [8]; Uusi-Kämppä, 2005 [9]
	no-tillage	Puustinen et al., 2005, 2007 [8]; Turtola et al., 2007^{TS} [9]
	reduced tillage	Puustinen et al., 2005, 2007 [8]; Turtola et al., 2007^{TS} [9]
	contour tillage	Puustinen et al., 2005, 2007 [8]
	drainage	Warsta et al., 2009* [10]
France	cover crops	Messer, 1980 [54]; Viguier, 1993 [56-58]
	mulching	Ballif, 1989 [53]; Viguier, 1993 [56-58]
	buffer strips	AREDVI, 2003 [55]; Le Bissonnais et al., 2004 [52]
	no-tillage	Messer, 1980 [54]; Viguier, 1993^{TS} [56-58]
	reduced tillage	Ballif, 1989 [53]
Germany	cover crops	Emde, 1992 [25]; Jung and Brechtel, 1980 [20-23,26-27]
	buffer strips	Jung and Brechtel, 1980* [20-23,26-27]; Voss, 1978 [24]
	reduced tillage	Jung and Brechtel, 1980* [20-23,26-27]
	contour tillage	Jung and Brechtel, 1980* [20-23,26-27]
Greece	terraces	Koulouri and Giourga, 2007* [91]
Hungary	contour tillage	Hudek and Rey, 2009* [34]; Pinczés, 1982 [36]
	contour bunds	Pinczés, 1982 [36]
	geotextile	Kertesz et al., 2007 [35]; Kertesz, pers. comm. [35]
Israel	soil amendment	Agassi and Benhur, 1991 [86]; Agassi et al., 1990 [86]
Italy	cover crops	Bini et al., 2006* [49]; Caredda et al., 1997* [50]
	mulching	Rivoira et al., 1989* [51]
	no-tillage	Postiglione et al., 1990 [46]
	reduced tillage	Basso et al., 1983a^{TS} [46]; Chisci and Zanchi, 1981* [47]; Chisci, 1989 [47,48]; Postiglione et al., 1990 [46]
	drainage	Chisci and Zanchi, 1981 [47]
	geotextile	Zanchi, 1983 [48]
Jordan	mulching	Abu-Zreig, 2006 [87]; Abu-Zreig et al., 2011 [87]
Lithuania	cover crops	Jankauskas and Jankauskiene, 2003b* [13]; Jankauskas et al., 2007* [13]
	no-tillage	Feiza et al., 2007 [13]
	reduced tillage	Feiza et al., 2007 [13]
	geotextile	Jankauskas et al., 2008 [14]
Morocco	reduced tillage	Heusch, 1970 [80]
	contour tillage	Laouina et al., 2003^{TS} [79]
Norway	reduced tillage	Børresen, pers. comm. [34]; Grønsten and Lundekvam, 2006^{TS} [34]; Lundekvam, 2007^{TS} [2,5-6]
	drainage	Øygarden, 1996 [1]; Øygarden et al., 1997* [1]

Table 7.1: *Continued*

Country	SWCT	Source
Palestinian territories	terraces	Abu Hammad et al., 2004, 2006 [85]; Al-Seekh and Mohammad, 2009 [84]
Portugal	cover crops	Roxo et al., 1996* [60]
	no-tillage	Oliveira, 2005* [59]
	contour tillage	Oliveira, 2005* [59]
Romania	no-tillage	Nistor and Ionita, 2002 [37]
	reduced tillage	Nistor and Ionita, 2002 [37]
	terraces	Teodorescu and Badescu, 1988 [38]
Serbia	terraces	Đjorović, 1990 [44]
Slovakia	mulching	Chomanicová, 1988 [33]
	no-tillage	Fulajtár and Janský, 2001 [31]
	reduced tillage	Chomanicová, 1988 [33]; Suchanic, 1987 [32]
	deep tillage	Chomanicová, 1988 [33]; Suchanic, 1987 [32]
	soil amendment	Chomanicová, 1988 [33]
Spain	cover crops	Gómez et al., 2009 [62]; Schnabel et al., 2001* [61]
	mulching	Albaladejo et al., 2000 [78]; Bautista et al., 1996 [74]
	buffer strips	Aspizua, 2003* [64]; Francia Martínez et al., 2006 [65]; Gómez et al., 2004 [63]; Martinez Raya et al., 2006* [65]
	exclosure	Castillo et al., 1997 [78]; Martinez-Mena et al., 1999 [76]
	reduced tillage	Bienes et al., 2006* [67-70]; Francia Martínez et al., 2006 [65]; Gómez et al., 2004^{TS} [63]
	contour tillage	Schnabel et al., 2001* [61]
	soil amendment	Ingelmo et al., 1998 [72]; Lopez-Bermudez et al., 1991 [77]; Romero-Díaz et al., 1999 [77]
	terraces	Llovet et al., 2009* [73]; Llovet, pers. comm. [73]; Martínez-Casasnovas and Ramos, 2006* [71]; Ramos and Martínez-Casasnovas, 2006, 2007 [71]; Sanchez et al., 1994 [75]; Williams et al., 1995* [66]
Sweden	cover crops	Ulén and Kalisky, 2005 [7]
	no-tillage	Ulén and Kalisky, 2005 [7]
	reduced tillage	Ulén and Kalisky, 2005 [7]
Syrian Arab Republic	cover crops	Bruggeman et al., 2005 [89]; Masri et al., 2005 [89];
	exclosure	Shinjo et al., 2000 [90]
	reduced tillage	Bruggeman et al., 2005^{TS} [89]; Masri et al., 2005 [89];
	terraces	Ali, 2007* [88]
Tunisia	contour tillage	Kaabia, 1995 [83]
Turkey	strip cropping	Köse and Taysun, 2002 [93]; Köse et al., 1996 [93];
	contour tillage	Köse and Taysun, 2002^{TS} [93]; Köse et al., 1996^{TS} [93]; Oguz et al., 2006^{TS} [92,94-103]; Oguz, pers. comm. [92,94-103]
U.K.	mulching	Brown, 1996 [16]
	reduced tillage	Brown, 1996 [16]; Quinton and Catt, 2004 [17]
	contour tillage	Quinton and Catt, 2004 [17]
	geotextile	Bhattacharyya et al., 2008, 2009 [15]; Mitchell et al., 2003 [15]

SWCTs have been tested on runoff plots in Europe and the Mediterranean since 1956 (Jung and Brechtel, 1980). Since then, the number of plots gradually increased and peaked between 1987 and 1997 (Fig. 7.2a). This peak is considerably later than the majority of runoff plot measurements in the USA (between 1930 and 1942; Laflen and Moldenhauer, 2003) and the majority of catchment sediment yield studies in Europe (Vanmaercke et al., 2011b). The decline in the number of plots after 1997 is partly due to the termination of some large research projects (e.g. MEDALUS). Furthermore, recently collected plot data may not have been published yet. However, measuring runoff and soil loss from plots is labour- and cost-intensive and the decline in plot data collection may also be linked to the substitution of field measurements studies by modelling studies.

Nevertheless, interest in field-measured data on the effects of SWCTs remains high as shown by the increase of the number of publications in recent years. Recent publications also review and re-analyse plot data collected over long periods which were never published internationally (e.g. Rousseva et al., 2006). Fig. 7.2b shows that, while soil management (i.e. mainly tillage) and mechanical methods have been studied continually respecitively since 1956 (Jung and Brechtel, 1980) and 1962 (Pinczés, 1982), crop and vegetation management methods have only been studied significantly since 1984. This most likely reflects changing attitudes towards sustainable agricultural practices. The increase in interest in the effects of mechanical methods since 1990 is reflected in an increase in research on the effects of terracing on R_a and SL_a in reforested land (Llovet et al., 2009; Sanchez et al., 1994; Williams et al., 1995), and to the use of (biological) geotextiles (Bhattacharyya et al., 2008, 2009; Jankauskas et al., 2008; Kertesz et al., 2007; Mitchell et al., 2003).

Fig. 7.3 shows the frequency distribution of the number of plots and number of plot-years over different plot slope lengths, slope gradients and annual precipitation depths. Soil management techniques (i.e. mostly tillage techniques) were mainly tested on gentle slopes while mechanical methods have been tested on steeper slopes. The frequency distribution of plot slope lengths and slope gradients is similar to the distribution for plots where no SWCTs are applied (Maetens et al., 2012b), and is likely determined by experimental preferences, as the modal plot length and slope gradient are close to the standard (R)USLE plot which has a plot length of 22.13m and a slope gradient of 9% (Renard et al., 1997). The concentration of plot lengths and slope gradients around these values also implies that effects of plot length and slope gradient on SWCT effectiveness in reducing R_a and SL_a are difficult to assess for longer and steeper slopes.

Other important environmental data (e.g. soil characteristics such as texture, organic matter content) and climatological data (precipitation distribution and intensity) were not reported systematically in the reviewed literature and no global analysis of these variables was possible. Furthermore, the number of plots for some of the studied SWCTs is limited, so no further division is made according to land use types (e.g. cropland, tree crops, vineyards) as this would reduce the plot sample size. These data limitations imply that there is likely a large variability in R_a and SL_a which can not be explained with the environmental data currently available, and any observed trends should be interpreted with care, considering possible effects of these unknown factors.

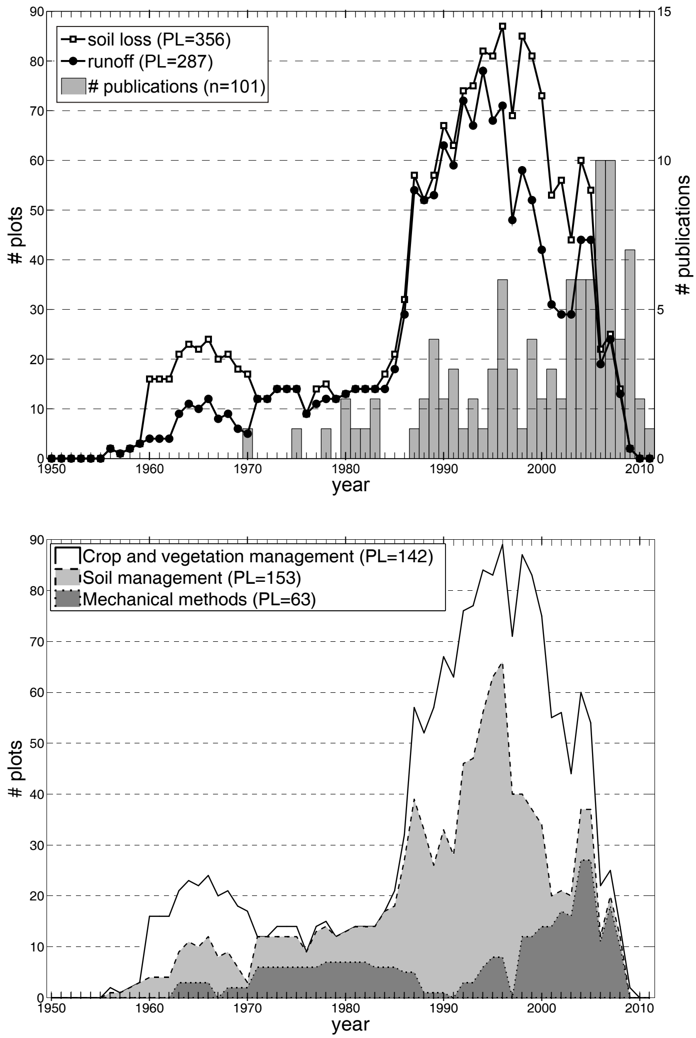


Figure 7.2: (a) Per-year evolution of the total number of erosion plots (# plots) set up to test soil and water conservation techniques (SWCTs) for which annual runoff and soil loss were recorded based on the individual plot (IP) database for Europe and the Mediterranean (Table 7.1). Grey bars represent the total number of publications (# publications) per publication year from which runoff and/or soil loss data were extracted. **(b)** Evolution of number of plots (# plots) per year for which annual soil loss and runoff were recorded on plots with a SWCT based on the IP database. The full black line indicates the total number of plots, while dashed lines and grey tones indicate the relative proportions of the three major SWCT groups (Table 2.1). n= number of publications, PL= number of plots.

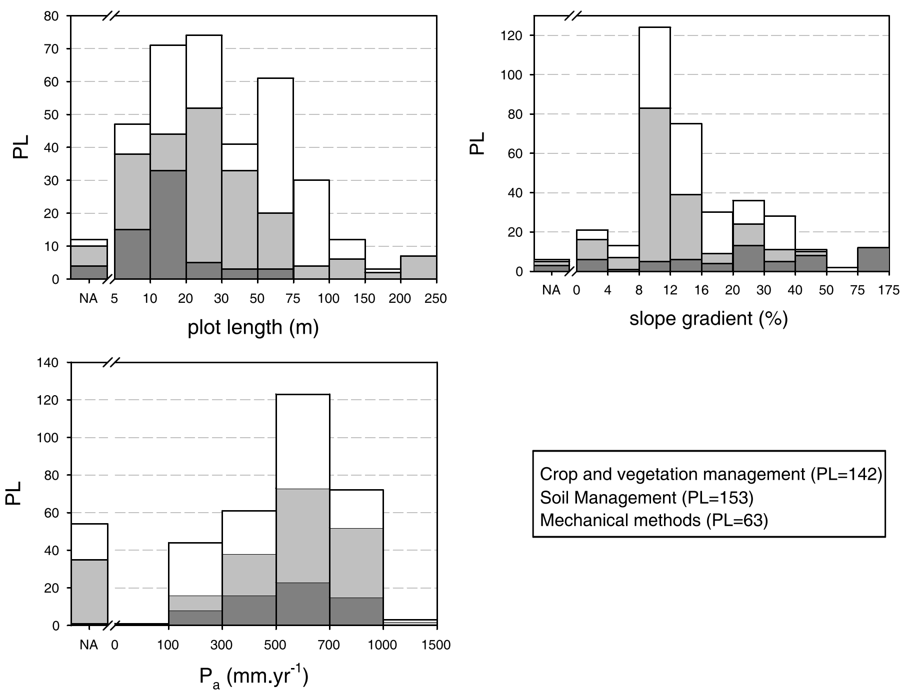


Figure 7.3: Frequency distribution of the number of plots (PL) included in the individual plot database for Europe and the Mediterranean for which annual runoff and/or soil loss data are available as a function of: **(a)** plot length, **(b)** slope gradient and **(c)** annual precipitation depth (P_a). A distinction is made between the three major SWCT groups (Table 2.1). PL: number of plots. NA: not available, i.e. plot length,slope gradient or Pa were not reported.

7.4 Effectiveness of SWCTs in reducing annual runoff and soil loss

In order to obtain a general overview of the effects of SWCT, the frequency distributions of annual runoff coefficient (RC_a, %) and SL_a, from all plots where SWCTs were applied were compared to frequency distributions of RC_a and SL_a measured on plots where local conventional land management practices were applied (Fig. 7.4). The latter were derived from a previously established erosion plot database (Maetens et al., 2012b). This approach allows a broad assessment of the overall effects of SWCTs on RC_a and SL_a, but cannot be used to quantify the effectiveness of individual SWCTs. Therefore, all plots with SWCTs for which a paired plot under local conventional management existed (i.e. a plot on the same measuring site with the same characteristics but with application of the local conventional management instead of a SWCT) were selected from the full SWCT plot database (individual plot database, IP) and grouped together in a separate subset (paired plot database, PP).

7.4.1 Overall effectiveness of SWCTs

Fig. 7.4 does not show a clear effect of the application of SWCTs on the exceedance probability distribution of RC_a. While RC_a-values for plots with a (semi-)natural vegetation cover are clearly smaller than RC_a-values for bare plots or plots under conventional cropland practices, this difference in RC_a is not observed for cropland plots where a SWCT was applied. On the other hand, Fig. 7.4b shows that the presence of a crop on the land already substantially reduces the exceedance probability of any given SL_a rate compared to the exceedance probability for SL_a from bare plots, while the application of SWCTs reduces the exceedance probability even further.

The observed exceedance probabilities correspond to the trend observed by Montgomery (2007) in a worldwide study of SL_a rates. Nevertheless, the reduction of the exceedance probability of any given SL_a rate by the application of SWCTs is generally smaller in Europe and the Mediterranean (Fig. 7.4b) than the reduction observed by Montgomery (2007). The latter reported that the exceedance probability of a soil loss tolerance level (T-value) of 5 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ drops by 75% when SWCTs are applied, compared to SL_a rates on cropland without SWCTs (Table 7.2). However, for Europe and the Mediterranean (this study) a reduction of only 14% is observed (Table 7.2). The larger effect of SWCT application on SL_a rates in the global study (Montgomery, 2007b) is due to the fact that SL_a rates on cropland reported by Montgomery (2007b) are generally much larger than those observed in this study (chapter 3).

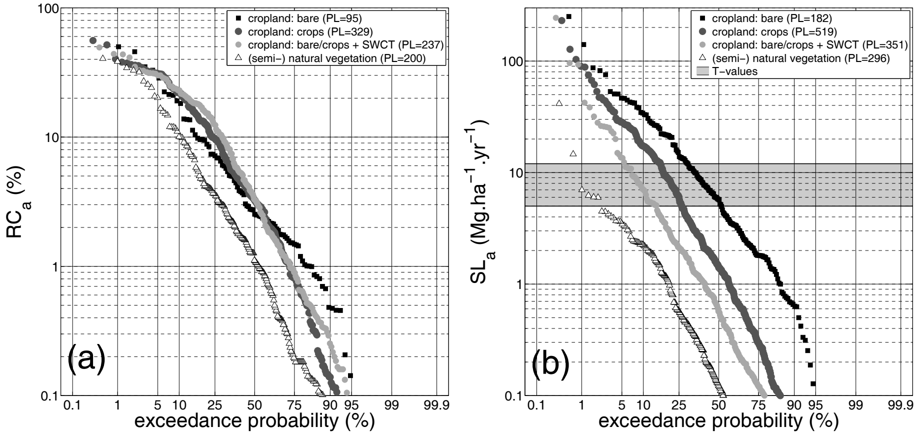


Figure 7.4: Exceedance probability distributions for the combined individual and paired plot databases for **(a)** annual runoff coefficients (RC_a) and **(b)** annual soil loss (SL_a) for bare plots, cropland plots (annual crops, olive orchards and vineyards) without soil and water conservation techniques (SWCTs), plots on land with semi-natural vegetation (forest, shrubland and grassland; obtained from the plot runoff and soil loss database described in (Maetens et al., 2012b), and all cropland plots with SWCT application (this study, Table 7.1). For SL_a, the range of tolerable soil losses (T-values, i.e. 5-12 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$) as defined by (Montgomery, 2007b) is indicated in grey. PL= number of plots.

These larger SL_a rates may be attributed to the inclusion of more erosion-prone sites in the global study (Montgomery, 2007b) as compared to the erosion plot sites in Europe and the Mediterranean (this study), but also to differences in measuring methods and spatial scale as the study by Montgomery (2007b) was not restricted to soil loss plots. In addition, the reported SL_a rates from land where SWCTs were applied are in general somewhat smaller in the global study by Montgomery (2007b) than in this study (Table 7.2). This difference may be attributed to the comparatively smaller number of plots where SWCTs were applied in the study by Montgomery (2007b), and thus the lower probability of observing extreme SL_a rates in that study.

7.4.2 Effectiveness per technique

To quantify the effectiveness of specific SWCTs, runoff ratios (RR) and soil loss ratios (SLR) were calculated for all paired plots (PP) in the database. Note that in these equations a plot with conventional land use management is used as a reference, whereas for RUSLE P-factors (Renard et al., 1997), bare plots

Table 7.2: Exceedance probabilities of annual soil loss on cropland without SWCTs and cropland with SWCTs for the worldwide study by Montgomery (2007b) and for Europe and the Mediterranean (this study), corresponding to two soil loss tolerance values (T). (see also Fig. 7.4).

	T= 5 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$		T= 12 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$	
	World (Montgomery, 2007b)	Europe and the Mediterranean (this study)	World (Montgomery, 2007b)	Europe and the Mediterranean (this study)
cropland without SWCT	85%	27%	62%	16%
cropland with SWCT	10%	13%	0%	4%
difference	75%	14%	62%	12%

are used as a reference to compare against.

$$RR = \frac{R_{aSWCT}}{R_{aCONV}} \quad \text{(Eq. 8.1)}$$

$$SLR = \frac{SL_{aSWCT}}{SL_{aCONV}} \quad \text{(Eq. 8.2)}$$

where:

R_{aSWCT} = annual runoff ($mm{\cdot}yr^{-1}$) measured on the plot with SWCT.

R_{aCONV} = annual runoff ($mm{\cdot}yr^{-1}$) measured on the plot with conventional practice.

SL_{aSWCT} = annual soil loss ($Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$) measured on the plot with SWCT.

SL_{aCONV} = annual soil loss ($Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$) measured on the plot with conventional practice.

Except for deep tillage and terraces, the median effectiveness of all other SWCTs in reducing R_a is much smaller than for SL_a (Fig. 7.5). Deep tillage is a technique aimed mainly at reducing runoff through increasing infiltration capacity (Chomanicová, 1988; Suchanic, 1987). For terraces, the higher median effectiveness in reducing R_a than in reducing SL_a is explained by the fact that the terraces were established as water harvesting technique in an arid climate (Al-Seekh and Mohammad, 2009). The higher effectiveness of SWCTs in reducing SL_a than in reducing R_a can be attributed to the fact that there are physical limitations to the increase of soil infiltration rates and thus reduction of R_a by SWCT, while the reduction of SL_a is not limited physically. The ranking of effectiveness for the individual SWCTs is different for R_a and SL_a reduction. Nevertheless, mechanical methods and crop and vegetation methods are generally more effective in reducing both R_a and SL_a than soil management methods. Exceptions to this trend are exclosures and strip cropping, which are less effective than the other SWCTs in the crop and vegetation management group.

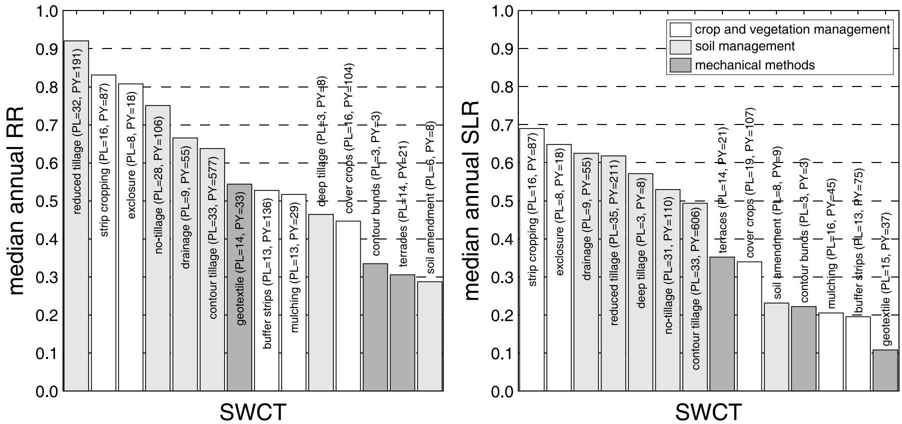


Figure 7.5: Median annual runoff ratios (RR, Eq. 8.1) and soil loss ratios (SLR, Eq. 8.2) for different soil and water conservation techniques (SWCTs) included in the paired plot database. SWCTs are ranked from left to right in order of increasing effectiveness. PL= number of plots, PY= number of plot-years.

The relative order of SWCTs in terms of RR and SLR, underlines the large effectiveness of vegetation cover (cover crops, buffer strips), soil contact cover (mulching, geotextiles), or physical barriers (terraces, contour bunds) to reduce R_a and SL_a. While tillage methods are less effective in reducing R_a and SL_a than crop and vegetation management or mechanical methods, they may be better suited where competition for water is an issue (Unger and Vigil, 1998) or mechanical methods are not compatible with agricultural practices (e.g. when mechanised tillage would be obstructed by terraces or stone bunds). Comparing the total number of plots for individual SWCTs with the median effectiveness of these techniques (Fig. 7.6) also shows that larger efforts have been made for the assessment of the effectiveness of no-tillage, reduced tillage and contour tillage. However, these are generally not the most effective SWCTs (Fig. 7.5). The limited number of plots for the more effective SWCTs also is an indication of the limited range of environmental conditions for which these more promising SWCTs have been tested in Europe and the Mediterranean.

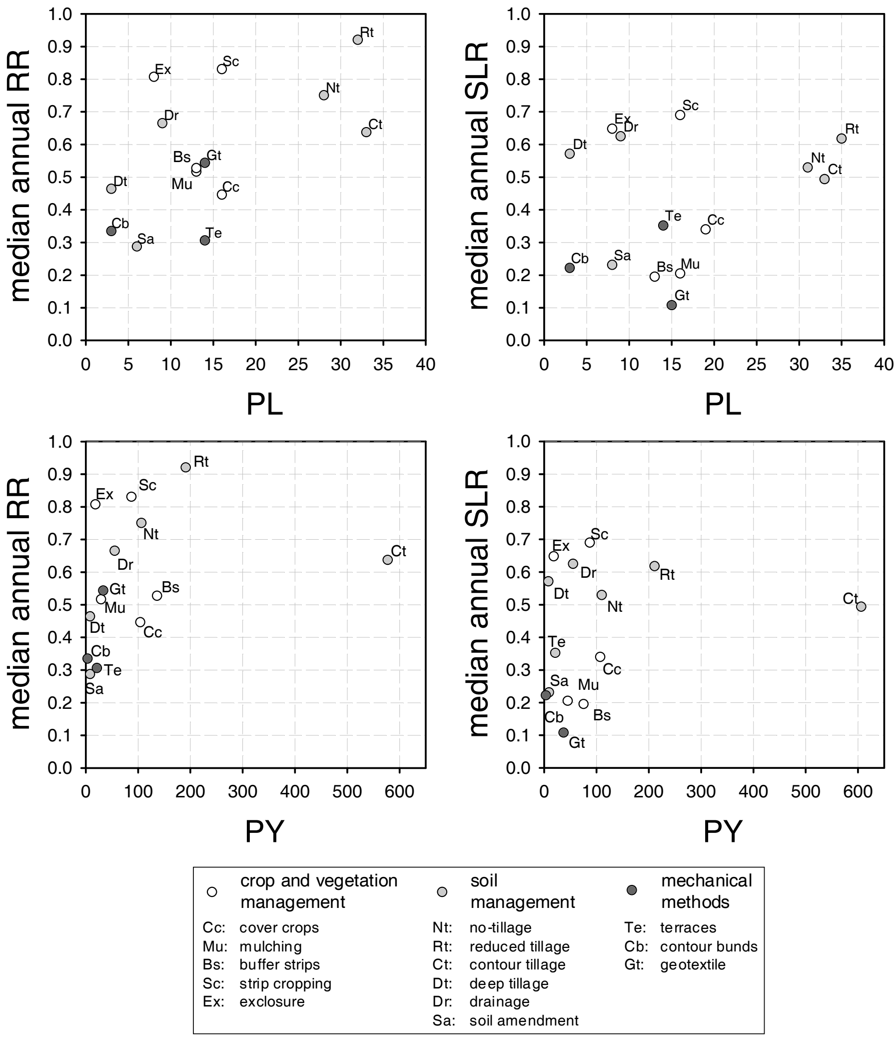


Figure 7.6: Median annual runoff ratio (RR, Eq. 8.1) and annual soil loss ratio (SLR, Eq. 8.2) for different SWCTs versus the number of plots (PL) and number of plot-years (PY) used to test the corresponding SWCTs.

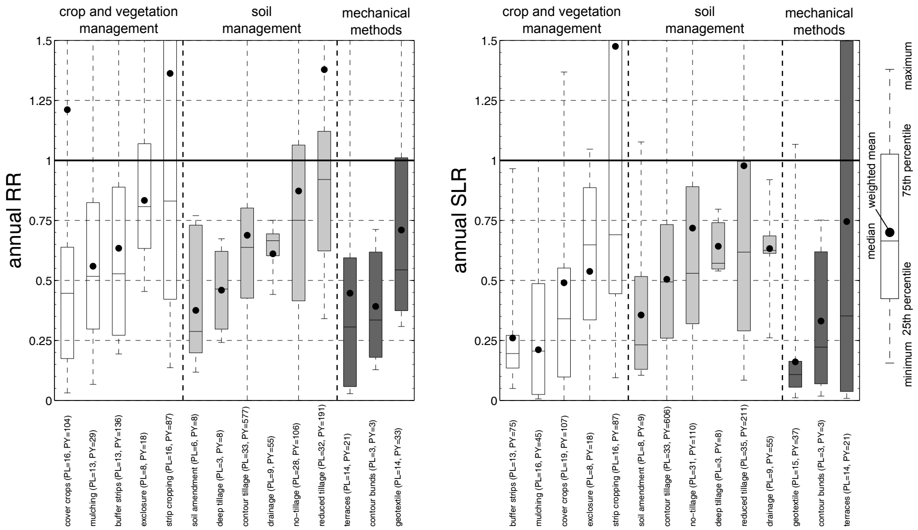


Figure 7.7: Frequency distribution and mean annual runoff ratio (RR, Eq. 8.1) and annual soil loss ratio (SLR, Eq. 8.2) for different soil and water conservation techniques (SWCTs) included in the paired plot database. SWCTs are ranked according to increasing median RR and SLR, within each of the 3 major SWCT groups (Table 2.1). PL= number of plots, PY= number of plot-years.

Weigthed mean RR and SLR values were calculated for each SWCT (Table 7.3). In accordance with the central limit theorem (Tijms, 2004), means were calculated by weighting the RR and SLR for each paired plot by the square root of the number of plot-years. Fig. 7.7 shows the weighted mean and distribution of SLR and RR for each SWCT. This figure illustrates that the range of SLR and RR for a given technique can be very large, indicating an important effect of local environmental and experimental conditions. Hence, the mean and median reduction ratios for a specific SWCT should be interpreted with caution as they may be subject to important variability. Furthermore, the frequency distributions of the reduction ratios are positively skewed, with a limited number of high RR and SLR values strongly affecting the weighted mean values (Fig. 7.7). Nevertheless, both the median and weighted mean values show a similar pattern: most SWCTs are less effective in reducing R_a than in reducing SL_a (Table 7.1).

Table 7.3: Number of plots (PL), number of plot-years (PY), weighted mean, median and coefficient of variation (CV) for annual runoff ratios (RR) and annual soil loss ratios (SLR) per soil and water conservation technique (SWCT) for all data in the paired plot database. Weighting for the mean is done according to square root of the number of plot-years. * Drainage: all reported results refer to surface R_a and SL_a only, hence not to runoff and soil loss measured from the pipe outflow.

	RR					SLR				
SWCT	**PL**	**PY**	**mean**	**median**	**CV**	**PL**	**PY**	**mean**	**median**	**CV**
Crop and vegetation management										
cover crops	16	104	1.21	0.45	1.31	19	107	0.49	0.34	0.84
mulching	13	29	0.56	0.52	0.71	16	45	0.21	0.21	1.41
buffer strips	13	136	0.63	0.53	0.83	13	75	0.26	0.20	1.02
strip cropping	16	87	1.36	0.83	1.42	16	87	1.48	0.69	0.95
exclosure	8	18	0.83	0.81	0.43	8	18	0.54	0.65	0.68
Soil management										
no-tillage	28	106	0.87	0.75	0.78	31	110	0.72	0.53	1.00
reduced tillage	32	191	1.38	0.92	0.93	35	211	0.98	0.62	1.44
contour tillage	33	577	0.69	0.64	0.96	33	606	0.50	0.49	1.41
deep tillage	3	8	0.46	0.46	0.47	3	8	0.64	0.57	0.22
drainage*	9	55	0.61	0.67	0.18	9	55	0.63	0.63	0.28
soil amendment	6	8	0.38	0.29	0.75	8	9	0.36	0.23	0.95
Mechanical methods										
terraces	14	21	0.45	0.31	1.08	14	21	0.75	0.35	1.38
contour bunds	3	3	0.39	0.33	0.76	3	3	0.33	0.22	1.15
geotextile	14	33	0.71	0.54	0.84	15	37	0.16	0.11	1.62

Relations between RR and SLR are given in Fig. 7.8. Regarding crop en vegetation management techniques, mulching and buffer strips are generally more effective in reducing SL_a than cover crops, but the latter are slightly more effective in reducing R_a (Fig. 7.8). With respect to tillage techniques, deep tillage seems less effective in reducing SL_a than contour tillage and no-tillage, but more effective in reducing R_a. For some plots, no-tillage or reduced tillage was found to reduce SL_a but increase R_a. Increased R_a after no-tillage or reduced tillage is attributed to increased soil crusting in some soils after no-tillage or reduced tillage, which reduces soil loss due to the increased resistance of the surface crust to erosion, but also limits infiltration and hence increases R_a. Vermang (2012) found that increased compaction by tractor wheels when reduced tillage is applied in wet conditions can also cause increased R_a compared to conventional mouldboard tillage. In these cases, soil conservation techniques are not necessarily soil and water conservation techniques. Hence, whether a SWCT reduces mainly R_a, SL_a or both is an important factor to consider when selecting the most suitable SWCT to address degradation problems on a specific site.

7.5 Factors controlling the effectiveness of SWCTs

The large variability of RR and SLR values for individual SWCTs (Fig. 7.7) prompts an analysis of the effect of environmental and experimental variables on the effectiveness of SWCTs. Due to a lack of data, this analysis was restricted to the variables reported in almost all publications from which data were extracted (Table 7.1): i.e. RC_a and SL_a from the reference plot, plot slope length, slope gradient and annual precipitation depth. R_a and SL_a are known to be strongly affected by soil characteristics (e.g. Bradford et al., 1987; Torri et al., 1997) as well as by rainfall distribution and intensity (e.g. Nearing et al., 2005). These factors may hence also affect SWCT effectiveness. However, these characteristics were insufficiently reported to allow a global analysis.

The absolute reduction of RC_a (ΔRC_a, %) and SL_a (ΔSL_a, Mg·ha^{-1}·yr^{-1}) by the application of SWCTs was calculated as:

$$\Delta RC_a = RC_{aCONV} - RC_{aSWCT} \qquad \text{(Eq. 8.3)}$$

$$\Delta SL_a = SL_{aCONV} - SL_{aSWCT} \qquad \text{(Eq. 8.4)}$$

where:

RC_{aSWCT} = annual runoff coefficient (%) measured on the plot with SWCT.

RC_{aCONV} = annual runoff coefficient (%) measured on the plot with conventional practice.

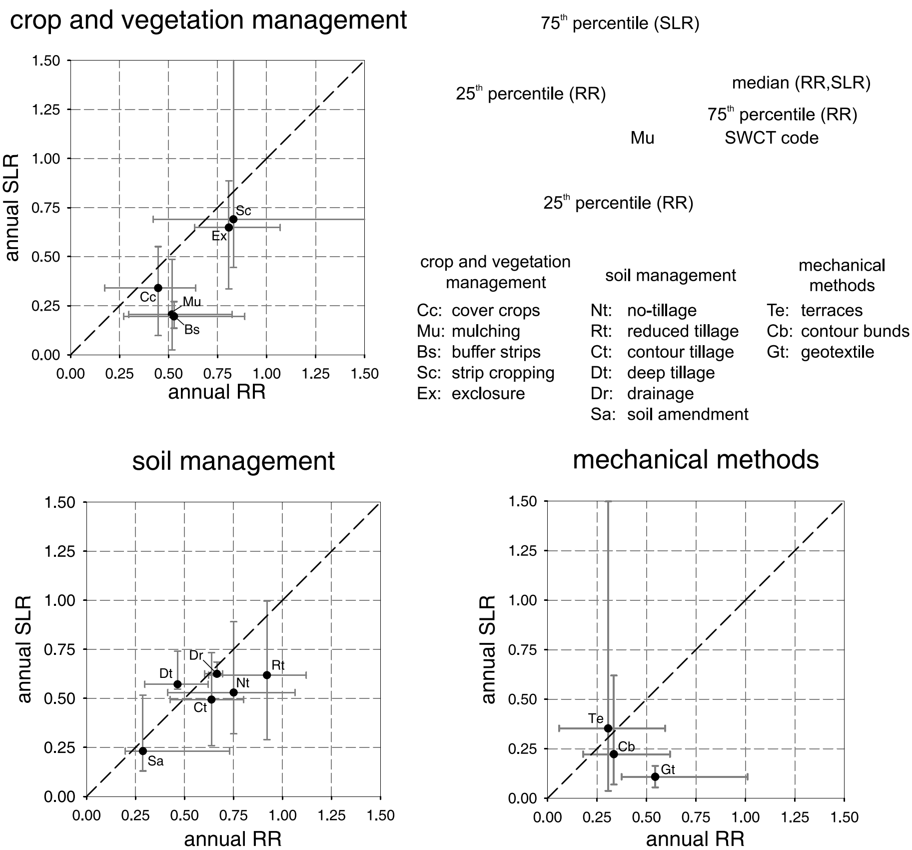


Figure 7.8: Relation between annual runoff ratio (RR, Eq. 8.1) and annual soil loss ratio (SLR, Eq. 8.2) for different soil and water conservation techniques (SWCTs) in the paired SWCT plot database. SWCTs plotting below the 1:1 line indicate soil loss reduction is stronger than runoff reduction.

Fig. 7.9 shows that negative absolute reductions in SL_a and RC_a due to the application of SWCTs occur almost exclusively on plots where the RC_a and SL_a rates on the conventional plot are low. For RC_a, these observations generally occur at RC_{aCONV} rates below 20%. For SL_a, most negative ΔSL_a-values occur for SL_{aCONV} rates below 2 Mg·ha^{-1}·yr^{-1}. This is attributed to the large variability in SL_a and R_a intrinsic to plot studies (e.g. Nearing et al., 1999) when the absolute RC_a and SL_a rates are low. As a result, the estimated low effectiveness of SWCTs for low absolute RC_a and SL_a values may be partly caused by the natural variability of R_a and SL_a plot studies. Nevertheless, some negative ΔRC_a values were also observed for higher RC_a values (Fig. 7.9). These values were derived from different studies where reduced tillage or no tillage techniques were applied (Chisci, 1989; Puustinen et al., 2005, 2007). Also for ΔSL_a, 8 negative values were noted for SL_{aCONV} rates larger than 2 Mg·ha^{-1}·yr^{-1}. Again, these points correspond to paired plot studies where tillage techniques (no-tillage, reduced tillage and contour tillage) were tested at different locations and under different land uses. This indicates that the lack of effectiveness is inherent to the tillage techniques, rather than depending on environmental conditions or measuring uncertainties. Possible causes for this lack of effectiveness in reducing R_a and SL_a for tillage techniques are that it takes time to build up a better soil structure, and that reduced tillage operations are sometimes carried out in unfavourable (e.g. wet) soil conditions (Vermang, 2012).

To explore relations between annual RR and SLR and plot slope length, slope gradient and annual precipitation depth for individual SWCTs, weighted regression equations of the type:

$$Y = aX^b \qquad \text{(Eq. 8.5)}$$

were calculated, whereby each observation in the regression was weighted by the square root of the number of plot-years corresponding to the plot measurement. The limited number of regressions for which the b-value is significantly different from 0 (α=0.05) indicates that the effect of plot slope length, plot slope gradient and Pa on SWCT effectiveness is limited (Table 7.4). For instance, no slope gradient and slope length effects on contour tillage effectiveness were found, even though both factors are used in the calculation of the P-factor for contour tillage (Renard et al., 1997). Nevertheless, all exponents (b) of the significant regressions are negative (Table 7.4), indicating that these SWCTs become more effective in more erosion-prone conditions (i.e. longer and steeper plot slopes and higher Pa). This is in line with results reported by Smets et al. (2008b,a) who observed that various soil surface cover types (i.e. organic and inorganic mulches, vegetation) become more effective in reducing SL on longer plots.

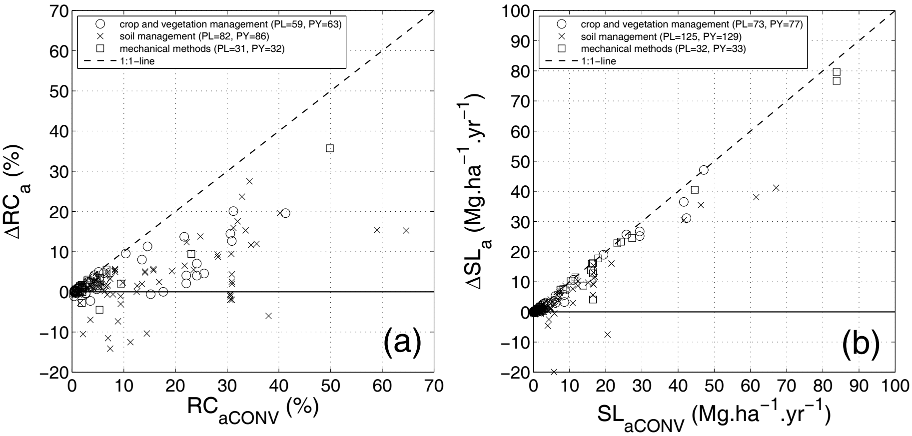


Figure 7.9: (a) Relation between the magnitude of annual runoff coefficients (RC_{aCONV}) measured on plots without soil and water conservation techniques (SWCTs) and the absolute reduction of annual runoff coefficients (ΔRC_a, Eq. 8.3) after applying a SWCT. **(b)** Relation between the magnitude of annual soil loss (SL_{aCONV}), measured on plots without SWCTs, and the absolute reduction of annual soil loss (ΔSL_a, Eq. 8.4) after applying a SWCT. PL= number of plots, PY= number of plot-years.

Table 7.4: Summary of the significant regressions ($Y=aX^b$, Eq. 8.5) between annual runoff ratio (RR), annual soil loss ratio (SLR) and plot slope length, plot slope gradient and annual precipitation for each of the soil and water conservation techniques (SWCTs). Only regressions significant at α=0.05 are given. PL= number of plots, a: regression coefficient, b: regression exponent, p: p-value; probability that the b-value is not significantly different from zero. r^2: coefficient of determination.

variable	SWCT	PL	a	b	p	r^2
			RR			
slope gradient						
	mulching	13	3.28	-0.88	0.02	0.42
	strip cropping	12	174.49	-1.83	0.01	0.55
annual precipitation						
	contour tillage	29	43.57	-0.66	$<$0.01	0.27
	geotextile	14	2.04×10^5	-1.94	0.01	0.43
			SLR			
plot length						
	cover crops	18	3.80	-1.06	0.04	0.25
	terraces	14	527.62	-2.67	$<$0.01	0.59
annual precipitation						
	mulching	14	861.36	-1.62	0.04	0.31
	no-tillage	24	424.32	-1.02	0.04	0.18
	terraces	14	3.6×10^8	-3.42	$<$0.01	0.63

The overall absence of clear relations between RR or SLR and plot slope length, plot slope gradient or Pa may be partly attributable to differences in other environmental and experimental conditions of the plot studies. These differences are hard to account for, because these conditions were rarely specified in the original data sources. For instance, rainfall distribution and intensity may have a large influence on RR and SLR, since R_a and SL_a often depend on a few low-frequency, high-magnitude events (e.g. de Figueiredo and Poesen, 1998). Unfortunately, insufficient rainfall distribution and intensity data were available to investigate potential relations with SWCT effectiveness. Likewise, soil characteristics may significantly control RR and SLR. However, no data were available to quantify this effect. Furthermore, the calculated regressions (Table 7.4) are strongly controlled by a few observations with a very high plot length or gradient. This further increases the uncertainty on the calculated relationships. As a result, our results remain inconclusive about the overall effects of environmental factors on the effectiveness of SWCTs.

It should be noted that the absence of a clear effect of plot length on RR and SLR does not imply that there is no scale effect in the effectiveness of R_a and SL_a. SWCTs are typically applied at the field scale and the effects of SWCTs on R_a and SL_a reduction at this scale may differ from the plot scale. In general, more research on the scale dependency of specific SWCTs (e.g. Leys et al., 2010) is needed before reliable extrapolations can be made. At the catchment scale, also the spatial distribution of the SWCT application has an effect on how effective they are, as was shown by (Verstraeten et al., 2006c) for riparian buffer strips.

7.6 Temporal variability and trends in the effectiveness of SWCTs: the case of tillage techniques

Mean measuring period for plots in the PP database was 4.9 yrs. (st. dev.: 11.2 yrs, median: 2 yrs., mode: 1 yr.). Hence, most studies focus on the quantification of the effectiveness directly after application of the SWCT, while temporal variability and evolution of SWCT effectiveness over time is largely ignored.

To explore the effectiveness of SWCTs during consecutive years, 65 time-series of annual paired plot R_a and/or SL_a observations were selected from the PP database. At these plots, the effectiveness of no-tillage, reduced tillage or contour tillage under cropland was tested for a period of at least 3 consecutive years. To increase the number of available data, also studies from alsewhere where the effectiveness of no-tillage, contour tillage or reduced tillage under cropland was evaluated were included. These other studies include time-series from 18 paired plots from the United States (Bosch et al., 2005; Cullum et al., 2007, 2010; McCool et al., 2008; McDowell and McGregor, 1984; Schreiber and Cullum, 1998; Shipitalo and Edwards, 1998) and 12 plot data time-series from Australia (Freebairn and Wockner, 1986).

Fig. 7.10 shows that there is considerable variability in RR and SLR for individual paired plots between the different years, independent of the time after SWCT application. The median trend of the RR and SLR over the consecutive years since the first application of the tillage techniques (Fig. 7.10) shows that for no-tillage, RR are generally larger than SLR throughout the measuring period and tend to increase over the years following first application. Hence, the effectiveness of these techniques to reduce runoff decreases over time. No such trend is observed for SLR, and both no-tillage and reduced tillage remained very effective in reducing SL_a throughout the entire study period. This is most likely attributable to increasing surface sealing when the soil is not tilled for several years. While this is beneficial for the reduction of SL_a through interrill and rill erosion, it also reduces infiltration capacity of the soil and thus enhances R_a (Bradford et al., 1987). Similar, but less clear trends can be noted for reduced tillage and contour tillage (Fig. 7.10). The number of sufficiently long time-series is limited however. As a result, these trends should be interpreted with caution as they may not be globally valid.

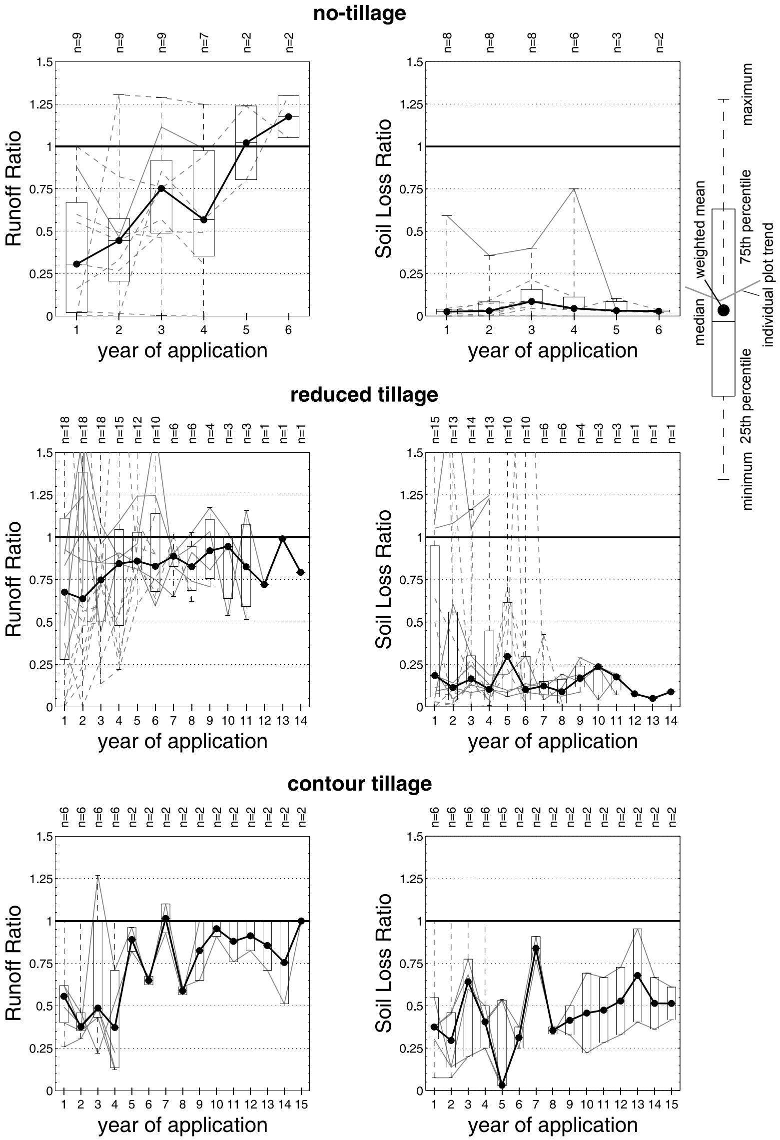


Figure 7.10: Frequency distribution (box plots) and median trend (black solid line) of annual runoff ratio (RR, Eq. 8.1) and annual soil loss ratio (SLR, Eq. 8.2) for consecutive years following the application of a soil and water conservation technique. All studies were conducted on plots under cropland. Thin grey solid lines indicate trends for individual plots in Europe and the Mediterranean, thin grey dashed lines indicate trends for individual plots in the U.S.A. or Australia. n = number of plots.

7.7 Conclusions

This study is the first to present a meta-analysis of field-measured plot data on the effectiveness of soil and water conservation techniques in reducing annual runoff (R_a) and annual soil loss (SL_a) for the Europe and the Mediterranean. While SWCTs are typically applied at the field scale, and effects of SWCTs at field or catchment scale may differ, the results of this study allow to estimate the effectiveness of different SWCTs in reducing both R_a and/or SL_a at the plot scale. Crop and vegetation management (buffer strips, mulching, cover crops) and mechanical methods (geotextiles, terraces, contour bunds) are more effective in reducing R_a and SL_a than soil management techniques (no-tillage, reduced tillage, contour tillage). Nevertheless, large research efforts have been invested in these soil management techniques, while potentially more effective techniques such as cover crops, mulching and buffer strips remain relatively underresearched. A key finding of this study is that SWCTs are generally much more effective in reducing soil loss than in reducing runoff at plot-scales. This is important to consider, since reducing runoff and promoting infiltration may be a much bigger concern than reducing soil loss. Both for soil loss and runoff, the measured effectiveness for a given SWCT varies widely between different studies, indicating an effect of environmental factors. However, due to a lack of data the factors controlling this variability could not be clearly identified. Some of the studied soil and water conservation techniques become more effective in reducing R_a and SL_a rates in more erosion-prone conditions, i.e. when larger R_a and SL_a are measured on the plot with conventional practice, on longer and steeper slopes or in areas with a higher annual precipitation depth. However, observed relations between these factors and RR and SLR are weak and future studies should aim to better identify factors controlling the effectiveness of SWCTs in reducing R_a and SL_a, as this is a key requirement for their successful implementation. Furthermore, the effectiveness of a SWCT may show important temporal variations. For no-tillage and, to a lesser extent, reduced tillage and contour tillage, clear indications were found that these techniques become less effective in reducing runoff after consecutive years of application. These longer-term trends, as well as the inter-annual variability of SWCT effectiveness are important factors to be further explored, as they will contribute to a better selection of these techniques to conserve both our soils and water resources. This will require studies evaluating the effectiveness of SWCTs over longer time periods (i.e. >3 years). Unfortunately, such studies are currently rare.

A APPENDIX: Characteristics of the SWCTs represented in the plot runoff and soil loss database.

In this appendix, an overview is given of the most important characteristics of the different soil and water conservation techniques (SWCTs) tested on runoff and soil loss plots in Europe and the Mediterranean that have been included in the plot database. This chapter should be considered in conjunction with the analysis of the effectiveness of SWCTs in reducing annual runoff (R_a) and annual soil loss (SL_a) given in chapter 7, as it presents essential details for the interpretation of these results, as well as an overview of the different SWCTs that have been tested on runoff and soil loss plots throughout Europe and the Mediterranean.

The division into major groups of SWCTs (i.e. crop and vegetation management, soil management and mechanical methods) is based on Morgan (2005). The plots where both no-tillage or reduced tillage in combination with a cover crop was tested were assigned to the cover crops category.

A.1 Crop and vegetation management

Crop and vegetation management techniques use the beneficial effects of vegetation cover on R_a and SL_a to reduce or prevent R_a and SL_a loss in-situ or trap runoff water and sediment at field borders to prevent off-site damage.

Cover crops

Conventional agricultural practices leave the soil bare during periods of the year (e.g. after harvest) or leave a proportion of the soil surface bare (e.g. the inter-row area in vineyards). Cover crops (Fig. A.1) reduce the period of the year that the soil is bare or keep the soil covered by vegetation year-round. For certain row crops sensitive to interrill and rill erosion (e.g.maize: Van Dijk et al., 1996), undersowing of a cover crop can be used. In permanent cultivation (i.e. vineyards and tree crops), cover crops protect the area of bare soil against R_a and SL_a. Cover crops should establish quickly, provide early and dense canopy cover, suppress weeds and have a deep root system to improve soil macroporosity (Morgan, 2005). A disadvantage to the use of cover crops is their water use, which can have negative effects on crop yield in drier areas. When killed, they may have a positive influence on the soil water balance however (Unger and Vigil, 1998). Many cover crops are leguminous species, which also act as green manures through nitrogen fixation (Peoples et al., 1995). Furthermore, also grass species are commonly used as cover crops. An overview of the different cover crop types included in the plot database is given in Table A.1.

Figure A.1: (a) Linseed (*Linum usitatissimum* L.) cover crop sown between rows of maize (*Zea mays* L.), United Kingdom (photo: www.ukagriculture.com), and **(b)** Yellow mustard (*Sinapis alba* L.) cover crop in November in Bertem, Belgium (photo: Jean Poesen).

Table A.1: Overview of the different cover crop types tested on runoff and soil loss plots in Europe and the Mediterranean and included in the plot database.

country	source	land use type	cover crop type
Algeria	Arabi and Roose, 1993	vineyard	Wheat-legume rotation with two tillage operations and fertiliser.
Austria	Klik, 2003, 2010; Klik, pers. comm.	cropland	Mixture of sweet pea (*Lathyrus odoratus* L.), vetch (*Vicia sativa* L.), buckwheat (*Fagopyrum esculentum* Moench), Berseem clover (*Trifolium alexandrinum* L.), Persia clover (*Trifolium resupinatum* L.), mustard (German: Senf) and Phacelia (*Phacelia* spp.), with direct seeding of the crop in the cover crop. Mixture of sweet pea (*Lathyrus odoratus* L.), vetch (*Vicia sativa* L.), buckwheat (*Fagopyrum esculentum* Moench), Berseem clover (*Trifolium alexandrinum* L.), Persia clover (*Trifolium resupinatum* L.), mustard (German: Senf) and Phacelia (*Phacelia* spp.), with mulchseeding of the crop after light tillage of the soil
Belgium	Laloy and Bielders, 2008, 2010	cropland	Winter rye (*Secale cereale* L.) and ryegrass (*Lolium perenne* L.) cover crop after reduced tillage with rotary cultivator
Denmark	Schjønning et al., 1995	cropland	Catch crop of ryegrass (*Lolium perenne* L.), NPK fertilizer applied in spring
Finland	Turtola et al., 2007	cropland	Undersown with timothy (*Phleum pratense* L.) and barley (*Hordeum vulgare* L.), mouldboard ploughed
France	Messer, 1980 Viguier, 1993	vineyard vineyard	Permanent grass cover Clover (*Trifolium* spp.) cover crop
Germany	Emde, 1992 Jung and Brechtel, 1980	vineyard cropland	Permanent grass cover Undersown with clover (*Trifolium* spp.) and drilled along the slope contour Undersown with clover (*Trifolium* spp.) and drilled perpendicular to the slope contour
Italy	Bini et al., 2006 Caredda et al., 1997	vineyard cropland	Permanent grass cover Winter forage crop of oats (*Avena sativa* L.), vetch (*Vicia sativa* L.) and clover after traditionally cultivated cereal crop
Lithuania	Jankauskas and Jankauskiene, 2003b Jankauskas et al., 2007	cropland	Cocksfoot (*Dactylis* spp.) - red fescue (*Festuca rubra* L.) cover crop Red clover (*Trifolium pratense* L.) - timothy (*Phleum pratense* L.) cover crop
Portugal	Roxo et al., 1996	cropland	Clover (*Trifolium* spp.)
Spain	Gómez et al., 2009 Schnabel et al., 2001	tree crops rangeland	Intercrop strip of ryegrasses (*Lolium rigidum* Gaudin and *Lolium Multiflorum* Lam.) Planting of forage shrubs along small ridges along the contour lines with a narrow furrow immediately behind the bushes Introduction of native species: subterranean clover (*Trifolium subterraneum* L.), bush clover (*Trifolium glomeratum* L.), yellow serradella (*Ornithopus compressus* L.)
Sweden	Ulén and Kalisky, 2005	cropland	Ryegrass (*Lolium perenne* L.), spring ploughed Grass ley, autumn tilling
Syrian Arab Republic	Masri et al., 2005; Bruggeman et al., 2005	tree crops	Vetch (*Vicia sativa* L.) cover crop after minimal tillage with Faddan (animal traction), two tillage passes per year

Mulching

Soil cover can also be provided by the application of organic or inorganic mulches (Fig. A.2). Organic mulches are either crop residue (e.g. (chopped) straw or maize stalks or leaves), natural products obtained off-site (e.g. pine needles, wood chips or bark), or of urban-industrial origin such as sewage sludge (Ingelmo et al., 1998) or urban solid refuse (Albaladejo et al., 2000). Inorganic mulches consist of rock fragments of different sizes. Mulches are particularly useful when the use of cover crops is not viable when insufficient rainfall or sudden onset of the rainy season prevent the early establishment of an effective soil cover. Also where water competition with the commercial crop is an issue, mulches may be an alternative. Nevertheless, organic mulches still use water and nitrogen during the decomposition process, which may have a negative effect on crop yield (Morgan, 2005).

The effectiveness of mulches in reducing R_a and SL_a has been shown to be determined by a number of key factors. The single most important factor is ground cover percentage, whereby R_a decreases linearly with increasing cover percentage and SL_a decreases exponentially with increasing cover percentage (Smets et al., 2008a). In addition, also mulch type, slope length (Cogo et al., 1984; Gilley and Risse, 2000; Smets et al., 2008a,b), and in the case of rock fragment mulches, position and depth of rock fragment placement (Poesen and Lavee, 1994) have an effect on mulch effectiveness. An overview of the different mulch types included in the runoff and soil loss plot database is given in Table A.2.

Table A.2: Overview of the different mulch types tested on runoff and soil loss plots in Europe and the Mediterranean and included in the plot database.

country	source	land use type	mulch type
France	Ballif, 1989	vineyard	Urban refuse mulch, with reduced tractor passages Urban refuse mulch, with traditional cultivation (± 20 tractor passages)
	Viguier, 1993	vineyard	60 $Mg \cdot ha^{-1} \cdot yr^{-1}$ straw or 30-40 $Mg \cdot ha^{-1} \cdot yr^{-1}$ compost
Italy	Rivoira et al., 1989	rangeland	mulching
Jordan	Abu-Zreig, 2006; Abu-Zreig et al., 2011	shrubland	Surface stone cover artificially increased to 15%, harrowed mannually to create a homogeneous surface Surface stone cover artificially increased to 5%, harrowed mannually to create a homogeneous surface
Slovakia	Chomanicová, 1988	cropland	Incorporation of staw into the topsoil
Spain	Albaladejo et al., 2000	shrubland	Organic urban solid refuse: 65, 130, 195, 260 $Mg \cdot ha^{-1} \cdot yr^{-1}$, no tillage Organic urban solid refuse: 65, 130, 195, 260 $Mg \cdot ha^{-1} \cdot yr^{-1}$, tilled by rotovator
	Bautista et al., 1996	post-fire	Barley straw (surface cover: 200 $g \cdot m^2$)
United Kingdom	Brown, 1996	cropland	Straw chopped and incorporated, non-inversion tillage Straw chopped and incorporated, mouldboard ploughed

Figure A.2: Different types of organic mulches (1: straw, 5: maize leafs, 6: maize straw, 7: wood chips, 8: bark, 9: oak leaves, 10: pine needles) and inorganic mulches (2: crushed rock fragments, 3-4: river gravel) (source: Smets et al., 2008b).

Strip cropping

In strip cropping systems, strips of erosion-sensitive crops (e.g. row crops) are alternated with strips more erosion-resistant crops (Fig. A.3). Strips are best aligned along the contour so R_a and SL_a occurring in the erosion-sensitive strips is trapped in the erosion-resistant strips and flow accumulation is prevented or limited. Interspacing of the strips is an important factor determining the effectiveness of this technique. A combination with grassed waterways to evacuate excess water can also greatly increase the effectiveness in reducing R_a and SL_a (Morgan, 2005). Strip cropping has the drawback that mechanised cultivation is more difficult.

Only one plot measuring station where strip cropping was tested was found; R_a and SL_a from 16 plots at Manisa, Turkey with alternating strips of wheat (*Triticum* spp.), chickpea (*Cicer arietinum* L.) and a grass mixture with different widths (10, 20 and 40m) are reported by Köse et al. (1996) and Köse and Taysun (2002).

Figure A.3: Strip cropping along the contour lines in Perieni, Romania (photo: Jean Poesen).

Buffer strips

Buffer strips are strips with erosion-resistant plant cover located along the lower edges of fields or along waterways (Fig. A.4). They consist mostly of grassed strips to minimise the obstruction of mechanised cultivation along the edges of fields, but also hedges and woody vegetation are used along waterways. Buffer strips do not reduce on-site R_a and SL_a but prevent it from leaving the field or entering a stream. The runoff and sediment trapping effectiveness of a buffer strip is dependent on buffer strip characteristics such as strip width, slope gradient and vegetation composition, stiffness, height and sowing density. But also external factors such as rainfall depth, runoff volume and velocity and sediment concentration and grain size have an important effect (Verstraeten et al., 2006c). Buffer strips are very effective in trapping R_a and SL_a on uniform rectilinear slopes where R_a and SL_a are equally distributed over the entire length of the buffer strip. Nevertheless, buffer strips are much less effective when runoff flow converges and runoff and sediment are concentrated in small areas while the remainder of the buffer strips receives little to no runoff and sediment (Verstraeten et al., 2006c). As an alternative to (grassed) buffer strips, also double sowing of the cereal crop in concentrated flow zones can be used to increase the resistance against R_a and SL_a(Gyssels et al., 2002). An overview of the characteristics of buffer strip plots included in the runoff and soil loss plot database is given in Table A.3.

Figure A.4: Grassed buffer strips with woody vegetation along the lower end of fields. The grassed strip in the middle acts as a grassed waterway, Belgium (source: LNE, 2001).

Table A.3: Overview of the characteristics of runoff and soil loss plots with buffer strips tested on in Europe and the Mediterranean and included in the plot database.

country	source	land use type	buffer strip characteristics
Algeria	Arabi and Roose, 1993	tree crops	unspecified buffer strip (French: bandes d'arrêt)
Bulgaria	Biolchev, 1975; Malinov, 1999 Rousseva et al., 2006	fallow	forest strips; 10, 20 and 40m wide grass strips; 5, 10, 20m wide
Finland	Puustinen et al., 2005, 2007 Uusi-Kämppä, 2005	cropland cropland	Timothy (*Phleum pratense* L.) grass buffer strip, 14m wide Timothy (*Phleum pratense* L.) and fescue (*Festuca* spp.) buffer strip, 10m wide, vegetation cut and removed in summer Shrubs and hardwood trees buffer strip, 10m wide vegetation not cut
France	AREDVI, 2003 Le Bissonnais et al., 2004	vineyard cropland	Ryegrass (*Lolium perenne* L.) buffer strip, 3 and 6m wide Ryegrass (*Lolium perenne* L.) buffer strip, 3 and 6m wide
Germany	Voss, 1978	cropland	Grass buffer strip, 1m wide
Spain	Aspizua, 2003	tree crops	Vegetative strips, 2m to 4m wide, combined with no-tillage or reduced tillage
	Francia Martínez et al., 2006	tree crops	Barley (*Hordeum vulgare* L.) buffer strips, 4m wide, no-tillage
	Gómez et al., 2004	tree crops	Ryegrass (*Lolium rigidum* Gaudin and *Lolium multiflorum* Lam.) strips in between trees
	Martinez Raya et al., 2006	tree crops	Barley (*Hordeum vulgare* L.) buffer strips (3mx6m strips) Lentil (*Lens culinaris* Medikus) buffer strips (3mx6m strips) Thyme (*Thymus* spp.) buffer strips (3mx6m strips)

Exclosure

Exclosure is the closing of land to grazing and agriculture to allow natural vegetation in the area to recover (Fig. A.5). As an alternative to exclosure, management of the stocking rate (i.e. the number of grazing animals per hectare) can be used to reduce R_a and SL_a to negligible rates (Morgan, 2005). The use of exclosures as a SWCT in Europe and the Mediterranean is relatively limited (Table A.4, Mosquera Losada et al., 2005). However, exclosure and stocking rate management are still relevant in the silvopastoral *dehesa* systems in the Southern Iberian peninsula (Castillo et al., 1997; Martinez-Mena et al., 1999) and pastoral systems in the Soutern and Eastern Mediterranean (Fig. A.5, Mazour, 1992; Morsli et al., 2004; Shinjo et al., 2000).

Effectiveness of exclosures is mostly determined by the total vegetation cover (Descheemaeker et al., 2006, 2008). Various authors report threshold total vegetation cover between 30% (Nortcliff et al., 1990) and 70% (Gutierrez and Hernandez, 1996) as sufficient to substantially reduce or prevent R_a and SL_a.

Table A.4: Overview of the characteristics of runoff and soil loss plots where the effects of exclosure were tested in Europe and the Mediterranean and included in the plot database.

country	source	land use type	exclosure characteristics
Algeria	Mazour, 1992	rangeland	grazed plot and exclosure, managed so as to allow grazing in the future
			grazed plot and exclosure plot, enriched in leguminous plants (*Medicago* spp.), managed so as to allow grazing in the future
	Morsli et al., 2004	rangeland	grazed plot and exclosure plot with natural vegetation
Spain	Castillo et al., 1997	shrubland	undisturbed natural vegetation plot and plot with manually clipped vegetation
	Martinez-Mena et al., 1999	shrubland	undisturbed natural vegetation plot and plot with manually clipped vegetation
Syrian Arab Republic	Shinjo et al., 2000	rangeland	plot in grazed area and plot in area with exclosure

Figure A.5: Grazed land (left), next to exclosure (right), Morocco (photo: A. Augustin).

A.2 Soil management

SWCTs in the soil management group aim to minimize soil disturbance through the implementation of alternative tillage practices. Conventional tillage consists of mouldboard ploughing which inverts the topsoil and is followed by the breaking up of clods by e.g. disc harrowing. This tillage practice has proven to be a very effective way to prepare the soil for crop cultivation and to control weeds (Phillips et al., 1980). However, conventional tillage creates a loose, friable seedbed (Phillips et al., 1980) vulnerable to splash erosion. Furthermore, traditional tillage up-and down the slope increases vulnerability to interrill and rill erosion and in addition also causes tillage erosion (Van Oost et al., 2006). Therefore, a wide range of alternative tillage techniques that leave the soil less vulnerable to erosion have been developed (Kassam et al., 2009). These include "conservation tillage", defined as any practice that leaves at least 30 percent soil cover after planting (Morgan, 2005), including both no-tillage and reduced tillage. Other tillage techniques that are used as SWCTs are contour tillage and deep tillage. Apart from conservation tillage, the soil can also be made more resistant to R_a and SL_a by improving soil structure and cohesion through the application of soil amendments, or by drainage (Morgan, 2005).

No-tillage

In no-tillage systems, soil disturbance is restricted to what is necessary for seed planting, which is done by drilling the seeds directly in the stubble residue of the previous crop (Fig. A.6), while weeds are controlled by the use of herbicides. No-tillage leaves 50 to 100% of the soil surface covered with crop residue (Morgan, 2005). The combination of minimal soil disturbance and increased soil surface cover is often found to substantially reduce R_a and SL_a. In addition, the elimination of the need for tillage passes by tractor reduces the adverse effects of tractor wheel ruts on R_a and SL_a (Fullen, 1998). No-tillage also has positive effects on soil organic matter content and aggregate stability of the topsoil (e.g. Blevins et al., 1998). Nevertheless, no-tillage is less suitable for soils that easily develop surface crusts as it may cause an increase in R_a (Jones et al., 1994). There are also indications that the effectiveness of no-tillage in reducing R_a and SL_a is scale-dependent and the technique is less effective on longer slopes (Leys et al., 2010). Effects of no-tillage application on crop productivity are as of yet not well understood, but Van den Putte et al. (2010) found in a meta-regression analysis of published studies that the application of no-tillage reduced crop yield by 8.5% on average, but this yield loss is financially compensated by lower machinery, fuel and labour costs (Phillips et al., 1980), thereby making the economic balance of no-tillage application often positive. An overview of the

Figure A.6: Maize grown in a no-tillage system in Germany. Maize was drilled directly in the stubble residue of the preceding cereal crop, Germany (source: SoCo, 2009, photo: J. Epperlein).

characteristics of runoff and soil loss plots where the effect of no-tillage on R_a and SL_a was tested is given in Table A.5.

Table A.5: Overview of the characteristics of runoff and soil loss plots where the effects of no-tillage were tested in Europe and the Mediterranean and included in the plot database.

country	source	land use type	no-tillage details
Albania	Grazhdani et al., 1999	cropland	no-tillage, both with stubble removed and stubble retained
Algeria	Arabi and Roose, 1993	vineyard	no-tillage, weed control by herbicides
Belgium	Laloy and Bielders, 2008, 2010	cropland	no tillage, stubble left on field followed by direct drilling of maize
Croatia	Basic et al., 2001, 2004	cropland	no-tillage, crops sown in residue, weed control by herbicide no-tillage, crops sown in residue, weed control by herbicide, double cropping
Finland	Puustinen et al., 2005, 2007 Turtola et al., 2007	cropland cropland	no tillage, direct drilling in straw residue and stubble no-tillage after harvest, light seedbed preparation before crop sowing
France	Messer, 1980 Viguier, 1993	vineyard vineyard	no-tillage, weed control by herbicides no-tillage, weed control by herbicides
Germany	Jung and Brechtel, 1980	cropland	no-tillage, crop drilled along the slope contour
Italy	Postiglione et al., 1990	cropland	no-tillage, direct drilling
Lithuania	Feiza et al., 2007	cropland	no-tillage after harvest, weed control by herbicides, light seedbed preparation before crop sowing
Portugal	Oliveira, 2005	tree crops	no tillage
Romania	Nistor and Ionita, 2002	cropland	no-tillage, wheat stubble left on field
Slovakia	Fulajtár and Janský, 2001	cropland	no tillage
Sweden	Ulén and Kalisky, 2005	cropland	no-tillage, organic fertilizer added

Reduced tillage

Reduced tillage or minimum tillage comprises a wide range of non-inversion tillage, using different tillage implements such as the chisel plough and disc plough or harrow (Fig. A.7). Also the number of tillage operations varies, depending on local conditions. Other applications of reduced tillage are weed control and breaking up of soil lumps (Louwagie et al., 2009a). Reduced tillage may have a positive effect on the soil organic matter content and is also less energy and labour demanding than conventional tillage (Davies and Finney, 2002). Nevertheless, in some instances the reduced breaking up of the soil surface may reduce soil porosity, resulting in higher R_a and SL_a (Morgan, 2005). An overview of the characteristics of runoff and soil loss plots where the effect of reduced tillage on R_a and SL_a was tested is given in Table A.6.

Table A.6: Overview of the characteristics of runoff and soil loss plots where the effects of reduced tillage were tested in Europe and the Mediterranean.

country	source	land use type	reduced tillage details
Albania	Grazhdani et al., 1999	cropland	disc tillage
Belgium	Laloy and Bielders, 2008, 2010	cropland	disc cultivator tillage after harvest (0-15cm)
Denmark	Schjønning et al., 1995	cropland	harrowed, crop sown up and down the slope
Finland	Puustinen et al., 2005, 2007	cropland	shallow stubble tillage (5-10cm) in straw residue reduced tillage (10-15cm) in straw residue
	Turtola et al., 2007	cropland	shallow stubble cultivation in autumn, seedbed preparation by harrowing, fertilized
France	Ballif, 1989	vineyard	limited cultivation (reduced number of tractor passages)
Germany	Jung and Brechtel, 1980	bare	soil broken up before winter, no other treatment
Italy	Basso et al., 1983b	cropland	minimum tillage (herbicide + harrowing)
	Chisci and Zanchi, 1981	cropland	minimum tillage (not further specified)
	Chisci, 1989	cropland	minimum tillage (not further specified)
	Postiglione et al., 1990	cropland	tilled by rotovation
Lithuania	Feiza et al., 2007	cropland	chiselling (to 20cm), cultivation (to 8cm) and harrowing (to 5cm), treated with herbicide
Morocco	Heusch, 1970	cropland	tilled by scratch plough (French: araire)
Norway	Grønsten and Lundekvam, 2006; Børresen, pers. comm.	cropland	spring tillage
	Lundekvam, 2007; Børresen, pers. Comm.	cropland	spring tillage
Romania	Nistor and Ionita, 2002	cropland	chisel tillage
Slovakia	Chomanicová, 1988	cropland	socketing (a spiked metal cylinder is pulled over the soil, increasing surface roughness and infiltration capacity)
	Suchanic, 1987	cropland	socketing
Spain	Bienes et al., 2006	cropland	minimum tillage (not further specified)
	Gómez et al., 2004	tree crops	minimum tillage
	Francia Martínez et al., 2006	tree crops	minimum tillage
Sweden	Ulén and Kalisky, 2005	cropland	reduced tillage, direct sowing in spring no-tillage, disc harrowing in autumn spring tillage
Syrian Arab Republic	Masri et al., 2005; Bruggeman et al., 2005	tree crops	two tillage passages with Faddan (animal traction) with biannual sheep manure and annual fertilizer application
United Kingdom	Brown, 1996	cropland	non-inversion poughing, Dutzi drilled and fertilized
	Quinton and Catt, 2004	cropland	tine tillage (to 10cm) along the slope contour tine tillage (to 10cm) perpendicular to the slope contour

Figure A.7: Reduced tillage using disc harrowing in a cereal crop, Germany (source: SoCo, 2009, photo: S.H. Gay).

Contour tillage

By ploughing and planting along the slope contour (Fig. A.8), the maximum soil roughness is created perpendicular to the direction of runoff flow (i.e. downslope), thereby limiting runoff flow connectivity. Non-infiltrated rainfall is held in the furrows where also soil eroded from the ridges is deposited. However, it is often not possible to perform contour tillage exactly along the contour, allowing R_a to flow along the ridges to lower parts, where it accumulates and ridges can eventually overtop or break through and cause rill or gully erosion (Renard et al., 1997). Therefore, the effectiveness of contour tillage varies with slope length and gradient (Morgan, 2005), which is also reflected in the way the support practice factor (P-factor) for contour tillage is calculated in the (R)USLE (Renard et al., 1997; Wischmeier and Smith, 1978). Contour tillage can be combined with strip cropping or grassed waterways to increase the effectiveness of the technique. An overview of the characteristics of runoff and soil loss plots where the effect of no-tillage on R_a and SL_a was tested is given in Table A.7.

Figure A.8: Contour tillage in an olive grove near Aleppo, Syria (photo: F. Turkelboom).

Table A.7: Overview of the characteristics of runoff and soil loss plots where the effects of contour tillage were tested in Europe and the Mediterranean and included in the plot database.

country	source	land use type	contour tillage details
Algeria	Mazour et al., 2008	cropland	tilled parallel to the slope contour
	Mazour, 1992	cropland	tilled parallel to the slope contour
Croatia	Basic et al., 2001, 2004	cropland	tilled (30cm deep) parallel to the contour subsoiled (60cm deep) and tilled (30cm deep) parallel to the contour deep ploughing (60 cm deep) parallel to the contour with single-bottom plough
Finland	Puustinen et al., 2005, 2007	cropland	tilled parallel to the slope contour
Germany	Jung and Brechtel, 1980	cropland	tilled parallel to the slope contour
Hungary	Hudek and Rey, 2009 Pinczés, 1982	bare grassland vineyard	tilled parallel to the slope contour tilled parallel to the slope contour tillage ridges parallel to the contour at every row in the traditional stake stystem tillage ridges parallel to the contour every five rows in the traditional stake stystem water retaining furrows ploughed parallel to the contour every five rows in the traditional stake support system
Morocco	Laouina et al., 2003	cropland	tilled parallel to the slope contour
Portugal	Oliveira, 2005	tree crops	tilled parallel to the slope contour
Spain	Schnabel et al., 2001	rangeland	strip tillage along the contour in dehesa farm
Tunisia	Kaabia, 1995	cropland	tilled parallel to the contour and perpendicular to the contour in rotation
Turkey	Köse et al., 1996; Köse and Taysun, 2002 Oguz et al., 2006; Oguz, pers. comm.	bare bare	tilled parallel to the slope contour tilled parallel to the slope contour
United Kingdom	Quinton and Catt, 2004	cropland	tilled parallel to the slope contour

Figure A.9: Subsoiler in combination with disc plough for non-inversion tillage (photo: SMS Agricultural Machines).

Deep tillage

Deep tillage or subsoiling is the breaking up of the soil to greater depths than in conventional tillage (>50 cm) (Fig. A.9). This practice is mostly applied to break up plough pans or impermeable layers in the soil and improve infiltration rates. Deep tillage may be combined with conventional tillage (Chomanicová, 1988).

R_a and SL_a rates from cropland plots were deep tillage was applied were reported for three plot measuring stations (i.e. Muska, Ploske, and Krušiny) in Slovakia by Chomanicová (1988) and Suchanic (1987).

Drainage

Heavy clay soils in cropland are prone to periodical waterlogging which can seriously affect crop yield (Cannell et al., 1980). Drainage of these soils prevents waterlogging and thereby also reduces surface R_a and SL_a. Nevertheless, drain outflow water can still cause off-site problems and is an important source of sediments (Stone and Krishnappan, 1998; Verachtert et al., 2011) and pollutants (Turtola and Paajanen, 1995).

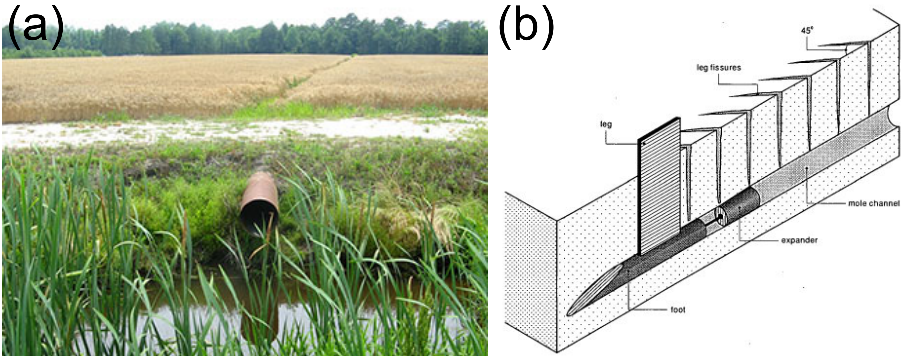


Figure A.10: (a) Drain pipe outlet (photo: North Carolina State University), and **(b)** Creation of a mole drainage channel by mole plough (source: Cavelaars et al., 1994).

Commonly used drainage techniques include networks of permanent drainage pipes (Carter and Berg, 1983; Cavelaars et al., 1994; Dierickx, 1993), mole drains (Cavelaars et al., 1994; Christen and Spoor, 2001; Leeds-Harrison et al., 1982) and surface drainage (Holden et al., 2007) (Fig. A.10). The implementation of a specific type of drainage depends on its costs, the drainage density and depth required, the desired lifetime of the drainage system and practical limitations. Tile drainage systems are applied in heavy clay soils and consist of drainage pipes laid in backfill material which are long-lasting and provide effective drainage, but are costly and are prone to clogging. Mole drains are unlined channels in the soil created by a mole plough (Christen and Spoor, 2001). While they are relatively cheap to implement and larger drainage densities are possible than with tile drainage, mole channels are vulnerable to roof collapse (Spoor and Ford, 1987) and piping erosion (Verachtert et al., 2011).

In Europe and the Mediterranean, the effect of drainage on plot-scale R_a and SL_a was tested in Finland (Warsta et al., 2009), Norway (Øygarden, 1996; Øygarden et al., 1997), Italy (Chisci and Zanchi, 1981), and Albania (Grazhdani et al., 1996).

Soil amendment

Soil amendments (Fig. A.11), soil stabilisers or soil conditioners make the soil more erosion-resistant by improving the soil structure. Various types of soil amendment are used, depending on the type and condition of the soil: e.g. gypsum, organic matter or synthetic polymers (e.g. Polyacrylamide). Most soil amendments are too expensive and short-lived for wide-scale agricultural use,

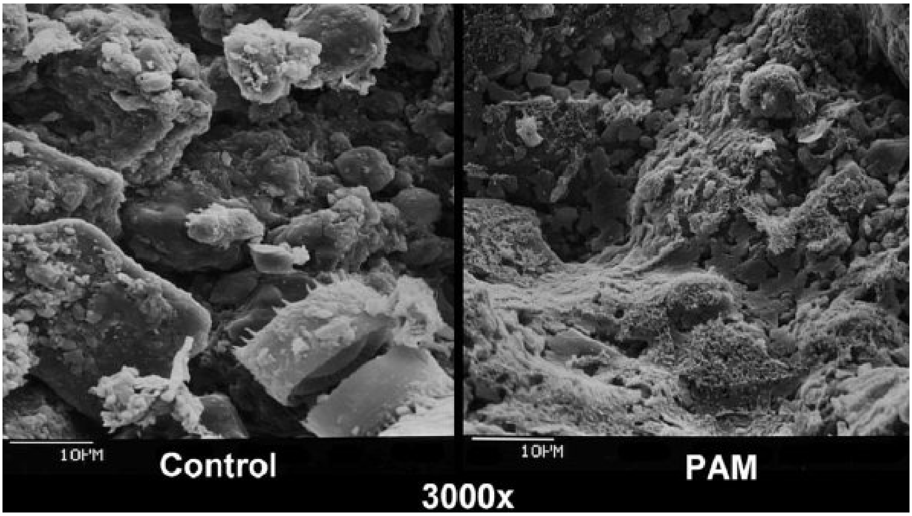


Figure A.11: Scanning electron photographs showing the effect of polyacrylamide applied by fertigation on soil structure (right), compared to a control treatment without polyacrylamide application (left) (source: Entry et al., 2002).

but they can be of use in high-value applications such as construction sites or roadcut sites (Agassi and Benhur, 1991; Agassi et al., 1990). An overview of the characteristics of plots where the effect of soil amendments on R_a and SL_a was tested is given in Table A.8.

Table A.8: Overview of the characteristics of runoff and soil loss plots where the effects of soil amendments were tested in Europe and the Mediterranean and included in the plot database.

country	source	land use type	soil amendment type
Bulgaria	Kroumov and Malinov, 1989	grassland, grazed	NPK fertilizer
		grassland, ungrazed	NPK fertilizer
Israel	Agassi and Benhur, 1991	roadcut embankment	5 $Mg \cdot ha^{-1}$ phosphogypsum
Slovakia	Chomanicová, 1988	cropland	vinilacetate-acrylate copolymer incorporated in the topsoil)
Spain	Ingelmo et al., 1998	shrubland	sludge (unspecified)
	Lopez-Bermudez et al., 1991	shrubland	treated with polymers (40 $g \cdot m^{-2}$)

A.3 Mechanical methods

Terraces

Terraces have been used for centuries and occur as native SWCTs all over the world (e.g. Dunning and Beach, 1994), serving various functions such as making cultivation possible on steep slopes, reduction of R_a and SL_a, or water harvesting (e.g. Al-Seekh and Mohammad, 2009). Terraces can be classified into three main types: diversion, retention and bench (Dorren and Rey, 2004; Morgan, 2005). Common features of all terraces are earth or stone embankments constructed along the slope contour (Fig. A.12). Runoff infiltrates behind the embankment or is evacuated through a drainage channel at lower, non-erosive velocities, thereby causing sediment to be deposited. Over time, the original slope profile is transformed into a step-like slope profile with steep embankments, covered with grass or reinforced with stones to prevent erosion. An overview of the characteristics of runoff and soil loss plots with terraces is given in Table A.9.

The effectiveness of terraces in reducing R_a and SL_a is determined mainly by terrace design and the horizontal and vertical spacing of embankments. Also construction and maintenance costs, and the obstruction of cultivation practices are important factors to consider in terrace construction. Truncation of the soil in the upper part and accumulation of sediment in the lower part of the terrace can also create gradients of soil fertility (Dercon et al., 2003; Vancampenhout et al., 2006). Maintenance of stone terraces is labour-intensive (Dorren and Rey, 2004), but failure to maintain terraces may cause the stored sediment to be eroded (Koulouri and Giourga, 2007; Lesschen et al., 2008).

Figure A.12: Terraces in an olive orchard, Syria (photo: F. Turkelboom).

Table A.9: Overview of the characteristics of runoff and soil loss plots with terraces in Europe and the Mediterranean and included in the plot database.

country	source	land use type	terrace characteristics
Greece	Koulouri and Giourga, 2007	tree crops	maintained terraces
			terraces abandoned for 5 and 20 years
Palestinian territories	Abu Hammad et al., 2004, 2006	cropland	stonewalled terrace, 50 years old
	Al-Seekh and Mohammad, 2009	cropland	earthen contour ridges
			stone terraces
		shrubland	stone terraces (Jessour) in a wadi bottom
			earthen semi-circle bunds
			stone terraces
Romania	Teodorescu and Badescu, 1988	tree crops	unspecified
Serbia	Đjorović, 1990	cropland	bench terraces
Spain	Llovet et al., 2009; Llovet, pers. comm.	forest	old terraces
		grassland	recently (5-8 years) abandoned terraces
		post-fire	old terraces
			recently (5-8 years) abandoned terraces
	Martínez-Casasnovas and Ramos, 2006	vineyard	broadbased terrace each 8 rows of vines
	Ramos and Martínez-Casasnovas, 2006	vineyard	broadbased terrace each 8 rows of vines
	Ramos and Martínez-Casasnovas, 2007	vineyard	broadbase terrace every 8 rows, high disturbance by land levelling
			broadbase terrace every 8 rows, low disturbance of the soil
	Sanchez et al., 1994	post-fire	terraces along the slope contour, followed by afforestation
	Williams et al., 1995	forest	bench terraces
Syrian Arab Republic	Ali, 2007	shrubland	continuous microcatchment vallerani terraces parallel to the contour
			intermittent microcatchment vallerani terraces parallel to the contour
			semi-circular manually constructed microcatchment terraces

Contour bunds

Contour bunds are earthen or stone banks constructed along the contour (Fig. A.13) and sometimes reinforced with vegetation, thereby effectively shortening the slope length (Morgan, 2005). Most contour bunds are constructed manually and are therefore labour-intensive. Hence, they occur mostly on smallholdings in developing countries. Although few guidelines for contour bund construction exist (Morgan, 2005), contour bund height and interspacing were found to depend on slope gradient and stone availability (Nyssen et al., 2007). Due to the trapping of runoff and sediment, contour bunds generally have a positive effect on crop production. In this study, only one plot measuring station with contour bunds was found in Tokaj, Hungary where contour bunds placed at five-row intervals in a vineyard with a traditional stake support system were tested (Pinczés, 1982).

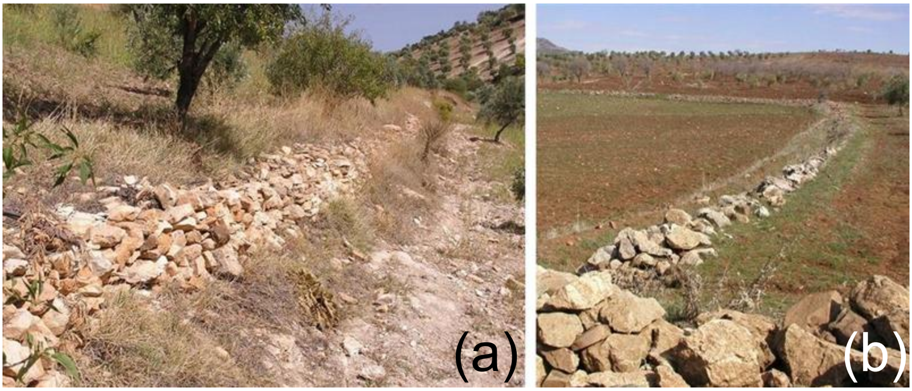


Figure A.13: Stone contour bunds in **(a)** an olive orchard and **(b)** cropland, Syria (photo: F. Turkelboom).

Geotextiles

Geotextiles have can provide effective protection against interrill and rill erosion on vulnerable soils such as recently established roadcut sites (Smets et al., 2011b). Geotextiles are available in a wide range of permeable textiles or woven mesh (Fig. A.14) made from natural (e.g. jute, coir, palm leaves, maize stalks) or synthetic (e.g. polypropylene) fibres (John, 1987; Morgan, 2005). The natural geotextiles are bio-degradable and provide protection during the period that the bare soil is vulnerable to interrill and rill erosion until vegetation can establish to protect the slope against erosion. Applied on top of the soil, geotextiles protect against splash erosion as well as overland flow. Buried in the soil, geotextiles provide little protection against rill erosion and no protection against splash and interrill erosion but reinforce the soil and increase slope stability. Important factors that determine the effectiveness of geotextile in reducing R_a and SL_a are the material, mesh size and degree of contact with the soil (Smets, 2009). An overview of the characteristics of the different types of geotextile tested on runoff and soil loss plots included in the database is given in Table A.10.

Figure A.14: Testing the effectiveness of different types of geotextiles, Lithuania (photo: B. Jankauskas).

Table A.10: Overview of the characteristics of runoff and soil loss plots where the effects of geotextiles were tested in Europe and the Mediterranean and included in the plot database.

country	source	land use type	geotextile type
Hungary	Kertesz et al., 2007	vineyard	plot bottom covered by 5m wide geotextile strip (Buriti and Borassus palm)
		tree crops	plot bottom covered by 5m wide geotextile strip (Buriti and Borassus palm)
Italy	Zanchi, 1983	bare	ground covered with a net (0.8x1mm mesh)
Lithuania	Jankauskas et al., 2008	bare	partially buried geotextile (Buriti and Borassus palm) on truncated roadside slope, natural regrowth of grasses and herbs
		bare -grassland	partially buried geotextile (Buriti and Borassus palm) on truncated roadside slope, grass sown
		bare	straw-coir on truncated roadside slope
United Kingdom	Bhattacharyya et al., 2008, 2009	bare	plot bottom covered by 1m wide geotextile strip (Buriti and Borassus palm)
			entire plot by geotextile (Buriti and Borassus palm)
	Mitchell et al., 2003	geotextile	jute geotextile net
			jute mat, later replaced by jute mat reinforced with jute net

Chapter 8

Conclusions

The overall objectives of this research are to assess the effects of different land use types on plot-measured annual runoff (R_a) and annual soil loss (SL_a), and the effectiveness of several soil and water conservation techniques (SWCTs) in reducing R_a and SL_a over the whole of Europe and the Mediterranean (i.e. the continental scale). Special attention is given to spatial and temporal variability in R_a and SL_a, and to environmental factors controlling R_a and SL_a under different land use types and application of different SWCTs.

8.1 The runoff and soil loss plot database for Europe and the Mediterranean

First, a database of plot runoff and soil loss containing R_a and SL_a data from 1 409 plots, corresponding to 9 297 plot-years from 239 plot-measuring stations throughout Europe and the Mediterranean was compiled. The database contains R_a data for 804 plots (corresponding to 5 327 plot-years) and SL_a data for 1 056 plots (corresponding to 5 327 plot-years) under conventional land management practice (see chapter 2). Furthermore, also R_a data for 287 plots (corresponding to 1 713 plot-years) and SL_a data for 356 plots (corresponding to 2 035 plot-years) where SWCTs were tested were collected (see chapter 7).

Due to a lack of a coordinated research effort at the European and the pan-Mediterranean level, plot measurements in Europe and the Mediterranean are dispersed in the literature. Many plot runoff and soil loss data have been published in the "grey" literature (i.e. project reports, governmental reports, PhD. theses) that are not easily accessible and are hence under-utilised. Also less details such as individual storm data or even individual annual data are available for many of the studies that report plot runoff and soil loss measurements in Europe and the Mediterranean.

These data have never been compiled for a global analysis. While still more plot runoff and soil loss data likely exist for Europe and the Mediterranean, this study is a substantial improvement over previous review studies (Table 1.1), both with respect to the total number of plots and plot-years included, as well as with respect to the systematic inclusion of R_a data and data from plots where SWCTs were tested. Furthermore, this study is also an improvement over previous review studies with respect to the study area. Most review studies focus on the country or sub-continental scale, while the inclusion of the whole of Europe and the Mediterranean (i.e. the countries around the Mediterranean Sea) in this study allows the assessment of R_a and SL_a over a much wider range of climatic conditions, especially as many of the semi-arid regions around the Mediterranean have never been included in previous reviews. As indicated in chapter 1, these semi-arid regions are especially vulnerable to soil degradation and desertification and hence the inclusion of these areas in the database presented in this study is a valuable addition to already existing review studies.

The database compiled in this study was then used (1) to assess the effects of different land use types on R_a and SL_a, (2) to evaluate the effectiveness of different SWCTs in reducing R_a and SLa, (3) to better understand the annual precipitation (P_a) - annual runoff (R_a) relation and (4) to assess spatial and temporal variability in plot-measured R_a and SLa.

8.2 Effect of land use type on annual runoff and soil loss

Land use type, and especially the vegetation cover associated with it, have been widely shown to be important factors controlling R_a and SL_a (section 1.2.1, section 3.1). Furthermore, the role of land use in soil protection is extremely sensitive to anthropogenic disturbance. Important land use changes have taken place in Europe and the Mediterranean, and processes of land use change and changes of land management (e.g. field-scale expansion and intensification in agriculture) have contributed strongly to land and soil degradation. On the other hand, the land use type is one of the factors in erosion prevention that can easily be managed and can be used very effectively in soil conservation strategies, provided that the effects of different land use types on R_a and SL_a are well understood over a wide range of environmental conditions.

Several knowledge gaps with respect to the effect of land us type on R_a and SL_a over the whole of Europe and the Mediterranean are addressed in chapter 3. Where previous reviews of the effects of land use type on R_a and SL_a are mostly limited to the country or regional scale, this is the first continental review considering both plot-measured R_a and SL_a rates. This allowed to evaluate R_a and SL_a rates over a much wider range of environmental conditions, which is essential knowledge when applying research results in other areas than those where they were measured. Especially with respect to runoff, such analysis was lacking.

The analysis in chapter 3 confirmed the important control of vegetation cover on R_a and SL_a rates, with marked differences in both R_a and SL_a between heavily disturbed land use types (i.e. construction sites), cultivated land (i.e. cropland, fallow plots, vineyards, tree crops), and semi-natural vegetation (i.e. shrubland, rangeland, forest, post-fire and grassland) for the whole of Europe and the Mediterranean. Generally, there is a good correspondence between R_a and SL_a for the different land use types, but at the regional scale, differences were found between R_a and SL_a rates. Mean SL_a values were smaller in the Mediterranean than in temperate and cold climatic zones, and mean annual runoff coeffient (RC_a) rates were generally higher in the cold climatic zone than in the temperate and Mediterranean zones for similar land use types.

Nevertheless, each land use type also comprises a wide variability in plot-measured R_a and SL_a (section 3.3.2, section 3.4.2). Annual precipitation depth (P_a) was shown to be an important controlling factor of R_a and SL_a (section 3.3.5, section 3.4.4). Nevertheless, it was not possible to further explore the climatic effects of precipitation distribution as expressed by the Modified Fournier Index, despite the fact that intra-annual precipitation distribution is likely to be an important factor controlling R_a and SL_a. Only weak relations were found between R_a and SL_a and other environmental factors that are generally considered important determinants of R_a and SL_a at the local scale such as plot length and slope gradient (section 3.3.1, section 3.4.1, chapter 5), indicating that these factors explain only a small part of the large variability in R_a and SL_a that is observed at the continental scale.

8.3 Effectiveness of SWCTs in reducing annual runoff and soil loss

A change of land use is not always feasible (e.g. for economic reasons), and some environments such as roadcut sites require specific protection measures against soil erosion by water. SWCTs have long and widely been used as measures against excessive R_a and SL_a. Nevertheless, very few review studies of interrill and rill erosion focus specifically on a quantitative assessment of the effectiveness of SWCTs in reducing R_a and SL_a. Most studies reviewing SWCT effectiveness are dedicated to specific techniques for the sake of comparison with the researcher's own results. Furthermore, the reduction of R_a by plot-scale (i.e. on-site) SWCTs is less studied than the reduction of SL_a. As a consequence, application of SWCTs in the field is often based on quantitative experience and the incorporation of SWCTs in erosion models remains difficult.

The review study of R_a and SL_a on plots where various SWCTs were applied in Europe and the Mediterranean (chapter 7) showed that most SWCTs are on average more effective in reducing SL_a than in reducing R_a, with the exception of drainage and deep tillage, which are SWCTs applied specifically where R_a is the major problem (section 7.4.1). Furthermore, the importance of vegetation cover as a factor controlling R_a and SL_a was further confirmed by the finding that crop and vegetation management (i.e. buffer strips, mulching, cover crops) are more effective in reducing R_a and SL_a than soil management techniques (i.e. no-tillage, reduced tillage, contour tillage).

Surprisingly, the most studied SWCTs (i.e. no-tillage, reduced tillage and conservation tillage) are on average not the most effective SWCTs, as crop and vegetation management techniques (i.e. cover crops, buffer strips, mulching) are generally more effective in reducing R_a and SL_a (section 7.4.2). This observation can in part be attributed to the fact that crop and vegetation techniques are already relatively reliable in reducing R_a and SL_a, and hence there is a lesser need to establish how effective these SWCT are and in which circumstances. Furthermore, buffer strips are mostly used at the field scale and are hence more difficult to assess using runoff and soil loss plots. In addition, a higher effectiveness is not always equal to higher efficiency (i.e. the cost-benefit ratio of applying these SWCTs). For instance, buffer strips are very effective in reducing SL_a at the plot scale, but can not always be applied easily as closely spaced in the field as on the runoff and soil loss plots (i.e. an upslope sediment source area of 10 to 20 meters length per buffer strip). Therefore, the practical use of these closely interspaced buffer strips is limited to land use types where they do not interfere too much with tillage practices (e.g. in olive orchards), which may contribute to the relative lack of plot-measured data for buffer strips.

Nevertheless, the effectiveness of individual SWCTs in reducing R_a and SL_a was found to be highly variable, suggesting several controlling factors that are unaccounted for. An important effect of the R_a and SL_a rate measured on control plots with conventional treatment was found (section 7.5), and especially for smaller R_a and SL_a rates, effectiveness of the SWCTs was more variable. Effects of environmental factors such as plot length, slope gradient or P_a on SWCT effectiveness could not be clearly identified (section 7.5). Despite practical importance with respect to e.g. the duration over which SWCTs should be applied, temporal variability of SWCTs has been largely neglected in the literature. The preliminary analysis (section 7.6) showed that there is considerable inter-annual variability in the effectiveness of conservation tillage techniques. With respect to runoff reduction, the effectiveness of no-tillage techniques tends to decrease over the years.

8.4 Predicting annual runoff using the curve number method

So far, the study of plot-measured R_a (804 plots corresponding to 5327 plot-years) has received much less attention than SL_a (1056 plots corresponding to 7204 plot-years) in Europe and the Mediterranean, both with respect to the reported data, as with respect to the analysis of R_a data at a continental scale. A better understanding of the P_a-R_a relation can furthermore also contribute to a better understanding and modelling of SL_a (section 3.3.3). Therefore, a closer analysis of the annual rainfall (P_a) - annual runoff (R_a) relation by means of a modified form of the SCS Curve Number method is presented in chapter 5. While the SCS-Curve Number (CN) method (Hawkins et al., 2009; NEH4, 2004) is used to predict event runoff from event rainfall, the general form of the equation was found to be better suited to predict R_a from P_a data than using a linear regression. Nevertheless, the parameters used in the CN method had to be re-evaluated and the concept of an initial abstraction ratio was dropped from the equation by setting the λ parameter to zero.

Estimated annual curve numbers for the P_a-R_a relation were substantially lower than for the original event-based CN method. Nevertheless, large similarities between the event-based CN method and the modified annual CN method were found; the annual CN number reflects the effects of different land use types on the P_a-R_a relation with lower annual CN values corresponding to land uses that are less prone to runoff generation (section 5.3.1). Furthermore, also the important effect of topsoil texture on the P_a-R_a relation, as expressed by the hydrologic soil group was demonstrated, while no substantial effect of plot length or slope gradient was found.

An important effect of intra-annual precipitation distribution was expected, but this could only be demonstrated through simulation (section 5.3.2) and not in the plot-measured data due to (1) a lack of detailed daily or event precipitation data in the publications from which plot runoff and soil loss were extracted, (2) difficulties in obtaining the required precipitation records from climatologic databases and (3) difficulties to summarize the effects of event precipitation distribution into a single parameter such as the precipitation concentration index (Eq. 5.7) that relates well to R_a and SL_a.

8.5 Explaining variability in plot-measured annual runoff and soil loss

Experimental assessments of R_a and SL_a are affected by a large degree of variability, especially under field conditions (Nearing et al., 1999). This variability is mostly due to the complexity of runoff and soil loss generating processes. For plot runoff and soil loss studies under field conditions, several environmental factors that control R_a and SL_a are always unaccounted for and can not be controlled (e.g. soil microtopography, spatial and temporal heterogeneity in soil infiltration rates). This causes a significant amount of random variability in the plot-measured R_a and SL_a data, that may obscure the effect of other controlling factors and explains some of the observations made throughout this research. This is illustrated in Fig. 8.1 and can account for e.g. the trends observed in Fig. 4.6a and Fig. 6.4. The importance of randomness in soil loss assessment was also observed by (Nearing, 1998; Nearing et al., 1999), and greatly complicates the analysis of plot-measured R_a and SL_a rates.

Nevertheless, the analysis of methodological uncertainty, and temporal and spatial variablity associated with plot-measured R_a and SL_a is essential for improving our understanding of the potential effects of land use changes and SWCTs in reducing R_a and SL_a.

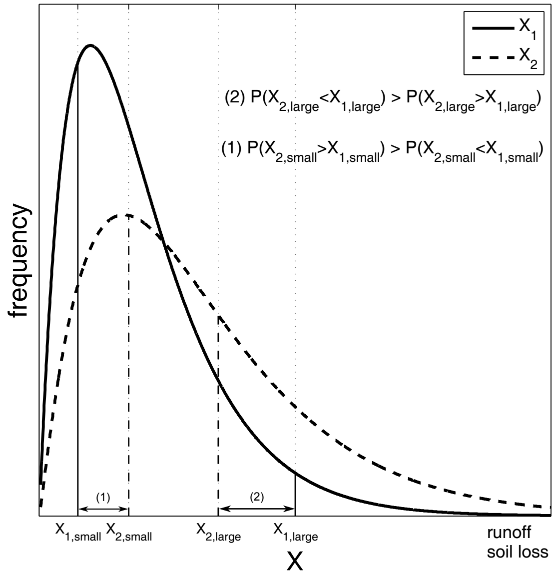


Figure 8.1: Two hypothetical frequency distributions of X (i.e. R_a or SL_a) for e.g. a time series of SL_a measurements in climate zone 1 (X_1) and a time series of SL_a measurements in climatic zone 2 (X_2). Due to the substantial random variability in R_a and SL_a measurements, small measured X_1 are likely to be larger in the X_2 measurement (situation (1)), while large measured X_1 are likely to be smaller in the X_2 measurement (situation (2)), making it difficult to detect differences in the distributions when only a limited number of observations is available. Due to the right-tailed nature of most R_a and SL_a frequency distributions, situation (2) is more pronounced than situation (1).

8.5.1 Temporal variability in annual runoff and soil loss

A first important source of variability in R_a and SL_a data is temporal or inter-annual variability, and this was addressed in chapter 4. Comparing plot-measured R_a and SL_a rates with short measuring periods (<5yrs.) can result in considerable uncertainty as there is a large inter-annual variability between individual years (section 4.3.1) and R_a and SL_a frequency distribution characteristics such as the coefficient of variation and skewness are strongly dependent on the measuring period (Fig. 4.4). As a large proportion of plot-measured R_a and SL_a data are measured over short measuring periods (mean: 6 yrs., median: 4 yrs.; chapter 2), this indicates that mean R_a and SL_a for most plots are not reliable estimators of long-term average R_a and SL_a. Temporal variability in R_a and SL_a were shown to be related (Fig. 4.5), but temporal variability in RC_a is generally smaller than temporal variability in SL_a (Fig. 4.12).

Furthermore, there are substantial differences between temporal variability in plot-measured SL_a and catchment sediment yield (Vanmaercke et al., 2012b), which can improve our understanding of differences in erosion processes between these spatial scales (section 4.3).

So far, most erosion studies have implicitly assumed that temporal variability is equal for different plots and plot-measuring stations when comparing results. This may not be the case however, and several factors potentially control temporal variability. Closer examination of several environmental factors (i.e. climatic zone, land use type, plot length, slope gradient and annual precipitation) which show that these factors explain little temporal variability, and indicate that a large portion of the observed variability may indeed be random. Uncertainties associated with random variability in plot runoff and soil loss measurements can introduce substantial random variability which makes the observation of actual temporal effects all the more difficult (Fig. 8.1). Furthermore, several authors have indicated the importance of the occurrence of low-frequency, high-intensity events (i.e. intra-annual variability) (e.g. de Figueiredo et al., 1998; Edwards and Owens, 1991; Larson et al., 1997; Poesen et al., 1996) in temporal variability of R_a and SL_a data, but this could not be clearly demonstrated with the available data.

8.5.2 Spatial variability in annual runoff and soil loss

Spatial variability in plot-measured R_a and SL_a comprises both scale dependency of R_a and SL_a and spatial patterns of R_a and SL_a. The range of spatial scales covered by runoff and soil loss plots is rather limited (Fig. 1.6) and all plots represent the effects of splash, interrill and rill erosion process at the hillslope scale. Nevertheless, even within the plot scale, scale dependency is known to occur with respect to contributing area for runoff and deposition of sediments within the plot before it reaches the collector system. The available R_a and SL_a data do not allow an in-depth analysis of these processes however. Nevertheless, relatively little variability in the observed R_a and SL_a rates could be attributed to plot length throughout this research, suggesting that scale dependency within the plot scale is relatively unimportant compared to the wide range of other factors controlling R_a and SL_a over the whole of Europe and the Mediterranean.

The importance of spatial variability was confirmed for the relation between P_a and R_a in chapter 5. Linear mixed effect modelling showed that between 30% and 93% of the variability in the P_a-R_a relation can be attributed to differences between plots and plot-measuring stations. Spatial variability was further assessed through the examination of regional differences in R_a and

SL_a between climatic zones (chapter 3), and by examining the effects of several environmental factors on R_a and SL_a throughout this research. Generally, the effects of several factors that are known to be important controls of R_a and SL_a in local plot studies did not explain a substantial part of the variability in R_a and SL_a observed at the continental scale. This suggests that other factors that are relatively homogeneous at the local scale (e.g. intra-annual precipitation distribution, soil texture and cracking behaviour), but vary considerable at the continental scale may be important causes of variability in R_a and SL_a observed in this research. This concept was shown for the effect of hydrologic soil groups, which are generally characteristic for a plot measuring station, on the P_a-R_a relation (section 5.3.1). An important effect of intra-annual rainfall distribution (i.e. event rainfall depth and concentration) on R_a and SL_a was also expected. This effect was explored through simulation of the P_a-R_a relation for different distributions of daily precipitation (section 5.3.2). However, the effects of intra-annual precipitation distribution on R_a could not be clearly demonstrated in the plot-measured data.

8.6 Recommendations for further research and applications

8.6.1 Experimental runoff and soil loss data

Given the labour-intensive, time-consuming and expensive nature of plot runoff and soil loss measurements, additional large coordinated research projects extensively using runoff and soil loss plots in Europe and the Mediterranean are unlikely to be set up. Specific research gaps such as quantification of runoff and soil loss from urban-industrial sites can be addressed with plot measurements. Nevertheless, more runoff and soil loss plot data likely exist, and also the inclusion of additional details (e.g. published event runoff and soil loss) in the databases described in chapter 2 and chapter 7 can have substantial added value to further investigate the research questions in this study, rather than conducting new runoff and soil loss plot experiments.

Also the integration of the plot runoff and soil loss database presented in this research with other climatological (e.g. E-Obs, CRU CL), soil (e.g. European Soil Database) and land use (e.g. CORINE, LANMAP) databases is a point for further research. This can give considerable added value to plot-measured R_a and SL_a data in new analyses, as was shown by e.g. Cerdan et al. (2010). Nevertheless, the analysis in chapter 5 and chapter 6 showed that this integration is not straightforward and requires more research.

Furthermore, also the relations between runoff and soil loss measured on field plots and runoff and soil loss measured using other methods such as rainfall simulations, soil surface lowering, radionuclide techniques and catchment sediment yield should be studied. While there are considerable uncertainties associated with the differences in methodology, all these methods quantify soil loss at different temporal and spatial scales and include different erosion processes. Hence, comparisons between different measuring methods can yield valuable insights in the dominant erosion processes (and hence targets for conservation efforts) at different spatial and temporal scales. An example of such comparative study between plot-scale soil loss and catchment sediment yield is given in Vanmaercke et al. (2012a).

8.6.2 A continental perspective on runoff and soil loss

As indicated in chapter 1, there is an increasing need to assess soil degradation rates at the continental scale in Europe and the Mediterranean, as these problems are increasingly seen as requiring a globally coordinated strategy and many policy initiatives originate at the continental scale in Europe. Plot runoff and soil loss measurements in Europe and the Mediterranean are typically conducted at a local scale however. Consequently, many of these studies have focussed on environmental factors that have been shown to have a strong control over R_a and SL_a at the local scale such as plot length, slope gradient, rooting density or microtopography and insights in the effects of these local factors on runoff and soil loss have become increasingly detailed.

Nevertheless, these are not necessarily the factors explaining most variability in plot-scale R_a and SL_a at the continental scale. This was illustrated in chapter 3 and chapter 5, where plot length and slope gradient were found to explain little variability in R_a and SL_a at the continental scale. At a continental scale, other factors such as land use characteristics (e.g. seasonality of vegetation cover), soil characteristics (e.g. texture and cracking behaviour) and precipitation characteristics (e.g. rainfall seasonality and intensity) are likely to be more important. Nevertheless, many of these factors are more or less homogeneous at the site scale and hence they are generally not well reported on in local scale studies. Better identification of the factors controlling R_a and SL_a at the continental scale can significantly improve predictions at the continental scale and possibilities to extrapolate local-scale models to larger areas.

8.6.3 Soil and water conservation techniques

The relation between runoff reduction and soil loss reduction has received little attention in global (review) studies. Nevertheless, studies of the effects of SWCTs on R_a clearly have many practical applications. Although it was shown in chapter 7 that SWCTs are less effective in reducing R_a than in reducing SL_a, on-site SWCTs still have considerable potential to act as buffers to runoff and to promote infiltration, thereby increasing the amount of plant-available water and reducing the need for costly off-site buffering measures such as retention basins.

There is currently no widely applicable erosion assessment procedure or erosion model that is quantitatively accurate to a degree that is acceptable for conservation planning, and that only requires input parameters that are feasible to collect for a typical conservation planning office. While such model may be ambitious, effective and economically efficient applications of SWCTs certainly require better knowledge on the reduction of R_a and SL_a rates by different types of SWCTs, and especially on the environmental factors that control variability in the effectiveness of SWCTs in reducing R_a and SL_a at a regional and continental scale. While the general difference in effectiveness between different SWCTs has been clearly shown in chapter 7, environmental factors that control the effectiveness of SWCTs in reducing R_a and SL_a could not be clearly identified. Nevertheless, a better knowledge on the factors controlling SWCT effectiveness is of key importance in translating the results in chapter 7 into practical guidelines or incorporating SWCT application in erosion models.

8.6.4 Quantifying variability and uncertainty

Long-term mean and regional mean runoff and soil loss rates have important applications in the identification of erosion hotspots and the assessment of the long-term effects of land and soil degradation. However, such long-term and regional means are mainly indicative of the problem of soil degradation, but contribute relatively little to the solution. The estimation of the (potential) temporal and spatial variability of runoff and soil loss rates is of key importance for several practical applications. Many plot-scale erosion studies have indicated substantial variability in R_a and SL_a, but few studies have specifically tried to further quantify this variability or explore the environmental factors controlling it. It is now well established that variability is an intrinsic part of soil erosion research, yet it remains generally poorly understood, especially at regional or continental scales.

As indicated in section 8.5, an investigation of the magnitude and factors causing variability in R_a and SL_a measurements is necessary. R_a and SL_a variability should not be considered a *"necessary evil"* precluding accurate model predictions, but should be a topic of research in its own right. Especially large-scale erosion models (e.g. regional or continental-scale models) would gain a lot in potential if they would incorporate a more probabilistic approach and predict likely ranges (i.e. frequency distributions) of R_a and SL_a rather than just mean values with a large, but mostly unknown, variability.

For some aspects of conservation planning such as flood prevention, maximum runoff and soil loss rates within a certain return period, rather than long-term average rates are important. This requires better knowledge on how to assess these maximum runoff and soil loss rates, not only on an annual, but also on the event scale (i.e. for different temporal resolutions: daily, annual and multi-annual averages). Hence, a detailed analysis of plot-measured runoff and soil loss rates at the event scale can contribute to the identification of the important factors that control temporal and spatial variability. As the number of studies with sufficiently detailed runoff and soil loss data for such analysis is relatively limited in Europe and the Mediterranean, research should be mainly directed at using widely available data such as precipitation records and distributions to accurately predict measured event-scale distributions of runoff and soil loss rates over a range of environmental conditions. This can contribute to a more pragmatic approach to erosion modelling, based on the prediction of extreme erosion events that are likely to occur within the design period for conservation planning, rather than to estimate long-term mean runoff and soil loss.

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