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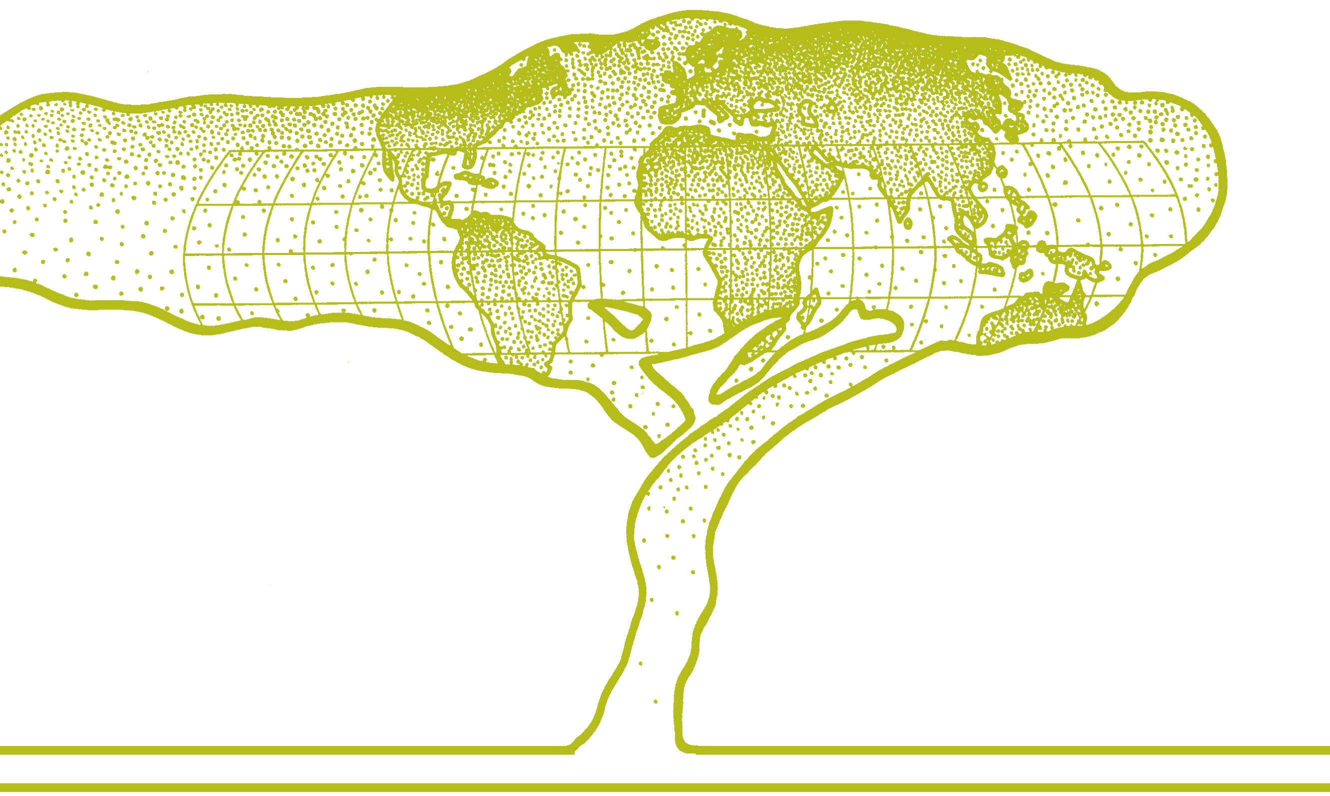

Ping ZHOU

Landscape-scale soil erosion modelling and ecological restoration for a mountainous watershed in Sichuan, China

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Landscape-scale soil erosion modelling and ecological restoration for a mountainous watershed in Sichuan, China

Ping ZHOU

Academic dissertation
for the Dr.Sc. (Agric.&For.) Degree

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ABSTRACT

Soil erosion control, at specific sites, requires quantitative evaluation of the potential soil erosion rate, as well as of the vegetation cover and its dynamics. The Revised Universal Soil Loss Equation (RUSLE) and geospatial data were used to model the soil erosion rate for the ultimate aim of soil conservation and vegetation rehabilitation in the Upper Min River (UMR) watershed of the Upper Yangtze River Basin, in Sichuan, China. The data used in this study for generating the soil loss were derived from Landsat Enhanced Thematic Mapper (ETM+) imagery, the Digitized Elevation Model (DEM), the soil erodibility value, and rainfall erosivity, as well as from a field inventory. The non-parametric *k*-nearest neighbor (*k*-NN) method was used to produce a vegetation cover management map by integrating the ETM images with vegetation coverage data measured on 625 field sample plots. The root mean square errors and the significance of biases at pixel level were evaluated in order to find optimal parameters. Raster maps were produced for describing the soil erodibility, rainfall erosivity, the slope length and steepness, and the cover management factor; and the soil loss risks were quantified by constructing a map indicating the soil erosion potential.

The restoration of the vegetation in a watershed needs to consider the natural vegetation distribution and dynamics. In the UMR watershed, the soil types, current vegetation distribution, vegetation dynamics and reforestation were studied by combining the information from the field inventory and from ancillary datasets. The study further investigated the relationship between vegetation types and soil orders, predicted, using logistic regression, the occurrence percentages of tree species that have a potential for forest landscape restoration, identified the priority areas for rapid restoration, and pinpointed the difficult areas for forest restoration where low precipitation is a constraint. The results showed that the vegetation types observed were well correlated with soil orders, and the latter could be used to deduce the potential restored vegetation in the areas of degraded secondary forest. Suitable tree species for restoration were suggested for different soil types at different elevations.

The study area in which different levels of human disturbance were also examined had an extent of 7 432 km^2. Based on the 625 sample plots studied in the field, the sites with forest cover were divided into four forests classes: (1) near-natural forests, (2) selectively logged forests, (3) natural regeneration forests (after clear-cutting), and (4) plantations. Forests at these four levels of human impact were analysed for the following quantitative characteristics: stand volume, basal area, weighted diameter, weighted height, and biodiversity indices for the woody plant species. The results imply that near-natural forests, with their higher biodiversity, can be used as references when developing new strategies for forest restoration.

The study demonstrated that a vegetation cover is essential for preventing excessive soil erosion in this mountainous watershed. The model developed for different vegetation cover scenarios also provided quantitative information on how the erosion rate could be reduced by different management interventions.

Keywords: Ecological restoration, Human disturbance, *k*-NN technique, Mountainous watershed, Revised universal soil loss equation, soil erosion, Sichuan, Upper Min River watershed

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PREFACE

This study was initiated under the auspices of the “Trees for the Yangtze River: Watershed management and ecosystem rehabilitation in Sichuan Province, China (WAMEC)” project (2004-2006), financed by the Academy of Finland. The work reported here was carried out at the Viikki Tropical Resources Institute (VITRI), Department of Forest Ecology, University of Helsinki, as part of this research project. The author also received scholarships for attending conferences and for finalising the thesis from the Graduate School in Forest Sciences (GSForest) and from the University of Helsinki. I express my gratitude to these organisations that provided the necessary facilities and a generous support for my research.

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I express my sincere appreciation to Prof. Timo Tokola, the co-supervisor of my studies, for his advice on sampling design and data analysis options, and technically checking the methods and results. I also express my gratitude to Juhana Nieminen, Toni Oliver, and Minna Hares for joint preparation of manuscripts. Thanks for many illuminating discussions! We shared many enjoyable moments when we worked on the manuscripts.

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I am utmostly indebted to my parents and grandparents for giving me their unlimited love and support. I am grateful to my sister-in-law Mingzhen Deng for her love and for kindly taking care of my son. Last but not least my deepest gratitude goes to my husband Xianbao Deng for his love and support, and to our lovely son Xun Jack Deng for his brilliant smile and for making my life wonderful.

Helsinki, April 2008

Ping Zhou

LIST OF ORIGINAL PAPERS

This thesis is based on the following original articles:

Study I
Zhou, P., Nieminen, J., Tokola, T., Luukkanen, O. & Oliver, T. 2006. Large scale soil erosion modeling for a mountainous watershed. Geo-Environment & Landscape Evolution II, Martin-Duque, J.F. et al. eds., 55-67. http://library.witpress.com/pages/PaperInfo.asp?PaperID=16212

Study II
Zhou. P., Luukkanen, O., Tokola, T., Nieminen J., 2007. Vegetation Dynamics and Forest Landscape Restoration in the Upper Min River Watershed, Sichuan, China. Restoration Ecology (Online Early Articles). doi:10.1111/j.1526-100X.2007.00307.x

Study III
Zhou, P., Luukkanen, O., Tokola, T., Hares, M. 2008.
Comparison of forest stands characteristics and biodiversity under different human impacts along an altitudinal gradient for mountainous watershed restoration in Sichuan, China. (Submitted).

Study IV
Zhou, P., Luukkanen, O., Tokola, T., Nieminen, J. 2008.
Effect of vegetation factor on soil erosion in a mountainous watershed. (Submitted).

In studies I - IV Ping Zhou designed the framework, organized experimental arrangements, collected the data, integrated the datasets into GIS, did statistical analysis, and prepared the manuscripts, which were revised and guided by Olavi Luukkanen. Timo Tokola gave advice on sampling design and data analysis options and technically checked the methods and results. In studies I, II, and IV, Juhana Nieminen was involved in field data collection and *k*-NN estimations. In study I, Toni Oliver assisted in double-checking the quality of input datasets and compared the digital elevation models interpolated by different methods. In study III, Minna Hares helped in preparing and revising the manuscript.

LIST OF MAIN ACRONYMS AND ABBREVIATIONS

AAP	Average Annual Precipitation
C	Cover management factor
DEM	Digital Elevation Model
EPIGPA	Environmental Protection Investigation Group of the Political Association
ETM	Enhanced Thematic Mapper
FAO	Food and Agriculture Organization of United Nations
GIS	Geographic Information Systems
GLCF	Global Land Cover Facility
GPS	Global Positioning Systems
IDW	Inverse Distance Weighting
ISRIC	International Soil Reference and Information Center
K	Soil erodibility factor
k-NN	kernel Nearest Neighbouring
L	Slope length factor
LDA	Linear Discriminate Analysis
LS	Slope length and steepness factor
P	Support practice factor
R	Rainfall erosivity factor
RMSE	Root Mean Square Error
RS	Remote Sensing
RUSLE	Revised Universal Soil Loss Equation
S	Slope steepness factor
SER	Society for Ecological Restoration International
UMR	Upper Min River
UNESCO	United Nations Educational Scientific and Cultural Organization
USDA	United States Department of Agriculture
USLE	Universal Soil Loss Equation
WEPP	Water Erosion Prediction Project

CONTENTS

ABSTRACT 3
PREFACE 5
LIST OF MAIN ACRONYMS AND ABBREVIATIONS 8
1 INTRODUCTION 11
1.1 Deforestation and re-establishment of forests 11
1.2 Soil erosion and its quantitative assessment 14
1.3 Ecological restoration and watershed rehabilitation 18
1.4 Objectives and structure of the study 21
2 MATERIAL AND METHODS 23
2.1 Material 23
2.1.1 Research site 23
2.1.2 Description of geospatial data 27
2.1.3 Sample techniques for field data collection 29
2.2 Data processing and analysis 31
2.2.1 Large-scale soil erosion modelling (I) 31
Slope length and Slope steepness factors (*LS*) 32
Rainfall and runoff erosivity factor (*R*) 34
Soil erodibility factor (*K*) 34
Cover management factor (*C*) 35
Support practice factor (*P*) 35
Soil loss intensity classification 36
2.2.2 Vegetation dynamics and human impact (II, III) 36
Vegetation class change 36
Human impact on forest stands 37
2.2.3 Identification of areas for restoration (I, II, IV) 37
Areas with high erosion risk 37
Identification of dry areas 38
Potential area for rapid restoration 39
2.2.4 Selection of woody species for restoration (II) 40
2.2.5 Effect of vegetation cover on soil erosion (IV) 40

2.3 Evaluation of predictions using empirical data 41
3 RESULTS 42
3.1 Geo-environment in the research area 42
3.2 Soil erosion (*I, IV*) 42
3.2.1 The R-factor 42
3.2.2 The C-factor 43
3.2.3 The LS-factor 43
3.2.4 The K-factor 46
3.2.5 Soil loss potential 46
3.3 Areas for restoration (*I, II, IV*) 47
3.3.1 High erosion risk areas 47
3.3.2 Dry areas 48
3.3.3 Potential areas for rapid restoration 50
3.4 Ecological restoration (*II, III, IV*) 51
3.4.1 Vegetation restoration scenarios for high erosion risk areas 51
3.4.2 Ecological restoration perspectives with different tree species 52
4 DISCUSSION 55
4.1 Large-scale soil erosion modelling 55
4.2 Effect of vegetation cover on soil erosion 58
4.3 Ecological restoration for soil erosion alleviation 59
5 CONCLUSIONS AND RECOMMENDATIONS 65
REFERENCES 67
APPENDICES 82
Appendix 1 Inventory records and equipment 82
Appendix 2 Inventory form 83
Appendix 3 Inventory codes for different records 84
Appendix 4 Vegetation class 85
Appendix 5 Tree species in the study area 86
Appendix 6 Equations for biodiversity 89
Appendix 7 Image before and after removing cloud and cloud-shadow. 90
Appendix 8 Ecozones 91
Appendix 9 Soil 92
Appendix 10 Elevation 93
Appendix 11 Slope 94
Appendix 12 Aspect 95

1 INTRODUCTION

1.1 Deforestation and re-establishment of forests

Forest resources are vital for the human life. Forests contribute to the livelihoods of about 1.6 billion people worldwide (World Bank 2004). Forests are valued for such global services as terrestrial biodiversity conservation, carbon dioxide sequestration, climate regulation, soil and water conservation, and natural disaster alleviation, as well as for locally providing various products, employment opportunities and environmental benefits. Forests are a critical component in the ecological balance, and their role has received increasing recognition with the worldwide awareness of global warming. However, the world lost three percent of its forests from 1990 to 2005, with an average decrease of 0.2 percent per year (UN 2007). Globally, the annual forest loss is now estimated at 13 million hectares (FAO 2007).

Deforestation is a serious problem in most developing countries. Deforestation is caused by over-cutting of trees, agricultural expansion, and other indirect causes, such as wars and conflicts, industrial activities, and urban development. In addition to the reduction of the area covered by forest, both primary and secondary forests have been degraded. The damaging consequences of this degradation commonly include losses in terms of ecological services, provision of goods, and the subsistence for forest-dwelling people (Lamb et al. 2005). An ideal situation would be to find a sustainable way to manage the forest resources and forested lands so as to meet the social, ecological, cultural and spiritual needs of present and future generations (UN 1992). However, the still prevalent signs of a rapid decline in the extent of tropical forests (ITTO 2002), losses of biodiversity (Dirzo and Raven 2003), and the expansion of degraded lands show considerable difficulties in achieving sustainability in forest management.

There has been a long period of over-cutting and illegal logging of natural forests in China (Bull and Nilsson 2004). From 1949 to the end of the 1970s, the policy of forest exploitation led to a significant decrease in the forest carbon storage, while a net increase in the carbon storage for the forests in the country has taken place since the late 1970s, mainly due to rapid establishment of forest plantations (Fang et al. 2001).

The Upper Min River (UMR) watershed, in the Upper Yangtze River Basin, is severely degraded due to deforestation. Figure 1 shows the long-term forest cover changes in this watershed. According to Marco Polo's travel notes, forests covered 50% of the watershed in the late 1200s (Li et al. 2006). However, the forest cover had declined to 30% by 1950 and to 18.8% by the 1980s (and, specifically to only 5%-7% along the main river); Up to 44% of the land in the UMR area has been described as degraded (Wu et al. 2003; Ye et al. 2003). The "Big Leap Forward" campaign, launched in 1958, encouraged the use of homemade furnaces for steel making and led to massive destruction of forests (Wang et al. 2004a). Large-scale logging in the UMR area reached its peak between 1950 and 1980 (Ye et al. 2003). Before a logging ban in 1998, the main focus of the forest sector in China was timber production, and by this time the total annual volume of timber extraction in the Aba prefecture in Sichuan province to which the UMR watershed belongs had reached 980 000 m^3, which was four times the annual increment, clearly indicating that deforestation had accelerated dramatically (EPIGPA 1990).

However, the forest cover in the UMR watershed increased a little, to 18.8 %, in the 1980s, and to 21% by 2004, due to the national-scale reforestation and afforestation efforts in China that in many areas had started to restore the forest cover since the 1970s (Fang et al. 2001; Wang et al. 2004a; Kauppi et al. 2006).

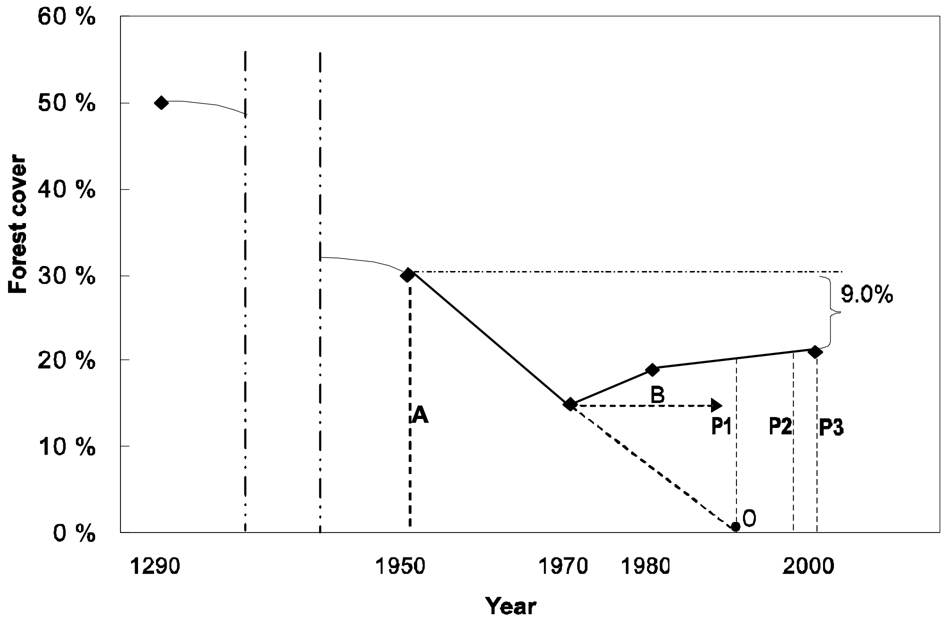

Figure 1. Forest cover changes in the UMR watershed. Net deforestation occurred before the 1970s, while recovery took place thereafter. An increase of 9 percentage points in forest cover is needed to reach the previous condition of the 1950s. A: the “Big Leap Forward” of the late 1950s and early 1960s; B: reforestation from the 1970s; P1: an ecological programme in 1989; P2: natural forest logging ban in 1998; P3: “Grain for Green” programme in 2000; O: Possible course of forest decline without reforestation activities.

The rehabilitation activities used during this period included aerial seeding, limiting of access to mountain areas, and planting of tree seedlings. In addition, other national programmes have contributed to improving the state of the forests in the UMR watershed. These programmes have included conservation of forests in the upper and middle reaches of the Yangtze River since 1989, a logging ban in natural forests in 1998, and a “Grain for Green” programme which has converted farmlands either to forest or to grassland in the upper Yangtze River basin since 2000 (Fig. 1).

After these reforestation activities the forest cover in the UMR watershed is still 9 percentage points lower than that at the beginning of the 1950s, i.e. prior to the acceleration of deforestation. With reduced vegetation cover, the runoff and soil erosion greatly increased, resulting in flooding and mudslides (Varis and Vakkilainen 2001; Sidle et al. 2004), that further damaged the vegetation and continued to threaten human lives and homes (Gurevitch et al. 2002). Therefore, scientific research is needed to identify suitable restoration methods and to delineate priority areas for achieving a sufficient forest cover which for this area has been set at 30%, i.e. at the level prior to the acceleration of deforestation. In such efforts, it is obviously essential to identify vulnerable areas with low vegetation cover and high erosion risk and to take advantage of the regeneration potential in the remaining forests for land rehabilitation.

1.2 Soil erosion and its quantitative assessment

Deforestation is the cause of various problems, such as accelerated soil erosion, loss of biodiversity, and an increase in the atmospheric carbon dioxide concentration. Soil erosion is a natural process that detaches and transport soil material through the action of an erosive agent (Foster and Meyer 1972). At the global scale, soil erosion by water is the most important land degradation problem (Eswaran et al. 2001). Water erosion includes splash erosion, sheet or interrill erosion, rill erosion, and gully erosion. Splash erosion occurs when soil particles are detached and transported as a result of the impact of falling raindrops. Sheet or interrill erosion removes soil in thin layers and is caused by the combined effects of splash erosion and surface runoff. Rill erosion is the disappearance of soil particles caused by concentrations of flowing water. Gully erosion occurs when the flow concentration becomes large and the incisions deeper and wider than with rills (Morgan 2005).

The total area of China affected by soil and water erosion is 3.67 million km^2 (38.2% of the total national territory), and the annual increase of the eroded area is about 10,000 km^2. Some 5 billion tonnes of soil is washed into reservoirs and lakes each year (Zhu 2000). The upper reaches of the Yangtze River and the upper section of the middle reaches of the Yellow River are the regions suffering from the most serious soil and water erosion in China. According to remote sensing survey results released by the State Council in 1990, the total land area affected by soil and water erosion in the Yangtze River catchments is 560,000 km^2, and 2.4 billion tonnes of soil is eroded here each year. Of this area, 352,000 km^2 is in the upper reaches, where the annual soil loss is 1.56 billion tonnes (MWR 1999a). Some of eroded soil from the Yangtze River catchments is deposited in reservoirs, side streams, and small and medium river channels, constituting threats to flood control, irrigation, water supply, and the hydro-electricity generation in small and medium-sized rivers. A catastrophic flood in 1998 in the Upper Yangtze region raised public attention to the problems of soil erosion and sedimentation (MWR 1999b).

Soil loss control calls for spatial erosion assessment at different scales. At point or plot level, soil erosion can be qualitatively assessed by certain criteria or directly measured by field devices. Hudson (FAO 1993) documented the methods to measure soil erosion rates in the field. Soil erosion can be qualitatively classified into the following four groups: slight, moderate, severe and extreme (FAO 2006). Slight erosion shows some evidence of damage to surface horizons while the original biotic functions of the soil largely remain intact. Moderate erosion is defined as bearing clear evidence of removal of the surface horizons, with the original biotic functions partly destroyed. Severe erosion refers to the surface horizons completely removed, the subsurface horizons exposed, and the original biotic functions largely destroyed. Extreme erosion refers to substantial removal of the deeper subsurface horizons (badlands) and to completely destroy of original biotic functions. Factors that regulate the soil erosion processes include climate, soil, terrain and ground cover (Lal 2001).

At a landscape level, the soil erosion can be evaluated qualitatively by remote sensing data or quantitatively by integrating spatial data on erosion factors. An eroded area can be identified from aerial photographs, and large and medium-sized gullies can be detected from various remote sensing data such as aerial photographs, Landsat, SPOT, or ASTER imagery (Langran 1983; Millington and Townshend 1984; Servenay and Prat 2003; Vrieling 2007).

Different methods have been developed to assess the soil erosion loss quantitatively, for instance, the Universal Soil Loss Equation (USLE; Wischmeier and Smith 1978), the Areal Nonpoint Source Watershed Environment Response Simulation model (ANSWERS; Beasley et al. 1980), the Modified Universal Soil Loss Equation (MUSLE; Smith et al. 1984), the Thorns model (Thorns 1985), the Agriculture Nonpoint Source Pollution model (AGNPS; Young et al. 1989), the Soil Erosion Model for Mediterranean Regions (SEMMED; De Jong 1994), the Water Erosion Prediction Project (WEPP) Hillslope model (Flanagan and Nearing 1995), and Revised Universal

Soil Loss Equation (RUSLE; Renard et al. 1997). Spatial information science and its various techniques including remote sensing (RS), geographic information systems (GIS), global positioning systems (GPS), and related internet technology have been widely applied to soil erosion monitoring and surveying since the 1980s.

Of the above models, the USLE and the RUSLE have provided a convenient tool for soil loss evaluation by taking the climate, geographical terrain, conservation support practice, soil, and vegetation simultaneously into consideration. The slope length and steepness factor, which reflects the terrain at a given site, can be computed from a DEM (Moore and Burch 1986a, b). The rainfall and runoff erosivity factor has been calculated in many studies based on the storm events and rainfall data (Renard et al. 1994; Millward and Mersey 1999; Angima et al. 2003; Ma et al. 2003). In a mountainous watershed, the effect of elevation on precipitation can be used to improve the geostatistical interpolation of the rainfall and runoff erosivity factor (Hevesi et al. 1992a, b; Goovaerts 1999).

The effect of vegetation cover on soil erosion is mainly assessed by three different ways. The first one is direct application of the C-factor from the RUSLE based on prior land use (PLU), canopy cover (CC), surface cover (SC), surface roughness (SR), and soil moisture (SM) sub-factors (Renard et al., 1997). The second one is to assign a C-factor according to a qualitative ranking of vegetation types (Wischmeier and Smith 1978; Morgan 1995). The third method is to calculate the C-factor from the Normalized Difference Vegetation Index (NDVI), defined as the near-infrared reflection minus the red reflection divided by the sum of the two (Tucker 1979; Thiam 2003; Wu et al. 2004).

However, it is difficult to apply the above methods to estimate the C-factor in cases similar to the present study area, because it is difficult to use the first method to assess PLU, CC, SC, SR, and SM simultaneously for a large area not mainly covered by agricultural land. The second method does not consider the variation in vegetation density within the same vegetation type, and the third method might lead to a poor relationship between Landsat-derived spectral indices and vegetation attributes.

The non-parametric k-nearest neighbour (k-NN) technique, widely used in a variety of forest inventory and biomass mapping applications over the years (Tokola 2000; Franco-Lopez et al. 2001; Katila and Tomppo 2001), provides a new method to map proportional vegetation cover. The *k*-NN technique is a nonparametric approach to predicting values of point variables on the basis of similarity in a covariate space between a given point and other points with observed values for the variables (Tomppo 1991).

A good vegetation cover is generally capable of preventing surface erosion, thus also reducing landslides. Removal of vegetation can greatly increase the runoff and soil erosion, particularly in mountainous areas (Gurevitch et al. 2002). Soil erosion control especially calls for forest restoration or rehabilitation, so as to reduce the erosion loss and to improve the soil stability.

1.3 Ecological restoration and watershed rehabilitation

As discussed above, the UMR watershed has been degraded due to deforestation, and it requires ecological restoration. Ecological restoration is an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability (SER 2004). A restored ecosystem should consist of indigenous species to the greatest practicable extent, sustain itself structurally and functionally, and be resilient to a normal range of environmental stresses and disturbances. Ecological restoration seeks to return destroyed, damaged or degraded ecosystems to their original state, both structurally and functionally as closely as possible, while rehabilitation aims at improving some functions of an ecosystem but not necessarily at fully restoring all its components (Roni 2005).

Forest restoration, which connotes a transition from a degraded state to a former near natural condition, encompasses concepts such as afforestation, reclamation and rehabilitation (Buck 2005). Forest restoration is addressed by commitments expressed by all major forest-related international conventions and processes, including Agenda 21 and the non-binding forest principles of the UN Conference on Environment and Development, the Intergovernmental Panel on Forests/Intergovernmental Forum on Forests, the UN Forum on Forests, the UN Convention on Biological Diversity, the UN Framework Convention on Climate Change, and the UN Convention to Combat Desertification. In global forest policy development, the Legally Non-Binding Instrument on all types of forests, adopted by the UN general assembly in December 2007, outlines the global objectives for sustainable forest management, as well as the direction for national policies and legislation for achieving them (UN 2008).

There are different approaches related to forest restoration, such as ecosystem management, the ecosystem approach, sustainable forest management, forest restoration with a landscape approach, and forest landscape restoration (Schlaepfer 2005). Forest landscape restoration, which is defined as an approach to restore the functions of forests across a whole landscape, brings stakeholders together to identify and put in place a mix of land-use practices (Pye-Smith and Saint-Laurent 2003). For practical reasons in forest landscape restoration, we need the knowledge of local species distribution at the landscape level, since natural biodiversity can be best maintained if forest restoration mimics the natural ecosystem processes (Fries et al. 1997).

Ecological restoration often requires the collection, movement, and mixing of huge amounts of plant material, typically seed (Mortlock 2000). The natural seed dispersal to restore plant communities requires connecting of seed sources with the restoration sites (Mouissie et al. 2005). Seed dispersal models have shown that the number of seeds occurring at a certain location decreases with the increasing distance to the seed source (Bullock and Clark 2000; Coulson et al. 2001). Thus, suitable buffers outside residual forests can be identified for rapid restoration by taking advantage of the available seed sources in the existing forests.

A watershed encompasses the total land area above some point at a stream or river which drains that point (Pereira 1973); it is made up of the natural resources in a basin, especially the water, soil, and vegetation factors (Talat 1977). The watershed is a hydrological unit often used both as a physical-biological unit and a socio-economic-political unit for the planning and management of natural resources (Sheng 1990).

The comprehensive development of a watershed, so as to make productive use of all its natural resources and also to protect them, is termed "watershed management". This includes land improvement, rehabilitation, and other technical measures, as well as human considerations (Talat 1977). Watershed management should recognize the connectivity provided by the stream system and the interrelationships among land use, soil and water, and the linkages between upland and downstream areas (Brooks et al. 1991).

Protection and reclamation of eroding areas are essential features of watershed restoration. If the soil is readily eroded, hydrological disturbances are also more likely to occur. The soil surface should be maintained with an erosion-resistant surface, such as that provided by vegetation cover. Vegetation and soil-related measures are thus key interventions in sustainable watershed management (Schiechtl 1985).

1.4 Objectives and structure of the study

The general aim of the present study was to analyze the processes of land degradation and rehabilitation in the Upper Min River watershed, in Sichuan, China, and to integrate the information on local ecosystem and landscape changes with the means of rehabilitation, particularly in areas of high soil erosion risk, so as to achieve sustainable watershed management. The specific objectives were as follows: (1) To evaluate the soil erosion loss using ArcGIS, by employing the method of Revised Universal Soil Loss Equation; (2) To identify the areas with high soil erosion loss in this particular watershed; (3) To examine the present level of vegetation degradation in the study area and to suggest a restoration pattern for a specified sub-watershed, in order to understand the interrelationships between vegetation and soil, by analyzing the vegetation distribution on landscape scale and deducing the potential vegetation type in order to establish a vegetation restoration strategy; (4) To identify priority areas for rapid ecological restoration; by taking advantage of the existing remnant forests; and (5) To analyze the relationship between soil erosion intensity and its affecting factors and thereby clarifying the effect of vegetation cover on erosion risk.

The study was divided into three phases: The first phase (I) aimed at quantitative evaluation of the soil erosion loss based on its component factors; this included processing of topographical and soil maps, remote sensing data, precipitation data, as well as field sample plots measurements to provide input raster maps for calculating the soil loss. In the second phase (II, III), the following factors were studied: forest stand characteristics under human impact, soil type, the current vegetation distribution, vegetation dynamics, possibilities for afforestation, and the choice of potential tree species for restoration. The third phase (IV) also included on the basis of a quantitative model of soil erosion, further clarification of the relationship between soil loss and its affecting factors, identification of areas with high erosion risk, and quantitative modelling of different vegetation cover scenarios as related to reduced soil erosion after vegetation restoration.

Together these steps of research were expected to improve our capability (1) To produce landscape-level soil erosion maps for a mountainous watershed with account of potential improvement measures; (2) To select suitable woody plant species for forest restoration in a given situation; and (3) To create procedures for landscape-level forest restoration that specifically focuses on problem areas, such as those with high erosion risk or low precipitation.

2 MATERIAL AND METHODS

2.1 Material

2.1.1 Research site

The Upper Yangtze River Basin is a mountainous region, which has an area of 1.04 million km^2, a mean annual runoff discharge of 435 million m^3, a mean sediment yield of 517 million tons, and a population of 140 million (Zhang and Wen 2004). The basin is one of the most severely eroded areas in China. In the eastern part of the basin, where the population density ranges between 100 and 800 people per square kilometer, arable land resources are limited and some of the slopes over 35° are still under cultivation. Water erosion on cultivated land not only results in on-site soil degradation and reduction in crop productivity, but also causes off-site problems related to downstream sedimentation (Zhang et al. 2003).

The Upper Min River, which is one of the most important tributaries of the Upper Yangtze River, is 341 km long with a drainage area of 23,040 km^2. The watershed is located in Sichuan Province, Southwest China, between 31º-33º N and 102º-104º E. The climate is governed by the northeast and southwest monsoons. A complex topography, with elevations ranging between 600 m and over 6,000 m, results in steep gradients of rainfall. The annual precipitation (P) ranges from 405 mm to 1950 mm in different parts of the watershed, the potential evapotranspiration (ET) ranges from 1,500 mm to 600 mm, and the P/ET ratio varies from 0.3 to 3 (Chen et al. 2005). The monthly precipitation shows distinct seasonal variation — about half of the precipitation falls in July, August and September (Fig. 2).

The forest cover amounts to 21% of the whole watershed at the present time. The UMR watershed has been divided into five ecozones: the Sub-tropical (1,300 – 2,200 m), Temperate (2,200 – 2,600 m), Sub-alpine (2,600 – 3,200 m), Boreal (3,200 – 3,600 m) and Arctic zone (3,600 – 5,700 m) (Editorial Board of Sichuan Vegetation 1980).

In the present study, a total of 625 inventory plots were randomly placed in the middle and upper reaches of the UMR watershed (Fig. 2), over an area of about 7,400 km^2, between 31º-33º N and 103º-104º E. The vegetation here ranged from subtropical evergreen broadleaved forest to alpine meadows. The study area was severely degraded due to deforestation and soil degradation; especially on either side of the Min River (Fig. 3). Some near-natural forests had remained in remote areas (Fig. 4). There were three soil orders with five major soil types as follows: Brown Forest Soil and Dark Brown Forest Soil (Alfisol order), Cinnamon Soil (Semi-Alfisol order), and Alpine Meadow Soil and Subalpine Meadow Soil (Semi-Aquatic order) (Guo and Ou 1991; Shi et al. 2005).

The study area has been inhabited for thousands of years by numerous ethnic groups, such as the Han, Tibetans, and Qiang. The main source of livelihood is agriculture (National Bureau of Statistics of China 2002). During the period of nomad immigration (A.D. 220 - 649), the upper limit of the subalpine forest moved downward and was replaced by expanding subalpine meadows (Fan and Zhao 2003). With an increasing number of farmers, much forest was gradually converted to farmland. During the time of the present field study in 2004-2005 when most farmers participated in the “Grain for Green” programme, some farmlands had already been abandoned and become forest or grassland again.

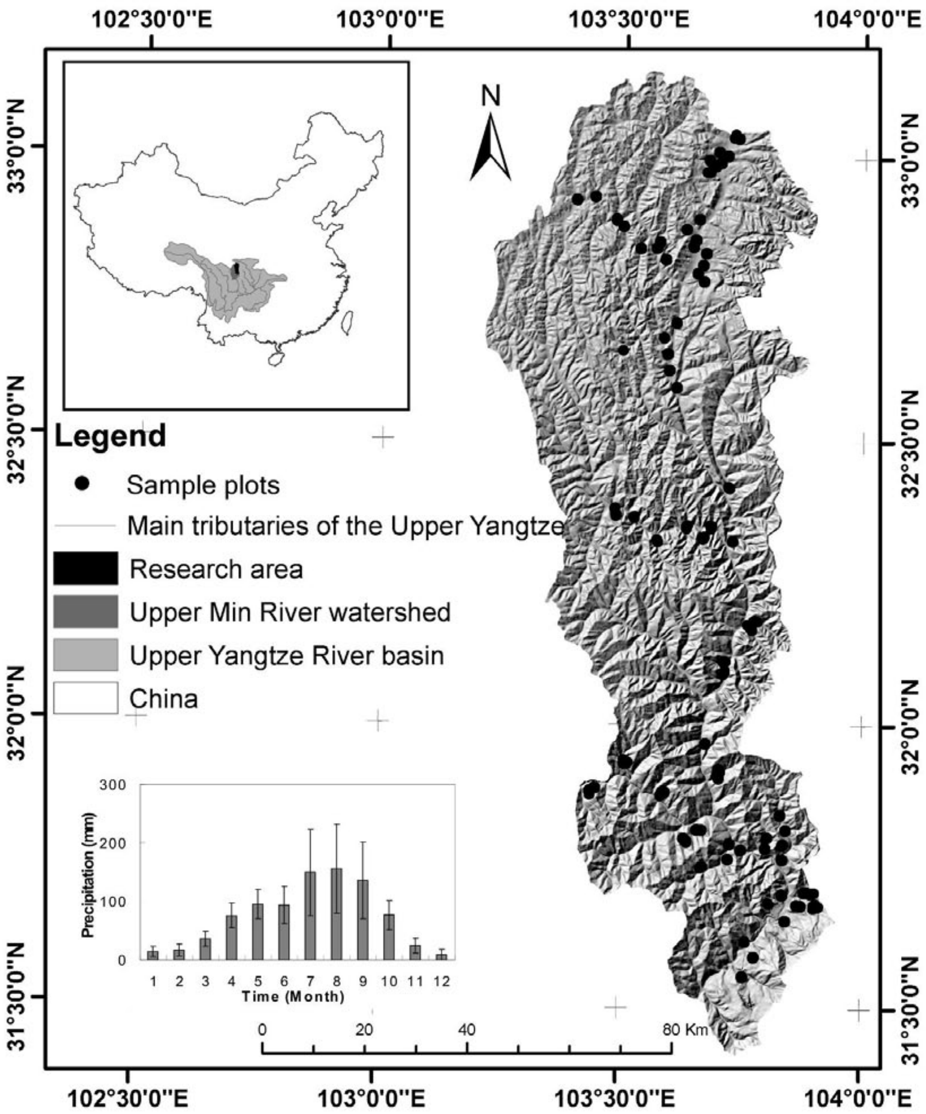

Figure 2. The study area located in the upper and middle reaches of the Upper Min River watershed, in the Upper Yangtze River Basin, Sichuan, China. Dots indicate the locations of 625 sample plots.

Figure 3. The Upper Min River watershed forms a transition from the Qinghai-Tibetan Plateau to the Sichuan basin, with high peaks and steep slopes. The vegetation and soils on either side of the Min River are severely degraded. (Photo: Ping Zhou).

Figure 4. In remote areas of the UMR watershed, with the least human interventions, the near-natural forest (here dominated by *Picea asperata*) remains. (Photo: Ping Zhou).

2.1.2 Description of geospatial data

Two consecutive ETM+ scenes, WRS2 130/037 & 130/038 (GLCF 2002) from the Landsat 7 satellite were used for vegetation cover estimation (Fig. 5A). Landsat ETM+ image data consist of eight spectral bands, with a spatial resolution of 30 m for bands 1 to 5 and band 7. The resolution for band 6 (thermal infrared) is 60 m and that for band 8 (panchromatic) is 15 m. Approximate scene size is 170 km north-south by 183 km east-west. ETM+ bands 1 (0.45-0.52 μm), 2 (0.52-0.60 μm), 3 (0.63–0.69 μm), 4 (0.77–0.90 μm), 5 (1.55–1.75 μm) and 7 (2.09-2.35 μm) were used in this study. The ETM+ images had been orthorectified at the time of image acquisition. To ensure compatibility between images and the ground data, each image was rectified and georeferenced to the Universal Transverse Mercator system (UTM): WGS_1984_UTM_Zone_48N. To minimize the effect of illumination differences on the surface reflectance, spectral bands were normalized using a Lambertian model (Teillet et al. 1982; Civco 1989; Conese et al. 1993). To remove clouds and cloud-shadows, a mask of cloud and cloud-shadow by classifying the pixels into cloud, cloud shadow, or noncloud-nonshadow, with a plurality of images, were used to generate a cloud-free and cloud-shadow free image. Appendix 7 shows an example of an image prior to and after removing clouds and cloud-shadows.

Scanned topographical maps (1:50,000) from the 1970s were used to derive the information on the vegetation 30 years earlier (Fig. 5B). A Vegetation map for 2002 was generated from satellite imagery; the data were obtained from the Chinese Academy of Forestry (Fig. 5C).

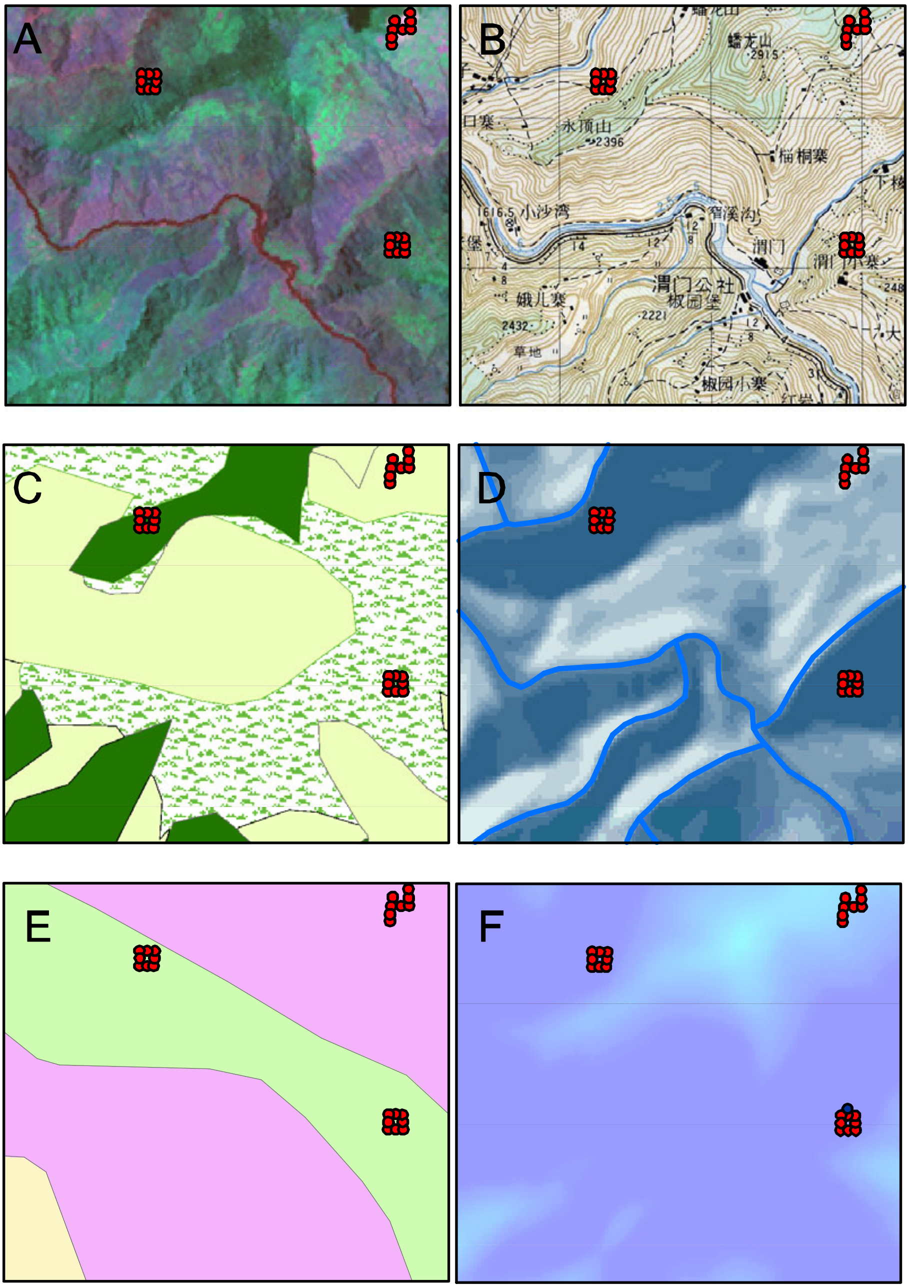

Figure 5. GIS datasets. (A) Landsat ETM+ image; (B) Topographical maps (1:50,000) from the 1970s; (C) Vegetation map for 2002; (D) Digital stream data and hillshade based on 25-m DEM; (E) Map of soil types; (F) Generated precipitation surface.

The Digital Elevation Model (DEM) used was based on a digital topographic map, with 100-m elevation contour lines (Fig. 5B) and stream data (Fig. 5D). It was interpolated into a 25 m cellsize grid with the Topogrid algorithm, which generates a hydrologically correct grid DEM using the contour lines and the stream data.

The soil map was inherently generalized as classes and separated by definite boundaries, which were refined from a 1:1,000,000 digital soil map (Fig. 5E). The map of soil types and the erodibility value for each soil type were used to generate a raster map for the soil erodibility factor.

The precipitation data were collected from 40 meterological stations with coordinates in and around the research area; the observation period was 5 years from 1998 to 2002. The stations were rather evenly distributed throughout the altitudinal range of 1,037 - 3,750 m. The point data were utilized to generate a precipitation surface using different geostatistic models (Fig. 5F). When cokriging was used to generate the average annual precipitation surface, precipitation data from only 38 stations were used (2 stations located nearby the research area that fell in another watershed were excluded).

2.1.3 Sample techniques for field data collection

A field inventory was done in 2004 using strata-delineated sampling with the aid of the global positioning system (GPS). A clustered systematic eight-plot sampling method made it possible to measure at least eight plots in one cluster per day, thereby decreasing the walking distance (Tokola and Shrestha 1999). The first plot of each eight-sample-plot cluster was randomly placed within a maximum of three hours' walking distance from a road. The distance between the two nearest plots was 100 m. Each plot had 1-5 subplots, depending on the vegetation. The sizes of the subplots in the research area were designed according to species-area curves and the optimum size of the plot for a given vegetation type (Cain 1938; Kent and Coker 1996). In total, 625 sample plots in 82 clusters were recorded for the vegetation and soil data. The plots were further divided into subplots as shown in the following:

- 2 m × 2 m for seedlings with height (H) ≤ 1.3 m
- 5 m × 5 m for saplings with H > 1.3 m, DBH ≤ 10 cm
- 10 m × 10 m for trees with 10 cm < DBH ≤ 20 cm
- 20 m × 20 m for trees with 20 cm < DBH ≤ 40 cm
- 30 m × 30 m for trees with DBH > 40 cm.

For each plot, recordings were made on GPS coordinates, stand conditions, tree species, shrub and main herbaceous plant species, tree diameters at breast height, sample tree height, and the tree quality defined as healthy, damaged, or dead, as well as on environmental factors such as elevation, slope and aspect, canopy closure, forest type, soil type, plot condition and land use (Appendices 1-3). Sample tree heights were measured with a clinometer. Missing tree heights were estimated using a height curve; it was computed using a two-parameter regression model for each species by plot separately. Tree volumes were computed for each tree species using volume functions (Zhang 2003; Wang et al. 2004b).

2.2 Data processing and analysis

2.2.1 Large-scale soil erosion modelling (I)

Steps taken during the data processing and analysis in Part I are illustrated in Figure 6, and the key points are further explained in the following paragraphs.

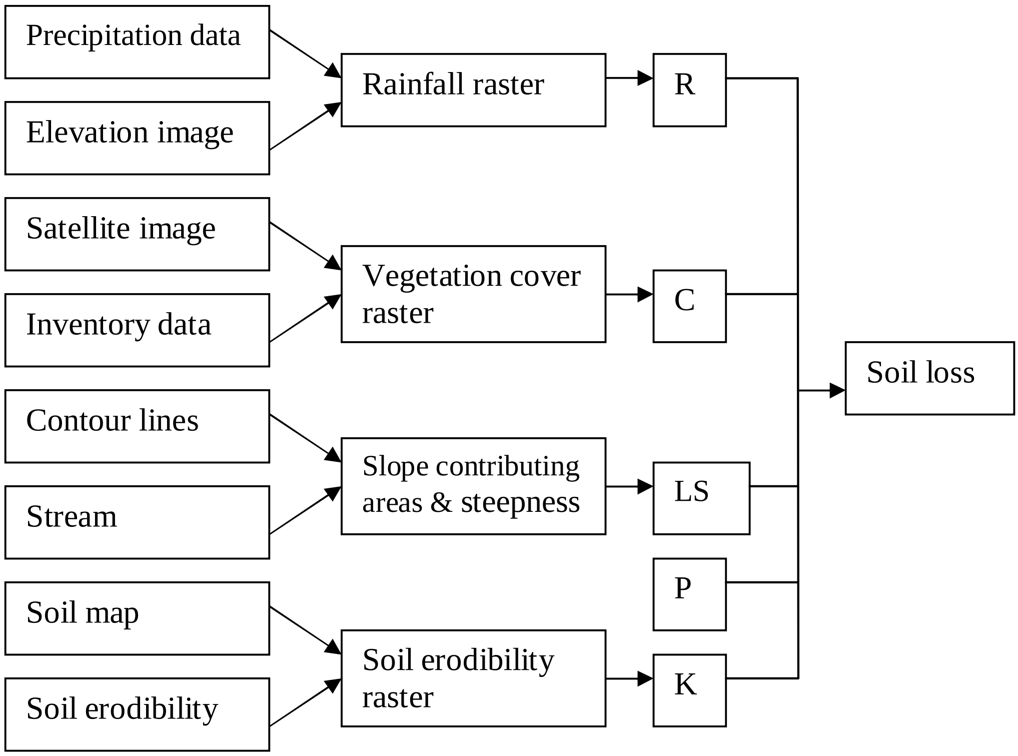

Figure 6. Flow chart for modelling of soil erosion loss caused by water.

The soil loss (A) due to water erosion per unit area per year (Mg ha^{-1} yr^{-1}) was quantified using RUSLE by the following equation:

$$A = R \times K \times LS \times C \times P \quad (1)$$

where A is the average soil loss due to water erosion, R the rainfall and runoff erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}), K the soil erodibility factor (Mg h MJ^{-1}

mm^{-1}), L the slope length factor, S the slope steepness factor, C the cover and management practice factor, and P the support practice.

Slope length and Slope steepness factors (*LS*)

The L-factor and the S-factor, which reflect the topographic erosion susceptibility on a given site, were computed together from the digital elevation model (DEM). The DEM used was based on a digital topographic map, with 100-m elevation contour lines and stream data. In order to achieve a geomorphologically realistic surface, it was interpolated into a 25-m cellsize grid with the Topogrid algorithm (Hutchinson 1989) which generates a hydrologically correct grid DEM using the contour lines and stream data.

In order to ensure flow continuity, small individual depressions caused by low DEM resolution were removed, rising their cell height values until a pouring point was achieved. The slope was calculated using the maximum downhill direction method, in which the slope value for each raster cell is obtained from the angle formed between the cell itself and the lowest neighbouring cell. The flow direction was calculated with the D∞ (infinite directions) method developed by Tarboton (1997), by which a dispersed or rilled flow is estimated for each cell from the slopes to the lower neighboring cells. The proportion of flow to pixel (i+1, j) is $\alpha_2 / (\alpha_1 + \alpha_2)$, and the proportion of flow to pixel (i+1, j+1) is $\alpha_1 / (\alpha_1 + \alpha_2)$. Flow direction is measured as the counter-clockwise angle from the East (Fig. 7).

Flow accumulation, i.e. the number of cells contributing with their flow to each particular cell, was calculated from the flow direction raster. The DEM sinks filling the slope angle, the flow direction, and the flow accumulation were calculated according to Tarboton (1997). For this project, an approach developed by Moore and Burch (1986a, b) was used to compute the LS-factor (Fig. 8).

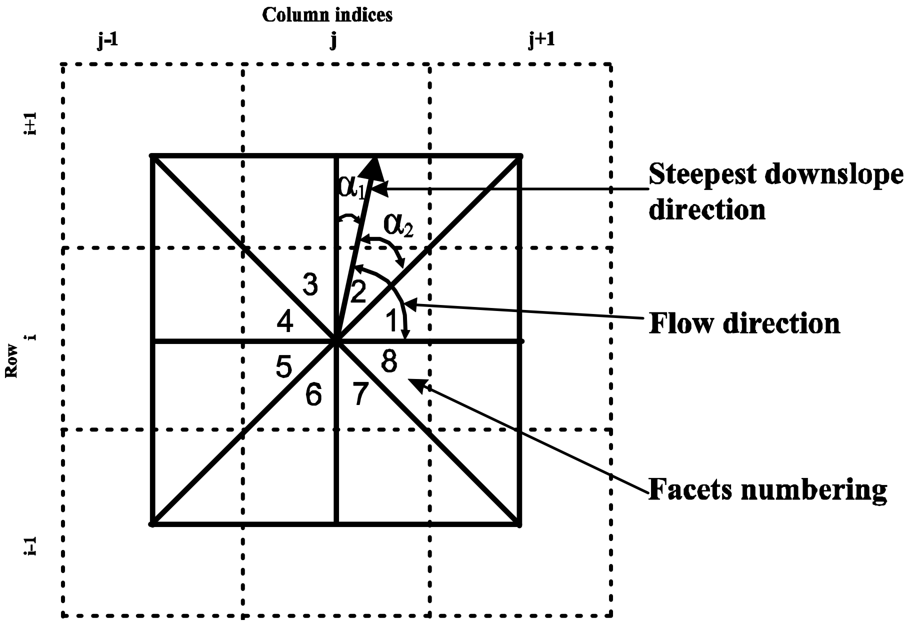

Figure 7. Flow direction as defined as steepest downwards slope on planar triangular facets on a block-centered grid (modified from Tarboton 1997).

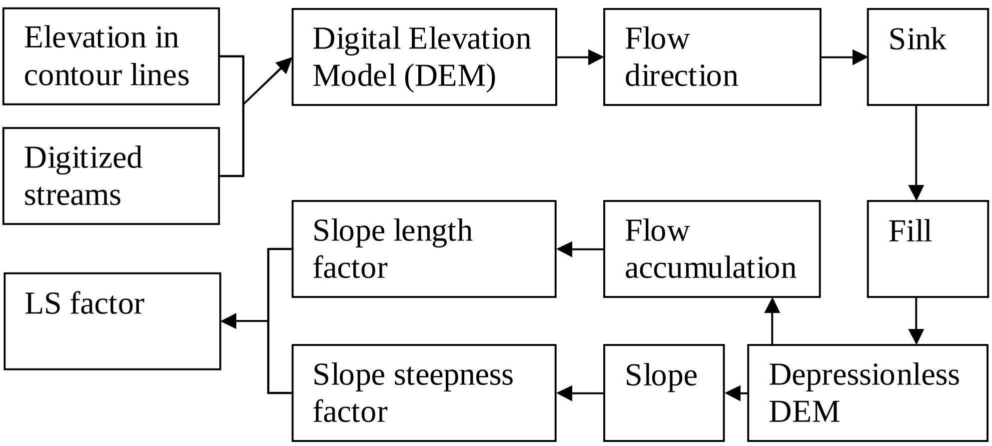

Figure 8. Flow chart for generating a raster map of the LS-factor based on elevation in contour lines and digitized streams.

Rainfall and runoff erosivity factor (*R*)

R is the long-term annual average of the product of event rainfall kinetic energy in MJ ha^{-1} and the maximum rainfall intensity in 30 minutes in mm per hour (Wischmeier and Smith 1978; Renard and Freidmund 1994). A regression equation adopted for application in the RUSLE model was used to calculate the R-factor based on the average annual precipitation (AAP; Renard et al. 1997).

For this mountainous watershed, the AAP and elevation data from 38 meteorological stations in the research area were obtained to check the correlation between precipitation and elevation. The AAP surface was interpolated with a multivariate geostatistic cokriging model. Cokriging is a multivariate geostatistical method that uses the spatial correlation between two or more variables to reduce the estimation variance when one of the variables is under-sampled (David 1997). The AAP surface was used to calculate the R-factor using the spatial analyst module of the ArcGIS software.

Soil erodibility factor (*K*)

The K-factor is a soil erodibility factor. It is a measure of the susceptibility of soil particles to detachment and transport by rainfall and runoff. Specifically, soil erodibility is a function of particle size distribution, organic matter content, structure and permeability (Renard et al. 1994; Renard et al. 1997). The erodibility value of each soil class can be calculated using an equation recommended by Wischmeier and Smith (1978). The raster map of the K-factor was produced according to a digital soil map and the erodibility value of each soil type.

Cover management factor (*C*)

The vegetation cover at three levels (for canopy cover, under-canopy cover, and surface cover) was recorded from the 625 sample plots of the study. The canopy cover was measured by a densiometer. The *k*-nearest neighbour (*k*-NN) method was used to produce the canopy cover map and the total vegetation cover map by integrating the Landsat ETM+ image information with the recorded vegetation coverage.

A set of parameters was chosen for the *k*-NN method in predicting the vegetation coverage map. The parameters used were the image features, the weight for each band, the distance, the number of nearest neighbours (the value of *k*), and the geographical reference area from which the nearest field plots were selected. A leave-one-out cross validation method was applied to calculate the root mean square errors (RMSE) and the average biases of predictions at the single pixel level for the different combinations of *k*-NN estimation. The RMSE and the biases were used as a measure of reliability of the continuous variables. The cover management factor (C) was then calculated from the vegetation coverage data using the equation recommended by Renard et al. (1997) and a regression equation built for the Upper Yangzte River Basin (Yang and Shi 1994).

Support practice factor (*P*)

P values range from 0 to 1, whereby the value 0 represents a very good man-made erosion resistance facility and the value 1 no man-made resistance erosion facility. In the study area there were some agricultural support practices, such as striped farmland and terraced farmland (Ma et al. 2003). However, most of the farmlands in the study area were small and consisted of self-managed lands. Since the spatial resolution of the ETM+ imageries was 30 m, it was impossible to distinguish the separate practices in the large-scale watershed from the available data. Therefore the value $P = 1$ was used for the whole area.

Soil loss intensity classification

Two kinds of intensity classification criteria were used to group the modelled soil losses into different groups. One was a visual interpretation and validation of the resulting erosion risk map for all sample clusters. The sites were given a subjective risk scale ranging from No risk – Low – Moderate – High – Extreme based general site characteristics. The other was a standard set provided by the Ministry of Water Resources, PRC for mountainous areas (MWR 1997). Soil loss intensities are typically divided into six classes, which also were used in the present study: negligible (< 500 t km^{-2} yr^{-1}), slight (500 – 2,500 t km^{-2} yr^{-1}), moderate (2,500 – 5,000 t km^{-2} yr^{-1}), severe (5,000 – 10,000 t km^{-2} yr^{-1}), very severe (10,000 – 15,000 t km^{-2} yr^{-1}), and extremely severe (> 15,000 t km^{-2} yr^{-1}).

2.2.2 Vegetation dynamics and human impact (II, III)

Vegetation class change

The vegetation class change was examined from the 1970s up to 2004. The vegetation classes in 2004 were recorded in the field inventory. There were four main different vegetation classes: forest, shrubland, agricultural areas, and grassland. In the forest class, four subclasses were recorded: closed forest, open forest, plantation, and other forests according to the criteria used by FAO (1996). The vegetation classes existing 30 years earlier were digitized from topographic maps from the 1970s. A matrix, with percentages of each class as well as losses and gains in different classes, was produced to illustrate the vegetation degradation and restoration trends.

Human impact on forest stands

According to the different levels of human impact, four classes of forests were observed and distinguished in the UMR watershed: (1) near-natural forests (forests that were characterized by low intervention and natural regeneration, without signs of management); (2) selectively logged forests (forests that had been selectively logged, with a low number of large trees, and usually with visible stumps); (3) natural regeneration forests after clear-cut (forests that had been clear-cut, leaving no large trees); and (4) plantations (planted trees in rows or arrays). Forests under different levels of human impact were analysed with the following quantitative characteristics: stand volume, basal area, weighted diameter, weighted height, biodiversity indices for tree species, and evenness of abundance.

The tree species diversity was described using the Shannon, Simpson, McIntosh and Berger-Parker's indices, and Evenness, Alpha, and Q-stat criteria (Peet 1974; Kempton and Taylor 1976; Magurran 1988; Appendix 6). Multiple range tests with non-normal distribution assumed were carried out using Statgraphic Plus 4.0 software in order to compare the calculated stand characteristics under different levels of human impact along altitudinal gradients. Linear Discriminant Analysis (LDA) was performed to calculate the coefficient of the linear combination of variables in different human impact classes.

2.2.3 Identification of areas for restoration (I, II, IV)

Areas with high erosion risk

The whole watershed was classified into areas with different erosion risks based on quantitative evaluation of soil loss. The areas with high, extremely high, severe, very severe, or extremely severe erosion intensity were reclassified as high-risk areas. Since there were five different vegetation zones and three different soil orders in the research area, the high-risk areas were further identified for each vegetation zone and each soil order. The high-risk areas were also statistically summed up for each vegetation type at the different elevation ranges by zonal statistics.

Identification of dry areas

Precipitation data were collected from 40 meteorological stations in and around the research area; the observation period was 5 years, from 1998 to 2002. The average annual precipitation values for different locations were used to generate a precipitation surface by Inverse Distance Weighted (IDW) in ArcGIS. IDW assumes that each input point has a local influence diminishing with distance, which weighs the points situated closer to the processing cell greater than those farther away. The correlation between the average annual precipitation (AAP) and the elevation data from 38 meteorological stations was calculated (the 2 exluded stations were outside the present watershed), and a regression equation was constructed.

The precipitation surface interpolated by IDW and the equation were used to produce a digital map of dry areas by including the areas with a predicted annual precipitation of less than 600 mm and excluding the areas above a threshold elevation. The threshold elevation was calculated by applying an upper limit of the estimated precipitation of 600 mm in the regression function for AAP vs. elevation (Fig. 9).

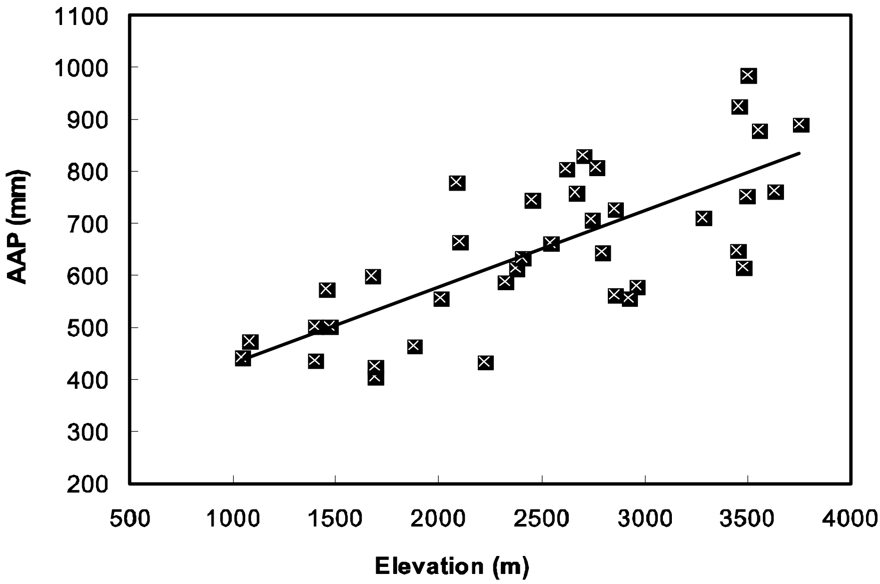

Figure 9. The average annual precipitation (AAP) as related to elevation. Equation: Y=286 + 0.146X; correlation: r = 0.74.

Potential area for rapid restoration

Using ArcGIS, all secondary degraded shrub lands earlier covered by forest were identified. The number of seeds occurring at a certain location decreases with an increasing distance to the seed sources. Because the slope of the corresponding regression line is not certain in mixed forests, five concentric buffers were delineated outside the remaining forests to the distances of 1 km, 2 km, 3 km, 4 km and 5 km, thus partly also dissolving the barriers between different patches. Priority areas for rapid restoration were determined as the overlap areas of degraded shrubland and concentric buffers of the existing forests.

2.2.4 Selection of woody species for restoration (II)

Logistic regression was used to predict the frequency and percentage of tree species at different elevations and on different soils. In logistic regression, the independent variable is dichotomous (Hosmer and Lemeshow 2000); in the present case, the species was either present or absent at a certain elevation range and on a certain soil type. Climate and soil are the most important factors for predicting the existence of certain vegetation (Kojima 1981). The microclimate in the present mountainous watershed was mainly influenced by the elevation. Therefore elevation was used as a surrogate for climate. The model assumes that the value of the unobserved continuous variable for the *i*th case is linearly related to the predictors.

The elevation data were divided into 5 classes based on the ecozones applied earlier to the UMR watershed (Editorial Board of Sichuan Vegetation 1980; Appendix 8). The soil data were divided into three orders (Guo and Ou 1991; Shi et al. 2005; Appendix 9). This Chinese soil classification system was used in the present study, because it was not possible to directly convert it to either of the two internationally recognized classification systems, the FAO system or the US Soil Taxonomy. However, the soil orders were cross-referenced to the US Soil Taxonomy (FAO/UNESCO/ISRIC 1990; USDA 2000).

2.2.5 Effect of vegetation cover on soil erosion (IV)

The cover management factor C is the ratio which compares the soil loss from under vegetation cover with that from bare soil. It is therefore a ratio with a value ranging from zero, when the soil is completely protected, to the value of one for bare soil. Through a series of computations using map algebra, maps were generated using ArcGIS software to show what effects could result from different cover management factors. First of all, the soil loss intensities under the current vegetation cover were compared to those under the scenarios of either no vegetation protection (C = 1) or good vegetation protection (C = 0.001). Five different vegetation cover restoration scenarios, with a restored vegetation cover of less than 40%, 50%, 60%, 70%, or 78.3% to each respective value, were simulated in order to calculate how different vegetation cover percentages affected the areas at high risk for soil loss. Zonal statistics were used to calculate the proportions of different soil erosion intensities under each scenario.

2.3 Evaluation of predictions using empirical data

A visual interpretation and validation of the resulting erosion risk map was performed for all field sample clusters. The sites were given a soil loss intensity scaled on the basis of the general site characteristics. A built-in cross-validation was applied for the vegetation cover estimation.

The elevation was used to aid a geostatistical analysis for generating predictions for AAP. Root mean square errors (RMSE) and standardized mean square errors (SMSE) were calculated to investigate the estimation accuracy.

The measured slope and elevation were used to evaluate the DEM generated by different methods, such as IDW, spline, kriging, topo to grid, and topo to grid with stream data. The best interpolation method, which was topo to grid together with the stream data, was chosen to generate the DEM for further calculations. Drainage networks modelled using flow accumulation data was compared with the real drainage network, in order to evaluate the methods.

3 RESULTS

3.1 Geo-environment in the research area

The UMR watershed was found to be a mountainous watershed with steep slopes and complex relief. The elevation ranges from 600 m to more than 6,000 m. The present research area in the UMR watershed occupied an area of 7,432 km^2, where the elevations ranged from 1,371 m to 5,527 m, and around half of the research area consisted of high mountains with elevations more than 3500 m (Appendix 10). Slope angles ranged from 0 to 77 degrees. Steep slopes with over 30 degrees of inclination occupied 53% of the watershed (Appendix 11). Slopes quite evenly faced to different directions, while the flat area only occupied 0.1% of the watershed (Appendix 12).

3.2 Soil erosion (*I, IV*)

3.2.1 The R-factor

The precipitation data and station elevations from 38 meteorological stations were used for estimating the average annual precipitation (AAP) over the entire watershed. The AAP showed a strong temporal and spatial bias. More than 50% of the precipitation was obtained in July, August and September. The AAP showed a significant correlation (p < 0.01, r = 0.74) with the station elevation (Fig. 9). A multivariate cokriging interpolation method was used in the analysis, since it takes into consideration the elevation that significantly affects the precipitation. The RMSE vulues were calculated so as to also investigate the estimation accuracy. The RMSE obtained by cokriging estimation was 86.88 mm, which indicated a reduction by 28.2% as compared to kriging estimation (121.2 mm). The estimated AAP was used for calculation of the rainfall and the runoff erosivity R-factor in ArcGIS. The R-factor varied from 1288 to 3342 MJ mm $ha^{-1}h^{-1}$ yr^{-1}.

3.2.2 The C-factor

A pixel-to-pixel canopy cover map and a total vegetation cover map were produced using the *k*-NN method. The tabulated data for *k*-NN estimations contained the following variables: the x and y coordinate locations of plots, their corresponding satellite image spectral values from six bands, and the canopy cover and the total vegetation cover. The RMSE and the average biases of predictions at the single pixel level were evaluated for each combination of parameters. The value of *k* (8), the distance (55 km), the bands (1, 2, 3, 4, 5, 7) and their optimal weights were chosen for a situation where the RMSE and the bias were minimal.

Based on the produced canopy cover map and the total vegetation map, the C-factor was further calculated using two methods. One was the equation developed by Renard et al. (1997), and the other was the regression equation developed from a nearby watershed in the Upper Yangtze River Basin by Yang and Shi (1994). The C-factor ranged from 0.015 to 0.892, and from 0 to 0.294, respectively (Fig. 10).

3.2.3 The LS-factor

The study area occupied a raster grid space of 7,700 rows by 2,736 columns, and elevations within it ranged from 1,260 m to 5,537 m. Approximately 94.7 percent of the watershed had slopes steeper than 9 percent. Slope angles ranged from 0 to 77.2 degrees, with a mean of 25.9 degrees and standard deviation of 12.0 degrees. As a result of applying Taudem, an ArcGIS extension developed

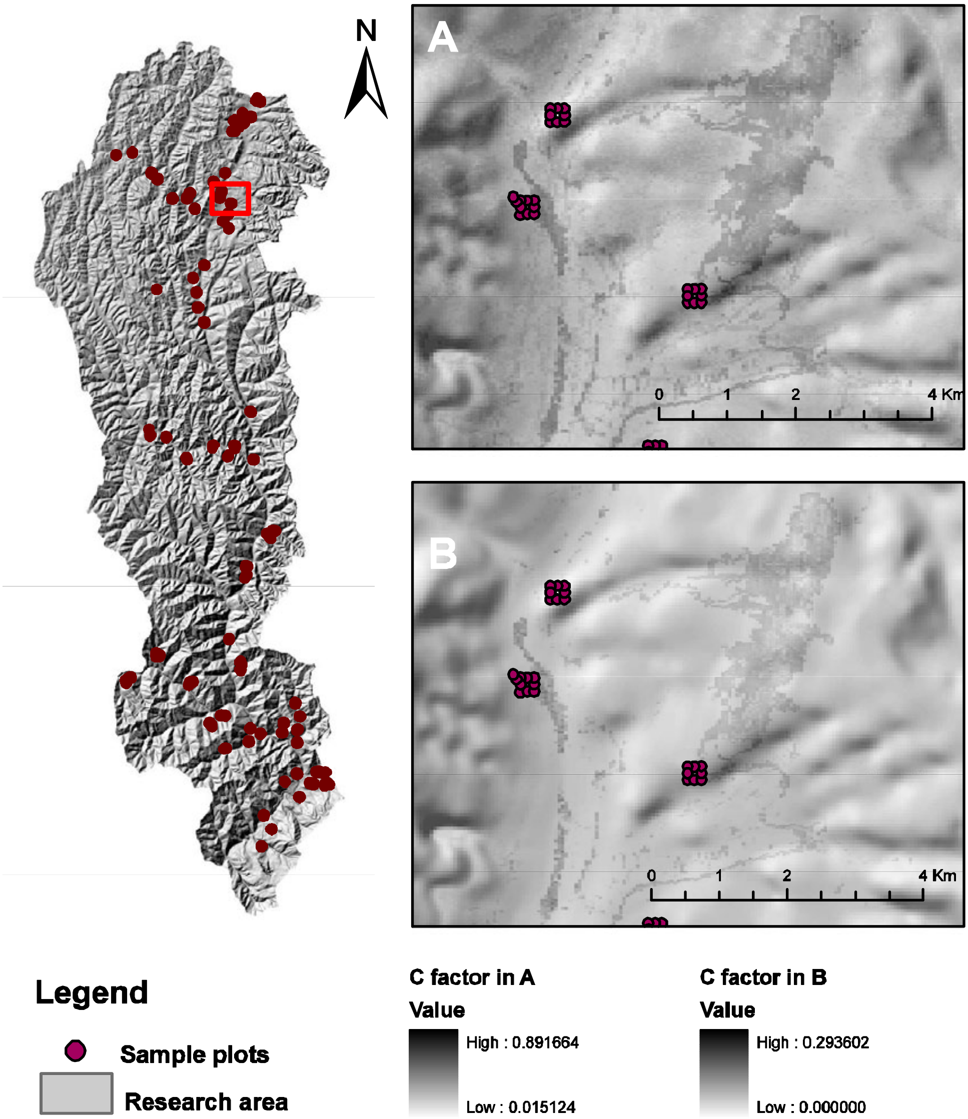

Figure 10. The C-factor as generated using two methods. A: based on equation developed by Renard et al. (1997); B: based on the regression equation by Yang and Shi (1994).

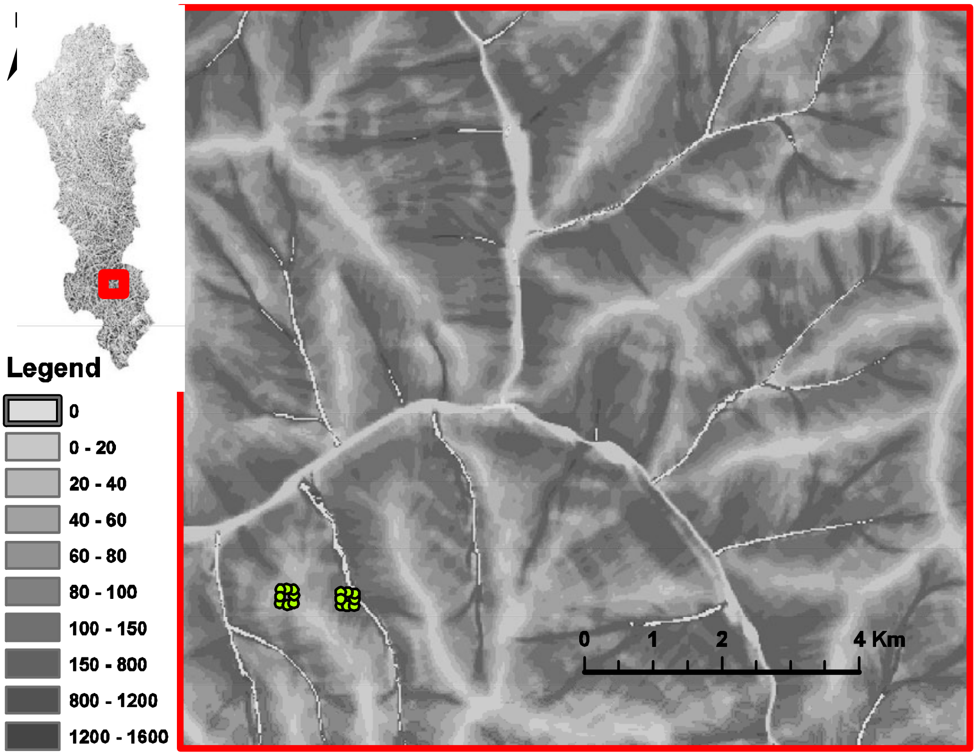

Figure 11. The predicted value of the LS-factor in a representative area of the experimental watershed.

by Tarboton (1997), the flow accumulation ranged from 1 to 15,496,180 m. The areas with flow accumulation value over 1,200 m were coincident with the main stream paths. The slope length factor ranged from 0 to 3,398, with a mean of 85.9, and 99.2% accounting to less than 304.7 m; this value is set in the RUSLE as a maximum critical slope length. The slope steepness factor S varied between 0.03 and 15.88, with a mean of 6.7 and a standard deviation of 3.1. The LS factor varied from 0 to 1600 (Fig. 11).

3.2.4 The K-factor

The soil erodibility factor (K-factor) is a quantitative description of the inherent erodibility of a particular soil type. The K-factor reflects the fact that different soils erode at different rates when the other factors that affect erosion remain the same. Soil texture is the principal cause affecting the K-factor, but the soil structure, organic matter content, and permeability also contribute. A map for the K-factor was produced based on the soil map and the erodibility value for each soil type. The K-factor in the present study area varied between 0.036 and 0.043 Mg h $MJ^{-1}mm^{-1}$.

3.2.5 Soil loss potential

The estimated soil loss from the present study area varied from 325 to 83,240 Mg ha^{-1} per year. According to the soil loss amount and field inventory results in general, the soils were divided into four ordinal classes representing the following situations: extreme risk (>10,000), high risk (3,000-10,000), moderate risk (1,000-3,000), low risk (<1,000) and "no data". The "no data" values were derived from two circumstances: firstly, they comprised the data with flow accumulation values higher than 1,200, which were coincident with the main stream paths; and secondly, they comprised the data with LS factor values higher than 1,600; this only happened in isolated cells with extremely high slopes and flow accumulating areas. Totally, 0.7% of the cells had a "no data" value, and 17.5% of the watershed showed high or extremely high erosion risk (cf. I, Table 2).

According to the classification criteria for water erosion intensity (MWR 1997), the predicted soil loss was classified into six erosion intensity categories: negligible, slight, moderate, severe, very severe, and extreme severe erosion intensity. Under the prevailing conditions, about 75% of the land in the watershed studied was classified as stable, 10% was at the level of slight or moderate erosion, while 15% showed severe, very severe, or extremely severe erosion loss (cf. IV, Table 3).

3.3 Areas for restoration (*I, II, IV*)

3.3.1 High erosion risk areas

In the present investigation, the soil erosion modelling in the experimental watershed was studied in Part I and Part IV. The soil erosion intensity was grouped into different classes using two separate methods. Both methods predicted around 15% of the total area having a high erosion risk. Figure 12 shows the areas with high erosion risk identified for urgent restoration on different soil types and at different elevations. These high-risk areas, which were found mainly along the main stream and in the foothills, had a distance of up to 2,300 m to the main stream of the UMR and were generally covered by sparse vegetation. Buffers or riparian corridors along the main stream appeared to play an important role in reducing the soil erosion.

Vegetation types showed distinct different proportions of high-risk areas at different elevation ranges (Table 1). The grassland at lower elevations showed the highest proportion of high-risk erosion areas. Most of the land dominated by grass was degraded as a result from deforestation and low precipitation. Agricultural land at lower elevation also showed a high proportion of high-risk erosion sites. This obviously was due to agricultural activities on steep slopes. Shrubs seemed to prevent the soil erosion quite well at low elevation. In contrast, at high elevations, shrublands seemed to be more sensitive to erosion.

The coniferous forest showed a relatively steady proportion of high erosion risk areas at different elevation ranges. The broadleaved forest was found to be sensitive to erosion at high elevations.

3.3.2 Dry areas

The research area was rasterized according to precipitation, then classified with dry areas determined as those having an annual precipitation of less than 600 mm. Low-precipitation areas were mainly found in Mianchi, from Wenchuan to Zhengjiangguan, and in Maoxian along the main course of the UMR.

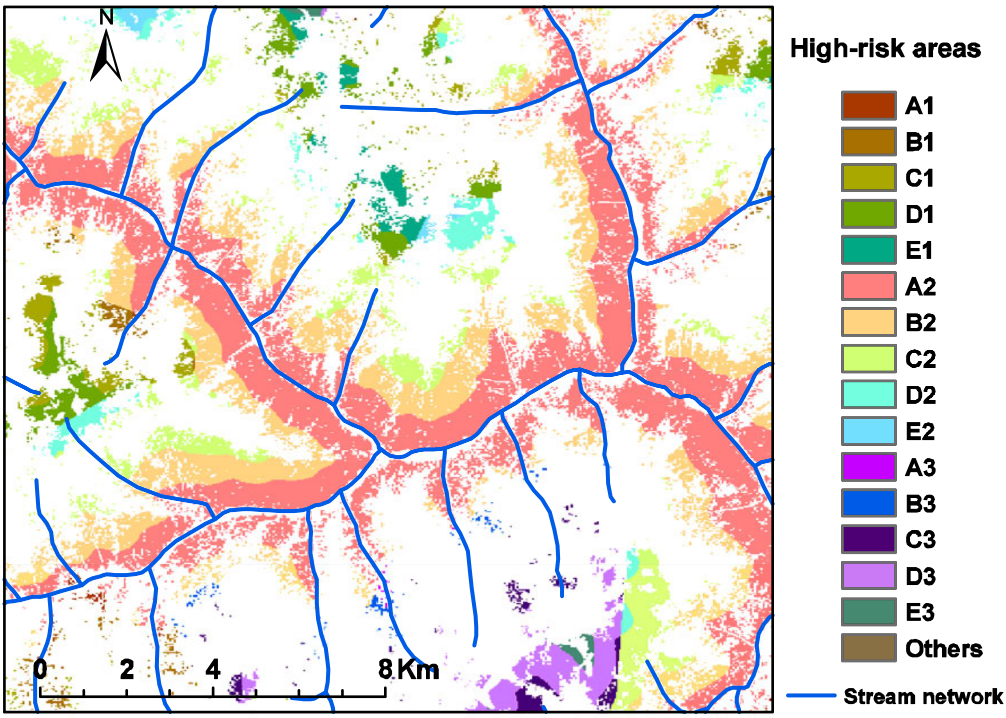

Figure 12. Example of high-risk erosion areas as identified for five different vegetation zones and three different soil types. Classes A1-E3 are combinations of vegetation zones and soil types.

Table 1. The proportion of high erosion risk areas, in different vegetation types at different elevation ranges.

Elevation range	Vegetation type	High-risk area (ha)	Total area (ha)	Proportion %
<2200 m	Conifer	11	99	0.12
	Mixed forest	4	28	0.13
	Broadleaved forest	2	28	0.06
	Shrub	20	437	0.05
	Grassland	322	711	0.45
	Agriculture	179	792	0.23
2200- 2600 m	Conifer	56	494	0.11
	Mixed forest	19	177	0.11
	Broadleaved forest	13	118	0.11
	Shrub	73	1084	0.07
	Grassland	186	590	0.31
	Agriculture	38	213	0.18
2600- 3200 m	Conifer	340	2512	0.14
	Mixed forest	59	569	0.10
	Broadleaved forest	82	585	0.14
	Shrub	430	3047	0.14
	Grassland	500	1917	0.26
	Agriculture	84	375	0.22
3200- 3600 m	Conifer	273	2823	0.10
	Mixed forest	19	442	0.04
	Broadleaved forest	71	303	0.23
	Shrub	468	3227	0.14
	Grassland	292	6385	0.05
	Agriculture	6	61	0.10

Accordingly, the total area of dryland (P<600 mm) in the whole UMR watershed was 1,670 km^2 (7%). Of this, 858 km^2 was within the present research area, corresponding to 12%. The largest dryland area was in Maoxian, covering 33% of it. Most of the drylands were degraded, and the main vegetation type found on them was sparse shrub or grass vegetation. Drylands with high erosion risk were assumed to be among the most difficult sites for vegetation restoration or land rehabilitation in the present study area. It was also concluded that forest restoration in this area was also limited by the absence of seed sources.

3.3.3 Potential areas for rapid restoration

Potential areas for rapid restoration were found to be the former forests that were now covered by shrubs but adjacent to existing forested land. The degraded lands dominated by shrubs were identified and buffered according to distances to forested land. It was also concluded that the shorter the distance was to a forest, the more seeds were dispersed to the area and, consequently, the easier would the restoration process be.

Of the potential areas for rapid restoration, the areas within 1 km, 2 km, 3 km, 4 km, and 5 km of existing forest covered 853 km^2, 191 km^2, 48 km^2, 22 km^2 and 10 km^2, respectively. The total area within 2 km of forest was 1,044 km^2 and that within 5 km of forest 80 km^2; these figures corresponded to 14% and 1% of the present research area, respectively.

3.4 Ecological restoration (*II, III, IV*)

3.4.1 Vegetation restoration scenarios for high erosion risk areas

The present study identified the total area with high erosion risk, which combined the severe, very severe and extremely severe soil loss classes. It occupied an area of 1,083 km^2 in the watershed. The high-risk areas had a significantly lower vegetation cover as compared to the areas with lower erosion risk. Under the present vegetation cover, around 15% of the area showed a high erosion loss (5% with severe, 3% with very severe, and 7% with extremely severe soil loss).

If all the land studied had been without any vegetation protection, then around 98% of the watershed would have suffered from extremely severe erosion loss. In such a case, mudslides and landslides could easily have occurred after a rainstorm. If all the land had been covered by dense vegetation, then a sharp decrease in the land area with severe or higher erosion loss would have occurred. In that case, only 0.4% of the watershed would have exhibited a severe or higher erosion loss. These scenarios suggested that an intact vegetation cover could efficiently protect the soil against erosion loss. A good plant cover is capable of preventing excessive soil erosion and reducing landslides as well. Consequently, removing vegetation can greatly increase the soil erosion rate, particularly in this kind of mountainous area.

The different vegetation restoration scenarios analysed in the present study showed how soil erosion could be reduced by increasing the vegetation cover. If the high-risk areas with a vegetation cover of less than 40% could be restored to a 40% cover, only a slight reduction in the soil erosion rate would occur.

Similar results were obtained when the vegetation cover was assumed to be 50% or 60%. Restoring the vegetation cover to 70% would have significantly reduced the extremely severe erosion loss. Restoring the vegetation cover to at least 78% on all the land in the study area have yielded a C-factor of at least 0.001, resulting in a sharp reduction in a high erosion risk and only a negligible soil loss on around 41% of the area, a slight soil loss on 56%, and a high erosion risk on only 0.1% (around 8 km^2) of the entire study area. This result suggests that in mountainous catchments such as the UMR watershed, a high soil erosion rate tends to occur when more than 30% of the soil is exposed and a vegetation cover of more than 78% can effectively prevent the excessive soil erosion caused by water.

3.4.2 Ecological restoration perspectives with different tree species

In this study, potential tree species with different predicted existence values were identified for the upper and middle reaches of the UMR watershed (cf. I; Appendix 5). They can be used for forest restoration on appropriate sites on degraded land, over a gradient of five ecological zones, ranging from 1,300 m to 5,700 m in altitude, and on soils representing three different soil orders, especially for restoration of degraded areas with high erosion risks.

The buffer areas closer to existing forests had a higher probability to receive seeds dispersed by wind or animals. Such areas could effectively contribute to the maintenance of natural tree populations. Areas covered by degraded shrubland were identified separately within 1 km, 2 km, 3 km, 4 km, and 5 km of the existing forests. Of these buffer zones, the fifth one obviously received the fewest seeds. However, if 64% of the degraded areas found within the 2-km buffer zone could have been successfully restored, the total forest cover would have increased by 9 percentage points, and the forest cover could have reached the earlier baseline level of 30%. Obviously, the local people who live inside or around these areas should be informed of how to maintain the near-by forests and how to become actively involved in the restoration of the 2-km zones of the degraded buffers.

On degraded drylands, several steps would be necessary, according to the present results. First of all, drought-resistant and nitrogen-fixing shrubs should have a priority in regeneration interventions. Local shrub species such as *Periploca sepium*, *Bauhinia brahycarpa* and *Caragana soogorica*, or local trees such as *Hippophae rhamnoides*, are known to establish themselves and grow well on degraded sites. Secondly, pioneer species such as *Salix sclerophylla, Salix heterochroma, Betula pendula, Populus davidiana*, and the exotic *Robinia pseudoacacia*, could be introduced. Finally, primary forest species such as *Pinus armandii* and *Cupressus chengiana* should be introduced as soon as the environment has been sufficiently modified by the pioneer and early successional species. In addition, measures that improve the soil moisture conditions, such as terracing, could be undertaken.

Especially if a finer scale for high erosion risk assessment could be applied, tree species could be selected for each specific site. First, the elevation range and soil type should be checked according to the location at the smaller scale; then tree species could be selected according to the elevation range and soil type, and the appropriate species could be introduced according to their adaptation to successional stages. The existing forests nearest to the degraded sites and the distance between those sites and the nearest village should be identified on a map; subsequently, the local people who live nearby could be encouraged, perhaps through government subsidies and incentives, to maintain the remaining forests and select suitable tree species for restoring the degraded sites.

In addition to restoring the degraded open lands that have resulted from deforestation and heavy soil erosion, efforts are also needed to restore the degraded forest lands. The existing forests have been affected by various man-made disturbances. It was found that the near- natural forests, selectively logged forests, and forests naturally regenerated after clear-cut varied in stand volume, biodiversity, and the degree of fragmentation (cf. III). They also varied in their capacity to become restored unaided by man, even assuming that further disturbances could be prevented. Forests tended to be more resilient at higher elevations, where less human disturbance had occurred. Special attention should be given to the degraded forests at lower elevations. Natural regeneration and tree planting could be combined in an efficient way, to enhance the speed and efficiency of restoration.

4 DISCUSSION

4.1 Large-scale soil erosion modelling

At the point or plot level, the soil erosion can be qualitatively assessed by certain criteria (FAO 2006) or directly measured by field devices (FAO 1993). At a landscape level, the soil erosion can be evaluated quantitatively by integrating the spatial data on erosion factors for each pixel. The extent of the study region, the spatial resolution (pixel size), and a framework for estimating and integrating the factors should be defined prior to large-scale soil erosion modelling. Spatial information sciences and techniques such as RS, GIS, GPS, and field surveying can be applied to estimate and integrate the factors affecting soil erosion.

The soil loss in the present study was predicted using the conceptual model of RUSLE (Renard et al. 1997). A mountainous topography and the great variation in precipitation and vegetation required a modification of the standard RUSLE factors and their derivation. The data used to generate the soil loss at a landscape scale consisted of ETM+ images, a DEM, a soil map, tabular precipitation data, and the field data.

It was found that slope calculations made with a maximum downhill method conserved the variability and the maximum slope values. This method produced no underestimation, since no averaging was used. Flow directions calculated by D∞ (infinite directions) improved significantly the water flow modelling, by allowing a dispersed flow to be modelled over the surface. This method calculated the flow direction from the lowest continuous neighbouring cells and fractionated the water flow between them, thus simulating a dispersed water flow and generating natural-looking flow maps. Other studies have mostly used the D8 approach method developed by O'Callaghan and Mark (1984). However, the D8 method produces an unrealistically rilled water flow, with an excess of straight lines in flow accumulation maps, because it can only produce eight different flow directions, to one of the neigbouring cells (in cardinal or diagonal direction).

It is difficult to assess the prior land use, canopy cover, surface cover, surface roughness, and the soil moisture, for a direct application of the C-factor from the RUSLE (Renard et al. 1997). The present result showed an overestimation, when the C-factor was computed from subfactors CC and SC, while PLU, SR and SM were set to 1. When a regression equation exploring the relationship between the vegetation cover and the soil loss in the similar mountainous watershed was used (Yang and Shi 1994), a better result was achieved, as compared to the field observations.

In the present case, the *k*-NN technique was used for the C-factor estimation. A calibration of the *k*-NN parameters was performed, as outlined by several articles dealing with forest estimation using *k*-NN methods (Tokola et al. 1996; Tokola et al. 2001; McRorberts et al. 2002). The RMSE and the significance of biases at sample plot pixel level were evaluated in order to choose the most optimal parameters, such as the numbers of *k*, the distance, and the band weights. The built-in cross-validation method of bias and error estimation was applied in all present calculations.

The soil loss usually shows wide variation, depending on the terrain, rainfall, vegetation and soils (Angima et al. 2003; Shi et al. 2004; Lu et al. 2004). The maximum soil loss that can occur and that still can permit the crop productivity to be sustained economically was found to be 2.47-12.36 Mg ha^{-1} yr^{-1} (cf. Renard et al. 1997). Morgan (1995) has argued that 10 Mg ha^{-1} yr^{-1} is an appropriate boundary value of soil loss over which agriculturists should be concerned. This value was used by Millward and Mersey (1999) to separate low and moderate erosion categories as predicted by RUSLE. Van Remortel (2001) has argued that erosion model can be used to derive patterns of erosion, but not necessarily the actual loss from erosion, because of the limitations of the methods used to derive some component factor values. Millward and Mersey (1999) found that the relative comparisons of soil loss among land areas were more critical than assessing the absolute soil loss in a particular cell.

A quantitatively modelled soil loss is usually grouped into several classes to indicate soil erosion intensity. In this study, two kinds of soil loss classification criteria were used. The one approach was for distinguishing five erosion intensity classes: no risk, and low, moderate, high or extreme risk, based on ground checking, and the other one was six erosion intensity categories: negligible, slight, moderate, severe, very severe, and extremely severe erosion, according to the classification criteria of water erosion intensity (MWR 1997). There is no one international standard to classify a modelled soil loss, yet.

A visual interpretation and validation of the resulting erosion risk map was performed for all sample clusters. A high or extremely high erosion risk mostly occurred on downhill gullies with long continuous slope lengths on either side of the main stream. Considerations should obviously be given to areas with extremely high erosion risk where landslides or mudslides could easily occur, according to the soil loss potential (Xia and Guo 1997).

The RUSLE is a factor-based model. An error in a factor value will produce an equivalent percentage error in the soil loss estimation (Wischmeier and Smith 1978; Renard et al. 1997). These errors are mainly due to inaccuracy components in each data layer and the limitations in the methods used to derive the component factor values (Millward and Mersey 1999). The accuracy of the predicted soil loss can be improved, if each factor layer is better estimated. For example, an R-factor surface can be better produced by using the multivariate cokriging method than the ordinary kriging. The LS-factor can be improved by a better generated DEM, maximum downhill slope and infinite flow direction. The C-factor can be improved by better estimation of the fractional vegetation cover. To assess the accuracy of the produced maps, validation with independent data is required. This can be obtained from field measurements, surveys, or high-resolution imagery (Vrieling 2007).

4.2 Effect of vegetation cover on soil erosion

The importance of each individual factor affecting the soil loss is not always the same; instead, it depends on regional characteristics, the specific erosion process under consideration, and the spatial and temporal scales studied (Vrieling 2007). In the present study, at the landscape level, the vegetation cover factor showed a significant positive relationship with the modelled soil loss. The presence of a vegetation cover can increase water infiltration and reduce the surface runoff, thus retarding the sheet erosion significantly (Woo and Luk 1990). With a reduced vegetation cover, the runoff and soil erosion can greatly increase, resulting in flooding and mudslides (Varis and Vakkilainen 2001; Sidle et al. 2004).

In semi-arid Tanzania, the overall most important variable affecting soil erosion has been found to be the vegetation cover density (Cristiansson 1981). It influences both the degree of soil protection and the soil moisture status, and thus also the runoff and soil loss. This tendency was also demonstrated in Kenya by Dunne et al. (1978). Lu et al. (2004) explored the relationships between soil erosion and land use or land cover distribution; they found that most climax and mature forests are found in low erosion risk areas. The present scenarios of the effect of vegetation cover on soil erosion showed that almost all the areas studied would be subjected to extremely serious soil loss by erosion without protection by vegetation under the prevailing geographical landform.

The soil loss by erosion can be greatly reduced by a denser vegetation cover. Therefore, a large-scale restoration of the vegetation cover would be a good way to improve the soil stability and to reduce the soil loss. This could be done in the present study area by means of selection of suitable woody plant species for different soils at different elevations, taking advantage of the existing forests (II). The soil loss hazards would be effectively alleviated, particularly if combined with conservation measures such as terraces, contour tillage, or contour hedgerows. This has already been suggested for a small watershed also in the Upper Yangtze River Basin by Shi et al. (2004).

4.3 Ecological restoration for soil erosion alleviation

The use of GIS, RS, GPS, and RUSLE enables quantitative spatial modelling of the soil erosion caused by water, whereby areas of high erosion risk can be identified for the implementation of soil conservation measures. As discussed above, the present scenarios of the effect of the vegetation cover on soil erosion also showed that the restoration of vegetation cover in high-risk areas could be an effective way to reduce the soil loss in the UMR watershed.

Various soil and water conservation programmes have been recently implemented in the Upper Yangtze River Basin. Even though none of them has explicitly focused on high-risks areas, they have to some extent already contributed to vegetation restoration in priority areas. The past programmes were parts of the conservation of forests in the upper and middle reaches of the Yangtze River in 1989 and included a logging ban in natural forests in 1998, and the “Grain for Green” program in 2000 (Ye et al. 2003). The first two actions prevented further man-made disturbances. A subsequent increase in the forest area demonstrated that the existing degraded forests have an inherent capacity for recovery. The “Grain for Green” programme was initiated to convert cultivated lands on slopes of 25° or more back to forest land or grassland, which could then reduce the soil loss on the steepest slopes.

In the present study, around 9% of the high-risk area was agricultural land, and the existing programmes were well addressing the restoration needs on this land. In comparison, about 26% of the high-risk area consisted of forest land. Most of the forests in the UMR watershed were subjected to heavy human impact. When the four levels of human impact on forests were separately studied, different patterns of tree species biodiversity were found in them along an altitudinal gradient. Near-natural forests with the lowest level of human disturbance were remaining only in remote areas above 2600 m elevation. Conventional forest management, mainly consisting of clear-cutting, affected the wood production in these forests negatively. Near-natural forests were partly defined and clearly distinguishable by their stand volume, in comparison to the other managed forests in the UMR watershed. A conclusion was that the near-natural forests could be used as baseline references for woody plant growth and biodiversity development, when new strategies for forest restoration and management are to be developed in the UMR watershed. This would follow the practice also used in boreal Fennoscandia (Kuuluvainen 2002).

As a whole, however, only a smaller proportion (35%) of the high-risk areas was found on forest lands or agricultural land, and a higher one (65%) was found on deforested land covered by sparse grass or shrubs. It is noteworthy that no programmes so far have addressed these deforested lands for controlling soil erosion. As a consequence of deforestation, the soil erosion on these lands has already led to a loss of topsoil, reduction in soil fertility, and a serious lack of native woody plants in the form of residual individual trees, natural seedling banks or soil seed banks. Obviously, it is almost impossible to restore the woody vegetation in these areas by the ecosystem resilience alone, especially in places where the systems have reached a new steady-state condition with the sites becoming occupied by sparse shrubs or grasses.

An ecosystem development threshold is commonly crossed when sites become occupied by grasses (Lamb et al. 2005). An ecosystem can be driven beyond the thresholds of resilience with an irreversible trajectory through time (du Toit et al. 2004). Large-scale reforestation by tree planting is one way to reduce the soil erosion and to restore an ecosystem in deforested high erosion risk areas that have crossed the ecological threshold, because those lands can not anymore be resilient by natural processes. The present study indicated that the potential vegetation on each soil order at different elevations can provide reference information for matching a tree species with a specific site (cf. II). Similar strategies have been used elsewhere for vegetation restoration on severely degraded areas where the original vegetation is partially or completely absent, e.g., for degraded land in Indonesia or degraded dry zones (Palmberg 1986; Sayer et al. 2004).

Previous tree planting activities in the UMR watershed have indicated efficient woody vegetation restoration on degraded shrublands. They have also showed a positive role in soil erosion control and carbon sequestration. However, these plantations have mainly consisted of single tree species. This kind of monoculture for restoration purposes has inevitably some drawbacks on biodiversity. Obviously, it is difficult to restore a functional or self-sufficient ecosystem by only using very few native species, even though they might significantly contribute to erosion control. In this study, the existence percentage values of different tree species were predicted for five ecological zones ranging from about 1300 to 5700 m in altitude and for soils representing three different soil orders based on the occurrence of species. A diverse range of native species including rare or endangered species was suggested for restoration purposes.

The ultimate way to select different species for ecosystem restoration remains debated. Restoration ecologists have tried to select species representing the following categories: (1) most sensitive species; (2) indicator species; (3) representative species; (4) umbrella species; (5) focal species; and (6) framework species (Perrow and Davy 2002; Elliott et al. 2003; Roberge and Angelstam 2004). Such approaches have been tested to be successful at least at some sites. At the landscape level, however, selecting only some of the species cannot ensure restoration of all the other observed species, which will inevitably lead to reduction of the biodiverstiy, especially when many species are limited by their specific ecological requirememts. SER (2004), however, has suggested that indigenous species should be used to the greatest practicable extent. The forest cover and the biodiversity will both increase, if the secondary forest is protected and if its connectivity is enhanced by reforestation using a diverse range of native species (Lamb et al. 2005).

Many researchers recommend the use tree material derived locally for restoration (Lesica and Allendorf 1999; Hamilton 2001; Wilkinson 2001; Jones 2003; Krauss and Koch 2004), because the local tree populations are better adapted to the prevailing environment, provide a better habitat for the fauna, maintain the genetic integrity of the site, and prevent any potential pollution of the local gene pool (Harris et al. 2006). Plantations of exotic tree species may develop a similar forest structure as the native species. However, the exotic species may cause more persistent population fragmentation in the rehabilitation process (Knight et al. 2001). For restoring highly disturbed sites where the local tree species and populations may not be well adapted, planting material from other locations may provide a better outcome (Lesica and Allendorf 1999). In such cases, some exotic species that have the least negative effects on ecosystem restoration can be considered. A leading basic principle for selecting woody plant species in assisted restoration would be to match a species with the site, and to use the local species as far as possible.

The adaptation ranges of tree species or populations might change under changing climatic conditions. The approach of a "bioclimatic envelope" is being extensively used to assess potential species range shifts in response to climate change (Bakkenes et al. 2002; Berry et al. 2002; Skov and Svenning 2004; Thuiller 2004; Araújo et al. 2005; Hijmans and Graham 2006).

Soil erosion and deforestation are the two main causes of land degradation in the UMR watershed. Other natural causes are landslides, mudslides, and earthquakes. The earlier land use, including clear-cutting of trees without appropriate regeneration measures, had already lead to a forest decline where large forest areas had developed a secondary vegetation cover mainly consisting of sparse shrubs and grass. Land had become more vulnerable to soil erosion, especially the steep slopes having the lowest vegetation cover. When deforestation with excessive soil erosion occurred on dryland, it was almost impossible to restore the previous forests by natural regeneration. These problems thus need further study so as to develop new suitable tools for forest ecosystem restoration in the UMR watershed.

On degraded drylands with a high erosion risk, different approaches should be used simultaneously. For instance, drought-resistant and nitrogen-fixing trees and shrubs could be given priority in planting activities. The present study identified *Periploca sepium, Bauhinia brahycarpa* and *Caragana soogorica* as suitable species for such purposes (II). The pioneer tree species could be specifically emphasized. Finally, late successional and climax tree species could be introduced as soon as the environment has been sufficiently modified by the other species. Measures that improve the soil moisture conditions also deserve serious consideration under such circumstances (cf. Schiechtl 1985; Bao and Chen 1999; Chow et al. 1999; Ma et al. 2004).

5 CONCLUSIONS AND RECOMMENDATIONS

The Upper Min River watershed is an ecologically and environmentally fragile area. There are two main environmental problems, the one being deforestation and the other soil degradation. The removal of vegetation has greatly increased the runoff and soil erosion, particularly in the mountainous slopes of the watershed. Degraded areas left after excessive soil erosion would be difficult for any measures aiming at ecological restoration. Almost no excessive soil erosion will occur when the vegetation cover approaches 80%. Proper watershed management in this region especially calls for vegetation restoration to improve the soil stability.

This study focused on large-scale soil erosion evaluation, identification of high erosion risk areas, and methods of vegetation restoration. The present results can support decision-making for watershed management and utilization, specifically by offering approaches for soil erosion assessment, forest stand evaluation for biodiversity conservation under various human impacts, priority area identification for quick rehabilitation, and for selection of suitable tree species specifically for various soil types and altitudes.

A total of 84 tree species were identified in the study area, of which almost one third were conifers. These mainly local species are recommended for restoration purposes on degraded lands, for use in their specific ecological zones within the range from 1,300 m to 5,700 m in altitude, and on soils representing three different classes corresponding to their native habitats.

A digital map was developed indicating priority areas for rapid restoration. Areas covered by shrubs and adjacent to existing forests have a high potential for forest restoration. The map can also be utilized for delimiting difficult areas for vegetation restoration, e.g., areas with low precipitation as a limiting factor. Another digital map that was developed can also be used for management decision to show the variation in soil loss intensity. Different scenarios of vegetation recovery that were run to display how the soil loss can be alleviated by vegetation restoration will also help in decision-making for watershed management purposes.

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APPENDICES

Appendix 1 Inventory records and equipment

Inventory records	Inventory equipment for each group
- Forest type	1 GPS equipment
- Stand condition	1 Precision compass
- Species	1 Diameter tape
- Height (m)	1 Measuring tape, 50m
- Diameter at breast height (cm)	1 HAGA clinometer
- Quality	1 Densiometer
- Soil	1 Shovel
- Age (year)	1 Jungle knife
- Basal area	1 Altimeter
- Aspect	1 Relscope
- Slope	1 Increment borer
- Elevation	1 Topographic map
- Land use	1 set of sheet holder, field forms, field instructions, pens, pencils, eraser
- Canopy closure (Trees)	
- Coverage (Shrubs and grass)	

Appendix 2 Inventory form

Date : _____________ Measurer: __________________ Recorder: _______________
Cluster: _____________Plot: __________________________Coordinates: _____________
Forest type: _____________Stand condition: ________________Soil: ___________________
Basal area: _____________Aspect: _______________________Slope: ________________
Elevation: _____________ Land use: ______________________Canopy closure: __________

Tree

No.	Stratified subplots	Species	DBH	Height	Quality	Age
1						
2						
3						
4						
5						
……						

Bamboo	No.	Stratified subplots	Species	DBH Clump	Height	Quality
Scattered bamboo	1					
	2					
	…					
Clumped Bamboo	1					
	2					
	…					

Shrub and Grass

	No.	Stratified subplots	Main species	Coverage
Shrub	1			
	2			
	…			
Total coverage				
Grass	No.			
	1			
	2			
	…			
Total coverage				

Appendix 3 Inventory codes for different records

1. Date: date according to calendar in form year, month, day (yy, mm, dd), for instance 04,06,18.
2. Measurer and recorder: give the number to different names: (1-13)
3. Cluster: different cluster has unique number different from each other
4. Plot: The plot has the number from 1 to 8 in different clusters
5. Coordinates: It has UTM _48N Coordinates for each plot
6. Forest type: The forests are classified into 16 forest types (Vegetation working group, 1980)
7. Stand condition: The stand in each sample plot is described according to different human impact

Code	Despcription
1	unlogged forest, near natural forests
2	logged forest, selectively logged forests
3	secondary forest, heavily logged forests
4	forest plantation
5	return agriculture to forest (new plantation)
6	grassland
7	natural rehabilitation after clear cutting
8	agriculture
9	others
10	shrub (climax)

8. Soil: There are 3 different soil orders and mainly 5 different soil types in the research area.
9. Basal area: Record the numbers measured by relascope
10. Aspect: Record the data (0-360) from Compass
11. Slope: Record the data (0-90) from HAGA clinometer
12. Elevation: Record the data (1500-4000) from Altimeter
13. Land use: (According to the classification from the Chinese Academy of Forestry) (1-11)
14. Canopy closure: Calculate the occupied area from the densitometer (0-100)
15. Species (Appendix 4)
16. DBH: Record the data from the tape
17. Height: Record the data from the tape or calculate from the distance and angle
18. Quality

Code	Description
1	healthy
2	damaged
3	dead

19. Age: Record the age measured by increase borer
20. Shrub and grass coverage: is estimated visually as a percentage

Appendix 4 Vegetation class

Vegetation class	**Area (km^2)**	**Proportion (%)**
Broadleaved forest	183.1705	2.5
Mixed forest	213.9315	2.9
Conifer	1092.5310	14.7
Shrub	1068.3790	21.7
Grass	3516.2005	47.4
Agriculture	230.6636	3.1
Others	569.0650	7.7

Appendix 5 Tree species in the study area

SortOrder	Code	Latin Name	Chinese Name	Family Name	Elevation Range (m)	Max Tree Height (m)
1	CF	*Cephalotaxus sinensis* Li	粗榧	Cephalotaxaceae	100 - 2000	
2	SJS	*Cephalotaxus fortunei* Hook.	三尖杉	Cephalotaxaceae	60 - 2500	20
3	HDS	*Taxus chinensis* Rehd.	红豆杉	Taxaceae	1500 - 2000	
4	GXB	*Cupressus duclouxiana* Hickel	干香柏	Cupressaceae	1400 - 3300	25
5	MJB	*Cupressus chengiana* S.Y.Hu	岷江柏	Cupressaceae	1200 - 2900	30
6	CEB	*Platycladus orientalis* Franco	侧柏	Cupressaceae	300 - 3300	20
7	YB	*Sabina chinensis* Ant.	圆柏	Cupressaceae	0 - 2300	20
8	DGYB	*Sabina tibetica* Kom.	大果圆柏	Cupressaceae	2800 - 4600	30
9	FZB	*Sabina saltuaria* Cheng	方枝柏	Cupressaceae	2400 - 4300	15
10	CIB	*Juniperus formosana* Hayata	刺柏	Cupressaceae	300 - 3400	12
11	HSS	*Pinus armandii* Franch.	华山松	Pinaceae	1000 - 3300	25
12	YOUS	*Pinus tabulaeformis* Carr.	油松	Pinaceae	100 - 2600	25
13	GSS	*Pinus densata* Mast.	高山松	Pinaceae	2600 - 3500	30
14	YNS	*Pinus yunnanensis* Franch.	云南松	Pinaceae	600 - 3100	30
15	YUNS	*Picea asperata* Mast.	云杉	Pinaceae	2400 - 3600	45
16	CXYS	*Picea likiangensis* var. *rubescens* Rehd.	川西云杉	Pinaceae	3000 - 4100	50
17	MDYS	*Picea brachytyla* Pritz	麦吊云杉	Pinaceae	1500 - 3500	30
18	ZGYS	*Picea prupurea* Mast.	紫果云杉	Pinaceae	2000 - 3800	50
19	QQ	*Picea wilsonii* Mast.	青芊	Pinaceae	1400 - 2800	50
20	MJLS	*Abies fargesii* var. *faxoniana* T.S.Liu	岷江冷杉	Pinaceae	2700 - 3900	40
21	LS	*Abies fabri* Craib	冷杉	Pinaceae	2000 - 4000	40
22	HGLS	*Abies ernestii* Rehd.	黄果冷杉	Pinaceae	2600 - 3000	60
23	TS	*Tsuga chinensis* Pritz.	铁杉	Pinaceae	1000 - 3200	50
24	YNTS	*Tsuga dumosa* Eichler.	云南铁杉	Pinaceae	2300 - 3500	40
25	SCLYS	*Larix mastersiana* Rehd.	四川落叶松	Pinaceae	- 2200	
26	HS	*Larix potaninii* Batalin	红杉	Pinaceae	2500 - 4000	50

27	DGHS	*Larix potaninii* var. *australis* Henry	大果红杉	Pinaceae	2700 - 4600	50
28	SMQ	*Acer mono* Maxim.	色木槭	Aceraceae	2100 - 2700	20
29	QZQ	*Acer davidii* Franch.	青榨槭	Aceraceae	500 - 1500	15
30	WJQ	*Acer maximowiczii* Pax	五尖槭	Aceraceae	1800 - 2500	12
31	ZHQ	*Acer sinensis* Pax	中华槭	Aceraceae	1200 - 2000	10
32	SRQ	*Acer tetramerum* Pax	四蕊槭	Aceraceae	1400 - 3300	12
33	MHQ	*Acer erianthum* Schwer.	毛花槭	Aceraceae	1800 - 2300	15
34	KDDQ	*Ilex franchetiana* Loes.	康定冬青	Aquifoliaceae	1850 - 2850	12
35	QS	*Toxicodendron vernicifluum* F.A.Barkl.	漆树	Anacardiaceae	800 - 3800	20
36	YQS	*Toxicodendron succedaneum* Kuntze	野漆树	Anacardiaceae	150 - 2500	10
37	YFM	*Rhus chinensis* Mill.	盐肤木	Anacardiaceae	170 - 2700	10
38	QFY	*Rhus potaninii* Maxim.	青麸杨	Anacardiaceae	900 - 2500	15
39	CM	*Aralia chiensis* Linn.	楤木	Araliaceae	0 - 2700	
40	HH	*Betula albo-sinensis* Burk.	红桦	Betulaceae	1000 - 3400	30
41	BH	*Betula pendula* Roth.	白桦	Betulaceae	500 - 4200	25
42	CPH	*Betula utilis* D.	糙皮桦	Betulaceae	2500 - 3800	30
43	HYJM	*Viburnum betulifolium* Batal.	桦叶荚迷	Caprifoliaceae	1300 - 3100	5
44	HLZ	*Cornus poliophylla* Schneid.	黑椋子	Cornaceae	1300 - 3100	10
45	SJ	*Hippophae rhamnoides* Linn.	沙棘	Elaeagnaceae	800 - 3600	18
46	DZ	*Eucommia ulmoides* Oliver.	杜仲	Eucommiaceae	300 - 2500	20
47	ZL	*Quercus dentata* Thunb.	柞栎	Fagaceae	0 - 2700	25
48	CDGSL	*Quercus aquifolioides* Rehd.	川滇高山栎	Fagaceae	2000 - 4500	20
49	BZL	*Quercus baronii* Skan	枹子栎	Fagaceae	500 - 2700	15
50	RCHL	*Quercus aliena* var. *acuteserrata* Maxim.	锐齿槲栎	Fagaceae	100 - 2700	30
51	GSZ	*Castanopsis delavayi* Franch.	高山锥	Fagaceae	1500 - 2800	20
52	YHT	*Juglans cathayensis* Dode.	野核桃	Juglandaceae	800 - 2800	25
53	MJZ	*Litsea pungens* Hemsl.	木姜子	Lauraceae	800 - 2300	10
54	CG	*Cinnamomum wilsonii* Gamble	川桂	Lauraceae	0 - 2400	25
55	CYXMJ	*Neolitsea confertifolia* Merr.	簇叶新木姜子	Lauraceae	460 - 2000	

56	CH	*Robinia pseudocacia* L.	刺槐	Leguminosae	-	25
57	XKYL	*Magnolia wilsonii* Rehd.	西康玉兰	Magnoliaceae	1900-3300	
58	XC	*Toona sinensis* Roem.	香椿	Meliaceae	1500-2300	25
59	S	*Morus alba* Linn.	桑	Moraceae	-	15
60	YYR	*Ficus heteromorpha* Hemsl.	异叶榕	Moraceae	-	5
61	GT	*Davidia involucrata* Baill.	珙桐	Nyssaceae	1800-2200	20
62	QLBLS	*Fraxinus paxiana* Lingelsh	秦岭白蜡树	Oleaceae	-	20
63	SL	*Pyrus pyrifolia* Nakai	沙梨	Rosaceae	-	15
64	CL	*Pyrus pashia* Buch.-Ham.	川梨	Rosaceae	650 -3000	12
65	ST	*Amygdalus davidiana* C. de Vos	山桃	Rosaceae	800 -3200	10
66	T	*Amygdalus persica* Linn.	桃	Rosaceae	-	8
67	L	*Prunus salicina* Lindl.	李	Rosaceae	400 -2000	12
68	MYT	*Cerasus tomentosa* Wall.	毛樱桃	Rosaceae	100 -3200	13
69	XCYT	*Cerasus serrula* Yu	细齿樱桃	Rosaceae	-	12
70	DKYT	*Cerasus pleiocerasus* Yu	雕核樱桃	Rosaceae	2000 -3400	
71	PG	*Malus pumila* Mill.	苹果	Rosaceae	-	15
72	SGHQ	*Sorbus koehneana* Schneid.	陕甘花楸	Rosaceae	2300 -4000	
73	SY	*Populus davidiana* Dode	山杨	Salicaceae	0 -4000	25
74	DGY	*Populus purdomii* Rehd.	冬瓜杨	Salicaceae	700 -3800	30
75	QY	*Populus cathayana* Rehd.	青杨	Salicaceae	450 -3980	30
76	CEL	*Salix fargesii* Burkill	川鄂柳	Salicaceae	600 -3900	
77	ZZL	*Salix heterochroma* Seemen	紫枝柳	Salicaceae	535 -4200	10
78	SZL	*Salix tetrasperma* Roxb.	四子柳	Salicaceae	300 -2800	10
79	FYYLS	*Koelreuteria bipinnata* Franch.	复羽叶栾树	Sapindaceae	300 -1900	20
80	CC	*Ailanthus altissima* Swingle	臭椿	Simaroubaceae	700 -2500	20
81	HD	*Tilia chinensis* Maxim.	华椴	Tiliaceae	-	15
82	DGJ	*Zelkova sinica* Schneid.	大果榉	Ulmaceae	800 -2500	20
83	LXS	*Cercidiphyllum japonicum* Sieb.	连香树	Cercidiphyllaceae	650 -2700	20
84	LCM	*Euptelea pleiosperma* Hook.	领春木	Trochodendraceae	900 -3600	15

Appendix 6 Equations for biodiversity

The Shannon or Shannon-Weaver (Weiner) Index is calculated as $H = -\sum p_i \ln p_i$, where p_i is the relative abundance of species and i is calculated from the number of stems per ha. Shannon's Index measures the average count per individual of samples taken from a population of species. The maximum value for the Shannon index occurs when the proportions are equal over all of the species.

The McIntosh index is calculated as $M = \sqrt{\sum n_i^2}$, where n_i is the number of individuals in the ith species in the sample. Simpson's (dominance) Index is calculated as $D = \sum\left(\frac{n_i(n_i-1)}{N(N-1)}\right)$, where n_i is the number of individuals in the ith species. Simpson's Index encompasses the richness (total number of species) and evenness (number of individuals per species of a given population). The higher *D*, the lower the diversity.

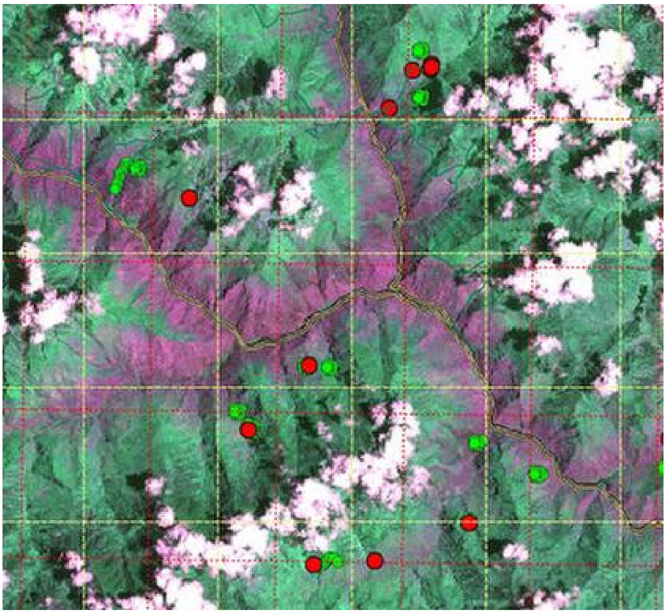
The Berger-Parker Index is calculated as $B = \frac{N_{max}}{N}$, where N_{max} is the number of individuals in the most abundant species. Like in Simpson's index, higher *B* means lower diversity, so the reciprocal is often used.

Evenness is a measure of how similar the abundances of different species are. When there are similar proportions of all species, then evenness equals one, but when the abundances are very dissimilar (some rare and some common species) then the value increases. One type of evenness index is derived from the Shannon-Weaver index: E = H / ln(S), where, S is the number of species in the sample, H is the Shannon-Weaver index.

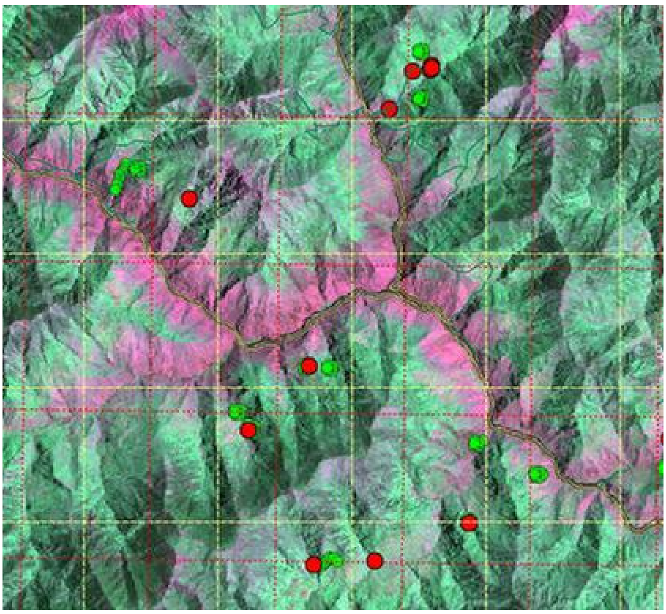
Alpha (α)is calculated by first estimating x from the iterative solution of $\frac{S}{N} = \frac{1-x}{x}(-\ln(1-x))$, where *S* is the number of species in the sample and *N* = the number of individuals in the sample, and then calculating Alpha from $\alpha = \frac{N(1-x)}{x}$.

Q-Statistic (*Q*) is a diversity index presented by Kempton and Taylor (1976). The index is based on the slope of a cumulative species curve in the mid-range of abundances. *Q* is less sensitive to the commonest species in the sample than e.g. the Simpson's index. Mathematical presentation is as follows: $Q = S/2 \ln(R_2/R_1)$, Where *S* = number of species in the sample, R_1 = lower quartile of the species abundance distribution, R_2 = upper quartile of the species abundance distribution.

Appendix 7 Image before and after removing cloud and cloud-shadow.



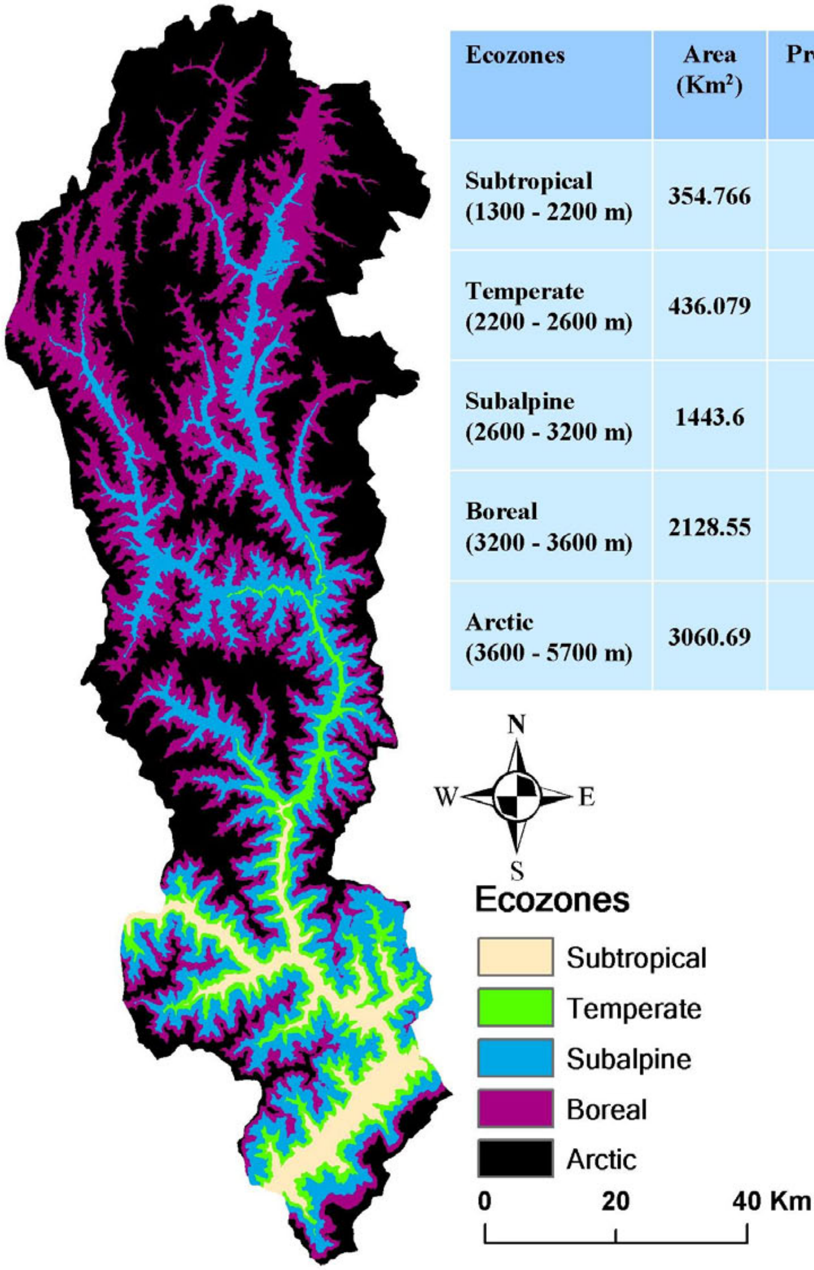
A. Image with cloud and cloud-shadow.



B. Image free of cloud and cloud-shadow.

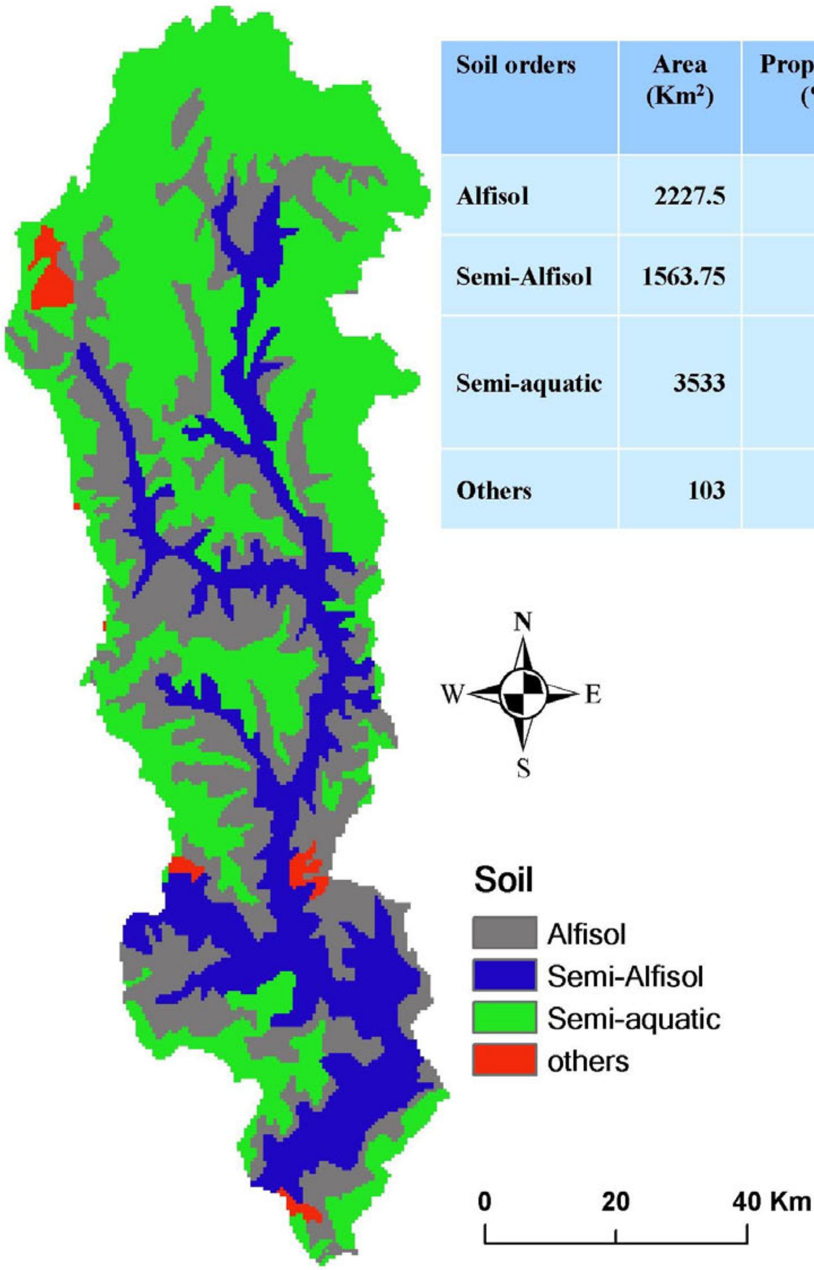
Appendix 8 Ecozones

Ecozones	Area (Km²)	Proportion (%)
Subtropical (1300 - 2200 m)	354.766	4.8
Temperate (2200 - 2600 m)	436.079	5.9
Subalpine (2600 - 3200 m)	1443.6	19.4
Boreal (3200 - 3600 m)	2128.55	28.7
Arctic (3600 - 5700 m)	3060.69	41.2

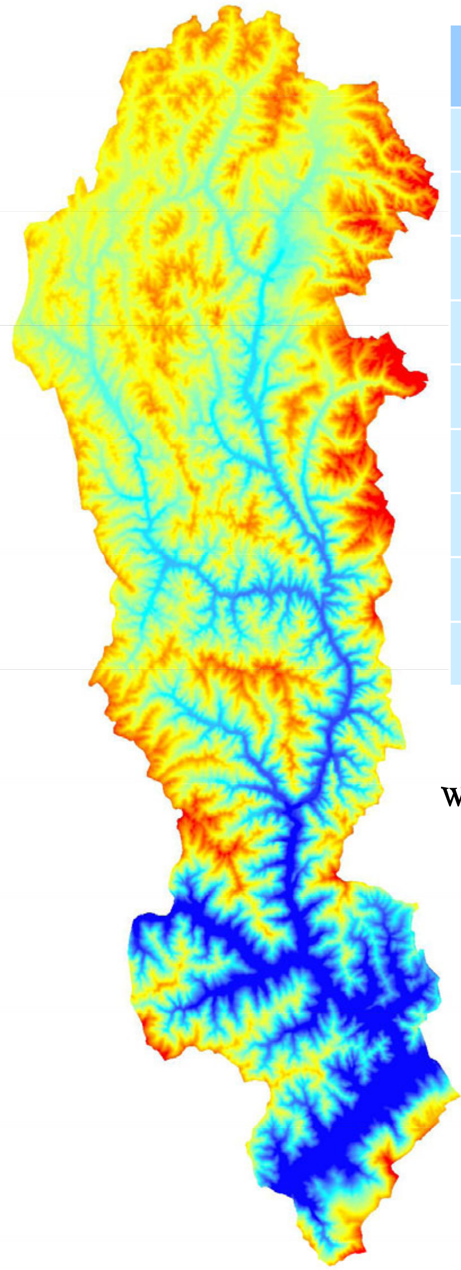


Appendix 9 Soil

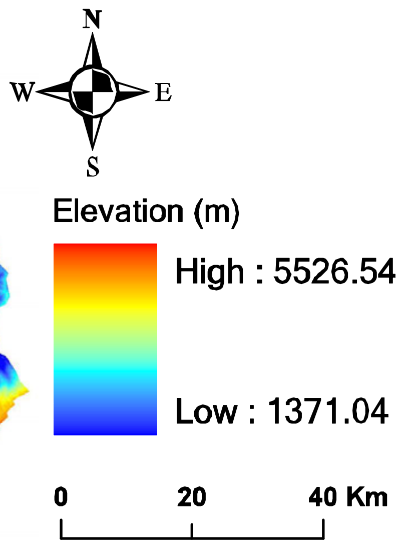
Soil orders	Area (Km^2)	Proportion (%)
Alfisol	2227.5	30.0
Semi-Alfisol	1563.75	21.1
Semi-aquatic	3533	47.6
Others	103	1.4



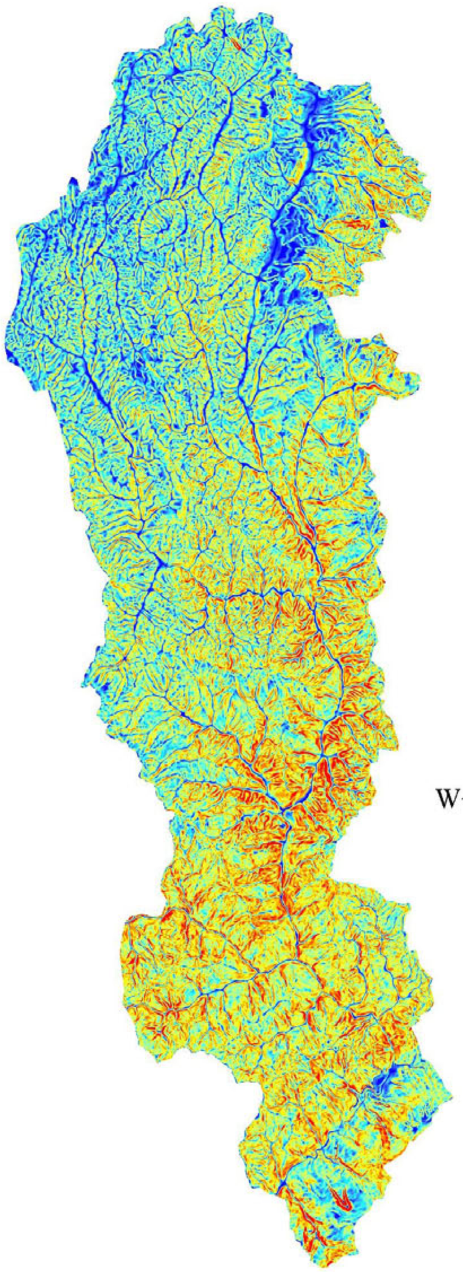
Appendix 10 Elevation



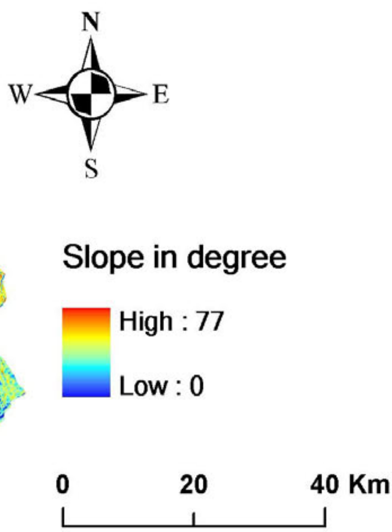
Elevation (m)	Area (km^2)	Proportion (%)
< 1500 m	7.100	0.1
1500 - 2000 m	206.079	2.8
2000 - 2500 m	440.559	5.9
2500 - 3000 m	907.668	12.2
3000 - 3500 m	2172.260	29.3
3500 - 4000 m	2758.630	37.2
3500 - 4500 m	852.134	11.5
4500 - 5000 m	75.223	1.0
> 5000 m	4.036	0.1



Appendix 11 Slope

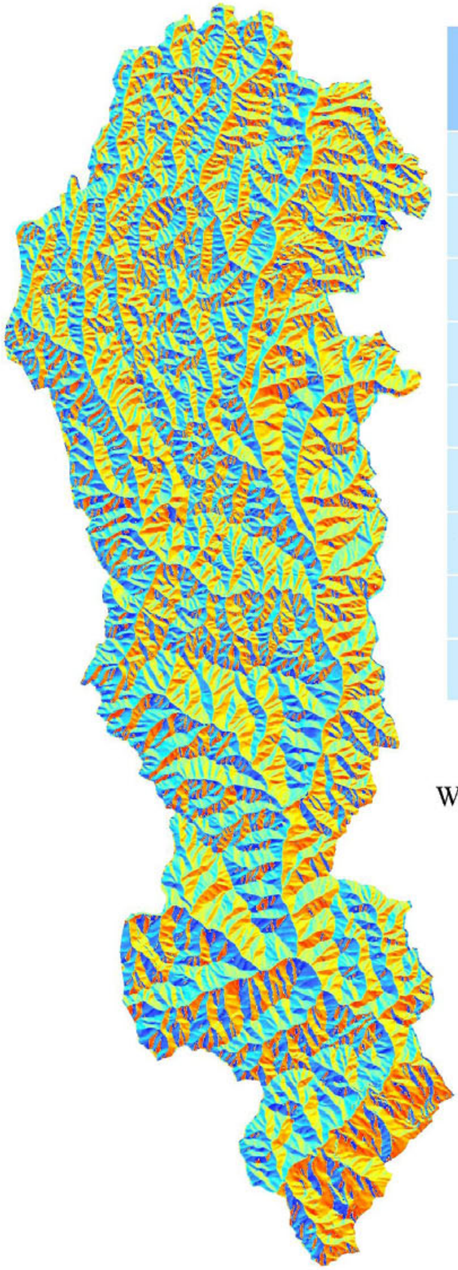


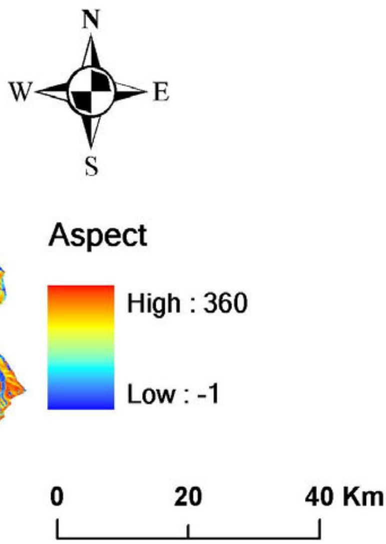
Slope in degree	Area (km²)	Proportion (%)
0-5	305.269	4.1
5-10	484.876	6.5
10-15	661.901	8.9
15-20	912.539	12.3
20-25	1143.200	15.4
25-30	1251.060	16.9
30-35	1144.710	15.4
35-40	830.640	11.2
>40	689.316	9.3



Appendix 12 Aspect

Aspect	Area (Km^2)	Proportion (%)
North	795.861	10.7
Northeast	910.815	12.3
East	967.550	13.0
Southeast	963.146	13.0
South	832.807	11.2
Southwest	950.554	12.8
West	1035.610	13.9
Northwest	962.131	13.0
Flat	5.526	0.1





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