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Grassland degradation in the Greater Caucasus: Ecological Relationships, Remote Sensing Promises and Restoration Implications

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3. Wiesmair M., Feilhauer, H., Magiera, A., Otte, A., Waldhardt, R. (2016): Remotely-sensed assessment of grassland degradation from vegetation cover and vegetation types. Ecological Processes (submitted)

Author's contribution:

In paper 1, I had the main responsibility for design, field work, data analysis and writing. The co-authors contributed valuable ideas and suggestions for this study. In paper 2 and 3, I had the main responsibility for design, field work, data analysis and performed the writing, while the co-authors provided statistical assistance, valuable ideas and suggestions for those studies.

Contents

Chapter 1:

Introduction

Upper Aragvi valley (view from Lomisi church), photo by Martin Wiesmair.

1.1 Grassland degradation

Grassland ecosystems provide multiple goods and services which are of high value for our society (White 2000; Zavaleta et al. 2010). Elementary are the contributions of the grasslands to the global food supply through milk and meat products of ruminants; thus the existing grasslands help to ensure food security for an increasing demand of a growing world population (O'Mara 2012). Local mountain tourism developments enhance economic growth that is founded on the grasslands' landscape setting for recreational activities such as hiking, backpacking, horseback riding and skiing (Debarbieux et al. 2014). Another value is the grasslands' capability of balancing greenhouse gas emissions; hence the ecosystem contributes to the mitigation of climate change (Food and Agriculture Organization of the United Nations 2010b; Soussana et al. 2007). One more grassland benefit is erosion control, which is provided by healthy, dense vegetation cover. Especially the fine, hairy plant roots bind to fungal hyphae, microorganisms, organo-mineral bonds and soil organic matter this creates aggregates that stabilize the soil (Bird et al. 2007; Jakšík et al. 2015; Oades 1984; Tisdall 1994). Furthermore, a high plant diversity increases the variety of root density and root depth which benefit the stabilization of mountain slope (Martin et al. 2010; Pohl et al. 2009). In addition to all the provided ecosystem services, grasslands are precious habitats for a wide range of organisms and contribute globally to a high proportion of biodiversity (White 2000).

Vast grassland areas have undergone degradation processes that evidently disrupt the provision of services from these ecosystems (Gang et al. 2014; Wen et al. 2013). Consequently, the globally increasing phenomenon of land degradation has negative impacts on the environment, society, and economy (Food and Agriculture Organization of the United Nations 2010). Due to the expansion of degraded grassland, researchers identified climate change and human activities as the decisive factors of grassland degradation (Xu et al. 2010; Zhou et al. 2013). The climatic factor is dependent on changes of temperature and precipitation regimes, whereas the human induced degradation can be observed on overgrazed grassland and recreational sites.

Grassland degradation is characteristic for developing and transition countries, where local populations exceptionally suffer from consequences of socio-economic losses and damaging natural disasters (Liu and Diamond 2005). The economic reduction of grassland productivity results from a reduced grass cover and density, increased abundance of unpalatable plant species and soil erosion (Liu et al. 2004).

Soil erosion is minimizing the soil aggregate stability on patches of reduced vegetation cover (Jakšík et al. 2015). Due to topographical properties, erosion processes have much higher consequences in alpine and mountainous terrain than in hilly areas (Stahr 1997). As a consequence, mountainous grassland patches, where the topsoil layer has been lost act as starting points for erosion and natural disasters (Kessler and Stroosnijder 2006). In mountain areas, soil erosion is a natural process which is in particular accelerated by inadequate land use management. Land use changes such as intensification or abandonment induce vegetation changes which influence soil stability and may enforce land degradation (Tasser et al. 2003).

The terrace-like cattle grazing trails increase slope texture roughness which decreases the potential of landslides as they halt the snow material (Leitinger et al. 2008). However, in steep, mountainous terrain, trampling by ungulates creates a net of horizontal, diagonal and vertical tracks where small damage spots can occur when animals are crossing between the passages (Riedl 1983). During heavy rain the water runoff increases on downward facing pathways and vegetation damage spots, which further results in an erosion of the soil layer (Dommermuth 1995; Riedl 1983). Accordingly, similar effects were observed on sites where the vegetation cover had been trampled by tourists (Klug et al. 2002). Consequently, if there is no management action against the water runoff taken, then the soil erosion will be followed by larger mass wasting events which remove the entire soil layer and expose rubble and scree of the parent rock material (Stahr and Langenscheidt 2015). Compared to the surrounding grassland, such habitats of scree display drastically altered site conditions and are further characterized by pioneer communities which establish a first stage of succession (Jenny-Lips 1930; Körner 2003; Zöttl 1952).

Due to climatic and topographic conditions, the natural soil formation on mountain slopes is an extremely protracted process. Hence, dense vegetation cover is a key prerequisite to balance the processes of rapid soil erosion and long lasting soil formation. In summary, the loss of vegetation cover and associated processes are based on complex interactions between land use, land use changes (intensification and abandonment) and regional mountain features, e.g. exposition, underlying parent- rock material, soil type and topography. Investigating all interrelated causes of degradation are therefore essential to understand these relationships. However, to prevent larger degradation events the detection of early erosion stages which are made visible by changes in vegetation is essential. Therefore a thorough knowledge of the vegetation which establishes under characteristic site conditions is mandatory for any site conservation efforts.

In the Caucasus, the failure of previous restoration efforts with unsuitable and exotic plant species indicates the need for information on the present vegetation and in which way it might advance under the impact of erosion events. A broad knowledge of early erosion stages and revegetation measures with indigenous, site specific seed mixtures has evolved for the European Alps (Florineth et al. 2002; Krautzer et al. 2013, 2011; Krautzer and Wittmann 2006; Krautzer et al. 2004) whereas nothing is known about the suitability of native plant species for restoration measures in the Caucasus region.

1.2 Remote sensing

Remote sensing (RS) is an observation without actually being present and is particularly beneficial for mountainous terrains where fieldwork is highly time-consuming and in some inaccessible regions even impossible (Curran 1980). Additionally, RS approaches provide a method to observe vegetation on a larger scale at multiple time points. The ability to observe diverse segments of the environment with means of RS is based on the recorded reflectance curves that vary from object to object due to biophysical properties (Carlson and Ripley 1997; Huete et al. 1985; Tucker and Miller 1977). Plant leaves contain pigments which absorb areas of the visible light for photosynthesis (Knipling 1970; Woolley 1971). Therefore, grass canopy reflectance has a distinct spectral reflectance curve (Tucker 1977; Tucker and Maxwell 1976). Increasing spectral and spatial resolution of space and airborne sensors broadens the options for remote sensing techniques. Hyperspectral sensors capture a high detail of the reflectance signal in very narrow ranges. Multispectral satellite sensors focus on specific spectral bands which cover a broad range of particular wavelengths, e.g. blue, green, red and infrared. As a substitute for space and airborne imagery, portable spectrometers offer the possibility to record hyperspectral data with a similar spectral coverage to test their applicability for a desired research question (Feilhauer et al. 2013).

1.3 Study area

The Republic of Georgia is situated in the Caucasus region and borders on the Black Sea. Due to high topographic and climatic gradients, Georgia consists of various major ecosystems such as evergreen and deciduous forests, dry mountain shrub lands, steppes, semi-deserts, wetlands, and high mountain habitats (Critical Ecosystem Partnership Fund 2004). The whole Caucasus region comprises the high amount of 2791 endemic plant taxa (Solomon et al. 2014) and is therefore declared as one of the global biodiversity hotspots (Myers et al. 2000). Mountain grassland in the high-montane and subalpine zone of the Georgian Caucasus includes different types of meadows and pastures mixed with subalpine tall herb vegetation (Lichtenegger et al. 2006; Nakhutsrishvili 1999). The present state of the Georgian landscape results from a long tradition of human land use which shaped the mountain regions. Archeological records of animal artifacts provide evidence of animal husbandry since ancient times (Lordkipanidse 1991). Additionally, animals play an essential role in religious myths, which reflects the population's historical dependency on livestock as a food source and working aid. One of those legends describes that the construction place of the Lomisi church was predetermined by an oxen. The Lomisi church lies on a mountain ridge, the Qsani-Aragvi watershed, south of Mleta. The village Mleta is divided into the two settlements Kvemo (Lower) and Zemo (Upper) Mleta which are situated on a talus fan in the upper Aragvi valley (Figure 1.1). This thesis focuses on the landscape of the upper Aragvi valley which was shaped by overgrazing, erosion and mass wasting events.

The history of the Aragvi valley which is situated in the Dusheti region is closely linked to its neighboring district, the Kazbegi region. Both districts are separated by the Crosspass (Jvari Pass, 2379 m a.s.l.) which used to be a great barrier for travelers. South of the Crosspass, along the Aragvi valley, people refer to themselves as Mtiuli, inhabitants of Mtiuleti. The Kazbegi region stretches north of the Crosspass along the Tergi river towards the Russian border. The traditional name for the inhabitants of the Kazbegi region is Mokheve, meaning people living in Khevi (georgian for gorge). Due to its localization, the Kazbegi region used to be particularly important for trade between Georgia and its neighboring countries. At present, settlements of the Mtiuli and Mokheve are located along the military road which connects the Georgian capital Tbilisi with Russia. Schmerling & Dolidze (1991) illustrate the military road's route and adjacent cultural monuments in their book "From Tbilisi to Caucasus". The authors describe old alternative routes before the recent military road was built. The previous routes already diverted at Kvesheti or Passanauri and did not come about Mleta. Furthermore, Kverashvili's (2012) book “Khevi and the Mokheve” clearly describes the connection between the construction of the military road and the benefits for the regional development of Mokheve and Mtiuleti. Since the steepest part of the military road is located on the opposite slope to Mleta, the village has played a key role after the construction of the military road in 1861. People benefited from the travelers which enabled further local economic growth and development of the mountain region.

About that time Nikiforov (1887) investigated the economic life of the Georgian state serfs. His work represents the oldest available census on population, livestock and land holdings of the Dusheti and Kazbegi region. For the Dusheti region, he stated that most arable fields belong to the aristocracy which are farmed by themselves or are given to families as a living. Unfortunately the report did not further differentiate between summer and winter rangelands. However, due to the limited grassland in combination with high livestock numbers, the need for additional fodder during winter times can be assumed.

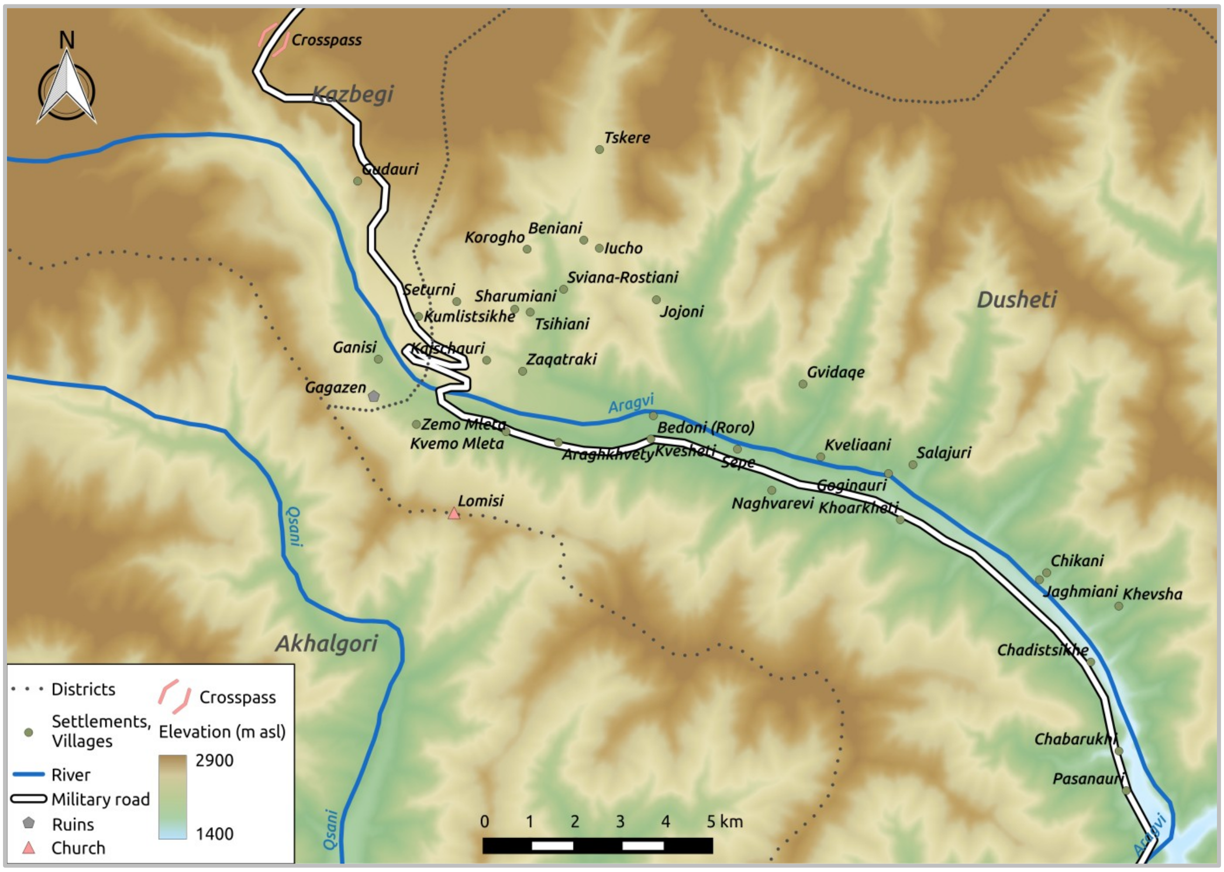

Figure 1.1 Map of the Upper Aragvi valley. (Map by Martin Wiesmair)

Particular interest was put on sheep husbandry and turned into a thriving business. Rcheulishvili's (1953) notes give insight into the large range of Georgian sheep breed and their particular uses. Accordingly, Itonishvili's (1970) "Familiy status of Mokheve" is a thorough review on the development of the sheep husbandry of the Kazbegi and Dusheti region. The author describes the importance of sheep husbandry for the mountain population due to the climatic and topographical settings. At first their number is likely to have been low due to the limited fodder resources during winter times. Due to the limited resources of mountain grassland for hay production as winter fodder, a transhumance system of pastoralism evolved with alternating grazing of northern grounds in the high-montane to alpine belt during summer times and winter grazing on the southern Georgian lowlands. Furthermore, the Russian annexation of Georgia in the early 19th century enabled the sheepherders to use grazing grounds in the northern Caucasian territory (Plachter and Hampicke 2010). In 1955, Rcheulishvili has already described the transhumance system which had evolved in the Georgian mountain regions and gave recommendations for the improvement of sheep farming. Later in 1971, Ketskhoveli described the negative impacts of land use to the landscape. For the Qsani gorge and the Lomisi ridge he stated that the relentless logging of timber as a fuel wood had created desert landscapes along the slopes. The author complains about the bad grassland conditions along the Lomisi ridge and in the Mleta area. He could already see large erosion gullies and found overgrazed pastures dominated by *Nardus stricta*. Furthermore, the author points out the importance of a dense vegetation cover to prevent erosion events. To his opinion, heavy rain events washed out the open soil which resulted in soil degradation and mass wasting events which reached a much more catastrophic character than it used to be. Ketskhoveli claims to reconstruct the natural vegetation by forest and native plant species to cover open soil, and to generally give more attention to the landscape. At that time, he could still meet a sheep herd which migrated from grazing grounds nearby the Caspian Sea to the Lomisi ridge. The author stated that about one third of the Georgian summer grazing grounds were in a bad condition and the other remaining rangeland were neither in good state. Particularly, the Aragvi and Tergi gorges, stream water heads and gorges showed a generally high degree of water erosion and degradation. Ketskhoveli's (1971) plant book, in which he was already complaining about the out-migration of the mountain population, can be considered as a claim for nature conservation.

Additionally to the literature review, personal conversations with inhabitants of Mleta gave me insight into the landscape development. Here is a summary of personal comments which I have encountered during my field stays in 2012 and 2013:

"Until the 1930ies the small settlement Gagazen existed north of Zemo Mleta. Ossetian sheepherders stayed there with their sheep from time to time. When the settlement was abandoned, the inhabitants settled in Zemo Mleta. To me Gagazen has always been a deserted settlement (born 1946). Kvemo Mleta was established after World War II. I was born in a previous settlement which was located down by the Aragvi River. Over time, the river extended after heavy rain events and the inhabitants left their housings in fear of flooding. Therefore, on an upper situated zone we established the village Kvemo Mleta at about 1950. Few stone walls still prove evidence of the previous settlement on the Aragvi river (Figure 1.2). Before that only housings for workers of the military road existed where Mleta is located nowadays and with the construction of the road further houses followed.

Annually, 60,000 sheep passed the Aragvi valley and another 24,000 sheep remained in the village. Those left for their winter grounds from September to May. Until 1970 no fences existed but way marks used to facilitate as a landmark for the herders. In the 1970ies, field clearance cairns were erected and protected by the herders. Until 1980 the canyon was used as a pass for sheep and other livestock, which has resulted in knee-deep erosion gullies. After a snowy winter, an avalanche cleared the whole passage which buried smaller houses and a cemetery at the bottom of the pass. During Soviet times, sheep husbandry used to be the main farming system which was predetermined by the Soviet's annual production quota. Since 1970, the number of sheep has been declining and only about 10% have remained since the 1990ies.

During Soviet period, people had several different breeds of cow (Swiss, milk etc.) and 5 large farms specialized on cattle farming and diary production. The dairy products were transported to Tbilisi. Nowadays only the Georgian mountain breed is used for cattle farming. Kvemo Mleta owns 23 cows which are daily driven to nearby grazing grounds and another 23 cows remain in the village and are fed by hay. The villagers alternate on a daily basis with their duty to drive home the cattle. Bull calves remain to the end of the summer on remote grazing grounds and are taken care of by cowboys. More distant pastures used to be mown at the end of the season but recently more often the hay is harvested from less inclined slopes. Therefore, Mleta cooperates with other neighboring villages.

During World War II, the whole region was deforested and what we see today is a secondary forest. At about 1950 the whole forest had been cleared. 20 years later we had a decent regrowth but in the 1990ies the forest was cut due to the need of fuel wood to substitute gas in winter times. As the gas price has been increasing since independence. In 1976 an avalanche cleared the gorge behind the school. The timber wood fulfilled the village's fuel wood supply for the coming 5 years. From 1989-93 there was a state-run reforestation with mountain ash (Sorbus aucuparia) and conifers (Pinus sp.). However most of the planted trees were damaged from grazing animals. During the reforestation process, the grazing was not stopped and therefore even 20 years old trees still look like shrubs.

During Soviet time, about 360 families lived in the village Mleta. Until 1990, Mleta profited a lot from tourism of horseback riding, camping, heliskiing and hiking. In the course of the South-Ossetian conflict, livestock was kidnapped and armed conflicts followed. As a consequence the air traffic was closed and tourism collapsed. Nowadays, there are about 200 families living in Mleta but many have migrated due to the lack of work."

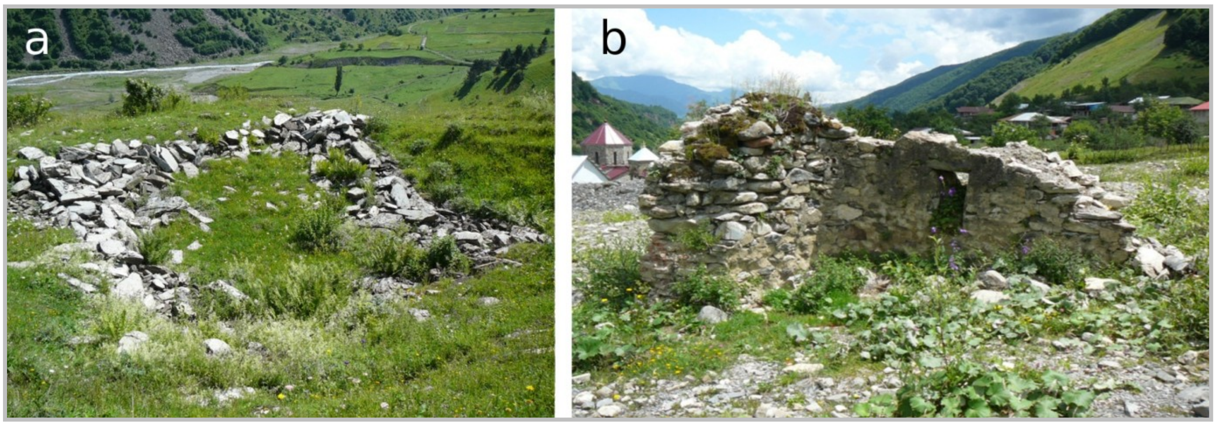

Figure 1.2 Previous settlements near Mleta: a) Signs of Gagazen (abandoned about 1930); b) Buried housewalls next to the Aragvi valley (abandoned about 1950). (Photos by Martin Wiesmair)

1.4 Objectives

The overall objective of this thesis was to study grassland degradation in the Greater Caucasus in order to develop site-specific methods to prevent further degradation in the Caucasus region. Therefore we implemented the commonly used feature of vegetation cover to assess the extent of grassland degradation by remote sensing imagery (chapter 1). However, to gain a deeper understanding we needed to better understand the impacts of overgrazing and erosion events on Georgian mountain grassland. To evaluate these impacts, we investigated the relationships between plant diversity, site conditions and vegetation cover (chapter 2). Based on those results we developed a list of potential plant species for grassland restoration measures. Furthermore we improved the detection of grassland degradation by multispectral satellite sensors as we implemented vegetation cover and vegetation types into a classification model (chapter 3).

1.4.1 Chapter 2: Estimating vegetation cover from satellite imagery

In this study, we developed a site-specific remote sensing approach to assessing grassland degradation based on vegetation cover. We photographed 93 plots within the high-montane and subalpine zone to determine their vegetation cover from image pixels. Further, we used a World View 2 satellite image and derived two vegetation indices, the modified soil adjusted vegetation index ($MSAVI_2$) and the normalized difference vegetation index (NDVI). In a random forest regression model, we tested each vegetation index as a predictor for vegetation cover to detect changes in the grassland canopy from high-resolution satellite images. From the data of the superior NDVI we mapped the high-montane and subalpine grassland cover for our area of interest. To evaluate the indices' ability to assess heterogeneous mountain terrain, we further determined the compositional cover values of rock, soil, and vegetation across varying degradation intensities.

1.4.2 Chapter 3: Plant diversity, site conditions, vegetation and grassland conservation

We described and quantified the mountain grassland vegetation which develops under characteristic overgrazed and eroded site conditions. We illustrated the vegetation composition, environmental variables and functional plant groups along a gradient of grassland vegetation cover. Further, we proposed potential native plant species for revegetation to restore and conserve valuable mountain grassland habitats.

The specific research questions were: a) Which environmental variables are related to the species distribution on overgrazed and eroded sites? b) How is species-richness related to the species distribution on overgrazed and eroded sites? c) Which species-richness and abundance of functional plant groups can be observed on sites of different vegetation cover? d) Which native plant species occur along a gradient of vegetation cover at high frequencies and can therefore be considered for the restoration of grassland ecosystems in the Caucasus?

1.4.3 Chapter 4: Enhanced remotely-sensed grassland degradation indication

In our third study we developed a novel tool to detect grassland degradation by multispectral satellite sensors. Therefore, we combined vegetation cover and vegetation types as indicators for the detection of grassland degradation from remote sensing. We used a hand-held field spectrometer to simulate the multispectral World View 2 sensor at a very high spatial resolution and calculated several multispectral vegetation indices. With random forest modeling we predicted vegetation cover and vegetation types from the simulated World View 2 bands, vegetation indices and environmental variables. Finally, we classified the grassland condition from the combination of vegetation types and threshold values of vegetation cover.

The specific research questions were: a) To which extent can spectral and environmental variables predict vegetation cover and grassland types and which predictor variables are most important? b) Can grassland degradation, represented by grassland types and coverage be detected in multispectral data?

1.5 Methods

For the remote sensing approaches (chapters 2 & 4), we used random forest modeling (Breimann 2001) with the spectral data as predictors. The random forest approach has previously been successfully used to analyze remote sensing data (Feilhauer et al. 2014; Lawrence et al. 2006; Rodriguez-Galiano et al. 2012; Stefanski et al. 2014). A random forest is an ensemble of individual regression trees (Grömping 2009), which are constructed by repeatedly splitting the dataset into homogeneous groups in order to explain the response variable (De'ath and Fabricius 2000). The significance of predictor variables is provided by the measure of variable importance. We used 100times bootstrapping with replacement to validate our model results.

To graphically display the similarity of data, we used non-metric dimensional scaling (NMDS) which is a widely used ordination technique among ecological studies. Therefore a distance measure is calculated which is stepwise placed into a multidimensional space to keep the original distances. The goodness of fit, or how well the configuration fits the data, is measured as stress (Kruskal 1964). To gain information about the indicator species of each vegetation cluster, we performed an indicator species analysis (Dufrêne and Legendre 1997).

In order to evaluate grassland degradation from multispectral data (chapter 4) we implemented the predicted vegetation cover values and vegetation types into a classification (Figure 1.3). Therefore, we set for each vegetation type a threshold of vegetation cover which indicates grassland degradation. For the pastures, we used the threshold of 70% vegetation cover which is a common restoration goal to secure mountain slopes (Krautzer and Klug 2009). For the poor grassland we defined the vegetation cover below 35% as being degraded. Sites which already display vegetation of eroded sites were generally assigned to the class of degradation and therefore the threshold of 100% vegetation cover was chosen.

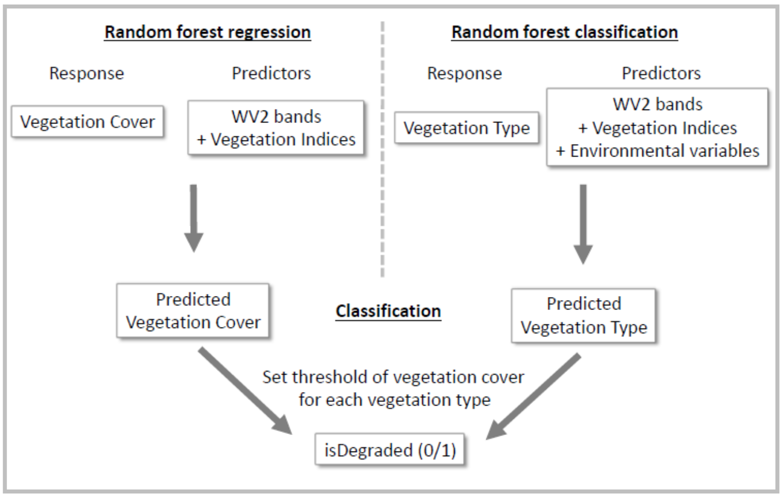

Figure 1.3 Classification scheme of grassland degradation (isDegraded 0/1) by including the predicted values of vegetation cover and vegetation type

1.6 Main results

The best results for an assessment of vegetation cover from satellite imagery were achieved by a random forest model with the NDVI. The produced vegetation cover map showed a low vegetation cover on pastures near the village of Mleta, which indicates more degraded areas and a higher pressure of cattle grazing. Similar developments have been observed within other former Soviet countries in Central Asia (Iniguez et al. 2005). As the animals remain longer in nearby areas, these grasslands are more intensively grazed (Suttie et al. 2005).

Particularly, the steep slopes near villages can be considered to be of higher risk for grassland degradation. Furthermore, we found low vegetation cover, indicating grassland degradation, along hiking trails.

The analyses of plant diversity, vegetation cover and site conditions (Chapter 3) showed four distinct vegetation types. The subalpine zone is characterized by tall herb vegetation, with a high infestation of ruderal pasture weeds. The high-montane sites comprises of nutrient-rich pastures, poor grassland, and scree vegetation. In total we revealed a median of 36 species per plot. Species-richness was highest for the high montane pasture and poor grassland, which differed significantly from the vegetation of the eroded sites. Considering the long lasting period of soil formation in mountainous regions, the long-term loss of diverse grassland and the development towards habitats of no conservation value has to be expected once the vegetation cover is removed. Due to the tolerance to varying site conditions of plant species which we could find within all vegetation types of the high-montane zone and a comparison to other species suggestions for restoration of mountain grassland (Krautzer et al. 2004), we suggested plant species for grassland restoration in the Greater Caucasus. The seed production and suitability of the proposed species for restoration measures in the Caucasus region should be further tested in field studies. Although some single species may possess the capability to quickly restore vegetation cover, the necessity to restore species-rich grassland for erosion control has been reported (Martin et al. 2010; Pohl et al. 2009).

To further improve the detection of grassland degradation by remote sensing we developed a classification model that includes the outcomes of our first two studies (chapter 4). From the implementation of several vegetation indices and WV2 wavebands we improved our previous results (chapter 2) for the detection of vegetation cover from remote sensing. Most important predictors for our random forest models were as follows: the enhanced vegetation index (EVI), the green atmospherically resistant vegetation index (GARI), red edge, near infrared1, near infrared2 and for the random forest classification of vegetation types, the environmental variables altitude and slope. We implemented the predicted values of vegetation cover and vegetation types into a classification. The classification of grassland degradation displayed an overall accuracy of 75%. The lowest accuracy was achieved for the poor grassland and the highest accuracy for the nutrient rich pastures.

1.7 Final conclusion

The cause for mountain grassland degradation involves complex interactions between biotic and abiotic environmental factors. On the one hand, geomorphologic and climatic settings predetermine the potential of erosion and mass wasting events. On the other hand, human activities influence the vegetation layer and its ability to stabilize the slope. In the upper Aragvi valley, most likely the overgrazing during Soviet period in combination with logging of protective forest destabilized steep mountain slopes and caused erosion events. Since Georgia’s independence, the land use changes have further impacted landscape development. However, due to a recent decline in gas prices the logging of protective forest has most likely come to a halt. Furthermore, the livestock numbers have declined and erosion events are mainly localized in the vicinity of settlements where an uncontrolled cattle grazing occurs. Furthermore, grazing has also positive impacts as the rough slope texture of grazing paths and short grasses prevent snow gliding and avalanches. However, if the vegetation cover is damaged by grazing animals or hikers, such damage spots can act as starting points for larger erosion events. On eroded sites, plant diversity is decreased and ecosystem services are lost. Additionally, such habitats of scree diminish the beauty of the mountain landscape. Due to the development of Georgian mountain regions from agricultural to touristic income, the scenic beauty could have further impact for theier future livelihood.

The outcomes of this thesis conclude to monitor grassland conditions. Due to the absent responsibility for the condition of collective rangeland, the establishment of a responsible person to oversee the grassland condition was advisable. The presented novel remote sensing method is a tool for the large scale assessment in addition to field observations. Due to the high costs of satellite images, we suggest assessing the sourroundings of villages, at a high spatial resolution. Before and after the grazing season the detection of vegetation damage spots is essential to prevent further erosion from rain and snow. The damaged spots need to be excluded from grazing and recreational activites, and revegetated with indigenous seed material. Therfore the seed material of the suggested plant species needs to be produced and harvested in Georgian mountain regions to ensure its conformity. The benefit of a revegetation with herbaceous plant material is a very quick regrowth which ensures that the sites can be used again after a few years. Whereas the large scale reforestation needed wide-ranging management installations. Previous reforestation efforts for slope protection have failed due to the lack of an appropriate grazing management. Consequently, to conserve precious Georgian mountain grasslands a sustainable landscape management for the collective mountain grasslands is mandatory. The result of this thesis serve for the implementation into sustainable agricultural and touristic development plans of mountain regions which suffer from grassland degradation.

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Chapter 2:

Estimating Vegetation Cover from High-Resolution Satellite Data to Assess Grassland Degradation in the Georgian Caucasus

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Abstract

In the Georgian Caucasus, unregulated grazing has damaged grassland vegetation cover and caused erosion. Methods for monitoring and control of affected territories are urgently needed. Focusing on the high-montane and subalpine grasslands of the upper Aragvi Valley, we sampled grassland for soil, rock, and vegetation cover to test the applicability of a site-specific remote-sensing approach to observing grassland degradation. We used random-forest regression to separately estimate vegetation cover from 2 vegetation indices, the Normalized Difference Vegetation Index (NDVI) and the Modified Soil Adjusted Vegetation Index ($MSAVI_2$), derived from multispectral WorldView-2 data (1.8 m). The good model fit of R^2 = 0.79 indicates the great potential of a remote-sensing approach for the observation of grassland cover. We used the modeled relationship to produce a vegetation cover map, which showed large areas of grassland degradation.

Keywords

Grassland degradation; erosion; overgrazing; NDVI; $MSAVI_2$; WorldView-2; Georgia; Caucasus

2.1 Introduction

Grassland ecosystems provide multiple goods and services such as food products from ruminants, erosion control, and recreation. Globally, vast grassland areas have undergone degradation that has been triggered by the impacts of climate change and anthropogenic activities such as overgrazing (Gang et al. 2014). Grassland degradation from overgrazing is common in developing countries, in which local populations suffer from the consequences of degradation such as socioeconomic hardship and increased natural disasters (Liu and Diamond 2005).

Similar processes can be observed in Central Asian and Caucasian countries where a transition in livestock management has taken place (Suttie et al. 2005). During the Soviet period, sheep husbandry was practiced with summer grazing in mountain sites and winter grazing in the lowlands. On their migration routes, large sheep herds damaged the vegetation layer of steep slopes (Körner 1980). Nowadays, in most parts of Georgia, migratory sheep husbandry has been replaced by localized cattle farming. Further, in the Georgian Caucasus, erosion is caused by unregulated cattle grazing and logging of protected forests; both have increasingly negative effects on soil stability (Ministry of Environment Protection et al. 2009). To control land degradation, the Georgian national risk assessment report defined areas in the Georgian Caucasus that are prone to natural disasters (CENN and Faculty of Geo Information Science and Earth Observation, University of Twente 2012). Restoration and sustainable use of pastures are urgently required. Furthermore, the growing popularity of hiking and downhill skiing requires sustainable management of sensitive recreational sites.

Approaches to recording the extent of grassland degradation in developing countries have emerged in China, where about 90% of grasslands are considered degraded due to overgrazing and other factors (Liu and Diamond 2005). Akiyama and Kawamura (2007) proposed grassland monitoring by means of remote sensing (RS) as a promising tool for restoring and sustainably managing affected regions. For a long time, the use of RS to monitor arid and semiarid grassland cover has been recognized as essential to determining livestock capacity in order to prevent desertification (Purevdorj et al. 1998).

The observation of vegetation cover on a larger scale at multiple time points makes RS approaches beneficial for monitoring purposes. Liu et al. (2005) used RS methods to estimate the vegetation cover of alpine grassland in Qinghai Province in China. Their results showed high accuracy levels, which indicate the applicability of RS methods for mountainous terrain.

Previous studies on the estimation of vegetation cover relied on examinations at a rather coarse spatial resolution of 30 m x 30 m. Such a scale is unlikely to show the heterogeneity of grass cover (Zha et al. 2003), as variations occur within a few meters in mountainous terrain (Asner and Lobell 2000). Consequently, there is a need to detect small-scale vegetation damage points, in order to prevent further erosion in mountainous regions (Alewell et al. 2008). Increasing the spatial resolution of space-borne sensors broadens RS options; resolution should be chosen in accordance with the spatial scale of the environmental pattern that is analyzed (Feilhauer et al. 2013). We chose imagery from WorldView-2, one of the multispectral sensors with the highest available spatial resolution for our area of interest. The applicability of vegetation indices for the estimation of vegetation cover has been tested with field spectrometers and satellite images (Gessner et al. 2013; Lehnert et al. 2015). From a wide range of vegetation indices, the Normalized Difference Vegetation Index (NDVI) and the Modified Soil Adjusted Vegetation Index ($MSAVI_2$) have been proposed as good predictors of arid and semiarid grassland vegetation cover (Purevdorj et al. 1998; Liu et al. 2007).

In this study, we developed a site-specific RS approach to assessing grassland degradation based on vegetation cover. This assessment can inform management of vulnerable grasslands in the upper Aragvi Valley, where grassland degradation, erosion, and mudflows frequently occur. We tested the 2 multispectral vegetation indices $MSAVI_2$ and NDVI for their appropriateness to detect changes in grassland cover from high-resolution satellite images. To evaluate the indices' ability to assess heterogeneous mountain terrain, we determined the compositional cover values of rock, soil, and vegetation across varying degradation intensities. From the data of the NDVI we mapped the high-montane and subalpine grassland cover for our area of interest.

2.2 Methods

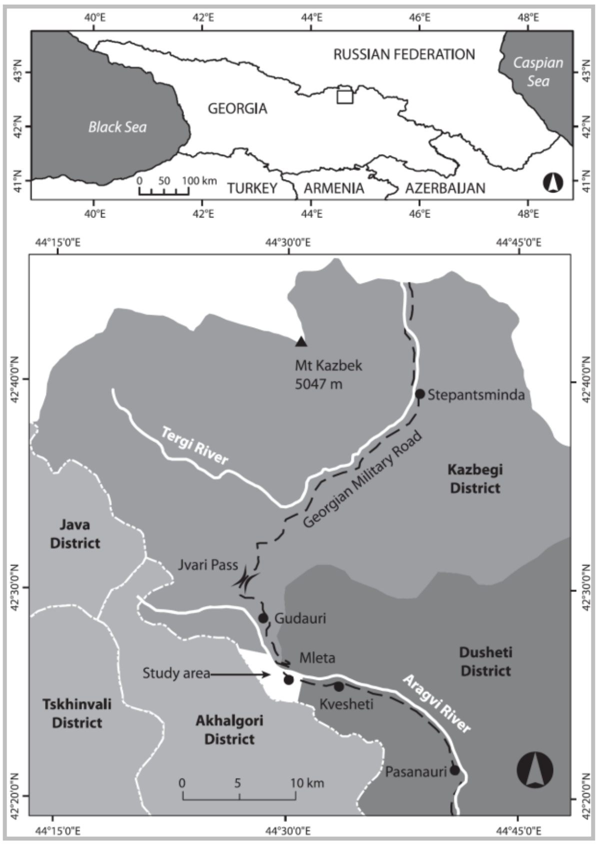
2.2.1 Study area

The study was conducted in the upper Aragvi Valley in the vicinity of the village of Mleta in the Greater Caucasus in Georgia (Figure 2.1). Mleta (42°25’52”N, 44°29’52”E, 1535 m above sea level [m a.s.l.]) is situated on the Georgian Military Road, which connects Tbilisi, the capital of Georgia, with Russia. Mleta consists of 2 parts, Zvemo (Upper) Mleta and Kvemo (Lower) Mleta. South of Mleta, at the bottom of the upper Aragvi Valley, lies the village of Pasanauri (42°21’8”N, 44°41’16”E, 1050 m a.s.l.). Climate data were contributed by the National Environmental Agency and modified by Ina Keggenhoff. The study area has a mean annual temperature of 8.2°C and a mean annual precipitation of 1011 mm. January, the coldest month, has a mean temperature of -3.3°C and 50 mm mean precipitation. The hottest month, July, has a mean temperature of 18.9°C and a mean precipitation of 103 mm.

The upper Aragvi Valley is formed by andesite-basalt in alternation with clay shale, shale marls, and enclosures of limestone and sandstone (Khetskhoveli et al. 1975; Gobejishvili et al. 2011). Close to Mleta, the upper Aragvi Valley is asymmetrically shaped. The slightly inclined, north-facing side is covered by loose sediment, which is prone to erosion and mudflows (Lichtenegger et al. 2006). In the Aragvi Valley, mountain meadow and forest soil can be found (Georgian Institute of Public Affairs 2007). According to the World Reference Base for soil (IUSS Working Group WRB 2007), soil types in the mountain meadows include Leptosols, Cambisols, and Cryosols. The mountain forest soil mainly consists of Dystric Cambisol. Along the river valley, alluvial deposits have built up Calcaric Fluvisols.

The slopes near Mleta range from the river valley bottom at approximately 1500 m a.s.l. to the ridges at about 2200 m a.s.l. The north-facing slopes are characterized by beech forests (*Fagus orientalis*), large erosion gullies, and grassland, which is mainly used for cattle grazing. Cattle tracks and erosion can be observed on the steep slopes of the grassland (Figure 2.2A). Due to anthropogenic impact and topographic features, no clear demarcation line can be drawn between the high-montane and subalpine zones of the Greater Caucasus (Lichtenegger et al. 2006; Nakhutsrishvili et al. 2006). We defined the high-montane zone border at about 1900 m a.s.l., where scattered rhododendron shrubs (*Rhododendron luteum*) indicate a transition to the subalpine zone. The high-montane grassland comprises grass species such as *Agrostis planifolia*, *Cynosurus cristatus*, *Festuca pratensis*, *Poa* *pratensis*, and *Trisetum flavescens* (Khetskhoveli et al. 1975; Lichtenegger et al. 2006). The subalpine grassland is characterized by *Astrantia maxima*, *Betonica macrantha*, *Festuca varia*, *Inula orientalis*, and a strong infestation of *Veratrum lobelianum* (Figure 2.2C).

Figure 2.1 Map of the study area and its location within the Caucasus region (Map by Martin Wiesmair)



2.2.2 Field data

In July 2012 and 2013, we sampled plots (25 m^2) of high-montane and subalpine grassland for vegetation cover, soil cover, and rock cover. In our study area, July is the month of peak plant development; thus, that period offered ideal conditions for vegetation sampling. In the plots, we arranged three 1 m^2 subplots in a triangle with the tip aligned uphill (Figure 2.3). We selected plots according to their total vegetation cover to sample a gradient of grassland coverage. All plots were located on the slope; the flat terrain was not sampled.

Vegetation and soil cover are essential indicators of grassland health or degradation (Zhang et al. 2013). Therefore, we visually estimated the percentages of vegetation, soil, and rock cover. However, due to observer estimation error, the vegetation cover estimates did not yield satisfying model results. To increase accuracy, we photographed the ground vegetation cover and further used these digital images to determine vegetation cover.

Therefore each subplot was photographed with a handheld digital camera (Panasonic LUMIX DMC-TZ1, 5 Megapixel). Photos were taken from a distance to the canopy height over plain ground at nadir 140 cm. We used the image processing program Photoshop CS5 version 12 (Adobe Systems, Mountain View, CA) to calculate the vegetation cover of each subplot. Within each subplot image, we identified pixels that represented vegetation and used the ratio of vegetation pixels to total image pixels to define the percentage of vegetation cover. We further distinguished between the covers of vascular plants and mosses, as mosses considerably contribute to the greenness of sparsely vegetated terrain (Karnieli et al. 2002, 1996). Finally, the plot vegetation cover was computed from the mean of the embedded subplot values calculated before. Altogether, 5 plots were detected as outliers and were removed from further analysis. The remaining 93 plots were then grouped into 4 classes of degradation intensity, based on their percentage of vegetation cover (Table 2.1), a classification comparable to those used in other studies. We used the Wilcoxon rank sum test with Bonferroni correction method for post-hoc class comparisons. All analyses were performed using the R Project statistical computing software (R Core Team 2014).

To extract spectral information from the satellite image, we sampled the geographic position of each plot. The 4 coordinates of our plot corners were recorded with a GPS device (Garmin GPSMap 62s) with a 3–5 m position accuracy. To increase geographic position accuracy, we repeated positioning on a different date, marking plot centers with magnetic markers to locate the plots with a metal detector (Figure 2.3). We further used the mean center function of ArcGIS10 (ESRI, Redlands, CA) to compute the geographic mean of 8 GPS points for each plot.

Table 2.1 Classification of degradation intensity of Georgian high-montane and subalpine grassland based on vegetation cover (modified from Purevdorj et al. 1998, Gao et al. 2006, and Liu et al. 2007)

Vegetation cover (%)	**Degradation class**
80 – 100	None
60 – 79	Light to moderate
30 – 59	Moderate to severe
0 – 29	Extreme

Figure 2.2 Grassland of the upper Aragvi Valley. (A) Cattle tracks and erosion from grazing on steep slopes near the villages; (B) cattle grazing on nondegraded, high-montane grassland; (C) subalpine grassland with an infestation of *Veratrum lobelianum*; (D) grassland degradation along a hiking trail (Photos by Martin Wiesmair)

2.2.3 Multispectral data and analysis

We chose the WorldView-2 satellite sensor, which provides 8 spectral bands from visible (400 nm) to near-infrared (1040 nm) at a spatial resolution of 1.84 m. The sensor provides a radiometric resolution of 11 bit and 16.4 km swath width with a revisiting time of 3.7 days (Digitalglobe 2013). Compared to other satellite sensors, WorldView-2 offers a very high spatial resolution (Ünsalan and Boyer 2011). Recently launched sensors such as WorldView-3 have an even higher spatial resolution but were not yet available when our studies took place. Our WorldView-2 image was acquired on 8 July 2011, during the period of highest vegetation density. The image was atmospherically corrected with the ATCOR 2 module of ERDAS 2013 (DLR, Wessling, Germany).

The vegetation indices $MSAVI_2$ and NDVI were calculated for our plots from the satellite image following Equations 1 and 2:

$$MSAVI_2 = \left(2r_{NIR} + 1 - \sqrt{(2r_{NIR} + 1)^2 - 8(r_{NIR} - r_{RED})}\right)/2 \quad (1)$$

$$\mathrm{NDVI} = \frac{r_{NIR} - r_{RED}}{r_{NIR} + r_{RED}} \quad (2)$$

where r_{NIR} and r_{RED} are the simulated reflectance values in near-infrared and red.

We used NDVI and $MSAVI_2$ separately as predictors for vegetation cover in our random-forest regression analyses. The random-forest approach has been successfully used to analyze RS data (Lawrence et al. 2006; Rodriguez-Galiano et al. 2012; Feilhauer et al. 2014). From the R-package "randomForest 4.6-7" (Liaw and Wiener 2002; Breiman and Cutler 2012) we chose the default setting for the number of predictors sampled for the splitting at each node. As suggested by Breiman (2003), we tested other values, but the default parameterization produced the best results. The number of trees to grow was set to 5000.

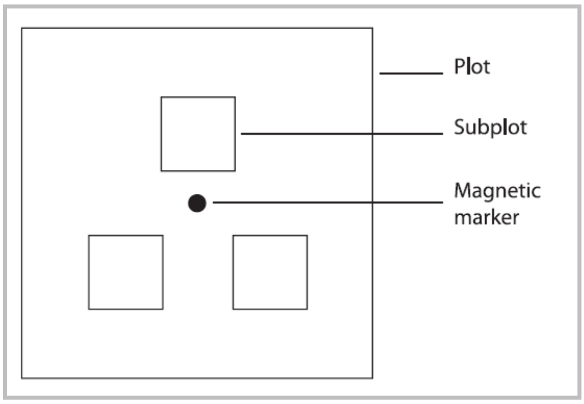
We used 100 times bootstrapping with replacement to validate the data sample. A predicted vegetation cover value for each plot was calculated from the mean of each bootstrap sample. The random-forest model fit was validated through a linear regression of the predicted versus the observed (ground truth) values. For each model we calculated the total root mean square error of prediction (RMSEP), a commonly used criterion for judging the performance of a multivariate calibration model (Faber 1999). For comparisons to other studies, we additionally extracted the RMSEP of each degradation class. All analyses were based on the continuous vegetation cover range. Afterward the classification levels were applied to the model results. The RMSEP was calculated following Equation 3:

$$RMSEP = \sqrt{\sum_{i=0}^{n}\left(\left(Y_i - \hat{Y}_i\right)^2/n\right)} \quad (3)$$

where X is the predicted value from the model, Y the observed value, and n the number of predictions.

A grassland vegetation cover map was predicted from NDVI values, which were extracted from the WorldView-2 satellite image. We applied a continuous vegetation cover scale to a map, where we masked out larger forested areas, streams, clouds, the Aragvi River bed, and settlements.

Figure 2.3 Arrangement of subplots and magnetic marker



2.3. Results

2.3.1 Grassland management

During our fieldwork, we witnessed the grassland management of the upper Aragvi Valley. Grassland is commonly used by all village inhabitants, mainly for cattle grazing on all vegetation cover densities. Most of the grassland area was used as pasture; only small parcels of meadows were fenced off to exclude grazing animals. In order to make use of the whole grassland area, some of the cattle remained close to the villages while others were driven to nearby grazing grounds each morning (Figure 2.2B). The cattle roamed freely during the day and returned to the village in the evening. Small herds of free-roaming horses were met on plateaus with dense vegetation cover. We observed controlled sheep herding on distant pastures southeast of Mleta near the village of Kvesheti. The hiking trails leading to a monastery on top of the mountain range attracted many tourists and pilgrims. The trails lie within the grassland, and we detected severe vegetation damage spots along them (Figure 2.2D). Minor work to restore parts of one hiking trail has been undertaken.

2.3.2 Site cover variables and vegetation cover models

Site variables are displayed as median values for each degradation class in Table 2.2. Soil cover ranged from 4 to 24% and rock cover from 0 to 50%. The soil and rock cover were lowest in sites of no degradation and highest in extremely degraded sites. All classes differed significantly, except that the soil cover of moderately to severely degraded sites did not differ from that of light to moderately degraded and extremely degraded sites. Soil and rock cover were strongly negatively correlated with vegetation cover.

Table 2.3 displays the validated results of both randomforest regression models with corresponding model errors within vegetation cover classes. The validation was calculated from bootstrapped predicted versus observed data. Values for each vegetation cover class were extracted from the model results, which were previously run from the full range of vegetation cover. NDVI and $MSAVI_2$ were calculated from a WorldView-2 satellite image. To visualize the model fits, we plotted values predicted by the model versus the observed values (Figure 2.4). We observed identical model fits for both vegetation indices at $R^2 = 0.79$. Minor differences in total errors or errors of individual degradation classes were observed between NDVI and $MSAVI_2$. The RMSEP for $MSAVI_2$ was 0.02% cover higher on severely and nondegraded classes. For extremely degraded sites, $MSAVI_2$ was 0.11% cover higher than NDVI and did not differ on moderately degraded sites. With decreasing vegetation cover, the model error increased for both indices.

We found the largest proportions of grassland degradation within the high-montane zone (Figure 2.5). Through visual interpretation we identified errors that corresponded to the given RMSEP values of about 15% cover on the extremely degraded sites, which are attributed to erosion gullies and zones of accumulation of debris flow.

Table 2.2 Median values of environmental variables for each degradation intensity class.

Environmental variables	Degradation classes				
	None	Light to moderate	Moderate to severe	Extreme	R^e
Vegetation cover (%)	91.5	70.2	42.2	23.7	-
Soil cover (%)	4.0^a	$12.5^{a,b}$	$20.0^{b,c}$	24.0^c	-0.73
Rock cover (%)	0.0^a	8.0^b	31.5^c	50.0^d	-0.87

[a,b,c,d] Significant variable differences for the Wilcoxon rank sum test of a post-hoc cluster comparison using the Bonferroni correction method ($p < 0.05$)
[e] Spearman correlation coefficient of vegetation cover to soil and rock cover at $p < 0.05$

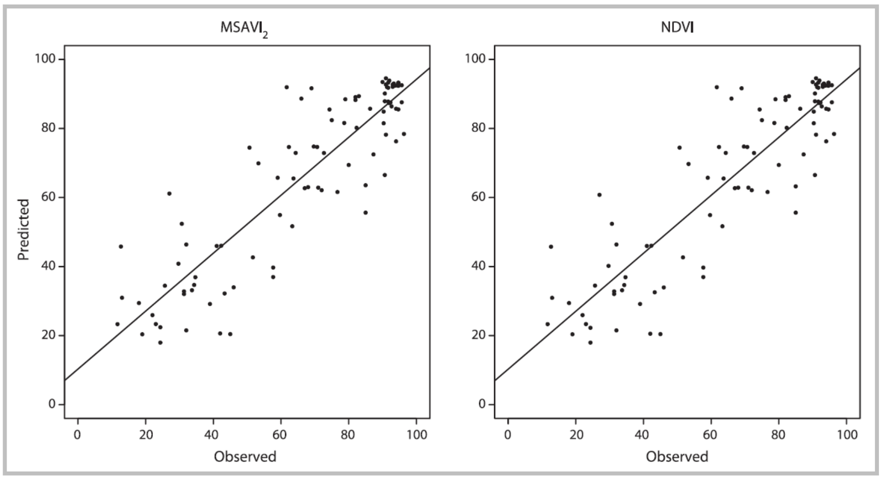


Figure 2.4 Model fits for NDVI and MSAVI2 based on predicted and observed vegetation cover values, given as percentage

2.4 Discussion

2.4.1 Grassland management

Our vegetation cover map indicates a higher pressure of cattle grazing on pastures near the village of Mleta, where we found more degraded areas. Similar developments have been observed within other former Soviet countries in Central Asia (Iniguez et al. 2005). As the animals remain longer in nearby areas, these grasslands are more intensively grazed (Suttie et al. 2005). In addition to land use, topographical conditions affect the severity of erosion. Tasser et al. (2003) found that a slope inclination of 30–40% increased the risk of alpine grassland erosion in the Alps. Therefore, steep slopes near villages can be considered to be of higher risk for grassland degradation. Slope inclination was not considered in our model but should be incorporated in future management plans.

The weeds *Veratrum lobelianum* and *Cirsium obvallatum*, which have been reported in grazing areas in the Caucasus (Callaway et al. 2000), primarily occur in the subalpine zone in areas with dense vegetation cover. Therefore, the influence of varying spectral characteristics of grazing weeds, as has been proposed by Liu et al. (2015), is mainly restricted to the subalpine zone. The subalpine zone of the study region is further interspersed with rhododendron shrubs, which might further contribute to variation in the spectral characteristics of dense vegetation cover. Nevertheless, the degradation spots along the hiking trails are well displayed on our vegetation cover map for the subalpine zone.

2.4.2 Vegetation cover assessment

Grassland in the study region showed higher proportions of soil and rock cover with increasing degradation intensity. This is in accordance with described erosion processes on steep slopes of the Alps. First, the vegetation layer is damaged, and then clods of soil are washed downward until the base rock layer becomes exposed (Stahr 1997). Although revegetation can be observed to some extent on these extremely degraded sites, the natural formation of a new soil layer on degraded mountain slopes is an extremely slow process.

Considering the differing site coverages of our study region, the differences in the spectral reflectance of rock, soil, and vegetation have to be considered for RS methods (Elvidge and Lyon 1985; Clark 1999). Purevdorj et al. (1998) showed that $MSAVI_2$ produced fewer errors than NDVI in the estimation of very low vegetation cover. In our model, differences between NDVI and $MSAVI_2$ were negligible, which is most likely attributable to different site conditions: Sampling plots in our study area included steep slopes up to 43° inclination, and the soil cover values did not exceed those for vegetation or rock cover. It is possible that the stronger topographic influence and high rock cover values interfered with the $MSAVI_2$, which therefore did not mitigate the soil background effect and did not strongly differ from NDVI. The similarity between the 2 vegetation indices at high vegetation cover has also been demonstrated by Qi et al. (1994). Furthermore, both indices were found to be strongly influenced by variations in spectral signals of rock–soil brightness (Elvidge and Lyon 1985).

Considering our model errors and map interpretation, the high rock cover within erosion gullies is most likely causing the higher errors in the prediction of vegetation cover < 30%. Even though Liu et al. (2007) and Purevdorj et al. (1998) showed high model accuracies for vegetation cover < 30%, our results indicate restricted applicability of the vegetation indices for very high rock covers in mountainous terrain. Novel approaches for grassland monitoring by means of multispectral reflectance incorporate several vegetation indices and performed well on the Tibetan plateau (Lehnert et al. 2015). Topographic correction methods, an incorporation of further vegetation indices, and advanced regression methods such as the support vector machine, which were presented by Lehnert et al. (2015), might further improve model results.

Our model's error rate is comparable to that of visual field interpretations, which can range from 10% (Kennedy and Addison 1987) to 15–40% (Tonteri 1990). We assume that the

NDVI's high sensitivity to changes in vegetation cover enabled the good model results. In our study, NDVI derived from multispectral reflectance was shown to detect grassland degradation at a high spatial resolution of 1.84 m, which seems to be appropriate to detect small vegetation damage spots in heterogeneous grassland terrain.

Table 2.3 Validated model fit of random forest regression models

		RMSEP				
Vegetation index	**R^2**	**Extreme degradation**	**Moderate to severe degradation**	**Light to moderate degradation**	**No degradation**	**Total**
NDVI	0.79	16.11	14.25	13.29	9.81	12.61
$MSAVI_2$	0.79	16.22	14.27	13.29	9.79	12.63

2.4.3 Practical implications

Our models proved to be most suitable for mapping vegetation cover of 30–100%. To control erosion in highmontane grassland, vegetation cover of at least 70% is needed (Moismann 1984). Therefore, our models' coverage range is of highest interest for early detection of grassland degradation to enable the implementation of appropriate grazing management and restoration practices.

The manual classification of vegetation cover from photographs of ground cover was highly time consuming, and automated classification methods have been presented as time-saving alternatives by other authors (e.g. Vanha-Majamaa et al. 2000; Zhou and Robson 2001). Although novel methods to retain the fractional vegetation cover from satellite images have been developed (e.g. Li et al. 2014), monitoring should always be supported by field surveys (Gintzburger and Saidi 2010).

Regarding the satellite acquisition date, our model results proved that the period of optimum vegetation growth is an appropriate time to differentiate vegetation cover from soil and rock cover. This has also been demonstrated for other regions with highest separability of green vegetation cover from soil/rock cover (Dennison and Roberts 2003; Marsett et al. 2006; Feilhauer and Schmidtlein 2011).

Because of their cost, WorldView-2 images can generally be applied only to small areas. Their use in transitional and developing countries can be limited to areas near villages that have been defined as vulnerable by larger assessments (such as the Georgian national risk assessment—CENN and Faculty of Geo Information Science and Earth Observation, University of Twente 2012), skiing slopes, and intensively used hiking trails.

For mountainous areas, general assumptions about grassland degradation based on vegetation cover should only be made after incorporating local knowledge about land use. For the upper Aragvi Valley, the loss of vegetation cover from land use and erosion has been well described (e.g. Khetskhoveli et al. 1975; Körner 1980; Lichtenegger et al. 2006). Additional impacts of overgrazing include reduction of plant diversity and infestation by unpalatable weed species (Liu et al. 2004). In the upper Aragvi Valley, these additional types of grassland degradation can be observed. This study, however, focused exclusively on loss of total vegetation cover. Its interrelationship with other degradation types was not tested in the study and would be a fruitful avenue for further research.

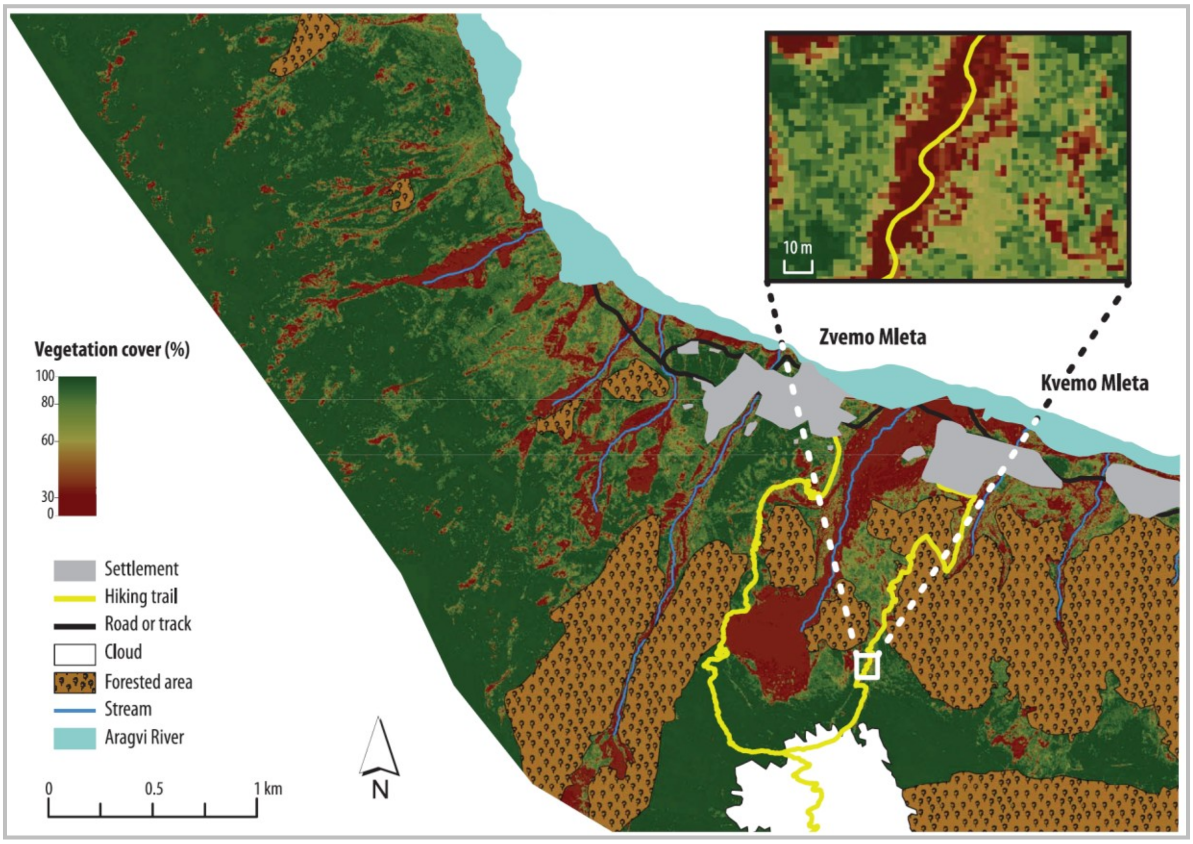


Figure 2.5 Vegetation cover predicted by NDVI for a high-montane and subalpine grassland in the upper Aragvi Valley in 2011. Inset shows degradation along a hiking trail (Map by Martin Wiesmair)

2.5 Conclusion

Transitional countries like Georgia have experienced substantial changes in land use, agricultural systems, and the tourism industry. Further development needs to take place in an environmentally sustainable manner. In order to reduce grassland degradation caused by uncontrolled grazing, the establishment of case-related, sustainable grazing management adapted to the vulnerable mountain grassland is urgently needed.

In the upper Aragvi Valley, the severe grassland degradation near the village of Mleta indicates that the local population is threatened by mass wasting events and the loss of available grazing grounds, and management measures are therefore necessary to prevent these risks. While the extremely degraded slopes require substantial revegetation efforts, more moderately degraded areas might be restored by better-regulated cattle grazing. In using RS to estimate grassland cover, uncertainties due to changes in plant composition and background signals have to be considered. Nevertheless, the RS method presented here can be used to detect changes in vegetation cover with an error rate that is comparable to the error rate of on-site field observations.

We propose the following site-specific management measures for the upper Aragvi Valley and mountain regions that face similar environmental problems:

- Take into account the whole range of vegetation cover.
- Accompany RS monitoring with field observations.
- Take information on slope inclination into account.

Maps of vegetation cover produced in the presented way can play a key role in the evaluation of current grassland degradation, the decision for potential tourist development, and the success of future management plans.

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Chapter 3:

Relationships between plant diversity, vegetation cover, and site conditions: implications for grassland conservation in the Greater Caucasus

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Abstract

Overgrazing, land use abandonment and increasing recreational activities have altered the vegetation of high-montane and subalpine grassland of the Caucasus. The failure of previous restoration efforts with unsuitable and exotic plant species indicates the need for information on the present vegetation and in which way it might change. Within the Greater Caucasus, we have described and quantified the mountain grassland which develops under characteristic overgrazed and eroded site conditions. Further, we have proposed potential native plant species for revegetation to restore and conserve valuable mountain grassland habitats. We used non-metric dimensional scaling ordination and cluster comparison of functional plant groups to describe a gradient of grassland vegetation cover. For our study region, we identified four major vegetation types with increasing occurrence of ruderal pasture weeds and tall herb vegetation on abandoned hay meadows within the subalpine zone. Within high-montane grassland a decline of plant diversity can be observed on sites of reduced vegetation cover. Due to a low potential of the grassland ecosystem to balance further vegetation cover damage, the long-term loss of diverse habitats can be expected. We conclude with management recommendations to prevent erosion and habitat loss of precious mountain grasslands.

Keywords

Land degradation; NMDS; overgrazing; functional plant groups; mountain grassland restoration

3.1 Introduction

The stability of mountain slopes is substantially influenced by site conditions such as vegetation cover, vegetation composition and species-richness (Martin et al. 2010; Pohl et al. 2009), which are in turn dependent on biotic environmental conditions and management practices (Tephnadze et al. 2014; Wellstein et al. 2007). Agricultural management practices under low stocking numbers and short grazing periods are of vital importance for a sustainable grassland management. Accordingly, a disregarded grazing regime harms the vegetation cover and induces grassland degradation processes. Furthermore, key plant species such as grasses can be replaced by unpalatable, invasive plants (Vallentine 2001) and some of them e.g. *Veratrum lobelianum* and *Cirsium obvallatum* are referred to as ruderal pasture weeds (Callaway et al. 2000). In steep, mountainous terrain, trampling by ungulates creates a net of horizontal, diagonal and vertical tracks where small damage spots can occur when animals are crossing between the passages (Riedl 1983). During heavy rain the water runoff increases on downward facing pathways and vegetation damage spots, which further results in an erosion of the soil layer (Dommermuth 1995; Riedl 1983). Accordingly, similar effects were observed on sites where the vegetation cover had been trampled by tourists (Klug et al. 2002). Consequently, if there is no management action against the water runoff taken, then the soil erosion will be followed by larger mass wasting events which remove the entire soil layer and expose rubble and scree of the parent rock material (Stahr and Langenscheidt 2015). Compared to the surrounding grassland, such habitats of scree display drastically altered site conditions and are further characterized by pioneer communities which establish a first stage of succession (Jenny- Lips 1930; Körner 2003; Zöttl 1952). Due to climatic and topographic conditions, the natural soil formation on mountain slopes is an extremely protracted process. In mountainous regions, erosion is a natural process which is further accelerated by a reduced vegetation cover. Dense vegetation cover is a key prerequisite to balance the processes of rapid soil erosion and long lasting soil formation. Hence, the loss of vegetation cover and associated processes are based on complex interactions between land use, land use changes (intensification and abandonment) and regional mountain features, e.g. exposition, underlying parent rock material, soil type and topography. Investigating all interrelated causes of degradation are therefore essential to understand these relationships. However, to prevent larger degradation events the detection of early erosion stages which are made visible by changes in vegetation is essential. Therefore a thorough knowledge of the vegetation which establishes under characteristic site conditions is mandatory for any site conservation efforts. Consequently, a broad knowledge of early erosion stages and revegetation measures with indigenous, site specific seed mixtures has evolved for the European Alps (Florineth et al. 2002; Krautzer et al. 2013). Immediate revegetation of open vegetation with indigenous, site specific seed mixtures is an essential prerequisite for the success of ecological restoration (Krautzer et al. 2004; Krautzer and Wittmann 2006; Krautzer et al. 2011). However, despite ongoing erosion processes nothing is known about the suitability of native plant species for restoration measures in the Caucasus region.

Altogether, overgrazing influences grassland condition which results in a reduced grass cover, an increased abundance of unpalatable plant species and soil erosion. On a mountain landscape level it has to be considered that these complex effects appear spatially and temporally interdependent. Previous studies have mainly independently researched effects of either vegetation cover (Martin et al. 2010; Pohl et al. 2009) or pasture weeds (Callaway et al. 2000) in relation to the diversity of degraded grassland. The first goal of our study was to describe and quantify the vegetation which develops under characteristic overgrazed and eroded site conditions. Therefore we have addressed the following research questions:

Which environmental variables are related to the species distribution on overgrazed and eroded sites?

How is species-richness related to the species distribution on overgrazed and eroded sites?

Our second goal was to give recommendations for a site specific ecosystem restoration in the Caucasus. Therefore we aimed to describe and quantify functional plant groups in order to maintain ecosystem functions after restoration measures. Further, we aimed to suggest potential plant species for revegetation. Therefore we addressed the following questions:

Which species-richness and abundance of functional plant groups can be observed on sites of different vegetation cover?

Which native plant species occur along a gradient of vegetation cover at high frequencies and can therefore be considered for the restoration of grassland ecosystems in the Caucasus?

3.2 Methods

3.2.1 Study region

The Republic of Georgia is situated in the Caucasus region and borders on the Black Sea. Due to high topographic and climatic gradients, Georgia consists of various major ecosystems such as evergreen and deciduous forests, dry mountain shrub lands, steppes, semi-deserts, wetlands, and high mountain habitats (Critical Ecosystem Partnership Fund 2004). The whole Caucasus region comprises the high amount of 2791 endemic plant taxa (Solomon et al. 2014) and is therefore declared as one of the global biodiversity hotspots (Myers et al. 2000). Mountain grassland in the high-montane and subalpine zone of the Georgian Caucasus includes different types of meadows and pastures mixed with subalpine tall herb vegetation (Lichtenegger et al. 2006; Nakhutsrishvili 1999).

The present state of the Georgian landscape results from a long tradition of human land use which shaped the mountain regions. Although archeological records of animal artifacts provide evidence of animal husbandry since ancient times (Lordkipanidse 1991), at first their number is likely to have been low due to the limited fodder resources during winter times (Itonishvili 1970). Over time, humans exploited the mountain forests and particularly replaced the subalpine birch forests by pastures, meadows, and arable fields (Nakhutsrishvili 1999). Nowadays, in the Aragvi and Tergi valley the once widespread deciduous, coniferous and mixed mountain forests of the high-montane and subalpine belt can only be found in small remaining patches within protected areas and remote places (Khetskhoveli et al. 1975; Parolly 2014). Due to the limited resources of mountain grassland for hay production as winter fodder, a transhumance system of pastoralism evolved with alternating grazing of northern grounds in the high-montane to alpine belt during summer times and winter grazing on the southern Georgian and Russian lowlands. Furthermore, the Russian annexation of Georgia in the early nineteenth century enabled the sheepherders to use grazing grounds in the northern Caucasian territory (Plachter and Hampicke 2010). In 1861, the construction of the military road was finished (Kerashvili 2012; Schmerling and Dolidze 1991), which connected Tbilisi, the capital of Georgia, and Russia and enabled further local economic growth and development of the mountain region. The sheep husbandry profited from infrastructural developments and many arable fields were turned into pastures and meadows (Itonishvili 1970). Due to the collectivization during the Soviet period, the sheepherders took care of large state-owned flocks. Herding very large sheep herds through the mountain regions had consequences for the landscape, as was observed on the steep slopes along the migration routes of the upper Aragvi valley (Körner 1980). The upper Aragvi valley is asymmetrically shaped by an east side (south facing) that consists of andesite-basalt and a west side (north facing) which comprises clay-shale, shale marls and enclosures of limestone and sandstone (Gobejishvili et al. 2011; Khetskhoveli et al. 1975). The slightly inclined, north facing side is covered by loose sediment which is prone to erosion and mudflows (Lichtenegger et al. 2006). For the upper Aragvi valley, a comparison of aerial images from the years 1958 and 2011 displayed a 10% increase in degraded and un-vegetated terrain (Klein 2011). This indicates ongoing erosion processes which are currently initiated by unregulated cattle grazing and logging of protected forests (Ministry of Environment Protection et al. 2009). The long lasting negative effects of the previous herding system collectivization followed by a de-collectivization, have been reported for several grassland ecosystems of former Soviet Union countries (Food and Agriculture Organization of the United Nations 2003). Additionally, recent development of an unrestricted increase in recreational activities (e.g. hiking) are further contributing to persisting erosion processes (Wiesmair et al. 2016).

In the Greater Caucasus, overgrazed grassland display vast areas of infestations by unpalatable plant species such as *Veratrum lobelianum* and *Cirsium obvallatum* (Callaway et al. 2000; Magiera et al. 2015). *Veratrum lobelianum* is closely related to *Veratrum album* which occurs in Central Europe and is, due to its acute toxicity and high abundance, a grazing weed of major concern (Schaffner et al. 2001). *Cirsium obvallatum* features sharp spines along its leaves and stems which protect the plant from being grazed (Callaway et al. 2000).

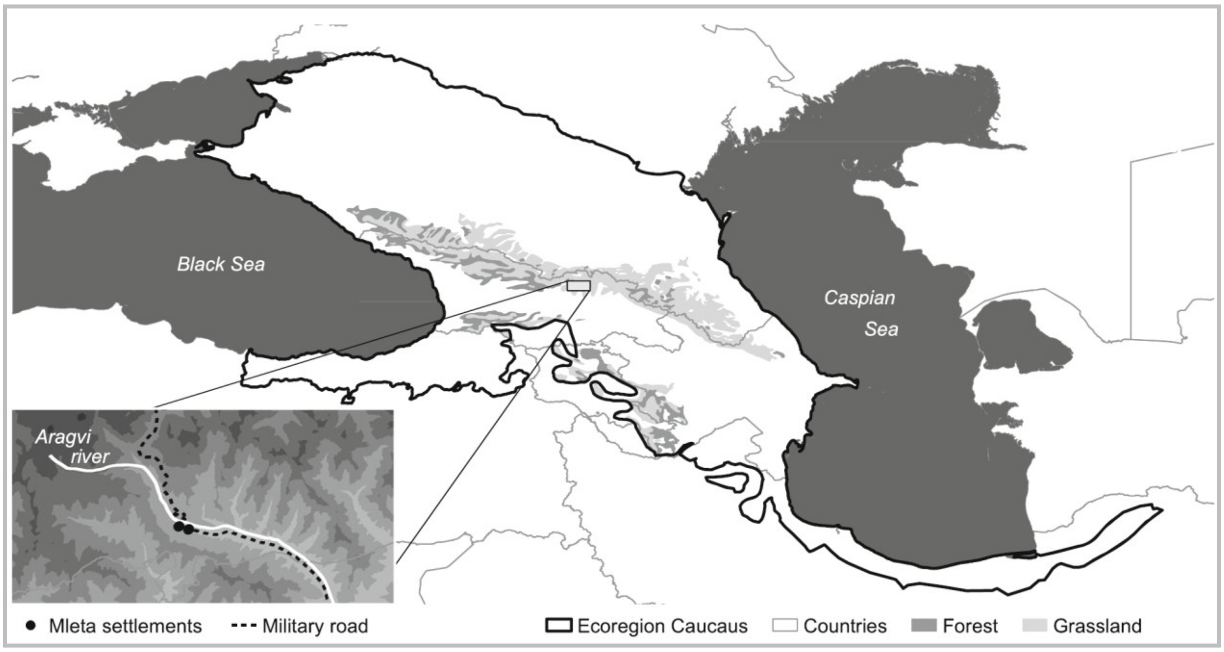

Figure 3.1 Localization of Georgia and the study region within the Caucasus biodiversity hotspot with visualizations of montane pine forests combined with montane, submontane fir and mixed fir forests (Forest), and subalpine vegetation together with alpine grassland (Grassland). Inlet showing a contour map of the upper Aragvi valley, with 500 m contour intervals starting from 1000–1500 m (light grey). Sources Caucasus Biodiversity Hotspot (Critical Ecosystems Partnership Fund), Map of the natural vegetation of Europe (BfN, Federal Agency for Nature Conservation Germany)

3.2.2 Study sites and field sampling

The village Mleta is divided into the two settlements Kvemo (Lower) Mleta (42°25’40”N, 44°29’52”E, 1455 m above sea level [m a.s.l.]) and Zvemo (Upper) Mleta (42°25’45”N, 44°29’23”E, 1535 m a.s.l.) which are situated on a talus fan in the upper Aragvi valley (Figure 3.1). A large net of animal tracks and vegetation damage spots which were caused by cattle and sheep grazing, can be observed on the slopes next to Mleta (Figure 3.2a). Minor construction works for hiking trails and forestation with maple (*Acer platanoides*) for wood production have been going on in the study region. Previous reforestation with coniferous trees (*Pinus* sp.) and leguminous bushes (*Amorpha fruticosa*) has failed to stop further land degradation and especially the planting of exotic plant species seems to be highly unsuitable for slope protection (Figure 3.2b). Due to its accessibility from the military road and the occurrence of mass wasting events on a large scale, Mleta is a perfect study region for erosion processes from overgrazing. Furthermore, grassland on the north facing slope next to Mleta consists of a uniform geological layer of slate with homogeneous soil layers of Cambisol under grassland and Leptosol on eroded sites.

In July 2012 and 2013, we sampled 97 plots next to the village Mleta for vegetation and environmental parameters. We selected the plots according to their total vegetation cover in order to sample a gradient of grassland coverage (0–100%). Due to the high correlation of vegetation cover to slope inclination, we used a digital elevation model to predefine sampling areas. For our study area we sought for the full range of slope inclination and vegetation cover. Heterogeneous topographic conditions determined sampling at a minimum distance of 5 m to bordering habitat patches. All plots were located on the slope (3–43° inclination) whereas the flat valley ground and wetland were not sampled. Each plot covered 25 m^2, within which we arranged three 1 m^2-subplots. Species abundance was estimated for each 1 m^2-subplot with the modified Braun-Blanquet cover-scale (Barkman et al. 1964). In order to include all occurring plant species on the plot without occurrences on the subplots, we assigned the lowest cover value (r) to species that were exclusively found on plot level. Vegetation, soil, rock and moss cover were visually estimated for each subplot. Maximum plant height was measured separately for grasses and herbs on subplot level. The subplot values were averaged to obtain a single plot value. Botanical nomenclature followed ‘‘The Plantlist’’ (The Plant List 2013).

Moreover, five samples of the top soil were randomly taken from each plot and pooled. Due to a very high stone content, we used a hand shovel to sample the upper soil layer (0–5 cm). The mixed samples were air- dried for 96 h. We sieved the soil samples to 2 mm to separate the coarse soil material from the fine soil. Coarse soil material was washed, all plant material removed, air-dried and weighed. We weighed the fine and coarse soil fractions in order to determine the percentage of stone content (> 2 mm). The fine soil fraction was used for further chemical analyses. Soil samples were analyzed for their C and N values at the Institute of Soil Science and Soil Conservation at the University of Giessen, to retain information on total nitrogen (N_t) and total carbon (C_t). Calcium carbonate ($CaCO_3$) was determined using the Scheibler apparatus method and was used to calculate the ratio of organic carbon (C_{org}) to the total amount of carbon. C/N was calculated from the ratio of organic carbon to total nitrogen. Plant available phosphorus (P_{CAL}), potassium (K_{CAL}) and magnesium (Mg) were analyzed at the Agrofor Lab, Wettenberg, Germany. We determined pH-value in CaCl2.

For each plot we measured the inclination with the SUUNTO PM-5 C 360 clinometer and the four coordinates of our plot corners with a GARMIN GPS62s. We marked the plot center with magnetic markers to re-locate the plots. The geographic direction was recorded to calculate the exposition values of northness (cosine of aspect) and eastness (sine of aspect) for each plot.

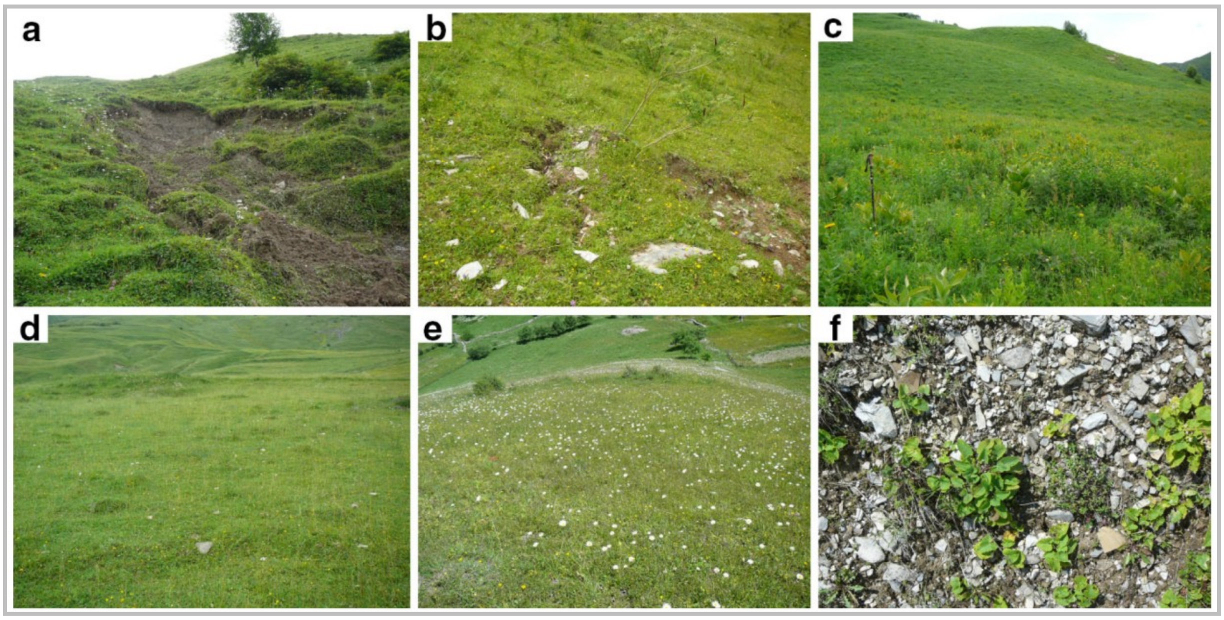

Figure 3.2 Degradation- and grassland types of the upper Aragvi valley. a) vegetation damage spot on highmontane pasture; b) open soil on a site with *Amorpha fruticosa*; c) *Seseli transcaucasica*-cluster with infestation of *Veratrum lobelianum*; d) *Cynosurus cristatus*-cluster; e) *Briza media*-cluster; f) *Parnassia palustris*-cluster

3.2.3 Analysis of functional plant groups

Following de Bello et al. (2010), we grouped plant species with similar responses to external factors and effects on ecosystem processes as functional plant groups. Therefore, the plant groups were first defined as herbs, graminoids (all grasses, including sedges) and woody plants (regenerating trees and shrubs). The herb group was further differentiated into *Fabaceae*, *Orchidaceae* and ruderal pasture weeds (*Cirsium obvallatum*, *Rumex obtusifolius*, and *Veratrum lobelianum*). *Fabaceae* play an important role in the input of nutrients due to their nitrogen-fixing capability. Plants of the *Orchidaceae* family belong to a group which has become endangered in many European grasslands due to the decline of suitable habitats (Calaciura and Spinelli 2008). Due to the high sensitivity of this plant family towards anthropogenic influences, conservation measures to protect these habitats would be as well necessary in Georgia (Akhalkatsi et al. 2003). Ruderal pasture weeds have the potential to replace grassland communities under intensive grazing when neglected by grazers (Vallentine 2001).

According to previous studies dealing with grassland conditions (Gao et al. 2006; Liu et al. 2007; Purevdorj et al. 1998), our dataset was classified into four vegetation cover intensities: 80–100% = dense, 60–79% = moderate, 30–59% = light, and 0–29% = sparely. We used the classification scheme to further analyze the abundance and species-richness of functional plant groups within a gradient of vegetation cover. Therefore we calculated the relative contribution of each functional plant group to the whole species number and abundance.

3.2.4 Analysis of vegetation and environmental data

We transformed the Braun-Blanquet cover values to percentage scale (r = 0.3, + = 0.5, 1 = 2.5, 2 m = 4.0, 2a = 8.75, 2b = 18.75, 3 = 37.5, 4 = 62.5, 5 = 87.5), which was further log-transformed. To reduce noise in the data (in total 171 species), we omitted species with less than three occurrences prior to ordination analyses. Three outlier plots were detected with nearest single linkage/nearest neighbor classification and were removed from further analyses. Following analyses were run with a dataset of 94 plots (1505–2183 m a.s.l.) and 127 species. We used the ‘‘vegan’’ R-package (Oksanen et al. 2013) for Ward’s classification and Non-metric dimensional scaling (NMDS) ordination, both was performed with Bray–Curtis distance. The settings for the NMDS were: global Multidimensional scaling using monoMDS; two convergent solutions reached after twelve tries; scaling of centering, PC rotation and halfchange scaling. A detrended correspondence analysis (DCA) was performed to receive information on gradient length. To gain information about the indicator species of each vegetation cluster, we performed an indicator species analysis (Dufrêne and Legendre 1997) using the ‘‘indval’’ function of the ‘‘labdsv 1.7’’ R-package (Roberts 2013). Species-richness, Shannon and evenness diversity indices were derived with the ‘‘vegan’’ R-package. We used the Wilcoxon rank sum test with Bonferroni correction method for posthoc class comparison after Kruskal–Wallis cluster comparison. All analyses were performed using the R project statistical computing software (R Core Team 2014). Environmental percentage and degree values (slope, vegetation, soil, rock, moss and stone content) were arcsine transformed prior to analyses of cluster comparison and NMDS.

3.3 Results

3.3.1 Vegetation types and site condition

The Ward classification resulted in four distinct clusters, which were plotted in an NMDS ordination (Figure 3.3). The classification separated subalpine (*Seseli transcaucasica*-cluster, Figure 3.2c) from high-montane grassland. Within the high-montane sites, the classification further differentiated between nutrient-rich pastures (*Cynosurus cristatus*-cluster, Figure 3.2d), poor grassland (*Briza media*-cluster, Figure 3.2e), and scree vegetation (*Parnassia palustris*-cluster, Figure 3.2f). The NMDS included 127 plant species and revealed a stress level of 13.6. Environmental vectors were fitted against NMDS ordination and most important variables ($p < 0.001$) are shown in the ordination graph (Figure 3.3). The corresponding vector values for the correlations of environmental vectors to the first and second NMDS axis are given in Table 3.1. The first axis displays a gradient of cover values (vegetation, soil and moss) and nutrient availability. The second axis represents a gradient of species diversity. Other variables such as canopy height, altitude, slope and rock cover display vectors in-between both axes. Northness, eastness and total carbon do not significantly correlate with the first two NMDS axes. Due to an intercorrelation of the environmental variables and for better visualization some of the variables were excluded from the ordination graph, as follows (Spearman correlation index, $p < 0.05$): Calcium carbonate ($R = 0.80$) with stone content; Moss cover ($R = 0.52$) with rock cover; species-richness ($R = 0.91$) and evenness ($R = 0.64$) with Shannon diversity; Maximum height of graminoids ($R = 0.63$) with maximum height of herbs. The DCA showed a gradient length of 4.1, which indicates a complete species turnover.

The ordination shows that the nutrient-rich sites are restricted to the *Seseli transcaucasica*- and *Cynosurus cristatus*-cluster. These sites have higher contents of magnesium, potassium, phosphorus, total nitrogen, and organic carbon. The *Seseli transcaucasica*-cluster displays a higher maximum vegetation height. The *Briza media*- and *Parnassia palustris*-cluster display low nutrient values and can be characterized as nutrient-poor. These sites are characterized by a lower vegetation cover with conversely higher soil, rock, and moss cover values. Furthermore, these sites display higher stone contents with increased pH and $CaCO_3$ values. Additionally, these plots are characterized by a higher slope inclination. In total we revealed a median of 36 species per plot. Species-richness was highest for the *Cynosurus cristatus*- and *Briza media*-cluster, which differed significantly from the *Seseli transcaucasica*- and *Parnassia palustris*-cluster. The clusters differed significantly in 21 parameters (Table 3.2). A complete species list and environmental variables are shown in the Appendix (Electronic Supplementary Material).

Table 3.1 Direction cosines of the environmental vectors to the first and second axis of NMDS ordination and corresponding squared correlation coefficient (r^2)

	NMDS1	NMDS2	r^2	
Altitude	-0.68	-0.73	0.66	***
Slope	0.66	-0.75	0.52	***
Cover vegetation	-0.87	0.49	0.76	***
Cover soil	0.98	-0.20	0.58	***
Cover rock	0.80	-0.60	0.73	***
Cover moss	1.00	-0.02	0.21	***
Height forbs	-0.71	-0.70	0.29	***
Height graminoids	-0.91	-0.42	0.18	***
Content stone	1.00	-0.09	0.51	***
pH	1.00	-0.04	0.75	***
$CaCO_3$	0.97	-0.24	0.45	***
P_2O_5	-0.99	-0.13	0.49	***
K_2O	-0.89	0.45	0.25	***
Mg	-0.89	0.45	0.55	***
$N_{(total)}$	-0.96	0.30	0.59	***
$C_{(total)}$	-0.92	0.40	0.03	
$C_{(org)}$	-0.97	0.26	0.60	***
C/N	-0.65	0.76	0.11	**
Northness	-0.73	-0.68	0.07	
Eastness	0.60	0.80	0.04	
Shannon diversity	-0.13	0.99	0.21	***
Species richness	-0.14	0.99	0.17	***
Evenness	-0.01	1.00	0.10	**

* $p < 0.05$; ** $p < 0.01$; *** $P < 0.001$

The Indicator species analysis (ISA) identified 78 species as indicators of the four clusters. 29 species were assigned to the *Seseli transcaucasica*-cluster, 29 species to the *Cynosurus cristatus*-cluster, 9 species to the *Briza media*-cluster and 11 species to the *Parnassia palustris*-cluster. Species that occurred in the ISA with a relative frequency of ≥ 50% were used to describe our clusters (Table 3.3), as follows: the *Seseli transcaucasica*-cluster is characterized by species of the subalpine tall herb vegetation such as *Astrantia maxima*, *Betonica macrantha*, *Cephalaria gigantea*, *Geranium ibericum*, *Hypericum bupleuroides*, *Inula orientalis*, *Seseli transcaucasica*, and by ‘‘real’’ grassland species such as *Anthoxanthum* *odoratum*, *Bromus variegatus*, and *Dactylorhiza euxina*. Further, *Cirsium obvallatum*, *Rumex obtusifolius*, and *Veratrum lobelianum* are ruderal species indicative of overgrazed grassland. *Cynosurus cristatus*-cluster, the nutrient-rich high-montane pasture, is dominated by *Cynosurus cristatus* and further characterized by *Alchemilla caucasica*, *Carex caryophyllea*, *Plantago lanceolata*, *Prunella vulgaris*, and *Trifolium repens*. The nutrient-poor *Briza media*-cluster is characterized by *Anthyllis variegata*, *Briza media*, *Leucanthemum vulgare*, *Linum catharticum*, *Polygala transcaucasica*, and *Thymus collinus*. Typical for the scree sites of the *Parnassia palustris*-cluster are the pioneer species *Campanula alliarifolia*, *Lactuca racemosa*, *Parnassia palustris*, *Trisetum rigidum*, and *Tussilago farfara* which are characterized by a rapid establishment due to their abundant, anemochorious seed dispersal with further spread through water runoff.

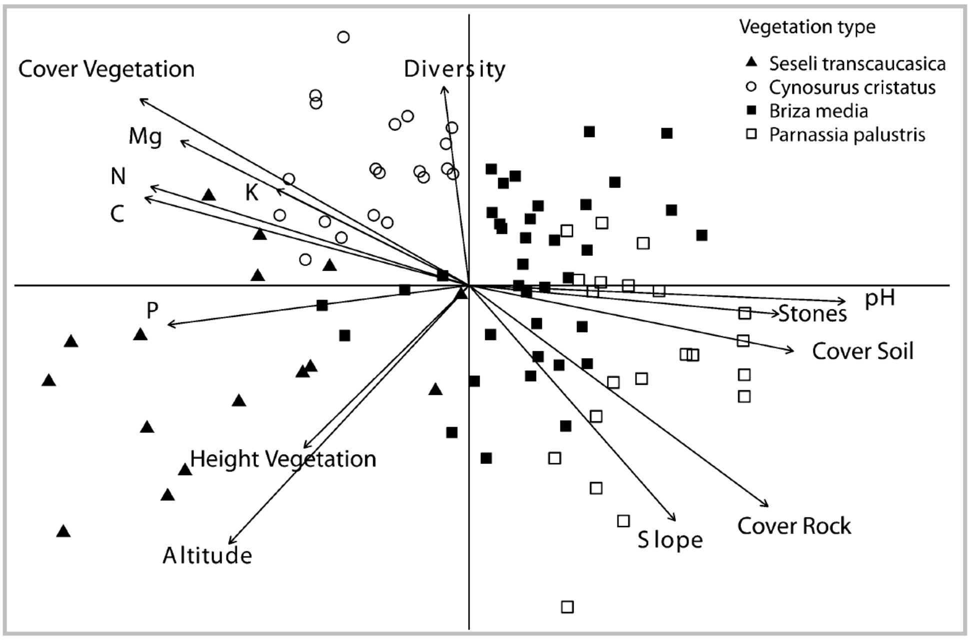

Figure 3.3 NMDS ordination of 94 grassland plots and 127 species, displaying the first and second axis. The stress level revealed at 13.6, environmental variables are indicated by arrows. Abbreviations: *C* Organic carbon, *Diversity* Shannon diversity, *Mg* magnesium, *N* total nitrogen, *P* phosphorus, stones stone content, *K* potassium, *height vegetation* max. height herbs

Species that occurred within the densely vegetated plots (*Seseli transcaucasica*- and *Cynosurus cristatus*-cluster) at a relative frequency of ≥ 50 % were *Anthoxanthum odoratum*, *Campanula glomerata* ssp. *caucasica*, *Centaurea salicifolia*, and *Luzula stenophylla*. Species that we found within the high-montane plots (*Cynosurus cristatus*-, *Briza media*-, and

Parnassia palustris-cluster) at a relative frequency of ≥ 50% were *Cruciata laevipes*, *Euphrasia* sp., *Leucanthemum vulgare*, *Linum catharticum*, *Phleum pretense*, *Prunella vulgaris*, *Trifolium pretense*, and *Thymus collinus*. Species that displayed a relative frequency of ≥ 50% within the nutrient-poor plots (*Briza media*- and *Parnassia palustris*-cluster) were *Anthyllis variegata*, *Briza media*, *Campanula alliariifolia*, *Parnassia palustris*, *Polygala transcaucasica*, and *Salvia verticillata*. Common grassland species which occurred within all plots at a relative frequency of ≥ 50% were *Agrostis vinealis*, *Festuca pratensis*, *Leontodon hispidus*, *Lotus corniculatus*, *Pimpinella rhodanta*, *Plantago lanceolata*, *Ranunculus caucasicus*, and *Veronica gentianoides*.

Table 3.2 Median values and quartiles of environmental variables from vegetation type clusters

Vegetation cluster	***Seseli transcaucasica* (n = 16)**	***Cynosurus cristatus* (n = 20)**	***Briza media* (n = 37)**	***Parnassia palustris* (n = 21)**
Altitude (m a.s.l.) *	1,989.5 ± 111.5 [a]	1,747.0 ± 266.5 [b,c]	1,689.0 ± 258.0 [b]	1,640.0 ± 252.0 [c]
Slope (°)*	22.5 ± 15.3 [a]	17.5 ± 7.5 [a]	37.0 ± 10.0 [b]	37.0 ± 5.0 [b]
Cover vegetation (%) *	100 ± 5 [a]	93.5 ± 6.5 [b]	60 ± 40 [c]	28 ± 31 [d]
Cover soil (%) *	0 ± 5 [a]	5 ± 3 [a]	15 ± 10 [b]	18 ± 7 [b]
Cover rock (%) *	0 ± 0 [a]	1 ± 5 [a]	15 ± 23 [b]	40 ± 17 [c]
Cover moss (%) *	0 ± 0 [a]	1 ± 2 [b]	5 ± 4 [b,c]	5 ± 9 [c]
Height forbs (cm)*	69.2 ± 18.3 [a]	28.3 ± 18.8 [b]	35.0 ± 15.0 [b]	31.7 ± 21.7 [b]
Height graminoids (cm)*	76.7 ± 8.3 [a]	39.2 ± 23.2 [b]	41.7 ± 20.0 [b]	45.0 ± 33.3 [c]
Content stone (%)*	4.1 ± 2.0 [a]	11.4 ± 8.8 [b]	19.3 ± 17.1 [b]	31.3 ± 13.8 [c]
pH ($CaCl_2$)*	4.49 ± 0.77 [a]	5.47 ± 2.17 [a]	7.11 ± 0.31 [b]	7.35 ± 0.39 [c]
$CaCO_3$ (%) *	0.00 ± 0.00 [a]	0.14 ± 3.58 [b]	13.40 ± 23.73 [c]	32.46 ± 25.82 [d]
P_2O_5 (mg/100g) *	3.04 ± 2.15 [a]	1.60 ± 1.91 [a]	0.42 ± 0.60 [b]	0.20 ± 0.38 [b]
K_2O (mg/100g) *	11.13 ± 3.40 [a]	7.88 ± 4.89 [a,b]	7.73 ± 5.36 [b]	5.56 ± 2.49 [c]
Mg (mg/100g) *	9.06 ± 3.46 [a]	9.23 ± 5.87 [a]	3.99 ± 1.80 [b]	2.94 ± 0.99 [b]
$N_{(total)}$ (%) *	0.74 ± 0.17 [a]	0.52 ± 0.16 [b]	0.34 ± 0.21 [c]	0.19 ± 0.09 [d]
$C_{(total)}$ (%) *	7.08 ± 1.38 [a,c]	4.98 ± 1.91 [b]	5.21 ± 2.66 [b,c]	5.44 ± 2.42 [b,c]
$C_{(org)}$ (%) *	6.83 ± 1.63 [a]	4.83 ± 1.64 [b]	3.12 ± 2.36 [c]	1.53 ± 0.68 [d]
C/N (%) *	9.41 ± 0.56 [a]	9.05 ± 1.07 [a]	8.91 ± 1.69 [a]	7.68 ± 1.31 [b]
Northness	0.53 ± 0.47 [a]	0.53 ± 0.14 [a]	0.53 ± 0.14 [a]	0.67 ± 0.47 [a]
Eastness	0.85 ± 0.85 [a]	0.85 ± 0.11 [a]	0.85 ± 0.11 [a]	0.75 ± 0.85 [a]
Shannon *	3.00 ± 0.21 [a]	3.15 ± 0.16 [b]	3.18 ± 0.31 [b]	2.95 ± 0.36 [a]
Richness *	34 ± 4 [a]	38 ± 7 [b]	39 ± 13 [b]	32 ± 9 [a]
Evenness *	0.84 ± 0.04 [a]	0.87 ± 0.02 [a,b]	0.87 ± 0.02 [b]	0.84 ± 0.04 [a,b]

[a, b, c, d] indicate significant variable differences for the Wilcoxon rank sum test of a posthoc cluster comparison using Bonferroni correction method ($p < 0.05$). * indicates significant differences between clusters from Kruskal–Wallis comparison ($p < 0.05$)

Table 3.3 Indicator species analysis of vegetation types ordered by relative frequency

Species name \ Cluster	I (n=16)	II (n=20)	III (n=37)	IV (n=21)	IndVAL
Agrostis vinealis Schreb.	88	100	70	62	44 **
***Seseli transcaucasica* (Schischk.) M.Hiroe**	**81**	10	5	0	77 ***
***Betonica macrantha* C. Koch.**	**75**	0	3	0	75 ***
***Astrantia maxima* Pall.**	**75**	5	3	0	75 ***
***Anthoxantum odoratum* L.**	**75**	60	16	0	43 ***
***Dactylorhiza euxina* (Nevski) Czerep.**	**75**	25	22	19	42 ***
***Cirsium obvallatum* (M.Bieb.) M.Bieb.**	**69**	20	22	10	64 ***
Trifolium caucasicum Tausch	69	40	46	10	48 **
Centaurea salicifolia M.Bieb.	69	50	24	29	41 **
Carex umbrosa ssp. *huetiana* (Boiss.) Soó	69	35	19	5	29 **
***Cephalaria gigantea* (Ledeb.) Bobrov**	**63**	5	19	10	54 ***
***Rumex obtusifolius* L.**	**63**	5	0	0	51 ***
***Hypericum bupleuroides* Griseb.**	**63**	10	14	5	46 ***
***Bromus variegatus* M.Bieb.**	**63**	45	35	29	40 ***
***Veratrum lobelianum* Bernh.**	**56**	0	0	0	56 ***
***Inula orientalis* Lam.**	**50**	0	5	0	50 ***
***Geranium ibericum* Cav.**	**50**	15	3	0	49 ***
***Cynosurus cristatus* L.**	38	**100**	46	14	76 ***
***Trifolium repens* L.**	81	**100**	57	24	52 ***
***Festuca pratensis* Huds.**	56	**100**	70	71	49 ***
***Leontodon hispidus* L.**	81	**100**	95	100	38 ***
***Carex caryophyllea* Latourr.**	6	**95**	54	19	55 ***
***Plantago lanceolata* L.**	56	**95**	97	81	55 ***
***Alchemilla caucasica* Buser**	50	**95**	89	43	46 ***
Prunella vulgaris L.	31	95	84	71	41 **
Pimpinella rhodantha Boiss.	100	95	89	86	39 *
Gentiana septemfida Pall.	50	95	54	33	36 *
***Cerastium fontanum* ssp. *vulgare* (Hartm.) Greuter & Burdet**	38	**90**	27	14	57 ***
***Phleum pratense* L.**	31	**85**	59	52	49 ***
Trifolium pratense L.	25	**85**	62	57	44 **
Euphrasia sp.	0	**85**	62	90	37 ***
Poa alpina L.	25	**75**	70	48	36 **
***Achillea millefolium* L.**	6	**70**	43	29	48 ***
Festuca ovina L.	25	**70**	46	14	36 **
Phleum alpinum L.	50	55	14	0	25 *
***Nardus stricta* L.**	13	50	3	0	49 ***
Medicago lupulina L.	0	50	70	48	32 *
Luzula stenophylla Steud.	69	50	19	0	32 **
***Thymus collinus* M.Bieb.**	19	70	**97**	86	52 ***
***Leucanthemum vulgare* (Vaill.) Lam.**	6	90	**95**	86	40 ***
Linum catharticum L.	19	70	92	100	40 **
***Polygala transcaucasica* Tamamsch.**	6	45	**84**	86	40 ***
Briza media L.	25	45	81	76	39 **
***Anthyllis variegata* Grossh.**	0	15	**59**	52	45 ***
Trisetum rigidum (M.Bieb.) Roem. & Schult.	56	45	84	100	37 **
***Parnassia palustris* L.**	6	10	49	**86**	48 ***
***Campanula alliariifolia* Willd.**	0	0	68	**81**	47 ***
Lactuca racemosa Willd.	13	0	41	76	40 **
Alchemilla cf. laeta Juz.	38	0	16	71	28 *
***Tussilago farfara* L.**	0	0	19	**57**	53 ***

Table 3.3 continued

Species name \ Cluster	I (n=16)	II (n=20)	III (n=37)	IV (n=21)	IndVAL
No group assigned, non-significant indicator values					
Ranunculus caucasicus M.Bieb.	100	100	97	90	31
Lotus corniculatus L.	75	90	97	100	29
Veronica gentianoides Vahl	56	95	76	43	32
Rhinanthus minor L.	88	35	54	29	28
Campanula glomerata ssp. *caucasica* (Trautv.) Ogan.	50	60	32	5	22
Cruciata laevipes Opiz	44	50	76	62	26
Salvia verticillata L.	13	30	57	62	22
Origanum vulgare L.	25	40	43	57	18

Species with a relative frequency of ≥ 50% (grey shading) and highly significant indicator values (***) are shown as indicator species (bold). Boxes indicate cluster assignment of indicator species, cluster I: *Seseli transcaucasica*, cluster II: *Cynosurus cristatus*, cluster III: *Briza media*, cluster IV: *Parnassia palustris.* * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

3.3.2 Functional plant groups

The relative abundance and species-richness of functional plant groups were calculated for each of the vegetation cover classes (Figure 3.4). Due to a contribution of high-montane and subalpine vegetation to the plant composition within the densely vegetated class, we separately visualized the *Seseli transcaucasica*-cluster. The herbs' contribution to the overall abundance increases with decreasing vegetation cover from 47% on dense sites of the high-montane zone up to 62% on sparely vegetated plots. Further, the proportion of graminoid abundance decreases from 35% on dense sites to 22% on lightly vegetated sites, whereas the relative abundance of the *Fabaceae* group remains stable at 15–16% throughout the same vegetation cover ranges. Sparely vegetated sites display a different trend for *Fabaceae* (10% abundance) and graminoids (26% abundance). Woody species (*Betula litwinowii*, *Daphne glomerata*, *Fagus orientalis*, *Prunus cerasifera*, *Pyrus caucasica*, *Salix caprea*, *Salix kazbekensis*, *Sorbus aucuparia*), orchids and ruderal pasture weeds contribute each to less than 1% abundance for the high-montane zone. However, for the subalpine *Seseli transcaucasica*-cluster, ruderal pasture weeds display about 10% relative abundance. Further, abundance of the *Seseli transcaucasica*-cluster is comprised by the overall lowest abundance value for the *Fabaceae* group (9%) in addition to herb and graminoid abundance which is similar to the densely vegetated high-montane sites.

Generally, herbs contribute most to overall species-richness which indicates the abandonment of former grassland management (haymaking). The herb species-richness ranges from 57% for the densely vegetated sites of the high-montane zone up to 61% at the sparely vegetated sites and densely vegetated *Seseli transcaucasica*-cluster. Graminoids contribute to the second largest proportion of species-richness of 19–21% for open vegetation cover and 25% for the densely vegetated sites. *Fabaceae* contribute 8% of species-richness at the sparely and lightly vegetated sites and display the lowest value of 5% on the densely vegetated *Seseli transcaucasica*-cluster sites. *Orchidaceae* show the lowest contribution to the species-richness on sparely vegetated sites (3%), whereas the other vegetation cover classes display a relative species-richness of 5 and 4% for the *Seseli transcaucasica*-cluster. The highest shares of woody species to species-richness show the lightly (8%) and sparely vegetated sites (6%). Woody species (*Sorbus aucuparia*) contribute only 1% to the relative species-richness on the densely vegetated *Seseli transcaucasica*-cluster sites. However, the *Seseli transcaucasica*-cluster displays the highest proportion of ruderal pasture weeds (4%). Whereas the plant group of ruderal pasture weeds is absent on the sparely vegetated sites.

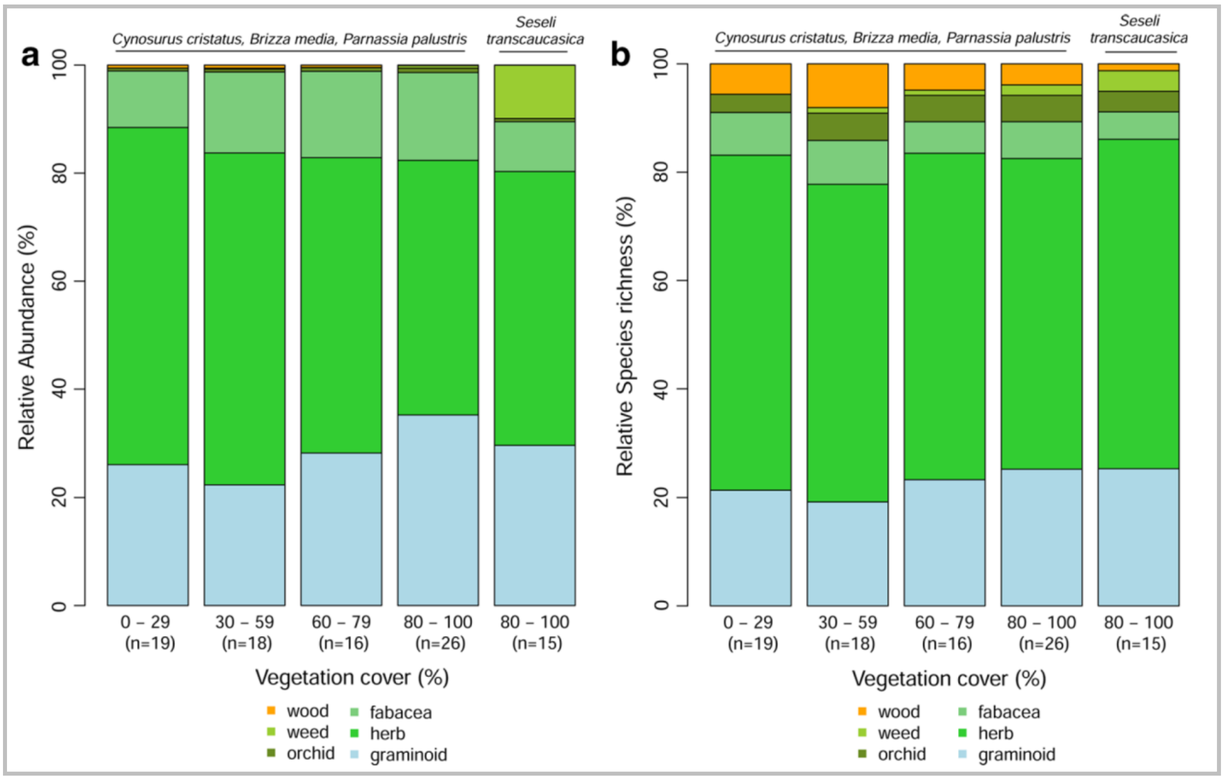

Figure 3.4 Relative abundance (a) and species-richness (b) within varying vegetation cover

3.4 Discussion

Our study has examined the vegetation which develops under characteristic site conditions of overgrazed and eroded sites in order to establish conservation measures for mountain grassland within the Caucasus biodiversity hotspot.

3.4.1 Site condition and vegetation types

Our ordination results indicate the changeover of species composition and site conditions of the subalpine and high-montane zone which is in accordance with the well-known altitudinal gradient of mountain vegetation (Ozenda 1988). In our data set, the altitudinal gradient can be further seen in an overlap of shared species within the nutrient-rich sites of the subalpine and high-montane zone. Nevertheless, the distinct position for most of the *Seseli transcaucasica*-cluster sites is further confirmed by the ISA that shows most of the subalpine species at highest frequency. Besides altitudinal influences, changes in land use have further affected the subalpine sites in our research area (Tephnadze et al. 2014). The laborious haymaking on remote slopes of the subalpine zone was replaced by hay production at more favorable locations near the villages which were former used as arable fields. Indicators for hay production in the subalpine belt are *Agrostis vinealis*, *Bromus variegatus*, and to a smaller extent *Hordeum brevisubulatum* which are typical hay meadow species of the Kazbegi region (Tephnadze et al. 2014; Magiera et al. 2013). Ruderal species such as *Cirsium obvallatum* and *Veratrum lobelianum* are zoochorously dispersed by grazing animals and are therefore characteristic for grazed habitats. Callaway et al. (2000) discovered so-called safe sites in the surrounding of these species where certain species (e.g. *Trifolium ambiguum* and grasses) benefit from the protection of herbivory. These safe sites on overgrazed grassland are characterized by high plant diversity whereas the subalpine grassland of our study region is dominated by highly competitive tall herbs such as *Seseli transcaucasica*, *Betonica macrantha*, *Astrantia maxima*, or *Cephalaria gigantea* which infest the abandoned hay meadows. However, we found a single specimen of *Traunsteinera sphaerica*, a considerably rare orchid species (Akhalkatsi et al. 2003), within the *Seseli transcaucasica*-cluster. Further, the subalpine meadows of the Kazbegi region have been described to substantially contribute to the phytodiversity on a landscape level (Tephnadze et al. 2014). Therefore future development of the productive subalpine grassland will be of interest as the low occurrence of meadow species indicates the loss of a valuable subalpine hay meadow habitat.

Our class comparison of environmental variables illustrates the changing site conditions along the slope of the high montane zone. Although we found a changing high-montane vegetation composition along the slope, these changes most likely contributed to a gradient of land use intensity. Tasser et al. (2003) identified effects related to a change in land use as an additional factor contributing to topographical factors causing the loss of vegetation cover and erosion events. We assume a higher grazing pressure within the slopes nearby the villages as it is known from former Soviet Union countries (Food and Agriculture Organization of the United Nations 2003). Furthermore, the altitudinal differences of vegetation types correspond to the distance from settlements and therefore the site conditions of the lower slope display lower vegetation cover. The location of erosion events on the lower part of the mountain slope was also found on steep overgrazed slopes of the Alps (Dommermuth 1995). In the high-montane zone, we observed the *Cynosurus cristatus* grassland vegetation which develops under less inclined, dense site conditions of the upper slope. This vegetation type displays a high vegetation cover density and high species-richness. As we further approach downslope, the *Briza media*-cluster indicates a transition towards the *Parnassia palustris*-cluster. As the slope ascents we found less vegetation cover on *Briza media* grassland and the lowest coverages on the lower slope at *Parnassia palustris*-cluster vegetation. Although the site conditions of slope inclination, stone content and phosphorus content are in accordance with the sparely vegetated sites, a higher vegetation cover and a lower rock cover distinguish the unique features of the *Briza media*-cluster from the *Parnassia palustris*-cluster. Due to its low nutrient values and high species-richness, the *Briza media*-cluster of our study area has to be recognized as a habitat as valuable as poor grassland of other mountain regions. This is further confirmed by the occurrence of one native *Gladiolus* population, a genius which has become particularly endangered in the European Alps. The *Parnassia palustris*-cluster sites carry the first stage of vegetation succession on habitats of scree. Other than the species-rich habitats of rock and scree of the Kazbegi region which are considered as valuable habitats for Georgia (Nakhutsrishvili et al. 2006), our described scree habitats are neither of any danger nor of conservation value. Considering the long-lasting period of soil formation in mountainous regions, the long-term loss of diverse grassland and the development towards habitats of no conservation value has to be expected once the vegetation cover is removed.

Seseli transcaucasica- and *Cynosurus cristatus*-cluster display acidic soil conditions whereas the high stone content on *Briza media*- and *Parnassia palustris*-cluster creates basic substrates. The content of calcium carbonate content indicates a mixture of calcareous and alkaline parent rock material, as it has been described for the study area (Gobejishvili et al. 2011; Khetskhoveli et al. 1975). Soil phosphorus content is generally very low, whereas potassium and magnesium values seem to be sufficient compared to grassland nutrient demands of managed European grassland (VDLUFA 1997, 1999). The strong decline of nutrients from vegetated to open sites that we observed was also reported from sites of the Alps (Florineth et al. 2002; Zöttl 1952). Nutrient values of our sparely vegetated study sites are comparable to site conditions which were recorded on a comparable study site in Italy (Florineth et al. 2002). However, phosphorus content within poor grassland (*Briza media*-cluster) and scree sites (*Parnassia palustris*-cluster) was several times lower compared to the Alps and similar low values were observed on study sites within the nearby Kazbegi region (Tephnadze et al. 2014). Additional factors such as the upper soil horizon thickness and fractions of coarse soil material where not incorporated into our soil analyses and might further influence the nutrient availability for plants.

3.4.2 Functional plant groups

Differences between the functional plant groups can be explained by the varying structural conditions of different vegetation cover classes. Graminoids play an important role to benefit the density of vegetation structure on densely vegetated sites where they are responsible for the higher abundance and species-richness. Accordingly, a sparely vegetated structure enables the establishment of higher herb abundance. The *Fabaceae* is a light-demanding plant group which therefore seems to be most affected by the structure and site conditions of sparely vegetated sites and densely vegetated *Seseli transcaucasica* cluster sites. *Fabaceae* play an important role in the revegetation of sparely vegetated sites due to their nitrogen-fixing capabilities (De Deyn et al. 2011). Due to their low competitiveness their relative abundance needs to be increased when considering restoration efforts. The decrease in the number of species of *Orchidaceae* on vegetation cover < 30% indicates the decline of a species group of high conservation value once the soil layer has been completely eroded. These outcomes support the demand for the protection of valuable and diverse grassland habitats in Georgia.

The occurrence of ruderal pasture weeds is largely restricted to the *Seseli transcaucasica*-cluster. We conclude for our study region that the high abundance of ruderal pasture weeds indicates a high competiveness to tall herb vegetation. The open vegetation structure of sparely vegetated sites seems to enable more shrub and tree species to regenerate. Further, we assume that a high grazing pressure prevents further reforestation as the abundance of woody species is generally very low. The occurrence of woody species (*Sorbus aucuparia*) in the subalpine zone is a remnant from the original forest which indicates the potential for reforestation in the subalpine belt. However, the distance to mature trees influences the process of seedling recruitment and was not tested within our study.

3.4.3 Potential plant species for ecosystem restoration

Due to the tolerance to varying site conditions of plant species which we could find within all vegetation types of the high-montane zone, we assume these species to be suitable for grassland restoration in the Greater Caucasus. Although some single species may possess the capability to quickly restore vegetation cover, the necessity to restore species-rich grassland for erosion control has been reported (Martin et al. 2010; Pohl et al. 2009). From the plant species' characteristics to grow on nutrient-poor sites, we suggest grasses which form the matrix of grassland such as *Agrostis vinealis*, *Briza media*, *Festuca pratensis*, and *Poa alpina*. Additionally *Festuca ovina* might be suitable for revegetation, as the closely related *Festuca valesiaca* has been found to secure erosion edges in subalpine grassland of the Georgian Caucasus (Caprez et al. 2011). Moreover we suggest the following herb species which also grow under poor soil conditions: *Anthyllis variegata*, *Campanula alliariifolia*, *Leontodon hispidus*, *Leucanthemum vulgare*, *Lotus corniculatus*, *Medicago lupulina*, *Pimpinella rhodantha*, *Polygala transcaucasica*, *Ranunculus caucasicus*, *Salvia verticillata* and *Trifolium pratense*. However, due to small biomass we assume that *Parnassia palustris*, and *Veronica gentianoides* can be neglected for revegetation. In comparison to other species suggestions for restoration of mountain grassland (Krautzer et al. 2004), we further assume that *Achillea millefolium*, *Koeleria luerssenii*, *Phleum alpinum*, and *Trifolium repens* might be appropriate species. Further, *Trifolium ambiguum*, a native *Fabaceae* species of the Georgian mountain region, is globally used for grazing in permanent pastures and forms dense structures once established (Cuomo et al. 2003). Although *Festuca varia* plays an important role for slope protection in the Alps we do not recommend this species for the Caucasus region as it has the potential to replace grassland communities under intensive grazing (Nakhutsrishvili 1999). We further identified *Trisetum rigidum* as being able to take the ecological position of pioneer on high-montane scree habitats as it is known for *Trisetum distichophyllum* in the subalpine region of the European Alps. The seed production and suitability of the proposed species for restoration measures in the Caucasus region should be further tested in field studies.

3.5 Conclusion for grassland management and conservation

We found four major vegetation types which have developed under the site conditions of overgrazing and erosion in the upper Aragvi valley. Our results indicate the loss of species-rich high-montane pastures once the vegetation cover is damaged. This is further confirmed by Tasser et al. (2003) who described larger erosion events on steep slopes after damage being done to the vegetation cover. Furthermore, Klug et al. (2002) reported little revegetation potential after the vegetation cover has been trampled. Due to the observed low individual numbers on sparely vegetated sites, we assume also a low potential for the natural succession through seed production on these sites. Therefore, indigenous, site specific seed material has to be produced to ensure restoration success. We strongly recommend applying our suggested native plant species and neglecting exotic species [e.g. *Chrysopogon zizanoides*] for revegetation. Therefore, for the *Cynosurus cristatus* pastures on steep slopes nearby the settlements we suggest focusing on early grassland revegetation once the vegetation cover gets damaged. Immediate seeding and grazing enclosures on vegetation damage spots are essential for grassland restoration success and slope protection (Krautzer and Wittmann 2006). Due to the high costs of large scale restoration methods we suggest putting emphasis on restoring recent small scale damages of pastures and recreational areas for the region of interest and other transition countries of the Caucasus region to stop land degradation and further habitat loss. The poor grassland vegetation of our study area showed the highest diversity and proved to also be habitat of specialist plants of high conservation value. Therefore the loss of these valuable stands needs to be halted. Further, these sites of *Briza media*-cluster displayed a median vegetation cover of 60% which should be the minimum goal for restoration management since Moismann (1984) proposed 70% vegetation cover to secure mountain slopes.

Due to site conditions, the highly productive *Seseli transcaucasica* grassland seems to be most suitable for grazing or haymaking. However, the high infestation with ruderal pasture weeds reduces the value as animal fodder. Therefore, strategies to reduce ruderal pasture weed abundance and to maintain the subalpine vegetation for animal use would be highly advisable. Further, on the more favorable, less inclined slopes of the subalpine zone cutting and haymaking should be again intensified towards the re-establishment of diverse *Hordeum* and *Bromopsis* community meadows in order to conserve habitat diversity. Overall, we suggest establishing an appropriate management plan on collective land of Caucasian grassland as it has been proposed for other countries which experienced collectivization (Food and Agriculture Organization of the United Nations 2003).

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Chapter 4:

Remotely-sensed assessment of mountain grassland degradation by grassland type and coverage

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Abstract

Vegetation cover has often been used as an indicator to detect grassland degradation in studies relying on remote sensing technologies. However, grassland degradation comprises not only the loss of vegetation cover but also the loss of phytodiversity, productivity, and a shift in vegetation composition. Due to the strong relationship between environmental conditions and species composition within mountain grassland a combination of environmental variables may improve the early detection of mountain grassland degradation. In our study region, the Caucasus, Georgia overgrazing and erosion events have degraded large grassland areas and methods to evaluate grassland condition on a large scale are needed. The aim of our study was to test the combination of vegetation cover and vegetation types to detect mountain grassland degradation with multispectral sensors of high spatial resolution. We combined vegetation cover and vegetation types as indicators for the detection of grassland degradation from remote sensing. Therefore, we used a hand-held field spectrometer to simulate the multispectral World View 2 (WV2) sensor in order to calculate multispectral vegetation indices. Further, we classified 139 grassland vegetation subplots according to species composition into 3 vegetation types with Ward clustering. We used random forest regression models to predict vegetation cover by WV2 wavebands and vegetation indices. Additionally, we included environmental variables into a random forest classification to predict the vegetation types. Finally, we combined the predicted vegetation cover and vegetation types to classify degraded sites. Therefore we set a threshold value for the vegetation cover of each vegetation type. We used NMDS ordination to display vegetation composition, vegetation types, environmental variables and predicted grassland degradation. From an overall accuracy of 75%, we assume that the combination of vegetation cover and vegetation types is a promising tool for the evaluation of grassland condition by multispectral remote sensing.

Keywords

Georgia; NMDS; random forest; remote sensing; overgrazing; Caucasus

4.1 Introduction

Grassland degradation comprises a loss of vegetation cover, plant diversity, productivity and changing vegetation composition. These interrelated processes lead to a strong modification of landscape patterns and services provided from grassland ecosystems. In particular, grassland degradation has a strong destabilizing consequence on mountain slopes where a dense vegetation cover with a diverse root system is an essential prerequisite for erosion control (Martin et al. 2010; Pohl et al. 2009). Though erosion events are a natural phenomenon within mountain landscapes, cultural, social and economic causes are contributing to the complex reasons for erosion (Holzner and Kriechbaum 2001) which are further accelerating grassland degradation processes. For instance, changes in grazing systems have contributed to overgrazing followed by grassland degradation in many former Soviet Union states and Asian countries (Food and Agriculture Organization of the United Nations 2003). In the democratic republic of Georgia, a former member of the Soviet Union, the past farming collectivization had long lasting impacts to the grassland ecosystem. Nowadays, the previously overgrazed pastures on less inclined plateaus are highly infested with ruderal pasture weeds (Callaway et al. 2000) and on steep slopes overgrazing resulted in heavily eroded sites. As the local mountain population strongly relies on the productivity of healthy grassland ecosystems, methods to detect degradation are needed in order to apply a sustainable land use management.

To detect the grassland condition of large mountain terrains, remote sensing (RS) approaches are particularly beneficial and highly time-saving compared to fieldwork assessments. Grassland degradation was successfully detected from fractions of vegetation cover with high resolution satellite images (Lehnert et al. 2015; Wiesmair et al. 2016). Furthermore, biomass loss or slope have been widely used as indicators to estimate grassland degradation (e.g. Pickup and Chewings 1996). However, Liu et al. (2015) suggested that vegetation cover may fail to assess grassland degradation. Particularly, unpalatable weeds which infest and degrade grassland ecosystems contribute to a high vegetation cover (Vallentine 2001). For the Georgian Greater Caucasus, Wiesmair et al. (2016b) described high vegetation cover and variance in slope inclination for subalpine grassland which was highly infested by ruderal pasture weeds such as *Veratrum lobelianum*. Furthermore, they described poor grassland which naturally occurs on sites of high rock and soil cover even if not degraded. Therefore the indication of grassland degradation from the single values of vegetation cover, slope inclination or vegetation type seems to be insufficient. Correspondingly, the combination of indicators, such as cover, biomass, proportion of edible plants and plant height, into a regional grassland degradation index has been presented for Asian grassland as a powerful tool to overcome such uncertainties and to further distinguish the grassland for degraded and non-degraded sites (Wen et al. 2010). Such an approach seems to be more precise as it is additionally including information of certain indicator species such as ruderal pasture weeds. However, the degradation intensity from vegetation cover has not yet been further differentiated for individual grassland types. To evaluate the grassland degradation processes of the Caucasus and other affected regions, more precise, regional information on the grassland status is needed. To increase the information of RS data, the good separability of hyperspectral reflectance has already been used to distinguish mountain grassland types (Magiera et al. 2013), grassland successional stages (Möckel et al. 2014), single grassland indicator plant species (Liu et al. 2006; Wang et al. 2010) and degrading Asian mountain grassland types (Liu et al. 2015). However, due to the current coarse spatial resolution of hyperspectral imagery and the absence of flight campaigns with hyperspectral sensors the practical utilization of such data is limited in the Greater Caucasus and other remote, high-mountain regions. A high spatial resolution is particularly important for the heterogeneous mountain terrain where variations occur within meters (Asner and Lobell 2000). Therefore, multispectral sensors which record reflectance at a very high spatial resolution have gained importance for RS assessments. Particularly, the implementation of several multispectral indices has been found beneficial for further discriminations of grassland degradation from vegetation cover at high altitude (Lehnert et al. 2015; Liu et al. 2015). As a substitute for airborne hyperspectral sensors, portable spectrometers offer the possibility to record spectral data with a similar spectral coverage to test their applicability for a desired research question. The aim of our study was to test the combination of vegetation cover and vegetation types to detect mountain grassland degradation with multispectral sensors of high spatial resolution. Therefore we used a hand-held field spectrometer to simulate the multispectral World View 2 (WV2) sensor which provides 8 spectral bands at a high spatial resolution of 1 m. In a landscape which is frequented by overgrazing, erosion and mass wasting events, we addressed the following research questions:

To which extent can spectral and environmental variables predict vegetation cover and grassland types and which predictor variables are most important?

Can grassland degradation, represented by grassland types and coverage be detected in multispectral data?

4.2. Methods

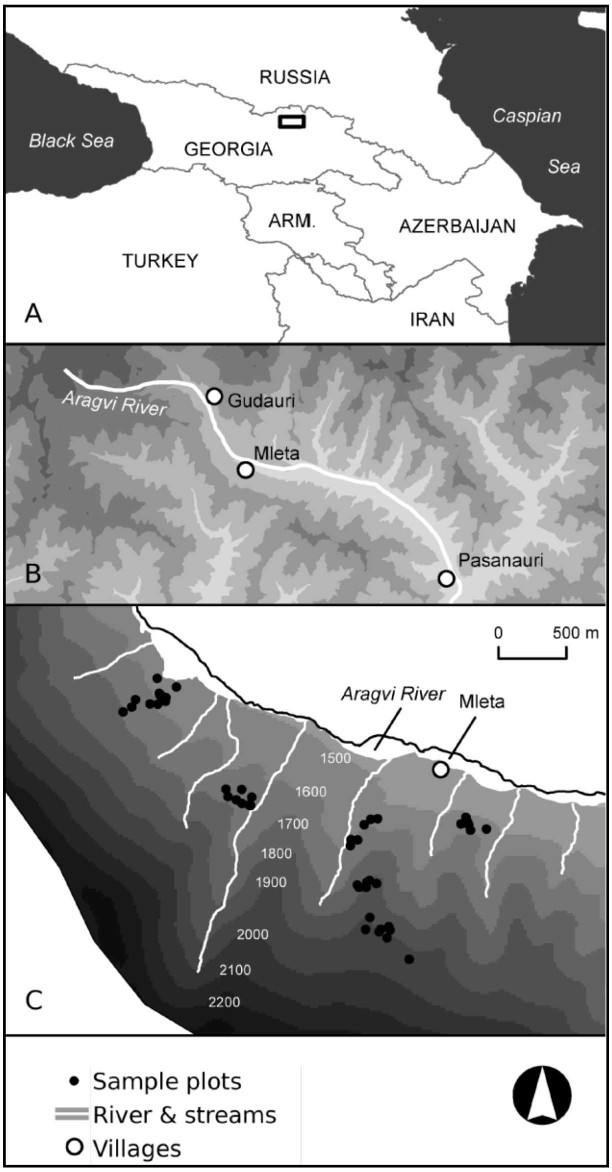
4.2.1 Study area

The study was conducted in the upper Aragvi valley in the vicinity of the village Mleta (42°25'40"N, 44°30'23"E, 1450 m above sea level [m a.s.l.]) in the Greater Caucasus in Georgia. The climate station at the valley bottom of the upper Aragvi valley (Pasanauri) displays a mean annual temperature of 8.2°C and a mean annual precipitation of 1011 mm. January, the coldest month, has a mean temperature of -3.3°C and 50 mm mean precipitation. The hottest month, July, is characterized by a mean temperature of 18.9°C and a mean precipitation of 103 mm. Climate data was contributed by the National Environmental Agency and modified by Ina Keggenhoff.

The upper Aragvi valley is built up from Andesite-Basalt in alternation with clay-shale, shale marls and enclosures of limestone and sandstone (Gobejishvili et al. 2011; Khetskhoveli et al. 1975). Close to Mleta, the upper Aragvi valley is asymmetrically shaped. The slightly inclined, north facing slope is covered by loose sediment which is prone to erosion and mudflows (Lichtenegger et al. 2006). Within the Aragvi valley, the grassland includes soil types of Leptosols, Cambisols and Cryosols.

The slopes next to Mleta range from the river bed at approximately 1500 m a.s.l. up to the mountain ridges at about 2200 m a.s.l. The north facing slopes are characterized by beech forests (*Fagus orientalis*), large erosion gullies and pastures which are mainly grazed by cattle and to a minor extent by sheep and horses. Uncontrolled cattle grazing can be observed close to the settlements (Wiesmair et al. 2016). High-montane grassland of our study site is characterized by strong gradients from densely vegetated pastures towards poor grassland and eroded sites of little vegetation cover. Pastures are characterized by *Cynosurus cristatus*, *Alchemilla caucasica*, *Carex caryophyllea*, *Plantago lanceolata*, *Prunella vulgaris* and *Trifolium repens*. Poor grassland is characterized by *Briza media*, *Anthyllis* *variegata*, *Leucanthemum vulgare*, *Linum cartharticum*, *Polygala transcaucasica* and *Thymus collinus*. These sites display a lower vegetation cover with patches of higher soil cover. Sites of highest rock cover and stone content can be considered as highly degraded and are vegetated by few pioneer plant species with high seed dispersal rates such as *Trisetum rigidum*, *Campanula alliarifolia*, *Lactuca racemosa*, *Tussilago farfara* and *Parnassia palustris*. These stands show the lowest plant diversity. However, some of the mentioned grassland species display a high variance of environmental condition which results in a gradual changeover from these vegetation types and complicates the definition of grassland degradation from individual indicator species. For further information of vegetation composition and related environmental variables of the study region see Wiesmair et al. (2016b).

Figure 4.1 Map of the study area. A) showing its location within the Caucasus region, the inset displays the upper Aragvi valley which is shown in B) at contour zones of 500 m C) displays plot allocation along the slope next to Mleta



4.2.2 Vegetation data and environmental variables

In July 2012, we sampled 48 plots (5 m x 5 m) next to the village Mleta for hyperspectral reflectance, vegetation composition and environmental parameters (inclination, altitude and total vegetation cover). In our study area, July is the month of peak plant development; thus, that period offered ideal conditions for vegetation sampling and we assumed highest separability of green vegetation cover from soil/rock cover. We selected the plots according to their total vegetation cover in order to sample a gradient of 0-100% coverage. Due to the high correlation of vegetation cover to slope inclination, we used a digital elevation model to predefine sampling areas covering the full range of slope inclination and vegetation cover. All plots were located on the slope (3–43° inclination) whereas the flat valley ground was not sampled. Each plot covered 25 m^2, within which we arranged three 1 m^2-subplots. For each plot we measured inclination with the SUUNTO PM-5 C 360 clinometer and altitude with the GARMIN GPS62s. We photographed the ground vegetation cover of each subplot and further used these digital images to determine vegetation cover. Therefore each subplot was photographed with a hand-held digital camera (Panasonic LUMIX DMC-TZ1, 5 Megapixel). Photos of the vegetation canopy were taken from 140 cm distance to the ground at nadir. We used the image processing program Photoshop CS5 version 12 (Adobe Systems, Mountain View, CA) to calculate the vegetation cover of each subplot. Within each subplot image, we identified pixels that represented vegetation and used the ratio of vegetation pixels to total image pixels to define the percentage of vegetation cover. Mosses considerably contribute to the greenness of sparsely vegetated terrain but moss cover cannot be used as an indicator of grassland degradation (Karnieli et al. 2002, 1996). Therefore, we further distinguished between the cover of vascular plants and mosses.

Vascular plant species abundance was estimated for each 1 m^2-subplot with the modified Braun-Blanquet cover-scale (Barkman et al. 1964). We transformed the Braun-Blanquet cover values to percentage scale (r = 0.3, + = 0.5, 1 = 2.5, 2m = 4.0, 2a = 8.75, 2b = 18.75, 3 = 37.5, 4 = 62.5, 5 = 87.5) which was further log-transformed. To reduce noise in the data (in total 136 species), we omitted species with less than three occurrences prior to ordination analyses. Following analyses were run with a dataset of 104 species. We used the “vegan” R-package (Oksanen et al. 2013) for Ward classification and Non-metric dimensional scaling (NMDS, Kruskal 1964) ordination, both was performed with the Euclidean distance which showed a better dispersal than Bray Curtis distance. The cluster tree was cut after visual inspection to result in three classes. The settings for the NMDS were: global Multidimensional scaling using monoMDS; 3 dimension; no convergent solutions - best solution after 20 tries; scaling of centering and PC rotation. NMDS is an ordination technique which is widely used in ecological studies to graphically display the similarity of data. Therefore a distance measure is calculated which is stepwise placed into a multidimensional space to keep the original distances. The goodness of fit, or how well the configuration fits the data, is measured as stress (Kruskal 1964). All analyses were performed using the R Project statistical computing software (R Core Team 2014).

4.2.3 Spectral data

To simulate the spectral canopy reflectance signal as measured by the spaceborne multispectral WV2 sensor at a high spatial resolution, we recorded hyperspectral reflectance with a portable spectrometer. With clear sky between 10h00 and 15h00 local time, we measured the canopy reflectance of our subplots. We used a FieldSpec® Hand-held 2 Portable Spectroradiometer (HH2, ASD Inc., Boulder, CO) which covers the spectral range from 350 nm to 1050 nm with a spectral resolution of less than 3 nm. We defined the internal averaging setting of the HH2 at 60 spectra for each measurement of reflectance, dark current and white reference. The reflectance spectra were measured relative to a white reference panel (Spectralon®, Labsphere Inc., North Sutton, NH), which was taken anew prior to each plot measurement. The height of measurements over ground was at nadir 140 cm. To capture the whole extent of a 1 m^2-subplot, we performed five measurements. For each subplot, we calculated the mean value of these five spectral measurements. Outliers were visually detected and removed from further analysis. The final analyses comprised the hyperspectral data out of 139 subplot measurements. Prior to further analyses we applied the Savitzky-Golay (Savitzky and Golay 1964) filter using the R-package “signal 0.7-4” (Signal developers 2013), fitted over 51 nm with a first order polynomial to smooth our dataset. The smoothing process with Savitzky-Golay filter is appropriate to reduce oscillating noise which results from outdoor conditions in the vegetation spectrum (Schmidt and Skidmore 2004). We used the hyperspectral signal to simulate the signal of the multispectral World View 2 sensor based on its band-specific spectral response functions. Therefore we implemented our spectral signal into the R-“simulatoR” function (Feilhauer et al. 2013). Further we used these simulated sensor bands to calculate the vegetation indices given in table 4.1.

Table 4.1 Spectral bands and vegetation indices used in the random forest modeling of vegetation cover and vegetation types

Full name	Abbreviation	Definition	Quotation
Spectral bands			
Coastal Blue	Blue1	400-450 nm	
Blue	Blue2	450-510 nm	
Green		510-580 nm	
Yellow		585-625 nm	
Red		630-690 nm	
Red Edge		705-745 nm	
Near Infrared Band1	NIR1	770-895 nm	
Near Infrared Band2	NIR2	860-1040 nm	
Vegetation Indices			
Atmospherically resistant vegetation index2	ARVI2	$-0.18 + 1.17(\frac{NIR2 - Red}{NIR2 + Red})$	Kaufman and Tanre 1992
Enhanced Vegetation Index	EVI	$2.5(\frac{NIR2 - Red}{(NIR2 + 6Red - 7.5Blue1) + 1})$	Huete et al. 1999
Green Optimized Soil Adjusted Vegetation Index	GOSAVI	$\frac{NIR2 - Green}{NIR2 + Green + 0.16}$	Cao et al. 2013; Rondeaux et al. 1996
Green Atmospherically resistant Vegetation Index	GARI	$\frac{NIR2 - (Green - (Blue1 - Red))}{NIR2 - (Green + (Blue1 - Red))}$	Gitelson et al. 1996; Gitelson et al. 2003
Green Soil Adjusted Vegetation Index	GSAVI	$\frac{NIR2 - Green}{NIR2 + Green + 0.5}(1 + 0.5)$	Sripada et al. 2006; Huete 1998
Modified Soil adjusted Vegetation Index	MSAVI	$\frac{2NIR2 + 1 - \sqrt{2(NIR2 + 1)^2 - 8(NIR2 - Red)}}{2}$	Qi et al. 1994
Normalized Difference Vegetation Index 690-710	NDVI 690-710	$\frac{NIR2 - Red}{NIR2 + Red}$	Gitelson and Merzlyak 1997
Normalized vegetation index	NDVI	$\frac{NIR1 - Red}{NIR1 + Red}$	Krieger et al. 1969; Rouse et al. 1973
Optimized soil adjusted Vegetation Index	OSAVI	$(1 + 0.16)\frac{NIR1 - Red}{NIR1 + Red + 0.16}$	Rondeaux et al. 1996
Simple Ratio 801/670 NIR/Red	NIR/Red	$\frac{NIR1}{Red}$	Daughtry et al. 2000
Simple Ratio NIR/Red Difference Vegetation Index	DVI	$\frac{NIR2}{Red}$	Jordan 1969
Soil Adjusted Vegetation Index	SAVI	$\frac{NIR1 - Red}{NIR1 + Red + 0.5}(1 + 0.5)$	Huete 1988

4.2.4 Random forest model

The dataset was analyzed with random forest (RF, Breimann 2001) regression and classification towards a prediction of vegetation cover and the vegetation types derived from Ward classification. The RF approach has previously been successfully used to analyze remote sensing data (Feilhauer et al. 2014; Lawrence et al. 2006; Rodriguez-Galiano et al. 2012; Stefanski et al. 2014). A RF is an ensemble of individual regression trees (Grömping 2009), which are constructed by repeatedly splitting the dataset into homogeneous groups in order to explain the response variable (De'ath and Fabricius 2000). By doing so, the out-of–the-bag (OOB) samples are left out of the model to test how the RF performs by variable predictor permutations and cross validation. The significance of predictor variables is provided by the measure of variable importance. Variable importance can be calculated from the error on the OOB data (error rate for classification, MSE for regression) or the decrease in node impurity from splitting on the variable over all trees (Gini index for classification and residual sum of squares for regression) (Liaw and Wiener 2002). The decrease in node impurity indicates the quality of homogeneity of one variable within its group (Breiman 1984). We calculated both importance variables in the following referred to as 'importance' and 'purity'. From the R-package "randomForest 4.6-7" (Liaw and Wiener 2002) we chose the default setting for the number of predictors sampled for the splitting at each node (m_{try}) (Breiman and Cutler 2012). The number of trees to grow (*n*tree) was set to 5000 trees which achieved stable values. For the RF regression model of vegetation cover we chose the 8 bands of WV2 in combination with several vegetation indices (see Table 4.1) which are appropriate for agriculture, vegetation and soil applications ([1]). Additionally, we included the environmental variables altitude and slope to our RF classification model of vegetation types.

We used 100 times bootstrapping to validate our model results. The training samples were drawn with replacement from the plot samples. The RF regression model fit was validated through a linear regression of the predicted vegetation cover versus the observed ground true values. The RF classification model was validated with a confusion matrix. Therefore we used the R-package "caret 6.0-71" (Kuhn et al. 2016). Further we calculated the total root mean square error of prediction (RMSEP) in order to evaluate RF regression accuracy. RMSEP is a commonly used criterion for judging the performance of a multivariate calibration model (Faber 1999). The RMSEP is calculated following equation 1:

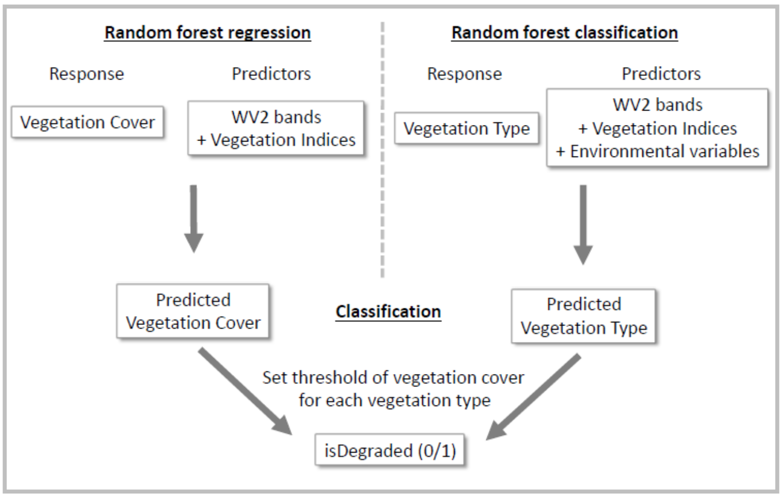
$$RMSEP = \sqrt{\sum_{i=0}^{n}((X_i - Y_i)^2/n)} \quad (1)$$

X is the predicted value from the model, Y the ground-true observed value and n the number of predictions.

4.2.5 Grassland degradation classification

In order to evaluate grassland degradation we implemented the predicted bootstrapped vegetation cover values and vegetation types into a classification (Figure 4.2). Therefore, we set for each vegetation type a threshold of vegetation cover which indicates grassland degradation. For the pastures, we used the threshold of 70% vegetation cover which is a common restoration goal to control erosion in high-montane grassland (slopes (Krautzer and Klug 2009). For the poor grassland we defined the vegetation cover below 35% as being degraded, which we determined from photographs. Sites which already display vegetation of eroded sites were generally assigned to the class of degradation and therefore the threshold of 100% vegetation cover was chosen. The grassland degradation classification with the predicted values was validated with the observed true values in a confusion matrix.

Figure 4.2 Classification scheme of grassland degradation (isDegraded 0/1) by including the predicted values of vegetation cover and vegetation type.



4.3. Results

4.3.1 Vegetation and environmental data

The gradual change of vegetation composition was visualized in the NMDS and revealed a stress of 14.6 (Figure 4.3). The Ward classification resulted in three distinct clusters which are in the following named after the indicator species of previous studies (Wiesmair et al. 2016b): *Cynosurus cristatus* (n = 21), *Briza media* (n = 56) and *Parnassia palustris* (n = 62). Environmental vectors were fitted against NMDS ordination and are shown in the ordination graph. The first axis displays a gradient of vegetation cover and slope. The second axis represents a gradient of altitude. The *Parnassia palustris*-type displays highest slope inclination and lowest vegetation cover. Plots of the *Cynosurus cristatus*-type occur on highest altitude at high vegetation cover. The *Briza media*-type appears on the transition between the other vegetation types as individual plots overlap with the other types.

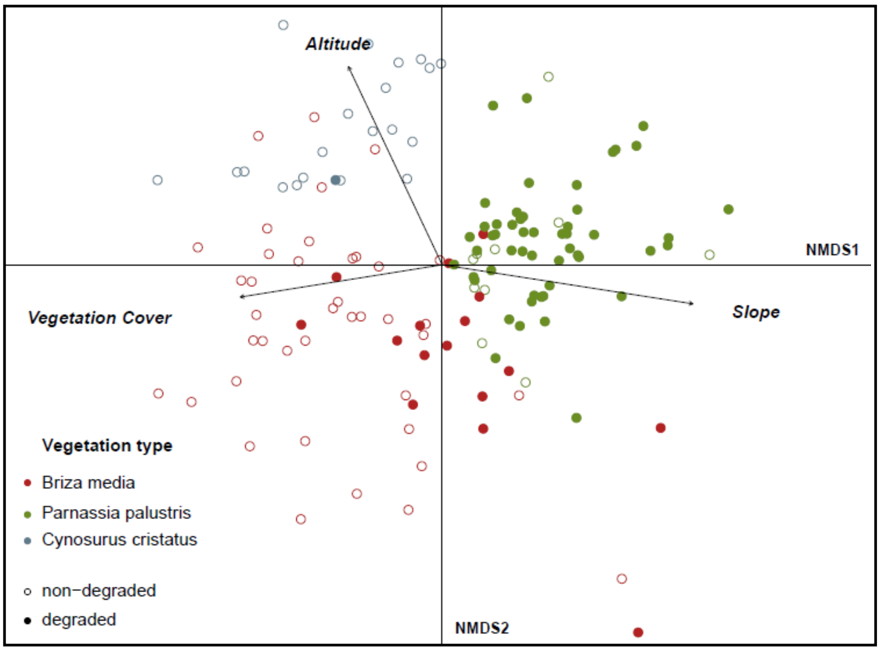


Figure 4.3 NMDS ordination of 139 subplots and 104 species displaying first and second axes, environmental variable gradients are indicted by arrows. Vegetation types are grouped by color and the prediction of grassland degradation is shown by symbols.

4.3.2 Random forest models

We used random forest regression and classification models to predict vegetation cover and vegetation types. The vegetation indices and band contribution to the RF models were visualized in figure 4.4. We found EVI and GARI to be most important vegetation indices for both models. The red edge and both NIR bands showed highest importance from the group of WV2 bands. For both models, the remaining wavebands (blue1, blue2, green, yellow and red) displayed a higher importance than the remaining vegetation indices. However, for the RF regression model EVI and GARI displayed highest variable purity whereas certain WV2 bands (blue1, blue2, green, yellow, red and red edge) showed lowest purity. Furthermore, for the RF classification of vegetation types, the environmental variables (altitude and slope) contributed most importance to the model. The same pattern was observed for the variable purity of the RF classification model.

We used only the above mentioned variable predictors of highest importance for our RF models. Validated results of RF regression models were calculated from all bootstrapped predicted versus observed data. The prediction of vegetation cover resulted in a model fit of $R^2 = 0.80$ and RMSEP = 12.25%. The RF classification model for the prediction of vegetation types had an overall accuracy of 76% at a Kappa statistic value of 0.60. The confusion matrix displayed a balanced accuracy between 76 and 87% (Table 4.2). The lowest accuracy was achieved for the *Briza media*-type and the highest accuracy for the *Cynosurus cristatus* grassland. Omitting predictors of least importance reduced our regression model (vegetation cover) results ($R^2 = 0.82$, RMSEP = 11.63%), while the classification model (vegetation types) results improved (74% overall accuracy, Kappa = 0.56).

Table 4.2 Confusion matrix of bootstrapped random forest classification model to predict vegetation types from simulated multispectral World View 2 bands, vegetation indices and environmental variables (altitude and slope). Overall accuracy = 76%, Kappa = 0.60

Vegetation type	Balanced Accuracy	Precision	Sensitivity
Briza media	76	72	72
Parnassia palustris	80	76	79
Cynosurus cristatus	87	90	76

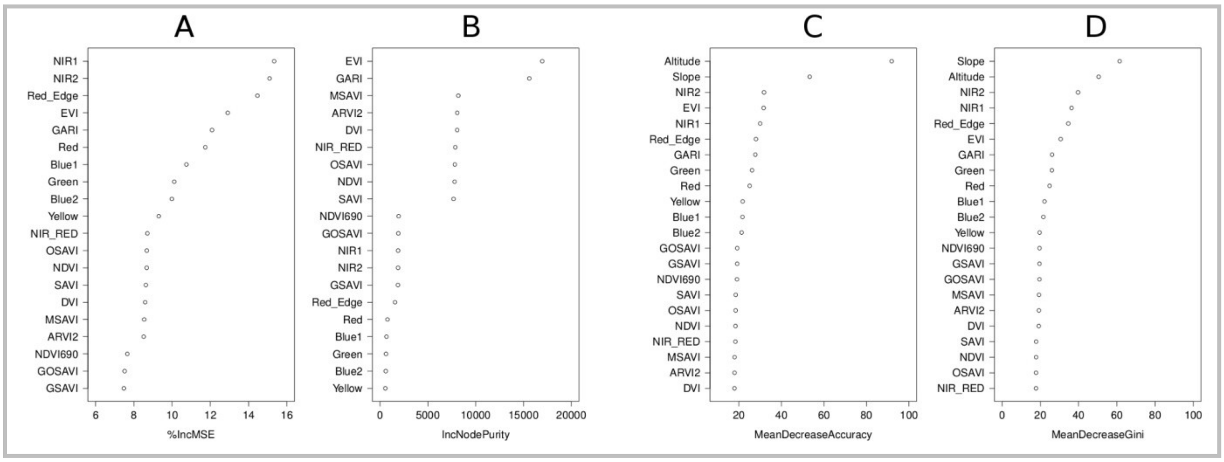


Figure 4.4 Importance (%IncMSE, MeanDecreaseAccuracy) and purity (IncNodePurity, MeanDecreaseGini) of vegetation indices, wavebands and environmental variables for random forest models to predict vegetation cover (A, B) and vegetation types (C, D). Full names of abbreviations are given in table 4.1.

4.3.3 Classifying grassland degradation

We implemented the predicted values of vegetation cover and vegetation types into a classification. The predictions were based on the variables of highest importance: EVI, GARI, red edge, NIR1, NIR2 and for the classification additionally altitude and slope. The classification of grassland degradation displayed an overall accuracy of 75%. The confusion matrix shows an accuracy of 72–79% for the given threshold levels within each vegetation type (Table 4.3). Lowest accuracy revealed the *Briza media*-type. To visualize the grassland condition of each subplot (Figure 4.3), grassland degradation was calculated from the average values of each bootstrap sample from predicted vegetation type (mode value) and vegetation cover (mean value).

Table 4.3 Confusion matrix of a grassland degradation classification from predicted values of vegetation types and vegetation cover

Threshold	Accuracy (%)
Briza media & < 35% cover	72
Parnassia palustris & < 100% cover	79
Cynosurus cristatus & < 70% cover	72
Overall	75

4.4. Discussion

Grassland degradation emerges as a process which alters the appearance of a landscape and can therefore be detected by RS methods. Vegetation composition and vegetation cover have been used as RS indicators to assess grassland condition on a large scale. Due to the heterogeneity of mountain landscapes, the discrimination of beginning small scale degradation is essential to apply early restoration management in the Caucasus region. Therefore, the aim of our study was to test the combination of vegetation cover and vegetation types to detect mountain grassland degradation with multispectral sensors of high spatial resolution.

From our good model fits we conclude that the gradient of vegetation cover was successfully predicted from the RF regression of multispectral bands and vegetation indices. Further, we could improve the results that were achieved by Wiesmair et al. (2016) when only single vegetation indices (NDVI or $MSAVI_2$) were used to map the vegetation cover of our study region. This is in accordance to Liu et al. (2015) and Lehnert et al. (2015), who proposed the combination of several indices to evaluate grassland condition. Lehnert et al. (2015) received similar results for their Partial least square regression (PLSR) model and could further improve their results by using support vector machine (SVM) regression. RF and SVM have both found to be very robust and non-parametric methods which are not prone to overfitting and to handle the multicollinearity of the predictor variables quite well (Belgiu and Drăguţ 2016; Mountrakis et al. 2011). However, RF is easier to use as fewer parameters need to be tuned (Chan et al. 2012).

In our study the vegetation classification resulted in three distinct vegetation types which were thoroughly described in a previous study (Wiesmair et al. 2016b). From the short ordination gradient and the overlapping vegetation types in the NDMS graph, we conclude a high share of plant species between the vegetation types which was further described by Wiesmair et al. (2016b). However, from the long vectors within our NMDS ordination we conclude a strong correlation of species composition to environmental gradients which is further clearly distinguishing our vegetation types. From our good RF classification results we assume that we could overcome the difficulties in RS to predict vegetation composition from short gradients due to the structural differences of our vegetation types. The accuracy of our RF vegetation type classification is in accordance to the results of Möckel et. al (2014) who received 77% accuracy in the classification of successional grassland stages. However, they used hyperspectral reflectance and could further increase the classification accuracy when the number of predictor bands was reduced with backward selection. This was not the case for our model when only highly important predictors were used. Möckel et al. (2014) found a high amount of important hyperspectral bands within the Short wave infra red (SWIR) region which was neither covered by the simulated WV2 bands nor by the original spectrometer data. Consequently, we assume that additional IR bands might improve our model as the multispectral WV2 bands in the red edge and NIR region were most important for our models. Additionally, the environmental variable importance of slope and altitude seems to be significantly improving our classification model as the highest importance was assigned to those predictors. The information of slope has been included into previous grassland assessments (e.g. Pickup and Chewings 1996) and was already suggested for management recommendations of degraded grassland by Wiesmair et al. (2016).

Grassland degradation was sufficiently assessed from the combination of vegetation cover and vegetation types. The vegetation types which are by itself considered as degraded, such as the *Parnassia palustris*-type, most likely reveal the highest classification accuracy. In such cases the decision if degraded or not is consistent to the accuracy of the vegetation type classification. For the vegetation types where a vegetation cover threshold for grassland degradation is set, in our case the *Briza media*- and *Cynosurus cristatus*-type, the classification precision is further reduced from the inaccuracy of the predicted vegetation cover. However, the low accuracy to predict grassland degradation on the *Briza media*-type supposedly results from the highest classification error rate of this vegetation type. Wen et al. (2010) overcame a similar problem, where *Ligularia virgaurea* occurred as a dominant and subdominant indicator species for degraded and non degraded sites, by the establishment of a grassland degradation index. To further distinguish the grassland condition, they calculated a weighted index from visible indicators such as cover, biomass, proportion of edible plants and plant height. However, as the RS assessment of biomass and cover proportions involves further inaccuracies we assume that our predictions of vegetation cover and vegetation types result in a more precise degradation index. Furthermore, our approach to include environmental variables into RF classification might decrease uncertainties from spectral similarity where ruderal pasture weeds simultaneously build up biomass and reduce grassland condition at high vegetation cover.

Hyperspectral reflectance and hyperspectral indices seem to be inadequate to represent the spatial scale of the environmental pattern that needs to be analyzed for grassland assessment. Rahman et al. (2003) determined the optimum spatial resolution at 6 m for a hyperspectral aircraft sensor, to represent most ecosystem functions of a Californian grassland. However, the costly hyperspectral aircraft sensors are unsuitable for transitioning countries of the Caucasus region where satellite sensors are beneficial for large scale grassland assessments. Moreover, our ordination results displayed strong environmental gradients which assist in the detection of beginning vegetation damage spots at a higher spatial resolution. Due to the increasing number of high resolution multispectral sensors, our study is contributing to future developments of landscape assessments by RS.

We conclude that a combination of spectral and environmental variables is essential to predict vegetation cover and grassland types, in particular of similar spectral reflectance. Further, we state that grassland degradation represented by grassland types and coverage can be detected by multispectral sensors. Moreover, we assume that the model accuracy will further increase with more advanced sensors such as the World View 3 (WV3). An improvement of the World View 3 sensor is its extension of spectral band information in the SWIR region and therefore we suggest testing the WV3 sensor for the modeling of vegetation cover and vegetation types. However, to put our approach into practice further tests are needed as the results of our study are based on a hand-held spectrometer. Therefore, the atmospheric transmission and topographic influence of steep slopes was not part of our study and needs to be studied in detail.

4.5. Conclusions for practicability

Vegetation cover is a strong indicator for grassland degradation from satellite images. Due to the increasing resolution of multispectral sensors, additional options to observe grassland condition have emerged. We recommend detecting grassland degradation from a combination of vegetation cover and vegetation types at high spatial resolution. Therefore we recommend including several multispectral bands such as NIR1, NIR2 and red edge, and the vegetation indices EVI and GARI to predict vegetation cover and vegetation types by means of remote sensing. Further, we suggest incorporating environmental variables, which display a strong gradient along the species composition and can be easily derived from elevation models such as slope and altitude, into the prediction of vegetation types. Therefore, we propose modeling by random forest or support vector machine with the first being easier to use. Due to the increasing resolutions of multispectral sensors we suggest testing World View 3 images with its additional SWIR bands for the detection of grassland degradation. The outcomes of this study represent important information to further enhance assessments of Caucasian grassland condition with means of multispectral remote sensing imagery.

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Summary

The Caucasus region is one of the global biodiversity hotspots which further comprises highly diverse mountain grasslands. These grassland ecosystems were shaped from a long tradition of human land use and provide multiple ecosystem services such as food supply for grazing animals, recreational sites and erosion control. However, changes of land use practices have induced grassland degradation in the Georgian Caucasus regions. Overgrazing during Soviet period and recent increases in recreational activities resulted in a reduced grass cover, an increased abundance of unpalatable plant species and soil erosion. Due to an expansion of grassland degradation, the loss of services provided by healthy ecosystems can be expected. To protect Georgian mountain grasslands, detailed information about ecological relationships within the ecosystem and methods to monitor grassland conditions are urgently needed.

This thesis investigated grassland degradation within the upper Aragvi valley of the Greater Caucasus, in the Republic of Georgia. Field studies were conducted in the vicinity of the village Mleta, in a landscape which is frequented by overgrazing, erosion and mass wasting events. The aim of this thesis was to develop site-specific methods to prevent further degradation in the Caucasus region. Therefore we implemented the commonly used feature of vegetation cover to assess the extent of grassland degradation by remote sensing imagery. We used random-forest regression to estimate vegetation cover from the Normalized Difference Vegetation Index (NDVI) derived from multispectral WorldView-2 data. The good model fit of R^2 = 0.79 indicates the great potential of a remote-sensing approach for the observation of grassland cover. The presented vegetation cover map shows grassland degradation on steep slopes close to human settlements and along hiking trails. Further, we investigated the relationships between plant diversity, site conditions and vegetation cover on overgrazed and eroded sites. We used non-metric dimensional scaling ordination and cluster comparison of functional plant groups to describe a gradient of grassland vegetation cover. For our study region, we identified four major vegetation types and performed an indicator species analysis. On abandoned hay meadows of the subalpine zone we identified tall herb vegetation with increasing occurrence of ruderal pasture weeds. Within high-montane grassland a decline of plant diversity can be observed on sites of reduced vegetation cover. Based on the results of the indicator species analysis, a list of 22 recommended native plant species to revegetate beginning small scale damage patches was elaborated and is presented in this thesis. In the last chapter, we improved the detection of grassland degradation by multispectral satellite sensors as we implemented vegetation cover and vegetation types into a classification model. Therefore, we used a hand-held field spectrometer to simulate the multispectral World View 2 sensor. A selection of predictors (vegetatation indices, spectral bands and environmental variables) was implemented into random forest models to predict the vegetation cover and vegetation types. The outcomes were further combined to estimate the grassland condition of our research area. Our results showed an overall accuracy of 75% and were displayed in an NMDS ordination graph.

To prevent further large scale erosion events and the loss of precious mountain grasslands we conclude that the presented remote sensing methods are promising tools for the early detection of beginning vegetation damage spots. Previous reforestation efforts for slope protection have failed due to the lack of an appropriate grazing management. Due to a low potential of the grassland ecosystem to balance further vegetation cover damage, the long-term loss of diverse habitats can be expected. Consequently, to conserve precious Georgian mountain grasslands a sustainable landscape management for the collective mountain grasslands is mandatory. The results of this thesis serve for the implementation into sustainable agricultural and touristic development plans of mountain regions which suffer from grassland degradation.

Zusammenfassung

Die Kaukasus-Region ist einer der globalen Biodiversitätshotspots. Innerhalb des Ökosystems wurde das Berggrünland durch eine traditionelle menschliche Nutzung geprägt und weist heute eine hohe Biodiversität auf. Das Grünland stellt für die Bevölkerung verschiedene Ökosystemdienstleistungen, wie z.B. die Nahrung für Weidetiere, menschlichen Erholungsraum und Erosionsschutz, bereit. Dennoch haben historische und aktuelle Landnutzungsänderungen eine Grünlanddegradation im georgischen Kaukasus verursacht. Während der Sowjetzeit wurden die Viehweiden stark überweidet, aktuell bewirkt ein Anstieg der Freizeitaktivitäten einen hohen Nutzungsdruck auf das Berggrünland. Dadurch wurden die Vegetationsdeckung reduziert, die Abundanz von Weideunkräutern erhöht und Bodenerosion gefördert. Wegen der Ausbreitung der Degradation kann eine Abnahme der vom Grünland zur Verfügung gestellten Ökosystemdienstleistungen erwartet werden. Um das georgische Berggrünland zu erhalten, sind detaillierte Informationen über die ökologischen Wechselwirkungen innerhalb des Ökosystems sowie Monitoringmethoden des Grünlandzustands dringend notwendig.

Gegenstand dieser Arbeit ist die Untersuchung der Grünlanddegradation des oberen Aragvital im Großen Kaukasus in der Republik Georgien. Die Feldstudien dazu fanden in der Umgebung des Dorfes Mleta in einer Landschaft, die durch Überweidung, Erosion und Massenabtragungen gekennzeichnet ist, statt. Ziel dieser Arbeit war die Entwicklung standortspezifischer Methoden zur Verhinderung weiterer Degradation im Kaukasus. Dafür wurde der häufig für Grünlanddegradation benutzte Indikator *Vegetationsdeckung* verwendet, um das Ausmaß der Degradationserscheinungen mit WorldView-2 Satellitenbildern abzubilden. Die Vegetationsdeckung wurde dabei durch den Normalized Difference Vegetation Index (NDVI) mittels random-forest Regression modelliert. Die guten Modellergebnisse von $R^2 = 0{,}79$ lassen auf ein großes Potential der Schätzung von Vegetationsdeckung mit Fernerkundungsdaten schließen. Die präsentierten Ergebnisse zeigen für das obere Aragvital eine niedrige Vegetationsdeckung auf Steilhängen nahe der Siedlungen und entlang von Wanderwegen an. Weiter wurde der Zusammenhang zwischen Phytodiversität, Standortbedingungen und Vegetationsdeckung auf überweideten und erodierten Flächen untersucht. Um den Gradienten der Vegetationsdeckung zu beschreiben, wurden Ordinationsverfahren (Non-metric dimensional scaling) und ein Vergleich

funktioneller Pflanzengruppen angewendet. Für die Untersuchungsregion konnten dabei vier Vegetationstypen unterschieden werden, für die eine Indikatorartenanalyse durchgeführt wurde. Innerhalb der Hochstaudenvegetation aufgelassener subalpiner Weiden zeigte sich ein vermehrtes Auftreten von Weideunkräutern. Für die Vegetation des hochmontanen Grünlandes wurde auf Standorten mit verringerter Vegetationsdeckung eine Abnahme der Phytodiversität beobachtet. Auf Grundlage der Indikatorartenanalyse wurde eine Liste mit 22 einheimischen Pflanzenarten, die ein Potenzial zur Begrünung dieser kleinen Schadstellen aufweisen, dargestellt. Im letzten Teil der Arbeit wurde durch die Verknüpfung von Vegetationsdeckung und Vegetationsklassen die Fernerkundungsmethodik zur Klassifizierung des Grünlandzustandes weiterentwickelt. Dafür wurden die Daten des multispektralen WorldView 2 Sensors aus den räumlich hochaufgelösten Reflexionsdaten eines tragbaren Feldspektrometers (Handheld 2 ASD) simuliert. Aus einer Vielzahl an Prädiktoren (Vegetationsindices, Spektralbänder und Umweltvariablen) wurden die Parameter Vegetationsdeckung und Vegetationstypen mit random forest modelliert, um aus dieser Information den Grünlandzustand zu schätzen. Diese Ergebnisse zeigen eine Genauigkeit von 75 % und wurden in einer NMDS Ordination dargestellt.

Um großflächige Erosion und den Verlust von artenreichem Berggrünland zu verhindern, werden die vorgestellten Fernerkundungsmethoden als Werkzeuge zur frühzeitigen Erkennung kleiner Vegetationsschadstellen empfohlen. Frühere Wiederbewaldungsmaßnahmen zum Erosionsschutz sind aufgrund von fehlendem Beweidungsmanagement gescheitert. Zusätzlich kann auf den Flächen, auf denen die Vegetationsdecke verletzt wurde, von einem langfristigen Verlust der Phytodiversität ausgegangen werden. Infolgedessen wird die Erhaltung des wertvollen georgischen Berggrünlands maßgeblich davon abhängen, ein nachhaltiges Pflege- und Managementkonzept für die gemeinschaftlich genutzen Bergweiden zu entwickeln. Die Erkenntnisse dieser Arbeit dienen dazu, sie in die nachhaltigen Entwicklungspläne für Landwirtschaft und Tourismus der von Grünlanddegradation betroffenen Bergregionen zu implementieren.

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Appendix

The electronic supplementary material can be downloaded from:

http://geb.uni-giessen.de/geb/volltexte/2017/12922

Erklärung gemäß der Promotionsordnung des Fachbereichs 09 vom 07. Juli 2004 § 17 (2)

„Ich erkläre: Ich habe die vorgelegte Dissertation selbständig und ohne unerlaubte fremde Hilfe und nur mit den Hilfen angefertigt, die ich in der Dissertation angegeben habe.

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