EFFECTS OF LAND USE CHANGE ON FIRE, VEGETATION AND WILDLIFE DYNAMICS IN ARID GRASSLANDS OF SOUTHERN RUSSIA

by

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A dissertation submitted in partial fulfillment

of the requirements for the degree of

Doctor of Philosophy

(Forestry)

Under supervision of Professor Volker C. Radeloff

At the University of Wisconsin-Madison

2010

Acknowledgements

This research would not be possible without my advisor Dr. Volker C. Radeloff whose bold ideas and vision has inspired his original proposal, the team of which I had luck to be part of. He took time to persuade me that continuing scientific carrier is worthwhile which I do not regret and always encouraged me in the hard moments. Volker was very productive and I always admired his ability to go through my cumbersome writings to gradually make them more clear. I enjoyed our discussions on different topics I always felt uplifted after leaving his office.

I am the most thankful for my initiation with south of European Russia to Anna Lushchekina, without whom my way through PhD would most likely be very different. I hope that our collaboration will continue. I am especially lucky that your interest and initiative brought me closer to my home town and I learned a lot about the area of origin.

I am grateful to the professors on my committee, Monica Turner, Tim van Deelen, Phil Townsend and Randy Jackson who kept me on the right track with their questions and comments, made me think more broadly and always stay on scientific way of knowledge. Courses I have taken with them, comments, suggestions were the highlights of my PhD and I learned a lot from them.

The help and support from SILVIS lab graduate students and staff was outstanding. Constant flow of help, humor, activities, beer and fresh veggies made my life in Madison very pleasant. Alexander Prishchepov made me less home sick, thank you for that, argument is essential part of our culture and you helped me stay with plenty of it.

During my research I met a lot of wonderful people who contributed in various ways, Peter Potapov, Tobias Kuemmerle were great examples of great scientists always encouraging me to do more. I would like to thank also Department of Forest and Wildlife Ecology which warm atmosphere helped me adapt to a new world, and being submerged in its scientific environment definitely my research better.

My family: Yulia, Katya and Artem, I love you all, thanks for your interest in my work and willingness to help. I know we've been through a lot these years, but I hope it was fun.

The last but not the least I want to thank NASA Earth Systems Fellowship Program for the support these years, it is hard to imagine what my life would be if you won't be with me all these years.

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Introduction

Human land use is widely acknowledged to be the most influential disturbance agent (Foley et al. 2005, MEA 2005). Changes in the disturbance regime profoundly affect components of natural and coupled socio-ecological systems such as land cover, ecosystem processes, and biodiversity (Turner 2010). In upcoming decades, land use will likely continue to be more important than other factors of change such as climate (Sala et al. 2000). This is worrisome, because excessive amounts of disturbance may lead to catastrophic shifts in the composition and function of communities, as their critical thresholds of disturbance are reached (Scheffer et al. 2001).

Human land use is both the cause and effect of current and past socio-economic conditions. Massive institutional changes can lead to changes in social and economic conditions of similar magnitude, and as the result, human land use may be altered as well. While often having degrading effects on the environment, massive institutional changes also represent a promising research opportunity, because they can provide natural experiments at an unprecedented scale (Diamond 2001).

After the breakdown of the USSR, land use in the arid grasslands of southern Russia changed substantially, the major change being a strong decrease in livestock numbers (FAO 2010). Like other natural systems, arid ecosystems are quite susceptible to human land use. While rarely used for row crops, arid ecosystems often suffer from degradation due to overgrazing that may lead to significant changes in vegetation and fauna, and can even result in desertification and famine (van de Koppel et al. 1997). The rapid rates of livestock decline in southern Russia in the 1990s were globally unprecedented, and matched by few other places in the world, including Kazakhstan (FAO 2010), which underwent the same institutional change. However, despite the massive extents and rapid rates of land use changes in the arid grasslands of southern Russia, the environmental consequences and effects of these changes have barely been studied, especially at broad spatial scales. This is unfortunate, because such declines in land use intensity represent a rare opportunity to understand the relationships among land use, vegetation, wildlife, and other disturbance processes in arid grasslands.

My study area is located in the semi-deserts of Southern European Russia and represents an expansive belt of arid ecoregions extending from Western Russia east through Kazakhstan to Mongolia. Typical vegetation communities representative of the arid grasslands of this region are dominated by feathergrass (*Stipa sp.*) and sagebrush (*Artemisia sp.*). Summers are hot and dry, and winters are cold with little snow. Annual precipitation ranges from 210 to 340 mm and summer droughts are common. About 10% of the study area is protected by the strict nature reserve Chernye Zemli ("Black Lands"). The study area covers about 30,000 km^2 (approx. 2 Landsat scenes).

The overarching goal of my study was to identify the effects of the recent socioeconomic changes on ecosystems patterns and processes, including fire, vegetation and wildlife. The project focused on the area shared by two administrative regions in southern Russia: Republic of Kalmykia and Astrakhan region, where land use changes have been widespread, and where the last European population of saiga antelopes is at risk of extinction. My general research questions were: how much, and why, have vegetation and fire regimes changed, and what are the relationships among land use, fire, vegetation and climate, and saiga habitat selection. The analysis was based on remotely sensed TM/ETM+, MODIS, and AVHRR satellite data as well as on a variety of field data and thematic maps. The dissertation consists of an introduction, which provides a summary of the entire dissertation, and four chapters, which examine specific research questions in detail.

Chapter 1

Research question: How did fire dynamics change before and after the breakdown of the Soviet Union?

Hypothesis: Fire occurrence increased significantly after the collapse of the Soviet Union.

Fire is a prominent disturbance factor in arid grasslands, but little is known about its history in my study area, ir in Central Asia in general. Furthermore, there has been no prior study anywhere that examined how fire dynamics change after a massive socio-economic shock, and the collapse of the Soviet Union may provide a case study that is relevant for arid grasslands globally.

To reconstruct burned area I used AVHRR data acquired from NOAA's Comprehensive Large Array-data Stewardship System (CLASS) for the period of 1985-2007 covering April until August annually. From the full set of images, two were selected for every year, with one representing the beginning and the other representing the end of the active burning season (late April – late August). Every image had low cloud cover and was free from geometrical and radiometric errors. Each image pair was stacked in annual eight bands composites, with four bands from the 'before' image and four bands from the 'after' image. For every year, burned and non-burned areas were identified using visual photo-interpretation based on specific spectral response and visible changes in two images. All samples were combined into one comprehensive training dataset.

Image classification was done using decision trees. Training data was used to parameterize the classification and the resulting decision trees were used to estimate burn probability for each pixel. Decision trees are a nonparametric method, i.e., no assumptions of underlying distribution of the data were made. To generate more stable models, a bagging approach was used to derive an ensemble of tree classifiers. Each tree classifier was trained separately using a 50% random subset from the comprehensive training dataset. The set of 30 trees was used to classify image stacks for every year.

Results provided strong evidence that burned area increased significantly in recent years compared to the beginning of the series in 1985. We observed a dramatic change in burned area starting in 1997-1998 and burning has remained widespread ever since. Since 1997, burning exhibited significant amount of variation among years compared to period before 1997.

Validation was conducted for 13 out of the 23 years, using separate burned area datasets, generated from various remote sensing datasets, such as Landsat/TM, ETM+ (1986-1989 and 1999), RESURS/MSU-SK (1996-1998), TERRA/MODIS (2000-2007). Overall, the AVHRR burned area map was about 60-70% accurate, but showed some underestimation of burned area, because of low-resolution bias of the AVHRR data. However, established temporal trends were robust, and matched official data well, highlighting even more the conservative nature of the latter. We conclude that institutional change can lead to significant changes in burning in arid grasslands and these changes that are lagged in time.

Related paper (published): Dubinin, M., Potapov, P., Lushchekina, A., & Radeloff, V.C. (2010). Reconstructing long time series of burned areas in arid grasslands of southern Russia by satellite remote sensing. Remote Sensing of Environment, 114, 1638-1648

Related paper (published): Dubinin, M., Lushchekina, A.A., & Radeloff, V.C. (2010). Assessment of modern burning regime in arid ecosystems using remote sensing data (case study of Chernye Zemli) [in Russian]. Aridnye Ecosystemy

Chapter 2

Research question: How is the annual burned area related to changes in livestock, vegetation and climate?

Hypotheses: The burned area in a given year were:

1) positively correlated with the amount of vegetation during the spring growing season immediately prior to the summer fire season, because of higher fuel availability;

2) positively correlated with the amount of vegetation during the secondary growing season in the fall of the previous year, again due to higher fuel availability;

3) positively correlated with climatic conditions that are favorable for vegetation growth during spring and fall of the previous year, also due to higher fuel availability;

4) significantly more correlated with climate conditions during the period after livestock numbers had declined and grazing pressure lessened, than during the period before.

The result of the first chapter highlighted the changes in burning regime that were linked to institutional change and the resulting land use changes. However, the actual environmental drivers of this change were not yet quantified. Though we knew that livestock numbers collapsed and vegetation started to recover, we did not know if land use or climate change was the main reason for the increase in fire activity, and how all these factors were interrelated. The main goal of my second chapter was thus to identify the interactions among these drivers and their effect on burning dynamics.

To address my research question, several statistical analyses of the significance of the relationships among annual burned area and different human and environmental variables were conducted. Burned area estimates from Chapter 1 were pair-wise correlated with a number of environmental variables, such as local precipitation, maximum and average temperature, and broad-scale climate indices (West Russia, Arctic and North Atlantic Oscillation indices), vegetation (represented by GIMMS NDVI for April, May and August aggregated over study area and their previous year values), as well as the main human driver, represented by livestock. To analyze the relationship of my burned area estimates with these different environmental variables before and after institutional collapse, the dataset was separated into to two periods and correlations were calculated for each period separately. This analysis was also coupled with regression modeling to try to assess if the slopes of the relationships were different before and after fires had started. Finally, derived pair-wise relationships were used to build a structural equation model that enabled the analysis of the whole set of relationships simultaneously, and removed weakly related variables.

When analyzed over the entire time period, burned area was significantly negatively correlated with livestock numbers and positively correlated with aggregated NDVI value in April (spring greenness). The highest positive correlation of burned area was with NDVI in May, meaning that more biomass in this month resulted in more fires. Moderate correlations were found with climatic variables such as precipitation in April and maximum August temperatures, highlighting the presence of direct and indirect effect of these factors on vegetation and fire.

When analyzing the two time periods separately, the amount of livestock was not an important factor anymore. Consistent with the overall analysis, aggregated NDVI values in April were highly correlated with burned areas both before and after the collapse. These results provided an interesting quantitative picture of the dynamics of the ecosystems and the relationships of its components with burned areas. The main determinant of burning was May NDVI (right before the burning season starts). Interestingly, although April NDVI was also significantly positively correlated with burning, the direct link with it was not significant, illustrating the fact that a) April pastures are heavily grazed, and b) vegetation in May is dependent of vegetation in April. None of the precipitation measures were directly significant in explaining burning, and this suggests that moisture was not important, probably due to precipitation during the fire season in the summer. On the other hand, precipitation was not eliminated from the structural equation models, and linked to vegetation in April and May.

With the exception of fall greenness, our results could not reject any of our hypotheses. Livestock represented an important background disturbance factor, but was not significant explanatory factor if two sub-periods were concerned separately. Interestingly, there was a very substantial increase in the correlation with May NDVI in the period after the collapse. This strong increase in correlation May NDVI suggested that because livestock removed most of the vegetation, NDVI was not a good predictor of burning before the collapse, but NDVI became an important predictor after the collapse as vegetation recovered.

Our results highlight a complex system of interacting drivers that can jointly explain the burning regime. Socio-economic changes unmasked the effect of climate on burning after the collapse, while during Soviet times, fires were heavily suppressed and fuel loads were low. Our analysis showed that the broad climate oscillation indices that are often used for explanation of burning regime should be used with care, as it they are not always related to local climate and thus the mechanisms of their effect on burning are unclear.

Related paper (in review): Dubinin, M., Lushchekina, A., & Radeloff, V.C. (2010). Relationships between climate, livestock, and fire in the arid grasslands of southern Russia. Ecosystems.

Chapter 3

Research question: What are the effects of fire and past land-use on vegetation communities in semi-deserts?

Hypotheses:

1) the area dominated by grasses increased and the area dominated by shrubs declined;

2) the area of bare, open sands decreased; and

3) areas where grasses replaced shrubs were associated with both wildfires and formerly open sands.

In Chapter 1 and Chapter 2, I found significant increases in fires and this increase was related to vegetation. However, I used only a proxy variable for vegetation (NDVI), and I did not know if the increases in fires resulted in – or were caused by – changes in the composition of the vegetation community. The goal of this chapter was to understand the effects of land use change on semi-deserts' vegetation communities.

Vegetation changes can reflect both recent changes and long-term land use legacies. The data on the extents and dynamics of disturbance regimes (fire and grazing) makes it possible to relate changes in the disturbance regime to vegetation composition changes. The underlying question was if the effect of these disturbances was substantial enough to result in changes in vegetation over a large area. However, the resulting vegetation patterns may also be affected by past land-use legacies, such as unsustainable row crops production attempts that resulted in rapid wind erosion and desertification. In this chapter I thus asked foremost what kind of vegetation community changes have occurred, and secondly if past land-use signals and fires were more common in areas now occupied by perennial grasslands.

To conduct the vegetation classifications we exploited the specific phenological patterns of the different vegetation communities. Annual grasses had more pronounced peak in greenness (NDVI) in early spring (March-April), perennial grasses in late spring (May), and sagebrush had a weaker NDVI signal overall, but its growing season was longer. We used these phenological features and carefully selected Landsat scenes (one for early spring and one for late spring for every year) to conduct classifications of the dominant vegetation types for 1985, 1998, and 2007.

The mapping of the burned areas in Chapter 1 allowed us to examine how fires affected vegetation patterns. Classification results showed considerable increase in perennial grass-dominated communities at the expense of shrub-dominated in the study area from 1985 to 2007. Conversion took place in almost 20% of the study area and more that 60% of these changes happened within boundaries of burned areas. However, a substantial portion of the study area showed an increase in perennial grass-dominated communities even without burning. Also, areas of change were related to former plowed fields, but not limited to them.

The increase in grasslands suggests a positive feedback between perennial grasses and fire. Areas with more fires were more likely to convert to grasslands, and grasslands were more likely to burn. Out results also support conclusions of plot-level vegetation succession studies conducted by the local scientists who reported that a type conversion from shrubs to grasses can also be caused solely by a release of grazing pressure. However, we found that the change was greatly facilitated by fire, and areas that burned more than three times since 1998 were more likely to be covered by perennial grassland vegetation compared to areas that burned less frequently.

Related paper (to be submitted): Dubinin, M., Lushchekina, A., & Radeloff, V.C. (2010). Rapid vegetation change after land use changes and increasing wildfire activity in Southern Russian semi-deserts.

Chapter 4

Research question: What constitutes suitable habitat for Saiga antelopes in our study area and how does it change among years and seasons?

Hypotheses:

1) there is no significant difference among years in terms of the amount and the patterns of suitable habitat;

2) there are substantial differences among seasons preferred in terms of the amount and the patterns of suitable habitat; and

3) vegetation type and productivity are among of the most important driving factors of habitat selection by Saiga.

Saiga antelope is the last free roaming antelope of Europe, but the last wild population in Russia is located in my study area, and has declined precipitously in recent decades. The sharp decline of Saiga population was usually attributed to poaching, but landscape change may also play an important role. However, how increases in fires and vegetation change affected Saiga habitat is unclear. For my fourth chapter, my goal was to describe habitat selection by Saiga especially in relation to the environmental changes that I reported in my second (fire) and third (vegetation) chapters.

My fourth chapter examines habitat selection at a local scale. The analysis was limited to a subset of the study area from the previous chapter which overlaps with two protected areas in the Republic of Kalmykia and in Astrakhan region. I used maximum entropy modeling and a used-versus-background approach to estimate probabilities of saiga presence. Saiga occurrence data was collected from 2003 to 2007 by the staff of the reserves and used together with maps of various human and environmental variables to estimate what variables explain the variability in the occurrence data the best, and how explanatory power varies among years and seasons. Among other variables, fire perimeters and fire frequency from Chapter 1, as well as the vegetation classifications from Chapter 3 were included.

I found a strong relationship between fires and habitat selection by Saiga. Areas that burned more often were consistently selected more frequently by saiga. Another very important factor that contributed substantially to most of the models was the distance to water sources. On the other hand, the effects of vegetation structure and composition were not particularly strong in my study, which is inconsistent with local field-based studies on saiga diet. Areas close to the farms were also avoided, indicating competition with livestock.

Our results highlight that changes in fire dynamics likely affected habitat selection by Saiga antelope. The clustering of Saiga in fire affected areas is probably an indication that poaching by humans and predation by wolves are still significant issues, even in the areas that are protected, so that Saiga antelopes select areas with high visibility in order to avoid threats. This information is new and has implications for management of this endangered species.

Related paper (in preparation): Dubinin, M., Lushchekina, A., & Radeloff, V.C. (2011). Habitat selection by Saiga antelope in Southern Russia and its inter- and intra-annual variability.

Significance

The results of my research add to the understanding of the relationships between land use change, fire, vegetation and wildlife dynamics after dramatic institutional changes. These results contribute to science and conservation in three ways: they provide new ecological insight, they led to management and conservation recommendations, and they made methodological advances.

Ecologically, my work addresses the general problem of coupled dynamics of different environmental factors, both human and natural as well as changes in the disturbance regimes. My study is of particular interest as it deals with an environment that underwent a substantial decrease in land use intensity and where the environment is currently recovering, which is rather rare, since land use is generally intensifying globally. My results help to understand historical and modern dynamics of fire, vegetation and grazing and provide statistical evidence for the drivers of these patterns in arid ecosystem. From the applied point of view, my research provides a foundation to study habitat selection of Saiga antelopes, which was not possible before, as neither fire nor vegetation were estimated explicitly over large enough areas in the region of study. More specifically my study:

- *Provides novel information about a unique ecosystem and its species*. My research fills an important information gap about arid ecosystem and species of southern Russia, some aspects of which were not represented in scientific literature before. Comparable ecosystems exist in other part of the world (for example, the mixed grasslands of the Great Plains, USA), but they have very different disturbance dynamics due to a much higher level of human domination.
- *Provides evidence of substantial and widespread environmental changes occurring at a regional scale.* Though my study area was small, it is representative for the vast arid

belt which traverses the countries of Central Asia and Mongolia, and all of which underwent similar socio-economic changes as Kalmykia. My research provides the first evidence of both an increase in fire activity and vegetation change following the collapse of the Soviet Union. Preliminary evidence suggests that the processes of change in burning regimes and vegetation were similar in the entire region.

- *Provides evidence for the existence of a restorative threshold effect*. Changes in ecosystem processes, especially fire, were rather abrupt, suggesting the presence of a tipping point, after which rare events increased in frequency and in the order of magnitude. Further monitoring of fires in the future could provide even more information about this basic ecological mechanism. Burned area might serve as an important indicator of coming future changes if increasing livestock numbers reach the tipping point again, causing ecosystem to revert back into the pre-fire state. A possible indication of that was the year 2010, which was exceptionally dry, and exhibited an unprecedented amount of burning throughout European Russia, but **not** in my study area potentially because livestock numbers have increased again.
- *Highlights interactions among disturbances*. My research highlights the importance of interactions among fire, climate, and livestock, as well as land use legacies. My results show that gradual changes in vegetation introduced by the decrease of one disturbance agent (livestock) can be amplified by a switch to another disturbance agent, and how vegetation interacts with particular past land use.

From the **management and conservation** point of view, ecosystem and species conservation can benefit from a better understanding of ecosystem structure and dynamics, and species habitat selection that my research provides. Some results of my project will have direct management implications. For example, I suggest establishing additional artificial water sources for saiga, to better control poaching, and to experiment with additional livestock grazing in the protected areas.

My research also highlights the need for better monitoring programs for fire, Saiga antelopes and vegetation to quantify the ecological patterns and processes. Better data could also improve the models that I have created and further improve understanding. Hopefully, my research will inform the development of conservation strategies aimed at Saiga antelope currently underway in the region.

Last but not least, some of the technical approaches introduced by this research represent **methodological advances** and provide new avenues to study changes in land cover and burning using remote sensing in semi-desert ecosystems. My results highlight that how machine-learning approaches such as decision trees, support-vector machines, and maximum entropy can make important contributions to both remote sensing analyses and wildlife habitat modeling in arid grasslands. The use of decision trees is promising for the analysis of long time series of remote sensing data. Common set of models applied uniformly to the sequence of images can help to reconstruct long-term times series in the more robust way, especially when dealing with data that is heterogeneous in quality. A phenological approach to image selection for classification provides effective methods for discrimination of vegetation associations, otherwise hard to distinguish, like grasses and shrubs. Structural equation modeling provides more comprehensive view of the ecosystem, because it integrates pair-wise correlations between sets of variables in an overarching framework.

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Chapter 1: Reconstructing long time series of burned areas in arid grasslands of Southern Russia by satellite remote sensing

Abstract

Fire is an important natural disturbance process in many ecosystems, but humans can irrevocably change natural fire regimes. Quantifying long-term change in fire regimes is important to understand the driving forces of changes in fire dynamics, and the implications fire regime changes for ecosystem ecology. However, assessing fire regime changes is challenging, especially in grasslands because of high intra- and inter-annual variation of the vegetation and temporally sparse satellite data in many regions of the world. The breakdown of the Soviet Union in 1991 caused substantial socioeconomic changes and a decrease in grazing pressure in Russia's arid grasslands, but how this affected grassland fires is unknown. Our research goal was to assess annual burned area in the grasslands of southern Russia before and after the breakdown. Our study area covers 19,000 km^2 in the Republic of Kalmykia in southern Russia in the arid grasslands of the Caspian plains. We estimated annual burned area from 1985 to 2007 by classifying AVHRR data using decision tree algorithm, and validated the results with RESURS, Landsat and MODIS data. Our results showed a substantial increase in burned area, from almost none in the 1980s to more than 20% of the total study area burned in both 2006 and 2007. Burned area started to increase around 1998 and has continued to increase, albeit with high fluctuations among years. We suggest that it took several years after livestock numbers decreased in the beginning of the 1990s for vegetation to recover, to build up enough fuel, and to reach a threshold of

connectivity that could sustain large fires. Our burned area detection algorithm was effective, and captured burned even areas with incomplete annual AVHRR data. Validation results showed 68% producer's and 56% user's accuracy. Lack of frequent AVHRR data is a common problem and our burned area detection approach may also be suitable in other parts of the world with comparable ecosystems and similar AVHRR data limitations. In our case, AVHRR data were the only satellite imagery available far enough back in time to reveal marked increases in fire regimes in southern Russia before and after the breakdown of the Soviet Union.

Introduction

Fire is one of the main disturbance agents in grasslands and savannahs. Fire shapes vegetation structure and composition, and represents an important land-management tool (Pyne 1984). Grasslands, woody savannahs, and savannahs represent more than 60% of the global burned area (Tansey et al. 2004), and in regions with high aridity, such as Central Asia, grassland fires account for 80% of all active fire counts (Csiszar et al. 2005). Fuel loads and emissions from grassland burning are relatively small (van der Werf et al. 2006), but grassland fires can foster the spread of invasive species (Brooks et al. 2004), affect wildlife (Archibald and Bond 2004), and cause air pollution that can spread as far as the Arctic (Stohl et al. 2007). Furthermore, interactions between grassland fires and human land use may result in ecosystem degradation, hydrologic changes, soil disturbance, and shrub encroachment (Archer et al. 1995). The restoration, conservation and management of arid grasslands thus require solid information on their fire regime and its change over time. Our goal here was to assess fire regime changes in the grasslands of southern Russia before and after the breakdown of the Soviet Union.

Fire regimes in general are largely determined by fuel moisture, fuel amount, and ignition sources (Bond and van Wilgen 1996). Arid grasslands are biomass-poor, and at least seasonally dry ecosystems, in which fuel amount and connectivity are the main limiting factors for fires (Meyn et al. 2007). Climate change can cause long-term changes in fuel amounts, but land use affects fuel amounts more directly and acutely. The main control of fuel amount in many arid grasslands is livestock (Bahre 1991). Unlike mesic grasslands, which evolved with intensive mammalian herbivory, arid grasslands are more sensitive to livestock grazing (Mack and Thompson 1982) often leading to overexploitation and degradation (Akiyama and Kawamura 2007; Zhang et al. 2007). Vegetation can recover when livestock grazing decreases, and the resulting increase in fuel amounts and connectivity may be able to sustain extensive wildfires (Liedloff et al. 2001). However, fire increases resulting from a decrease in livestock grazing have rarely been studied, and the long-term relationship between livestock grazing, burning dynamics, and environmental factors is not well understood. Thus, it is often inappropriate to study livestock-fire relationships with small plots and difficult to employ a classical experimental design at such broad temporal and spatial scales.

Where controlled experiments are impossible, natural experiments may offer insights (Diamond 2001). Significant socioeconomic disruptions and the resulting land-use changes can provide natural experiments to study accompanying environmental changes (Kuemmerle et al. 2007). Land-use change research typically examines how socioeconomics affect land use directly (Lambin and Geist 2006). However, it may be that indirect cascading effects, such as alterations of disturbance regimes, have a much larger effect on the ecosystem (Burcher et al. 2007). In the context of livestock grazing and grassland fires, the ideal natural experiment would comprise a strong decrease in livestock grazing, sufficient time for the resulting changes in fire regimes to manifest themselves, and relatively constant conditions for all other relevant factors. The breakdown of the Soviet Union in 1991 provided such a natural experiment.

The former Soviet Union including Russia experienced dramatic changes in livestock population in the arid grasslands, such as in the Republic of Kalmykia (Figure 1). In the early 1970s, under state-controlled plan-based economic policies, sheep numbers more than doubled, and stayed at over 800 thousand heads for about twenty years. During these years, the bulk of sheep industry was represented by large agricultural enterprises, which overexploited the grasslands. Grazing intensified up to a point where some pastures were grazed year-round. Intensive grazing caused widespread wind erosion and vegetation degradation (Zonn 1995). Due to overgrazing, southern Russia was called “Europe’s first anthropogenic desert” in the mid 1990s (Saiko and Zonn 1997). Some model forecasts predicted that even with protection, more than 66% of southern Russia's grassland would convert to bare sand by 2000 (Vinogradov 1995), and substantial governmental action was taken to combat desertification.

However, following the breakdown of the USSR in 1991 livestock populations dropped by almost an order of magnitude (Figure 1) and remained low until around 2000 (Brooks and Gardner 2004; ROSSTAT 2007). Large collective and state-owned farms were no longer subsidized after 1991, resulting in broad-scale de-collectivization and abandonment. The livestock declines allowed vegetation to recover and may thus have cause potentially an increase of grassy fuels. Thus the hypothesis is that the decrease in grazing pressure resulted in an increase in grassy fuels, and ultimately in an increase in grassland fires. The problem is that no fire data are readily available to test this hypothesis since official fire records for the region, similar to other regions are incomplete and inaccurate (Soja et al. 2004).

Remote sensing and satellite data have been used for more than 20 years to monitor fires in many different parts of the world (Csiszar et al. 2004). Coarse resolution satellite data are one of the best sources for historic burned area mapping because of their high temporal resolution and long history of acquisitions (Arino et al. 2001). Two basic types of algorithms exist to estimate fire areas from Advanced Very High Resolution Radiometer (AVHRR) or Moderate-Resolution Imaging Spectroradiometer (MODIS) data: a) detection of active fires, and b) mapping burn scars or burned areas. Though area burned correlates well with the number of fires, deriving burnt areas from active fire detections is error prone, especially in some ecosystems, because the detected hotspots can underestimate the actual burned area (Giglio et al. 2006; Hawbaker et al. 2008; Miettinen et al. 2007). In contrast, burned area assessments identify postfire disturbance, but not the actual fire event.

Typically burned area mapping involves band differencing and thresholding or classifying single or multitemporal data in the form of raw band values and derivative indices (Arino et al. 2001; Gong et al. 2006; Kučera et al. 2005). Burned area maps from coarse spatial resolution data are prone to several types of errors: low resolution bias, lack of imagery in certain areas, and geometric and radiometric errors (Barbosa et al. 1999; Boschetti et al. 2004). Most of these problems are more prevalent in AVHRR data, and have been solved or notably reduced in recent sensors such as MODIS on board of Terra and Aqua. However, MODIS images are only available since 2000 and this relatively short time series limits the analysis of long-term trends. Thus AVHRR remains the only remote sensing dataset capable of reconstructing long-term (20+ years) burning trends (Chuvieco et al. 2008; Kučera et al. 2005) at regional scale. The classification of satellite images for burned area mapping typically requires regular AVHRR observations, composited from daily data. Full resolution AVHRR data (Local Area Coverage, LAC, or High Resolution Picture Transmission, HRPT, both at 1.1 km at nadir) greatly improves the classification of burned areas (Pu et al. 2007; Razafimpanilo et al. 1995; Sukhinin et al. 2004).

Unfortunately, the amount and quality of AVHRR data vary greatly among different regions. Daily AVHRR observations at 1.1 km spatial high resolution are not always available, especially for those parts of the world where local archives had not been established until the late 1990s. Southern Russia is one such area, and only a limited amount of AVHRR data is available in international archives such as NOAA's Comprehensive Large Array-data Stewardship System (CLASS, http://www.class.noaa.gov/saa/products/welcome). Because of the limited data availability, existing remote sensing methods designed to map burned areas from AVHRR data can not be applied and there is a need to develop new approach to map burned areas in regions where only CLASS-type AVHRR data are available.

The main goal of this study was to assess changes in burning in the grasslands of southern Russia before and after the breakdown of the Soviet Union. Our hypothesis was that annual burned area increased substantially after the breakdown of the Soviet Union due to the decline of livestock populations and more abundant fuels. Our second goal was to develop and test a method to derive burned area trends from temporally sparse CLASS-type AVHRR data. Our study was motivated by 1) the absence of any solid information on distribution of fires throughout 1980s and '90s in the study area and throughout Central Asia, 2) substantial socio-economic changes in the region providing a unique 'natural experiment' to examine the effects of decreasing livestock grazing on fire regimes, and 3) the need for a remote sensing approach that can map long-term burned area trends in arid grasslands from sparse AVHRR data.

Methods

Study area

Our study area is located in the grasslands of Southern European Russia and occupies about 19,000 km2 of the Republic of Kalmykia and Astrakhan Region (Figure 2). The study area was defined in the west and east by the administrative boundaries of the Republic and in the north and south by the boundaries of the common area of available AVHRR overpasses. The study area is sparsely populated (population density 0.8 to 1.4 persons/km^2 (CIESIN and CIAT 2005) with no expectation for population growth (CIESIN et al. 2005).

The climate of the study area is arid, with hot, dry summers (mean daily temperature of +24°C in July; max +44°C, 280 days of sunshine per year on average). Annual precipitation is 150 to 350 mm (with a mean of 286 mm for 1985 to 2007). Summer droughts are common, and most of the precipitation falls in spring (43% of all precipitation), coinciding with the period of most vegetation growth (Walter and Box 1983). The topography of the study area is predominantly flat with a mean elevation of -15 m below sea level. The study area has a complex geological history of transgressions and regression of the Caspian Sea and soils characterized by a gradient from sandy aeolian deposits and sandy loams in the southeast corner of the study area to clay loam in northwest corner (Kroonenberg et al. 1997).

Vegetation associations are typical for northern Precaspian Plains and represent combinations of steppe and desert types. The main vegetation associations are shortgrass steppe (*Stipa* spp., *Festuca* spp., *Agropyron* spp., *Anizantha tectorum,* and other graminoids, **Figure 3**) and sage scrub (*Artemisia* spp., *Kochia prostrate,* **Figure 3**) (Golub 1994). Shortgrass steppe is characterized by a short growing season in April and May and rapid senescence in the dry summer. The grasses exhibit fire adaptation due to dense bunches which protect seeds and generate abundant fuels for burning. Sagebrush (*Artemisia* spp.) dominated shrublands have less biomass, but a longer growing season, and sometimes exhibit a second vegetation peak in the fall or even early winter (Kurinova and Belousova 1989). *Artemisia* spp. is more susceptible to fire because its buds are situated above ground and can be killed or damaged by fires. The lack of fire tolerance by *Artemisia spp.* might lead to its substitution by *Stipa* spp. and other graminoids. The primary human land use of the grasslands in the study area is as rangelands to support grazing for domestic livestock, mainly sheep and to a lesser extent cows and goats.

Study period

We studied burned area dynamics from 1985 to 2007. The choice of the study period was determined by remote sensing data availability and because of significant changes in land use after 1991. The 23 year fire record that we derived is one of longest for a burned area estimation with coarse resolution satellite data. Other burned area studies have either focused on forests or analyzed smaller time spans while studies of equivalent time span are particularly rare (Chuvieco et al. 2008; Kučera et al. 2005).

Data

Classification data

We used coarse spatial resolution AVHRR level 1B imagery in the form of digital numbers to estimate burned areas and three types of coarse- and medium- resolution satellite imagery for validation. We chose AVHRR for our burned area time series, despite its limited spatial and spectral resolution, because the length of the AVHRR data record matched our research goal to compare pre- and post 1991 burned areas. For the burned area classifications, we analyzed AVHRR LAC images acquired by National Ocean and Atmospheric Administration (NOAA) satellites (9,11,14,16,17). LAC data are stored onboard the satellite and downloaded at one of the NOAA receiving stations. All daytime images from 1985 to 2007 were downloaded from NOAA's CLASS. From these data, we selected a subset of images ranging from April 1st to September 20th of each year, corresponding to the beginning and the end of the growing season, and fully capturing the time of summer droughts and thus the fire season. From all images for each year, two images were selected according to the following criteria: i) the sensor angle was less than 45° degree off nadir, ii) no clouds obscured the study area, iii) the first image represented the peak of the growing season or close to it (April, May, or early June), and iv) the second image represented the end of the dry season and the end of the fire season (late August to early September). Most of the CLASS AVHRR LAC imagery had to be discarded. Typically, AVHRR images covering whole study area were available for every second day of the study period. For example, in 2006, out of 173-174 days (April-September), 90 were available. Among those 49 images were discarded due to excessive cloud amount, 10 due to scanner malfunction and 29 due to position of the study area 45° degree off nadir. Among 12 remaining images of good quality, 3 were available for May-Early June and 3 for August. We based our classification on the assumption that fires will raise reflectance, due to the removal of vegetation, and that after fire event brightness will remain elevated through the rest of the season. Persistent increase in brightness was confirmed by preliminary MODIS analyses (**Figure 4**). Unfortunately, due to limitations in data availability, the dates of the

AVHRR images were not identical among years. Image dates for the earlier image with the exception of 1998 (early April) were all from early May or the first week of June. Later image dates ranged from early (1994, 2000, and 2005) to late August and to early September. All satellite images were clipped to the study area boundaries.

The two selected images for each year were then composited into 12 bands, where bands 1-3 represented the AVHRR spectral bands from the spring image (620-670, 725-1100, 3550-3930 nm); 4-6 represented the AVHRR bands from the fall image; 7-9 were band differences where each fall band was subtracted from the corresponding spring band and offset value was added to avoid negative numbers; and 10-12 represented NDVI for the first and second image (NDVI1, NDVI2) and their difference (NDVI12). We removed band 5 as it was not present for all AVHRR instruments. Bands 4 was removed from the classification after preliminary analysis showed that this bands did not help differentiate burned from surrounding, non burned areas. Dark object subtraction was used to correct the images radiometrically (Song et al. 2001). The minimum observed value was selected from the Caspian Sea, and we subtracted the digital number observed over clear water for each band from all pixel values in each respective band.

Validation data

For validation, we used Terra/Moderate Resolution Imaging Spectroradiometer (MODIS), Landsat/Thematic Mapper (TM), Enhanced Thematic Mapper Plus (ETM+) and RESURS/Multi Spectral Scanner (MSU-SK) imagery. The MODIS Level 1B calibrated radiances product (MOD02QKM) with 250-m resolution was obtained from NASA's Level 1 and Atmosphere Archive and Distribution System (http://ladsweb.nascom.nasa.gov) and used for validation from 2000 to 2007. In addition to the validation, we also used MODIS data to estimate the temporal regime of burning (e.g., fire season length), because of MODIS' high temporal resolution. Landsat TM/ETM+ data were obtained from the USGS EROS Data

Center at 30-m resolution for 1986 to 1989 and for 1999. Unfortunately, no TM/ETM+ data were available in either public (USGS GLOVIS) or commercial archives (Euroimage) for the period of 1990 to 1998. Instead we used RESURS MSU-SK data obtained from R&D center Scanex (http://scanex.ru) at 150-m resolution and 4 spectral bands ranging from 540 to 1175 nm for 1996 to 1998. All validation data was collected for the period correspondent to period of the AVHRR data that we used (i.e., April – September). Last but not least, we obtained official fire statistics (Ministry of Emergency). Although the official fire statistics do not include all burned areas, the statistics provided comparison for the recent years of the burned area time series.

Data processing

Geometric correction

For every year in the sequence, the first AVHRR image was co-registered to the MODIS image of June 7th 2008 with a second-order polynomial transformation and nearest-neighbor resampling in ERDAS IMAGINE. The second image from a given classification pair was then co-registered to the first image using the same approach. The pixel size was set to 1.1 x 1.1 km. Semiautomatic tie point collection provided an initial set of ground control points (GCP). For each image from 7 to 10 GCPs were used for registration. Overall co-registration accuracy (root mean square error) was less than one pixel for each image. MODIS and Landsat data georeferencing accuracy is far superior to that of AVHRR and no additional georeferencing was necessary (Justice et al. 1998; Lee et al. 2004). Visual checking confirmed that the georeferencing of the validation data to the AVHRR data and 1 : 200,000 topographic maps was fine. RESURS data were co-registered to the June 7th 2008 MODIS image using the same methodology as for the AVHRR data. All imagery was reprojected to WGS84 coordinate system and equal-area Albers projection (central meridian at 45 degrees, first standard parallel 52 and second standard parallel 64 degrees).

Training data

Visual image interpretation provided training data for both burned and un-burned areas for the classification. For each AVHRR image pair, a set of burned and non-burned areas was digitized. Burned areas were identified based on one of two possible features. Areas that had burned late in the season and were still black were characterized by lower reflectance in the fall image, especially in AVHRR band 2 (Pereira 1999). Conversely, areas that had burned earlier in the season were highlight by vegetation removal and a high proportion of bare ground, causing higher reflectance in bands 1 and 2 of the fall image. For every year non-burned areas were also identified, including unburned vegetation, bare ground and water bodies. All samples were combined into one comprehensive training dataset, which consisted of 11,706 pixels (2,629 pixels for burned areas and 8,447 for non-burned). We used a combined training dataset, rather than keeping training data for each year separate, because we could not detected any visible burned areas in the AVHRR images in the 1980s.

Classification

Our classification method had to account for the fact that there were major limitations in the AVHRR image availability (see above). We used decision tree classification to calculate the certainty that a given pixel burned in a given year. Decision trees are a robust approach to classify satellite data (Friedl and Brodley 1997; Hansen et al. 2000) and are effective tool for burned area mapping in grasslands (Maggi and Stroppiana 2002; Stroppiana et al. 2003). Decision trees are nonparametric, i.e. no assumptions about the underlying distribution of the data are made, and decision trees can capture non-linear relationships between spectral data and different information classes (Breiman 1984). To create more generalized and stable classification models we used a bagging approach (Breiman 1996). Training data were sampled 30 times, each time extracting 15% of the comprehensive training dataset. Each selected sample was used to create a tree model resulting in an ensemble of 30 models. These models were applied to classify each of the 23 annual AVHRR data composites. The 30 classification results were then combined by averaging to estimate the probability of burning from 0 to 100%. Based on the probability, a simple majority threshold was used to tag the pixel as burned. Lastly, we used a 3 x 3 majority kernel to filter and smooth the classification results. This filtering may have removed some correctly mapped small fires, but we employed it to ensure conservative estimation of burned area and lower commission errors.

Validation

Burned area map validation

For the validation of the results we created an additional dataset of burned areas from higher resolution imagery (TM, ETM+, MODIS, MSU-SK). Red-green-blue composites were created for each image, using bands 2 (841-876 nm), and 1 (620-670 nm) from MODIS, bands 5 (1550-1750 nm), 4 (760-900 nm), and 3 (630-690nm) from TM/ETM+, and bands 4 (810-1000 nm), 2 (600-720 nm), and 1 (540-600 nm) from MSU-SK imagery. We validated our classifications for each year for which validation data were available.

Similarly to training data selection, validation data were obtained by visual image interpretation, shown to be at least as precise as automatic methods (Bowman et al. 2002). Delineating burned areas was straightforward due to distinct fire scar boundaries, often confined by linear features such as roads (Figure 7). The easy recognition of fires was further facilitated by the lack of other disturbances in the area that could have resulted in comparable patterns. In many cases fire scars remained visible for 2 to 3 years after burning, unless a new fire overrode an older scar (Figure 7). The elevated brightness of burned areas was also quite stable intra-annually (**Figure 4**). The validation dataset was rasterized to the 1.1 x 1.1 km pixels. Omission and commission errors were calculated for each year by combining binary burned classification and rasterized validation datasets. Producer's and user's accuracies were calculated from the four possible combinational classes (Congalton and Green 1998). Additionally, we calculated the Kappa coefficient, which is a proportion of agreement obtained after removing the proportion of agreement expected to occur by chance (Congalton and Green 1998).

Out of season validation

Because the AVHRR image dates were not uniform, we also checked if sub-optimal image dates caused omission of some fires. We tested this by comparison with numbers of active fire hotspots captured by both TERRA and AQUA satellites (MOD14A1 and MYD14A1). Based on the active fire data, we calculated the percentage of fires between August 20th and August 30th compared to the entire month of August, and the percentage of fires in August and September to the entire fire season for each year from 2000 to 2007. Though the number of active fire detections is correlated with the size of the burned area, this measure is not always reliable. Since some of the selected AVHRR images were collected in early August and we found some active fires in September that were not covered by the selected images, we extended the validation dataset until the end of September for the years 2002 to 2007. This allowed us to estimate how much burned area was missed due to the lack of optimally timed AVHRR images in some years.

Model transferability validation

Due to the coarse resolution of AVHRR satellite imagery, the visual estimation of training data may not always be reliable. Furthermore, in some years, no burned areas could be found for training. Using a comprehensive training dataset and a uniform set of decision tree models for all years captured the variability in burned area signals, and ensured that burning was not simply missed in a given year when visually interpreting the AVHRR data. However, the use of the comprehensive training dataset needed validation. To explore the transferability of the model to years for which there was no burned area training data, we removed training data for a set of years where amount of burning exceeded 15,000 km^2 (2000, 2002, and 2005 to 2007) from the comprehensive training dataset and repeated the classification. These tests quantified the extent to which uniform set of models was able to detect burning in one year in the absence of samples from this year in the comprehensive training set.

Results

Annual burned area

Burned area increased dramatically from virtually no burning in the 1980s and the first half of the 1990s, to up to 19% of the study area (3,600 km2) burned annually after 2000 (Figure 5 and Table 1). Among the 23 years that we studied, nine had substantial amounts of burned areas, while the other 14 had less then 2% of the area burned. Eight out of these nine high-burn years occurred after 1998 (Table 1). Large-scale burning started abruptly in 1997 and 1998, when 5% of the territory was burned and has continued regularly until today. However, burned area showed a high degree of variability, with one or two years of burning often followed by a year with negligible burned area.

Among the different AVHRR-derived bands that we used for the classification, the most important were the NDVI of the fall image and the difference between spring and fall values of band 1 and band 3. These metrics explained 47% of the variability; each contributed respectively 24%, 14%, and 9% to the decrease of deviance for all 30 decision tree models. Other metrics were equally valuable and contributed on average 6% of the deviance decrease.

Large fires contributed to the majority of the burned area. As estimated using MODIS data, burns larger than 250 km^2 represented 48% of total burned area. Though small fires (less than 60 km^2) represented 77% of the total number of fires since 2000, their share accounted only for 18% of total burned area (Figure 8). On average, the five largest fires in a given year represented 66% of the total burned area for that year. Since 2000, there was also a marked increase in the area of the largest fire, which grew from 58% of the total area burned in 2000 to almost 70% in 2006, and 71% in 2007 (1,930, 3,740, and 3,120 km^2 respectively).

Validation

Burned area map validation

Validation of the burned area maps relied on data from different satellite sensors. For burned area delineations we used a total of 1,066 MODIS images (a mean of 133 images per year, range 71-195), and 10 RESURS images (mean 3, range 2-4). MODIS data was available for the entire burning season; RESURS data also covered the entire burning season, except 1998, where the last RESURS image was acquired on August, 20. Validation with Landsat data was based on September or late August imagery, and we used only one late season image per year. Average producer's accuracy of the burned area class in years where at least 200 km^2 burned was 68% (Table 1). Average user's accuracy for the same years was 56% (Table 1). The Kappa coefficient for the same year was 52% or moderate, according to classification scale by Congalton (1996). It was not possible to validate independently 10 out of 23 years (1985, 1990-1995) due to the lack of MODIS, Landsat, and RESURS data. No signs of burned areas were found for 7 of the validated years (1986, 1988, 1996, 1999, and 2003). Official statistics, available since 2002, were well correlated with the estimated trend, but showed generally much less burned area than our classifications (Figure 5).

Temporal distribution and out of season validation

We estimated the temporal distribution of fires for 2000 to 2007 using MODIS active fire hotspots and our burned areas boundaries derived for validation. As reported previously (Carmona-Moreno et al. 2005) burning predominantly started in June and ended in August

(Figure 9). Few active fire hotspots were detected before June and in late August and September. The percentage of active fire counts in August compared to a 3-month total was 26% (32% annual average), and only 9% (12% annual average) of the total number of active fires were detected in September. In August, most of the burning happened in the beginning and the middle of the month. The percentage of active fires detected during the last ten days of August was only 12% of the total number of active fire counts during August (14% annual average).

Burn areas mapped using MODIS data that occurred after the AVHRR images were taken showed that only a small proportion of burns were missed due to the timing of images. With the exception of 2003, less than 2% of total burned area occurred after the AVHRR images were taken (20 to 90 km^2 in a given years). However, in 2003 only 3 fires occurred in total, and one of those 3 occurred after the AVHRR image date and covered 70 km^2 or 63% of the total burned area in that year. Some burning may have taken place outside the windows captured by the AVHRR in those years for which no MODIS data were available. However, field observations, expert opinion and examination of imagery for period before and after fire season indicated that such burning was overall very limited.

Model transferability

We also tested how well our pooled training data could classify burns in years for which no training data were available (or withheld to test the transferability of the training data). Overall, the transferability was moderate. The amount of burned area for a particular year detected without training data from this year ranged from 84% (2002), 52% (2006), 44% (2007), down to as low as 17% (2005), of the area predicted with a full sample. Transferred models were successful in detecting burns that led to the complete removal of vegetation resulted in big areas of bare ground. Poor transferability in 2005 was likely due to a somewhat abnormal fire season. Unlike other test years, most of the burned areas in 2005 happened late in the year and showed low reflectance in visible bands due to large amounts of remaining soot.

Discussion

Our goals were twofold. First, we aimed to estimate changes in burned area before and after the precipitous declines in livestock caused by socioeconomic changes following the breakdown of the Soviet Union. Second, we wanted to develop a method to map burned areas from AVHRR data suitable for the parts of the world for which only data from CLASS were available. Both goals were met successfully.

The main finding from our study was a dramatic increase in burned areas in the arid grasslands of Southern Russia starting in the late 1990s (Figure 5). The area burned each year jumped from almost no fires in the 1980s to large-scale burning, covering up to 20% of study area in a single year after the mid-1990s. Since 1998, on average 1,381 km2 burned per year (9% of the study area, Figure 10). At this rate, the entire area will burn every 11 years. Fires occurred almost exclusively during the driest season of the year (Carmona-Moreno et al. 2005). Unfortunately, no data on lightning occurrences and dry thunderstorms exist for the area, but our field experience suggests that they are rather rare. Thus we assume that most of the fires are human-caused, resulting from transportation, mainly to and from local herding enterprises, hunting activities, including illegal poaching for indigenous saiga antelope (*Saiga tatarica*), and carelessness (e.g., wide use of old machinery, smoking, etc.). There is no history of grassland burning for pasture management in the study area (Y. Arylov, personal communication).

Burned area trends, vegetation, grazing, and socioeconomic changes

The abrupt increase in burned areas that we observed followed a sharp decrease in livestock abundance around the time of breakdown of Soviet Union in 1991. We suggest that the main mechanism leading to the increase in fires was the removal of livestock, predominantly sheep. Reduced disturbance by livestock allowed vegetation to recover, and caused a gradual increase in biomass and fuel connectivity. Grass-dominated vegetation communities (*Stipa* spp.) have increased particularly rapidly (Neronov 1998). Our results suggest that this recovery of grasslands led to an increase of the amount of dry litter that permitted the consequent significant increase in burning.

Increase in burned areas was also concomitant with the reduction of fire suppression activities in Russia, resulting from economic hardship after the breakdown of Soviet Union in 1991, and the major economic crisis of 1998 (Shvidenko and Goldammer 2001). Unemployment caused an increase in the number of visits by locals to the steppe in search of other sources of income. Burning was used to assist poaching of the endangered saiga antelope and led to near obliteration of the last population of saiga in European Russia (Lushchekina and Struchkov 2001). Official burned areas statistics were only available after 2002. The trends in the burned area statistics correlated well with our AVHRR-based estimates, but official statistics highly underestimated the area burned. Underestimation of fires in governmental data is due to the fact that only fires that were actively suppressed were included in official statistics (Soja et al. 2004).

The strong increase of annual burned area in late 1990s in our study region is similar to other parts of the grasslands of Central Asia and Mongolia. Being part of the Soviet block, these countries showed similar socioeconomic changes and declines in livestock numbers. Though fire data are scarce for Central Asia, there has been a 6-fold increase in steppe and forest fires in Mongolia between the periods of 1985-1995 and 1996-1997 (Erdenesaikhan and Erdenetuya 1999; Velsen-Zerweck 2002), similar to the pattern found in our study.

The area that burned most often occurred in a strictly-protected nature reserve (Chernye Zemli Zapovednik) established in 1990, which obtained Biosphere Reserve status in 1993, and covers 1,220 km^2 (6% of the study area). The reserve allows research, conservation efforts, and education, but prohibits any other human activities, like livestock grazing or haymaking. The extra protection may have facilitated fuel buildup in the protected area. Together with the livestock declines in the surrounding areas, this may have caused increasing numbers of fires of medium and large size. Although there is no 'let burn' policy in Russia's protected areas, according to our estimation, the reserve burned almost entirely in 2000, 2002, 2006, and 2007, most likely due to the lack of funds for effective fire suppression.

Satellite data limitations

The only satellite dataset with annual data records spanned from 1985 until present that was available for southern Russia was CLASS-type AVHRR data. Despite the limitations of CLASS-type AVHRR data, we were able to develop a method to map burned areas and to detect long-term trends in annual burned area. However, our accuracy assessment highlighted the limitations of the available coarse-resolution satellite data. We could not use burned area mapping algorithms that require multiple (>2) images for a given year due to the lack of good data in the archives, excessive cloud cover, as well as geometric and radiometric artifacts. In our study area, daily AVHRR data that could be composited were not collected by local stations until the late 1990s. We suggest that the lack of data is largely responsible for the reported classification accuracies, and higher accuracies could have been obtained with if daily acquisitions were available. However, despite its limitations, AVHRR data often provides the only means possible to reconstruct dense time series of burned area estimates for long time spans, prior to the launch of MODIS in 2000. Other remote sensing data, while providing better spectral and spatial resolution, lack temporal resolution, spatial coverage, and the long-term data record necessary to give clear answer to the question of how burned area changed throughout the recent 20+ years. This is especially true for those regions of the world where remote sensing data records are sparse due to poor archiving or lack of acquisitions. Recent developments in remote sensing data distribution policy, such as the 'freeing' of the Landsat data archives, may make it possible to map burned areas with high resolution in time and space. However, we caution that many areas of the world have acquisition gaps, and methods like the one we present here will be needed to estimate burned area trends for long time series.

Validation of the burned area estimates and the burned area mapping approach

Our study dealt with a particularly long time series of satellite observations and various satellite sources were used for validation based on their availability. Though the accuracy of the estimates derived in this study was moderate, similar accuracies were found in comparable environments by studies using better data and more advanced methodology (Roy and Boschetti 2009). Differences among the imagery used for validation may have resulted in some inconsistencies in the reported accuracy rates, which we could not quantify. Generally, the dataset with highest temporal frequency (i.e., MODIS) was best suited to delineation burned areas for our validation dataset , followed by less frequent, but spatially more detailed Landsat/TM, and finally the only data available for late 1990s (i.e., RESURS/MSU-SK)..

Our analysis of the fire season with MODIS active fire data showed that a few fires could have occurred after the dates of our AVHRR images, but burned areas potentially missed were in all likelihood negligible. Our conclusion that fires were essentially absent before 1998 was also supported by our test of the transferability of the decision trees to years for which no training data for burned areas were available. On average of 42% of the burned areas detected with full training data were still detected when training data for a given year were omitted. That suggests that even if our visual interpretation of fires in the AVHRR data prior to 1998 missed fires erroneously, our models would have still captured a considerable portion of them.

We found that the largest fires captured up to 70% of the total area burned in a given year (Figure 6). Burned area classifications in Southern California (Minnich 1983), the Intermountain West (Knapp 1998), Mongolia (Erdenesaikhan and Erdenetuya 1999) and Australia (Yates et al. 2008) found similar patter with the most of the burned area attributed to few large contiguous burns. We caution though, that the share of large burned areas may have been overestimated due to the low resolution bias of AVHRR data, resulting from the 1.1 x 1.1 km pixel size (Boschetti et al. 2004), and our use of a majority filter. Omission of small burned areas with AVHRR data is especially critical in mosaic environments representative of crop production systems (Laris 2005). We cannot rule out that fires were just as frequent prior to 1998, but of such small size that AVHRR missed them all. However, this explanation for the observed increase in burned area is highly unlikely. First, size class analysis with higher-resolution MODIS data during the years since 2000, when fires were widespread, showed a clear dominance of large fires (Figure 8), supported by the contiguous nature of grassland vegetation and the absence of crop production. And second, comparable rates of burning expressed as much smaller fires would require much more dense population (to provide ignitions, but also to suppress fires before they got large) or fine patterns of land use (to prevent fires from getting large). Neither was present in the study area historically.

Lastly, our finding that there was essentially no fire prior to 1998 is supported by the only other remote sensing study in the study area. A change detection for 1989 to 1998 by Hoelzel (2002) did not mention any burned areas as well. Though not quantified, sudden and vast increase in burning in our study area after the mid-1990s has also been mentioned in other studies (Shilova et al. 2007) and is corroborated by members of local communities and other scientists working in the area (Bakinova T.I., Dzhapova R.R., Neronov V.V., personal communications).

Concluding remarks

Our study provides one of the first estimates of burned area trends for the arid grasslands of Central Asia, and represents one of the longest burn area time series obtained using remote sensing here and elsewhere in the world. Drastic socioeconomic changes often lead to changes of a particular disturbance agent. In arid grasslands, the decline in grazing might result in the transition of the ecosystem to the different state. However, changes in vegetation due to reduced grazing likely lead to the substitution of grazing by fire as the main disturbance. Interestingly, the replacement of the disturbance agents did not occur immediately, but exhibited a time lag. Knowing the time lag between disturbances is important in order to make accurate predictions, and we speculate that fire may again decline given continuing restoration of livestock numbers returning to their historic highs (Fig. 1). Whether we are seeing recovery of the ecosystem in the long term or transition to a new state remains a question.

Acknowledgements

We gratefully acknowledge support for this research by the NASA Land-Cover and Land-Use Change Program, the National Geographic Society, and by a NASA Earth System Science Dissertation Fellowship for M. Dubinin (07-Earth07F-0053). We also thank the R&D Center Scanex (Moscow, Russia) for providing RESURS imagery.

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Tables

Table 1. Annual burned area estimation and validation results.

Year	Burned area, km^2	Burned area % total	Producer's accuracy, burned	User's accuracy, burned	Producer's accuracy, unburned	User's accuracy, unburned	Kappa
1985	0	0.0	N/A	N/A	100.0%	100.0%	N/A
1986	25	0.1	0.0%	N/A	100.0%	99.9%	0.0%
1987	0	0.0	0.0%	N/A	100.0%	98.7%	N/A
1988	74	0.4	0.0%	N/A	100.0%	99.6%	0.0%
1989	21	0.1	0.0%	N/A	100.0%	99.9%	0.0%
1990	4	0.0	N/A	N/A	100.0%	100.0%	0.0%
1991	0	0.0	N/A	N/A	100.0%	99.5%	N/A
1992	266	1.4	N/A	N/A	100.0%	98.6%	0.0%
1993	90	0.5	N/A	N/A	100.0%	99.5%	0.0%
1994	0	0.0	N/A	N/A	100.0%	100.0%	N/A
1995	0	0.0	N/A	N/A	100.0%	99.8%	N/A
1996	0	0.0	0.0%	N/A	100.0%	99.4%	N/A
1997	518	2.7	18.5%	82.3%	99.9%	97.8%	29.4%
1998	933	4.9	92.5%	34.2%	90.8%	99.6%	46.1%
1999	0	0.0	0.0%	N/A	100.0%	99.9%	N/A
2000	1,602	8.5	78.2%	48.9%	92.4%	97.9%	55.5%
2001	417	2.2	62.3%	28.4%	96.4%	99.1%	37.1%
2002	2,586	13.7	82.6%	64.1%	92.7%	97.1%	67.2%
2003	34	0.2	28.2%	0.4%	82.5%	99.8%	1.7%
2004	567	3.0	67.8%	32.5%	95.6%	99.0%	41.4%
2005	2,193	11.6	52.0%	63.5%	96.1%	93.8%	50.5%
2006	3,781	20.0	86.5%	73.8%	92.3%	96.5%	74.2%
2007	4,397	23.3	72.8%	75.4%	92.8%	91.8%	66.4%

Table 2. Detection rates of burned areas in different size class categories.

	Burned area patch size (km^2)							
	<10	10-20	20-35	35-60	60-100	100-150	150-250	>250
Number of fires	144	64	48	39	31	19	16	14
Number of fires detected	31	27	23	30	24	16	16	13
Percent detected	22%	42%	48%	77%	77%	84%	100%	93%

Figures

Figure 1 Total sheep population in three administrative regions of the Republic of Kalmykia (Chernozemelsky, Iki-Burulsky, Lagansky) which closely correspond to the boundaries of our study area (ROSSTAT 2003, 2007).

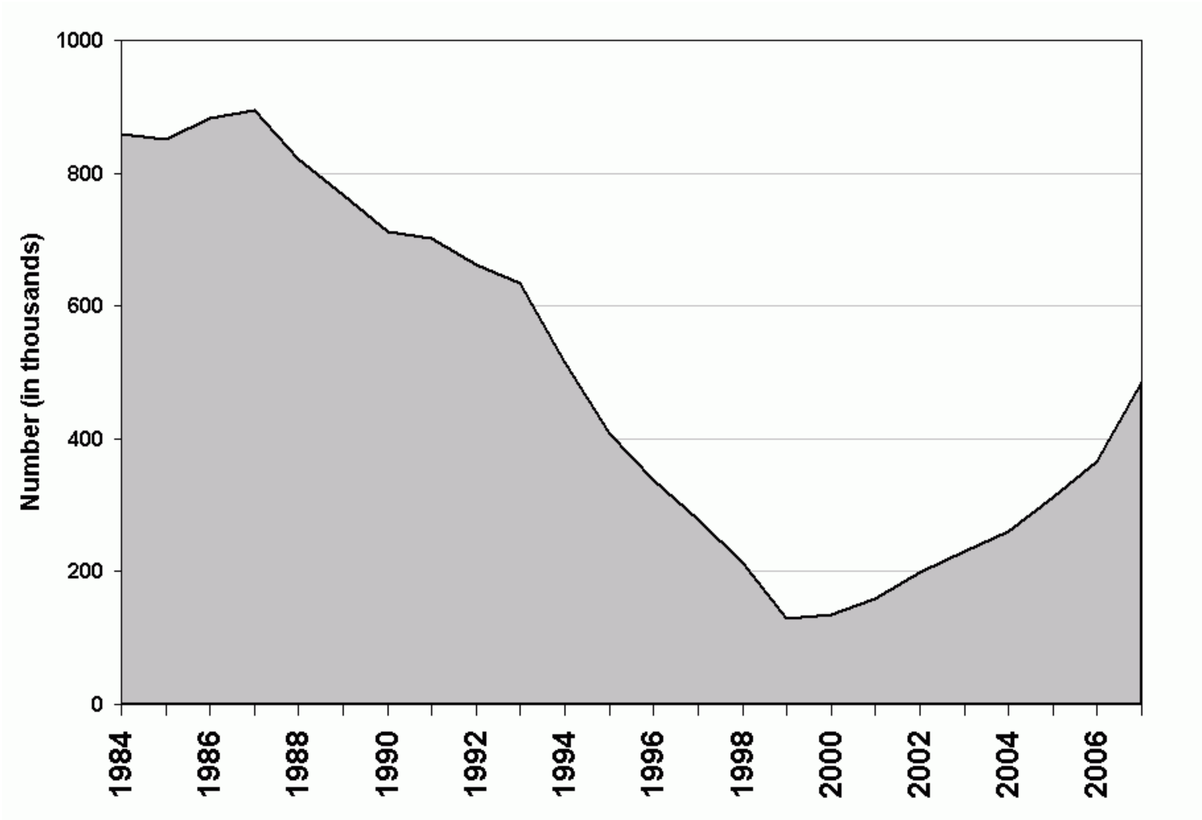


Figure 2. Study area location (hatched polygon with black outline, grey line – Republic of Kalmykia; the grayscale background represents shaded relief (SRTM30).

Figure 3. Feathergrass (*Stipa* sp.) (top) and sage (*Artemisia* sp.) (bottom) dominated communities in the beginning (left) and the end (right) of the vegetation season.

Figure 4. 16-day MODIS reflectance values in bands 1 (triangles) and 2 (squares) for a sample area (median of 9 pixels) affected by fire (hatched line denotes the fire date). Winter-time snow contaminated values are not shown.

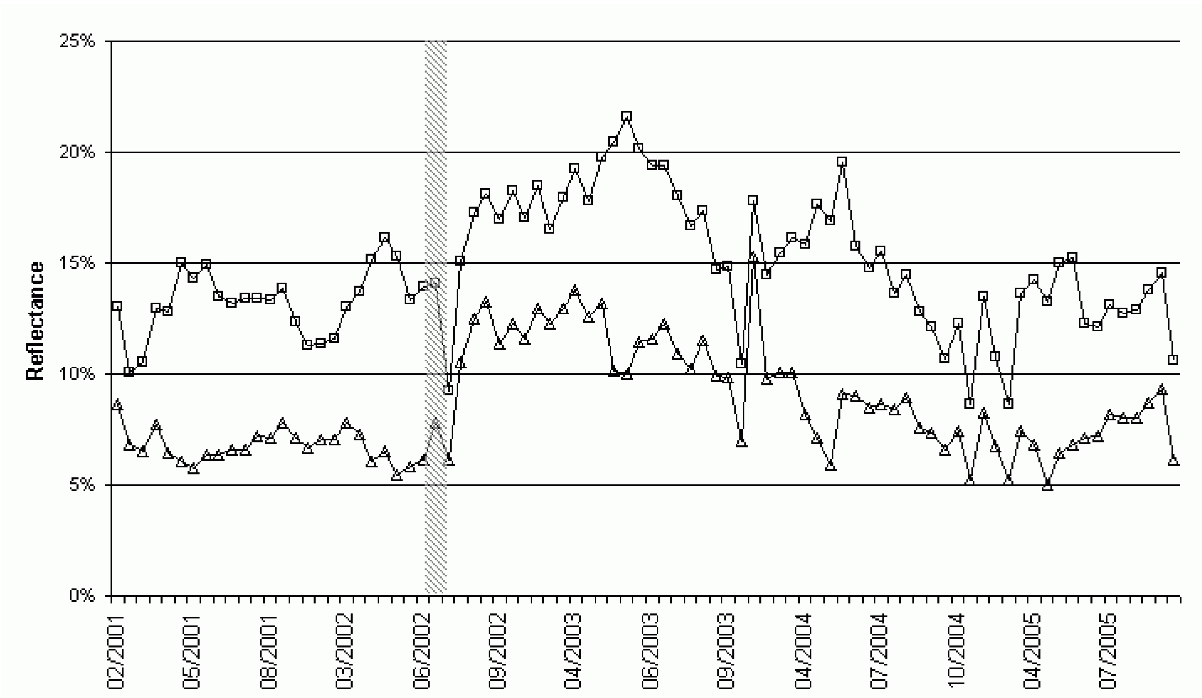


Figure 5. Estimated burned area according to the AVHRR analysis (solid line), validation dataset (dotted line, triangles) and official burned area data (dashed line, squares).

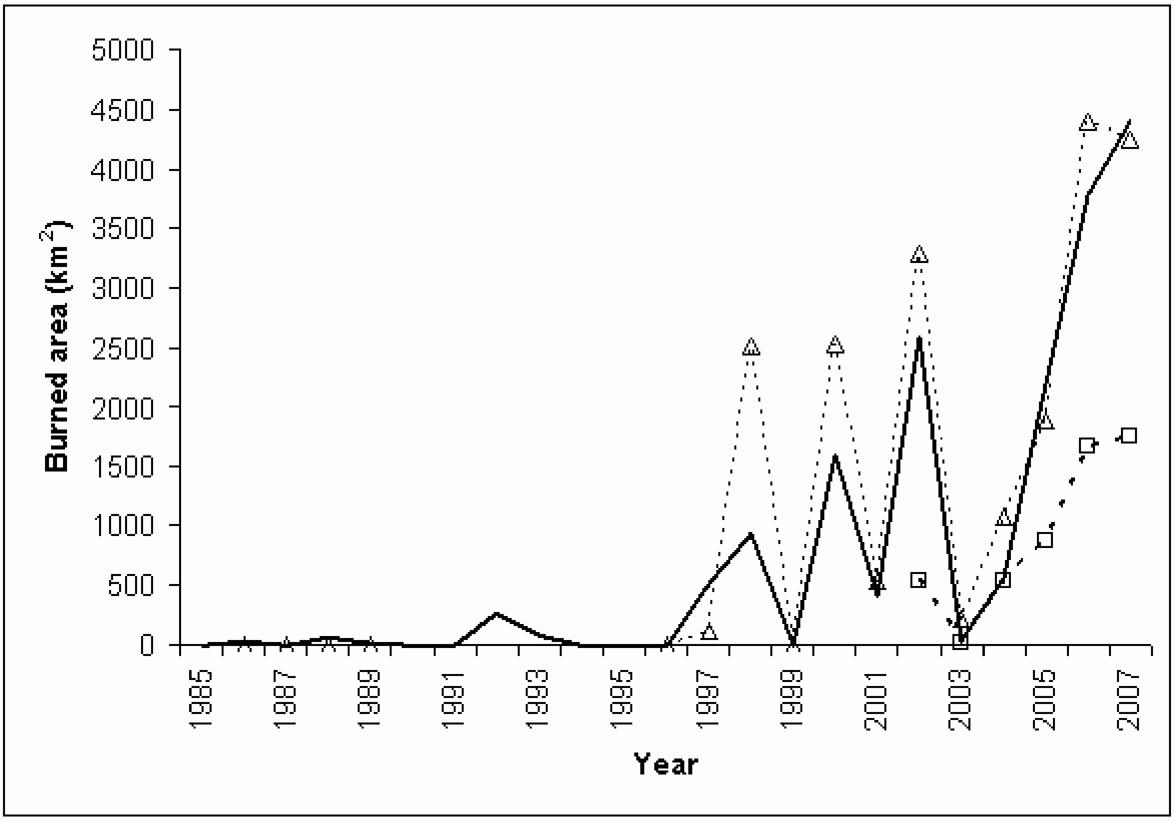


Figure 6. An example of AVHRR-based burned area map for year 2000 (red polygons) comparing with MODIS-derived burned area validation data (outlined in black) shown on top of the AVHRR image from Aug 1, 2000. The boundary of analysis region is shown in magenta.

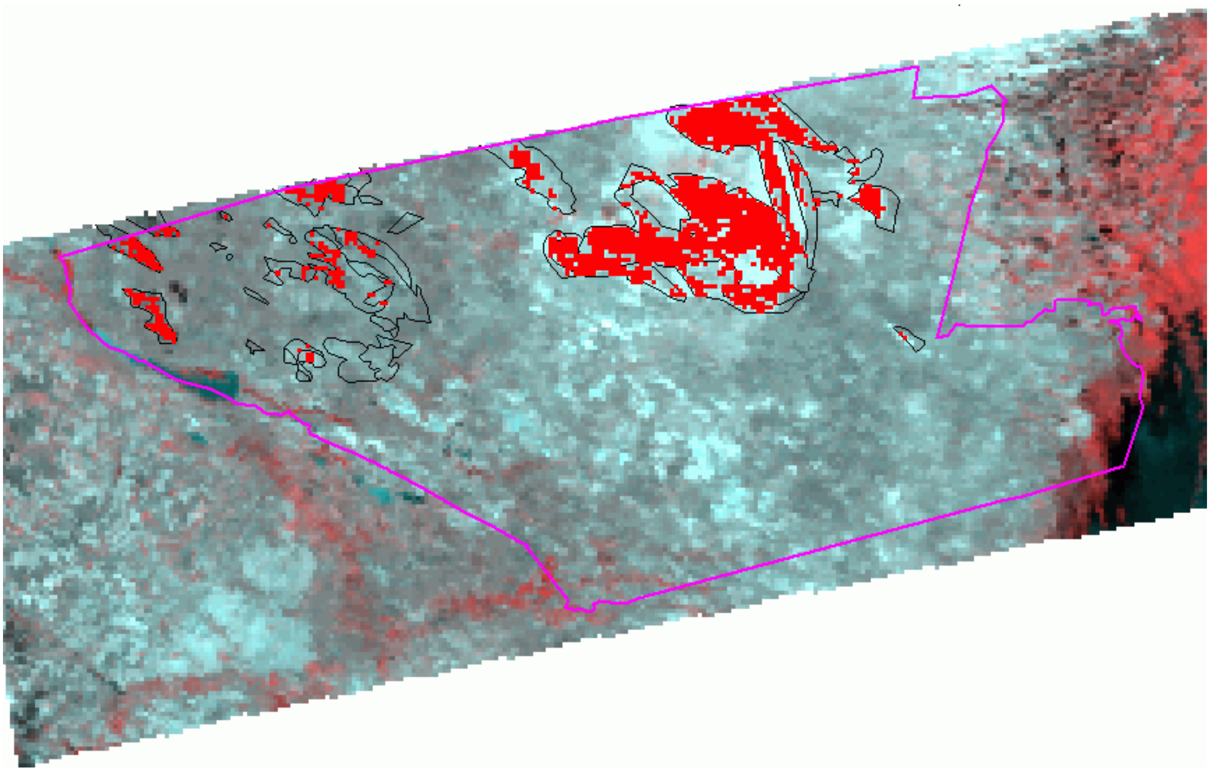


Figure 7. Example of before-burning and after-burning in a Landsat TM image of September, 1988 and an ETM+ image of July, 2001 (5-4-3 band combination). Ubiquitous fire-scars bounded by roads in the 2001 image, most of these scars are from previous years (marked with arrows). The photo represents an area burned in September, 4, 2005, photo taken March, 18, 2006

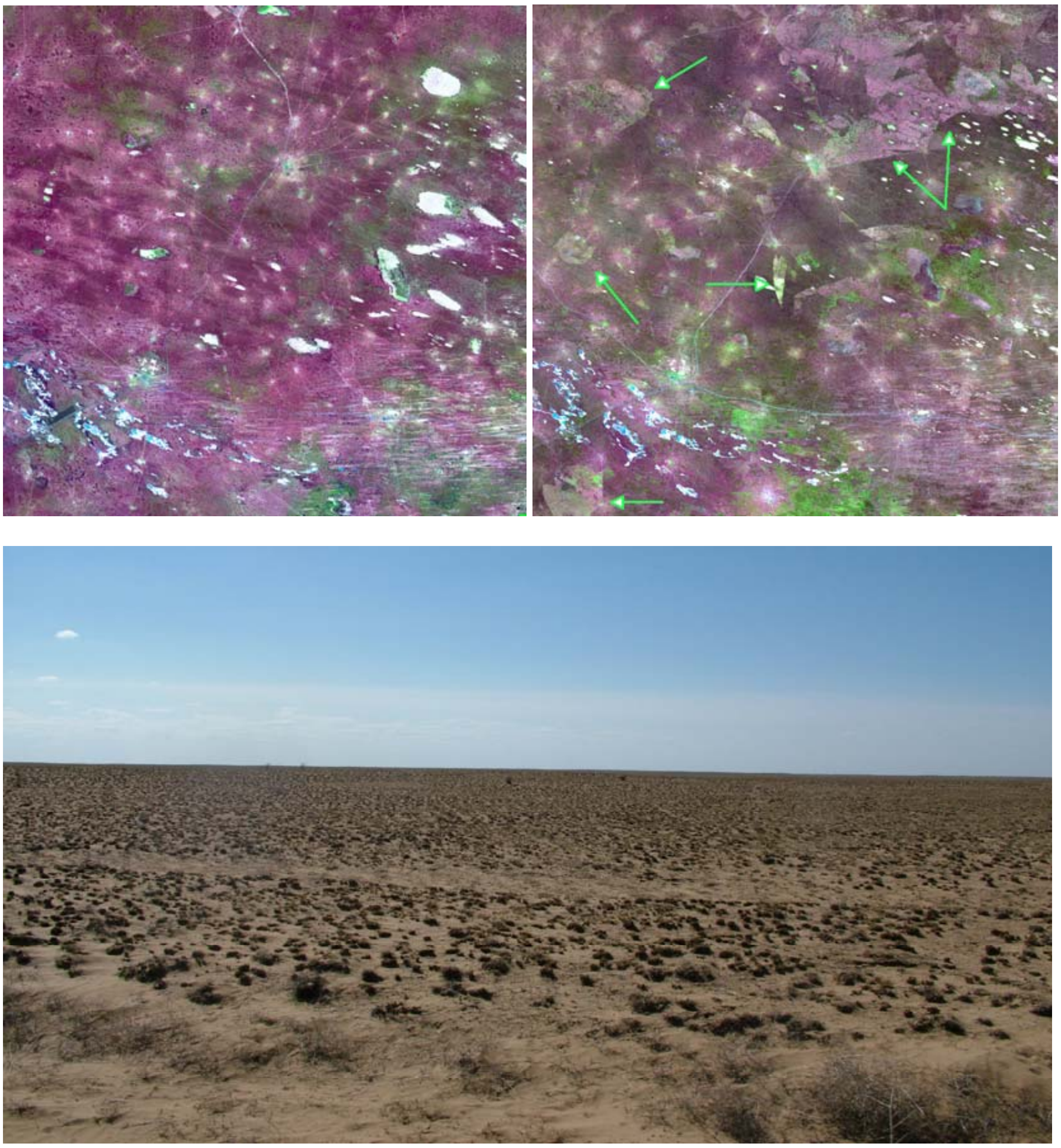


Figure 8. Number (line) and area (bars) of burn areas from 2000 to 2007 summarized by size class.

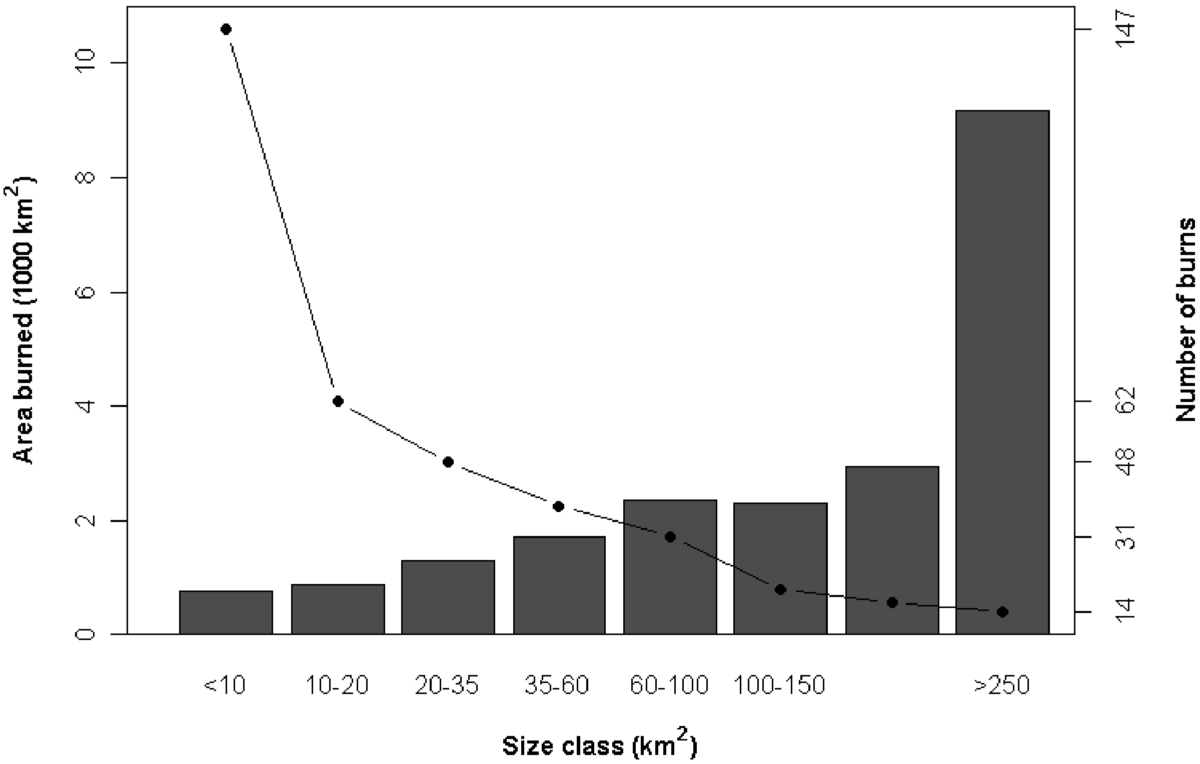


Figure 9. Fire seasonality, i.e., the burned area by month, based on the MODIS image interpretation.

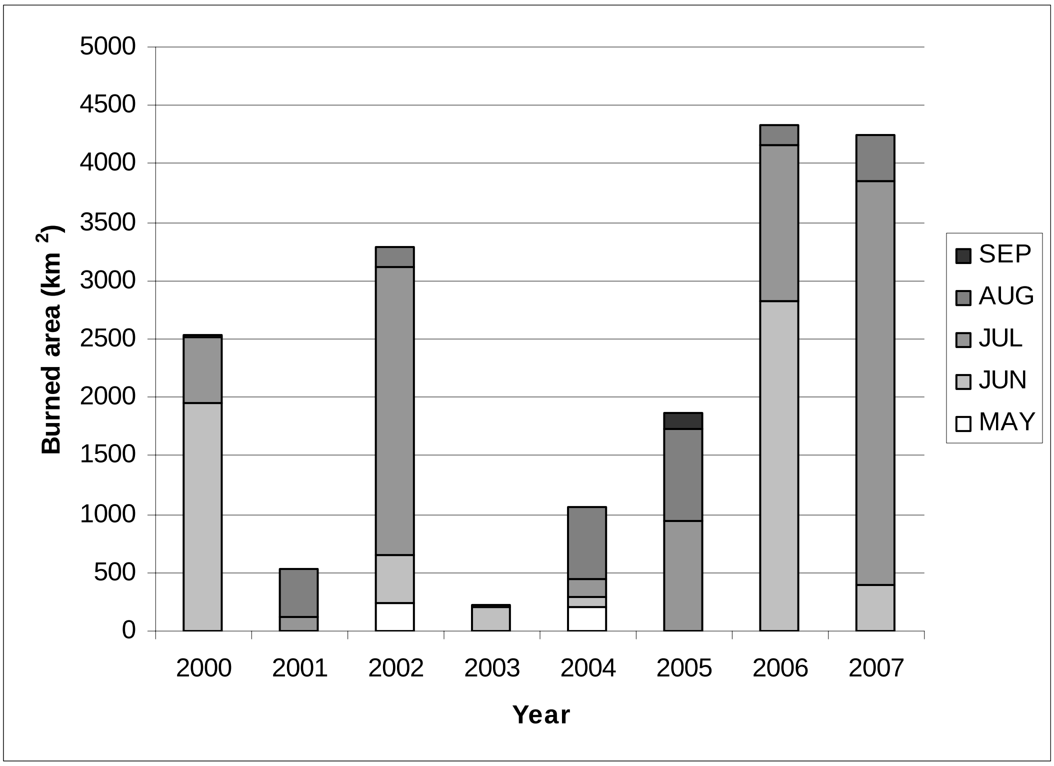
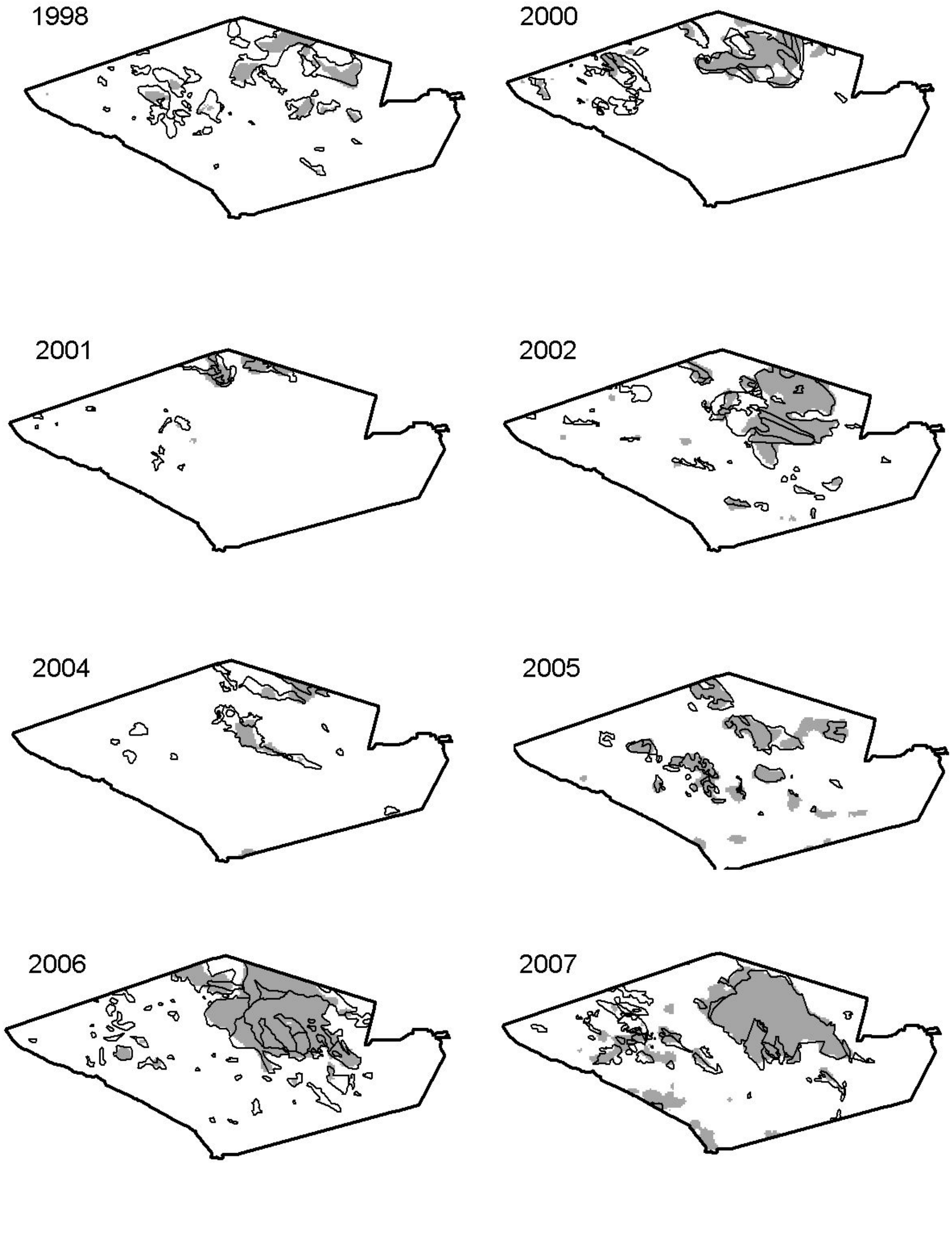


Figure 10. Burned area mapped for selected years with high fire activity. Gray shading – AVHRR-based burned area maps, black outlines – validation burned area maps (1998 – RESURS-based, others – MODIS-based).



Chapter 2: Relationships between climate, livestock, and fire in the arid grasslands of southern Russia

Abstract

Fire is an important natural disturbance process in arid grasslands but current fire regimes are largely the result of both human and natural processes and their interactions. The breakdown of the Soviet Union in 1991 spurred substantial socioeconomic changes and was ultimately followed by a rapid increase in burned area in southern Russia. It is unknown though if the increase in burned area was caused by decreasing livestock numbers, vegetation changes, climate change, or interactions of these factors. Our research goal was to identify the driving forces behind the increase in burned area in the arid grasslands of southern Russia. Our study area encompassed 19,000 km^2 in the Republic of Kalmykia in southern Russia. We analyzed annual burned area from 1985 to 2006 against a range of explanatory variables including livestock population, NDVI, precipitation, temperature, and broad-scale oscillation indices using best subset regression and structural equation modeling. Our results supported the hypothesis that the vegetation recovered within several years after the livestock declined in the beginning of the 1990s, to a point at which large fires could be sustained. Climate was an important explanatory factor of burning, but mainly after 1996 when lower livestock numbers allowed fuels to accumulate. Ultimately, our results highlight the complexity of coupled human-natural systems, and provide an example of how abrupt socioeconomic change may affect fire regimes.

Introduction

Fire is one of the main disturbance agents in grasslands and savannahs, and can be an indicator of ecosystem change. Fire shapes vegetation structure and composition and represents an important land-management tool (Pyne, 1984). Grasslands, woody savannahs, and savannahs contain more than 60% of the annual global burned area (Tansey et al., 2004), and in regions with high aridity, such as Central Asia, grasslands fires account for 80% of all active fire counts (Csiszar et al., 2005). Fuel loads and emissions from grassland burning are relatively small (van der Werf et al., 2006), but grassland fires can foster the spread of invasive species (Brooks et al., 2004), affect wildlife habitat (Archibald and Bond, 2004), and generate air pollution that can spread as far as the Arctic (Stohl et al., 2007). Furthermore, the interactions between grassland fires and human land use may lead to ecosystem degradation, hydrologic changes, soil disturbance, and shrub encroachment (Archer et al., 1995). At the same time, fire is an indispensable component of grassland ecosystems because fires stimulate germination and regrowth (Letnic, 2004), and this is why fires are often set deliberately to enhance pastures and raise the availability of nutritious forage for livestock (Guevara et al., 1999). However, management, conservation, and restoration of arid grasslands require an understanding of the driving forces of burning, and such an understanding can only be gained from long-term fire records. Our goal was to understand the drivers of fire regimes in southern Russia, and particularly to identify the reasons for the marked increase in annual burned area since the mid-1990s (Dubinin et al., 2010).

Wildfire patterns are determined by a multitude of factors, but ultimately the key factors are fuel availability (moisture and amount) and ignition sources (Bond and van Wilgen, 1996; Meyn et al., 2007). In southern Russia, like in many other areas, almost all ignitions are human-caused, and constant in number since human populations have remained fairly stable. Fuel availability, on the other hand, has changed considerably over time, both in terms of fuel amounts and flammability, which in turn is determined by moisture. While fuel moisture is largely controlled by climate, the amount of fuel in grasslands is highly dependent on land use, and especially grazing. Grazing influences the rate of vegetation recovery and directly reduces both amount and contiguity of fuels by consumption, wallowing, and trampling of vegetation, eventually leading to catastrophic shifts in vegetation structure and composition if grazing levels are not moderated (van de Koppel et al., 1997).

Most grasslands are experiencing intensifying land use, and long-term decrease in land use intensity is generally a rare phenomenon. However, periods of massive socioeconomic change may provide unique natural experiments (Diamond, 2001) to study the effects of decreasing land use intensity (e.g., lower grazing pressure) on fire patterns. Institutional changes in the former USSR have repeatedly led to changes in the vegetation, first after October Revolution and the Civil War of 1917-1920 and then after the end of World War II (1941-1945, Zonn, 1995b). Vegetation recovered after these events due to substantial decreases in human populations and livestock numbers, as well as the cessation of other agricultural activities. Vegetation changes were particularly evident in the arid grasslands of the Caspian plains because of the fragility of upper soil layer, largely residing on alluvial sands.

The collapse of the former USSR in 1990/1991 also resulted in a sharp decline in livestock numbers, and detailed livestock data provide a much more detailed picture of the changes in livestock numbers and hence grazing pressure for the 1990s than for prior periods of livestock declines (Figure 11). Soviet era livestock numbers were so high that they caused disastrous environmental effects (Vinogradov, 1995; Zonn, 1995a). Rangeland scientists had calculated that even on the best pastures sheep density should not exceed 28-30 heads/ha, but sheep density reached 130 heads/ha by 1975, and continued to increase until late 1980s (Zonn, 1995b). Overgrazing proved to be ecologically disastrous for the fragile arid ecosystems and ultimately led to desertification starting in the 1960s and continuing until the early 1990s (Saiko and Zonn, 1997). The subsequent sharp decline in livestock numbers was paralleled by other substantial changes in ecosystem processes, including fires. During the period of overgrazing, fires were virtually absent in our study area (Dubinin et al., 2010). However, within half a decade after the declines in livestock numbers, annual burned area increased very rapidly (Figure 11), especially on sites that had previously been highly degraded (Dubinin et al., 2010).

The relationship of climate and burned area can be both direct and indirect. It is generally assumed that in seasonally-dry, biomass-poor grasslands fuel amounts are more important drivers of fire than climate as fuels are desiccated during dry period regardless of climate during the year (Meyn et al., 2007). However, the relative importance of climate may increase as land use intensity weakens (Meyn et al., 2007). Higher precipitation, especially during fire season can impede burning due to higher fuel moisture, and thus lower fuel availability. On the other hand, high precipitation during the growing season prior to the fire season can indirectly cause an increase in burned area because the rain results in more abundant fuel (Flannigan and Wotton, 2001; Pausas, 2004). In other words, the amount of burning in a given year might depend on the weather during the previous year, which may determine fuel availability (Knapp 1998).

When examining the effects of climate on annual burned area it is important to consider global climate oscillation patterns as important explanatory factors (Flannigan and Wotton, 2001). Via teleconnection mechanisms, climate oscillations can be synchronized with fire weather conditions across large areas. For example, El Niño - South Oscillation (ENSO) and North Atlantic Oscillation events are associated with increased fire activity in different regions of the world (Le Page et al., 2008; Greenville et al., 2009). Similar linkages occur between fire activity and the Arctic Oscillation (Balzter et al., 2005). The suggested mechanism for the relationship between fire and broad-scale climate is via local climate. In semi-arid and arid ecosystems, where precipitation is the limiting factor for fuel production and hence fires, increased rainfall might result in a pulse of productivity and thus higher fuel availability in subsequent years (Holmgren et al., 2006). However, the relationship between broad-scale climate oscillations and local temperature and precipitation are regionally specific and additional analysis is required to establish how broad-scale climatic patters affect fire patterns (Schonher and Nicholson, 1989; Keeley, 2004).

However, any interactions between vegetation, climate, and burned area may be overridden by land use. Grazing pressure above a critical threshold can limit vegetation to such an extent that fires can no longer occur, irrespective of climatic conditions (van de Koppel et al., 1997). This means that climate probably did not predict burned area while livestock numbers where high under Soviet agricultural practices. However, once livestock numbers declined below a critical threshold vegetation recovered to a point where fires could be sustained, and it was after these thresholds had passed that climate exerted its effects on the burned area of a given year.

In summary, the overarching goal of our study was to understand what explains fire regimes during rapidly changing socio-economic settings. Our general hypothesis was that socio-economic changes that followed the breakdown of the Soviet Union lowered grazing pressure, allowed vegetation to recover, and increased fuel availability, thereby providing the necessary prerequisite for fires, the amount of which was then controlled by climate. From this general hypothesis stemmed several specific hypotheses which we tested. We hypothesized that the burned area in a given year was:

a) positively correlated with the amount of vegetation during the spring growing season immediately prior to the summer fire season, because of higher fuel availability;

b) positively correlated with the amount of vegetation during the secondary growing season in the fall of the previous year, again due to higher fuel availability;

c) positively correlated with climatic conditions that are favorable for vegetation growth during spring and fall of the previous year, also due to higher fuel availability;

d) significantly more correlated with climate during the period after livestock numbers had declined and grazing pressure lessened, because vegetation was able to recover to a point above which fires could occur;

e) negatively correlated with livestock numbers, because fewer livestock result in more abundant fuel;

f) and negatively correlated with higher precipitation and lower temperature during fire season because higher fuel moisture impedes fires.

In order to test these hypotheses, we conducted statistical analyses to identify the best set of variables to explain annual burned area from 1985 to 2006. In addition, we conducted our analysis separately for the two time periods before and after the breakdown of the Soviet Union to test if the relationships between burned area and livestock, vegetation, and climate were changing from the Soviet to the post-Soviet period. To our knowledge, no previous studies of comparable temporal extent have attempted to explain burning dynamics in the grasslands of southern Russia and Central Asia.

Methods

Study area

Our study area was located in the grasslands of Southern European Russia and occupied about 19,000 km^2 of the Republic of Kalmykia and Astrakhan Region (Figure 12). We had estimated annual burned area for this study area in previous research (Figure 11, Dubinin et al., 2010).

The climate of the study area is arid, with hot, dry summers (mean daily temperature of +24°C in July; max +44°C, 280 days of sunshine per year on average). Annual precipitation is 150 to 350 mm (with a mean of 286 mm from 1985 to 2007). Summer droughts are common, and most of the precipitation falls in spring and fall, coinciding with the two major growing seasons (Walter and Box, 1983). The topography is predominantly flat with a mean elevation of -15 m below sea level. The study area has a complex geological history of transgressions and regressions of the Caspian Sea. Soils are characterized by a gradient from sandy aeolian deposits and sandy loams in the southeast corner of the study area, to clay loam in the northwest (Kroonenberg et al., 1997).

Vegetation associations are typical for the northern Precaspian Plain and include both steppe and desert types. The main vegetation associations are shortgrass steppe (*Stipa* spp., *Festuca* spp., *Argopyron* spp., *Anizantha tectorum,* and other graminoids) and sage scrub (*Artemisia* spp., *Kochia prostrata*) (Golub, 1994). Shortgrass steppe is characterized by a short growing season in April and May and rapid senescence in the dry summer. The perennial grasses are well adapted to fire due to dense bunches, which protect seeds and basal meristems with tightly packed leaf-sheaths, and which can resprout after fire. Both annual and perennial grasses generate abundant fuels. Sagebrush (*Artemisia* spp.) dominated shrublands have less biomass, but a longer growing season, and sometimes exhibit a second vegetation peak in the fall and early winter (Kurinova and Belousova, 1989). *Artemisia* spp. is more susceptible to fire because its buds are situated above ground and can be killed or damaged by fires. The lack of fire tolerance by *Artemisia spp.* can lead to its substitution by *Stipa* spp. and other graminoids (Neronov, 1998). The primary human land use of the grasslands in the study area is for grazing by domestic livestock, mainly sheep, and to a lesser extent cows and goats. The study area is sparsely populated (population density 0.8 to 1.4 persons/km^2, (CIESIN and CIAT, 2005).

Broad-scale burning in the region can be traced as far back as 18th century (Pallas and Blagdon, 1802). However, little is known about spatial and temporal dynamics of fires both historically and in the recent past since practically no research has been done on fire patterns in southern Russia prior to our research and governmental statistics are largely inaccessible, incomplete, and biased (Dubinin et al., 2010).

Study period and data sets used

We studied burned area and its drivers from 1985 to 2006. The choice of the study period was determined by satellite image availability, and by the significant changes in land use that occurred after 1991. Our 23-year burned area record is one of longest derived from coarse resolution satellite data. Other burned area studies of such a long time span have either focused on forests and mesic grasslands or used indirect or approximate estimates of burning (Beckage et al., 2003; Mouillot and Field, 2005; Chuvieco et al., 2008).

Fire

Remote sensing data are the primary source of burned area information because they can provide both long and dense time series that can be used for statistical analysis (Kasischke and Penner, 2004). Historical and official records of burning in our study area are scarce and unreliable (Dubinin et al., 2010). Charcoal accumulation and tree-ring methods are not representative in grasslands because of low temporal resolution and low woody biomass. Our data set of burned area in Southern Russia consisted of 23 raster maps representing burned and non-burned areas for each year from 1985 to 2006. Burned area maps were generated based on National Oceanic and Atmospheric Administration (NOAA) Advanced Very High Resolution Radiometer (AVHRR) Local Area Coverage data (Dubinin et al., 2010). The resolution of the burned area maps was 1.1 km pixel size. Burned areas were mapped with supervised decision trees applied to an annual image pair representing one image taken during the peak growing season in the spring right before the fire season, and a second image taken right after the fire season in late summer (Dubinin et al., 2010).

Livestock

Livestock population data were obtained from Ministry of Agriculture of the Republic of Kalmykia and represented the total population of sheep and goats for the study area. Though goats are combined with sheep in official statistics, its share is small (less than 4%). Cattle numbers are an order of magnitude smaller than those of the sheep (Kalmstat, 2008).

Vegetation

In terms of vegetation data, we focused on vegetation indices that served as a proxy for fuel amounts. The Normalized Difference Vegetation Index (NDVI) is significantly correlated with aboveground net primary production (ANPP) in many ecosystems, including grasslands (Tucker et al., 1985; Paruelo et al., 1997). We thus summarized vegetation as the mean monthly NDVI averaged for the entire study area. NDVI data were extracted from the GIMMS NDVI dataset (Tucker et al., 2005). GIMMS NDVI data are available twice a month from 1981 to 2006 and have 8-km resolution. GIMMS data are well suited for vegetation studies in semi-arid regions (Fensholt et al., 2009).

Climate

Climate data were extracted from the Climate Research Unit Time Series product version 3 (Mitchell and Jones, 2005) which included 0.5° gridded mean (T) and maximum temperatures (TMX), as well as precipitation data (PRE) interpolated from meteorological stations (New et al., 1999). In addition to local climate measures, we also looked at broad-scale climatic oscillation indices. Because it was not clear which broad-scale climatic pattern might affect local climate in our study area we chose three the North Atlantic, the East Atlantic/Western Russia, and the Arctic oscillations index, abbreviated NAO, WR, AO respectively (Barnston and Livezey, 1987; Hurrell et al., 2003). The positive phase of AO indicates higher pressure in mid-latitudes, which results in more precipitation and enhanced greening in Central Asia during spring (Buermann et al., 2003). The positive phase of the NAO is associated with anomalous low pressure in the Subarctic, higher precipitation in the oceans between Scandinavia and Iceland and lower precipitation in Southern Europe (Hurrell et al., 2003). During positive phase of the WR, temperatures above North Caspian region and West Russia are drier then normal (Krichak et al., 2002). Climatic indices were averaged by season and year, thus each oscillation was represented by 5 variables. Additionally, we calculated seasonal and yearly precipitation and temperature summaries.

Statistical approach

Bivariate analyses

We tested our hypotheses about the relationships between burned area and potential predictor variables with Pearson's correlations indices and linear regression models with and without interaction terms. Log-transformed burned area was the response variable. Predictor variables included livestock, vegetation, and climate, both for the current and the previous year. Residuals were examined for possible temporal autocorrelation using Durbin-Watson tests and autocorrelative function plots (maximum time lag = 10 years). Heteroscedasticity was checked for via visual examination of normality plots. To identify the best predictors in our candidate models, we used best subsets analysis. The number of times that a particular variable entered the model was calculated for 20 best models. Models were ranked according to their adjusted r-squared values. All statistical analysis were performed in CRAN R (R Development Core Team, 2009).

Initial analyses showed that spring NDVI was a strong predictor of burned area, and this raised the question what determined spring NDVI. We used the same statistical approach as outlined above, but used spring NDVI as the response variable, which was then modeled with the same set of explanatory factors, except for measures made later than spring.

Last but not least, we studied the effect of climate oscillations on local climate. Initial analyses had shown strong correlations between climate oscillations at broad scales and local measures of temperature and precipitation.

Regression modeling for before and after periods

One of our hypotheses was that the relationship between climate and fire was fundamentally different before and after the breakdown of the Soviet Union, because the resulting declines in livestock numbers allowed vegetation to recover to a point above which fires were possible, and climate could exert control on burned area totals. In order to test this hypothesis, we stratified the data into two periods of time representing 'before' and 'after' the breakdown of the Soviet Union. We chose 1996 as a splitting year, resulting in 11 observations for the 'before' period and 12 observations for 'after'. We chose 1996 because previous research found that 5- to 6 years were required for the recovery of vegetation after grazing declined (Bananova, 1992; Zonn, 1995b). To determine if slopes of the regressions were significantly different between before and after periods we used regression models in the form of:

$$Y = 1+x+dummy+dummy:x$$

Where *x* was a given independent variable, *dummy* indicated the period (0: before, and 1: after), and *dummy:x* represented an interaction between the dummy variable and the independent variable. We checked for changing variance structure by using a mixed-effect model that included group-effects. ANOVA was used to check if the more complex model was significantly different from the more simple one (i.e., the null hypothesis was that variance was not significantly different before and after). If we were unable to reject the null hypothesis, then we used simpler model, and otherwise the grouped variance mixed effect model. We were particularly interested to test if there were significantly different slopes in the regression lines for particular variables in the 'before' and 'after' periods. Only variables which showed statistically significant correlations in either of the periods were included in these analyses.

We also selected the best 2-variable regression model to estimate coefficients for two separate regressions, one for 'before' and one for 'after'. We tested if coefficients were significantly different using Chow tests, commonly used in time series analysis to test for the presence of a structural break (Chow, 1960).

Structural equation modeling

As stated in the Introduction, our general hypothesis was that socio-economic change, grazing pressure, vegetation change, fuel amounts, and climate all were related, and that their interactions ultimately determined burned area. The challenge was to identify these complex relationships. Regression analysis is well suited to identify significant predictors of burned area, but less powerful to assess the relationships and interactions among the predictor variables (and strong relationship raise collinearity concerns).

Structural equation modeling can examine possible causal pathways among intercorrelated variables, identify associations among variables while statistically controlling for other model variables (i.e., to partition relationships), and examine the likelihood of alternative models given the data at hand (Bollen, 1989). To facilitate our analyses and interpretations, we developed a meta-model, which represented main linkages between system components without statistical details. Our bivariate correlation analyses functioned as an indicator selection analysis for the structural equation model, and helped us to identify one or more representatives for each entity of the conceptual model, which included global climate that influenced local climate. Local climate, livestock and burning all influenced fuels, and fuels in turn influenced burning. Significant variables indicated causal relationships and were used to construct a structural equation model (Grace, 2006). Nested models included a common set of variables and assumptions about dependencies (e.g., NDVI depends on livestock), but differed in the pathways deemed to be significant. We used maximum likelihood procedures to estimate coefficients and to evaluate model goodness-of-fit. Sequential application of single-degree-of-freedom $\chi 2$ tests was used to determine which pathways should be retained in the final model. Decisions to keep or remove a variable or path from the model were also informed by modification indices (Jöreskog and Sörbom, 1984; Grace and Bollen, 2006). The structural equation modeling was conducted in Amos 5.0 (Amos Development Corp, 2003).

Results

Our prior analysis of remote sensing data had shown that burned area increased very rapidly in southern Russia after 1996 (Figure 11, Dubinin et al, 2010). Statistical tests confirmed that burned area in the 1997-2006 period was significantly higher than in the 1985-1996 period even though there was a high variability among years (1044 km^2 (±1032) versus 37 km^2 (±66), $t = 3.08$, $P = 0.013$, Figure 13).

Livestock and fire

Livestock numbers were significantly lower in the 1997-2006 period, and showed the strongest percent change (Figure 13). Both current and previous year livestock were negatively correlated with burned area, but correlations were only significant when analyzed for the entire study period and not significant when the dataset was split into 'before' and 'after' periods. Burned area was more strongly correlated with livestock numbers in the previous year than livestock numbers in the current year. Both current and previous year livestock were among the five top bi-variate models and statistically significant at the $p < 0.05$ level (Table 4).

Vegetation and fire

Mean April NDVI was significantly higher in the 'after' period compared to the 'before' period, but while May NDVI showed the same trend, the difference was statistically not significant (Figure 13 and Table 3). However, April NDVI and even more so May NDVI were the most powerful explanatory variables predicting burned area (May NDVI, p=0.0012, r=0.66) (Table 3). Correlation was particularly high for the 'after' period. However, neither spring NDVI measure was significantly correlated with burning in the 'before' period only. Among our other vegetation measures, during the 'before' period, only September NDVI in the previous year was significantly correlated with burned area. Slopes of the relationships between May NDVI and fire were significantly different according to our models (Table 7). Best subsets analysis showed that May NDVI was present in 12 out of the 20 best candidate models (Table 8).

Climate and fire

Correlations of burned area with precipitation and temperature were generally weaker than with vegetation. However, we did find significant positive correlation for precipitation in April (Table 3). The only significant correlation with temperature was for August, which was positive. Interestingly, the daily temperature range in August was more strongly correlated with burning than maximum temperature itself. Furthermore, mean temperature, temperature range in August, and precipitation showed statistically significant differences between the 'before' and 'after' after periods, although the significance of the difference in precipitation was marginal (Figure 13). Summary climate averages were highly correlated with monthly values (i.e., the correlation between summarized maximum summer temperatures and maximum temperature in August was 71%). Correlations between burned area and climate variables were similar in the 'before' and 'after' periods, and correlations were not significant. Only summarized summer precipitation was found to be significantly negatively correlated with burning during period 'after'. Climate measures from the previous year were not significantly correlated with burned area either.

The relationships of burned area with climate oscillation indices were complex. Several climate oscillation indices were significantly correlated with burning either 'before' or ‘after’, but only the previous year fall WR index showed any evidence of significant change between the two periods (Table 3). However, out of 6 indices that were significantly correlated with burned area, 3 were previous year fall indices. Previous year average fall WR index was a particularly strong predictor and was included in 7 of the 20 best models (Table 8).

Relationships among spring vegetation, climate, and livestock

Vegetation productivity in spring was consistently positively correlated with spring precipitation and several fall and winter oscillation indices, and consistently negatively correlated with livestock numbers in the previous year (Table 6). Not surprisingly, NDVI in May was strongly correlated with NDVI in April. Interestingly, precipitation in the previous year was better correlated with spring NDVI during the ‘before’ period and the entire time series, and the correlation was statistically not significant during the ‘after’ period. We also found several strong correlations between spring NDVI and climate oscillation indices for the previous year, especially during the ‘after’ period.

Local and broad-scale climate

High correlations between burned area and climate oscillations did not correspond to strong relationships between climate oscillation indices and local measures of temperature and precipitation. Though there were some statistically significant correlations between precipitation and temperature and oscillation indices for particular periods, correlations were weak (Table 5). For example, we found a significant negative correlation between both spring and summer maximum temperatures versus the NAO index in April, and cumulative summer precipitation was significantly correlated with the August WR index. However, none of the fall climate oscillation indices, which were well correlated with burned area, were significantly correlated with any precipitation of temperature measures. At the same time, there were a number of other correlations that were hard to interpret. For example, the April NO index was significantly correlated with August temperatures, and the winter WR index was correlated with April precipitation. Spatial correlation fields calculated for fall precipitation and temperature and respective oscillation indices confirmed that correlation between broad scale climate and local climatic patterns in the study area were not statistically significant (Figure 14).

Multiple regressions

The model yielding the highest adjusted r^2=0.73 included previous year AO, NDVI5 and their interaction. However, previous year AO was not statistically significant by itself ($p = 0.1$) and the interaction term only marginally significant ($p = 0.05$) at the 0.05 level. The model where all three parameters were statistically significant at 0.05 level was ranked third, had a adjusted $r^2 = 0.58$ and included May NDVI, livestock population and their interaction (Table 2). Five best models with broad-scale climate indices omitted yielded smaller adjusted r^2 values and were also dominated by NDVI.

Structural equation modeling

The final structural equation model included 6 exogenous variables and one endogenous variable ($\chi^2 = 7.7$, df = 13, Probability level = 0.862, Figure 15). Results provided by structural equation modeling clarified a number of relationships that were not clear from the univariate and multivariate analyses. The structural equation modeling suggested that livestock mainly influenced vegetation productivity in April, but not in May. Vegetation productivity in April in turn was more influenced by the cumulative precipitation of the previous year than by precipitation in April itself. Structural equation modeling confirmed that vegetation productivity in May was dependent on April precipitation and on April NDVI. There was no indication that vegetation productivity in April by itself influenced burning. In models that included pathways between livestock and burned area, and between April NDVI and burned area, these pathways were all non-significant. The pathway between precipitation during the fire season and burned area was not significant either and we removed this variable. Because the bivariate analysis did not reveal any mechanisms via which climate oscillations influence burned area, we did not include oscillation indices in the structural equation modeling.

Discussion

Our study area experienced a rapid increase in burning after the collapse of the USSR, and our result highlight that these fire patterns were the result of interactions between decreasing livestock densities and recovering vegetation, modulated by climate. The structural equation models in particular highlighted the complexity of the system, where factor that influenced burning the most in their turn were dependent on a host of other factors.

Livestock

Livestock was the main determinant of the amount of vegetation, as measured by NDVI. Our analysis supported the hypothesis that livestock was significantly related to the increase in burned area through the amount of fuels. Though this relationship would seem logical, the decrease of livestock and following increase in burning had rarely been documented explicitly and evidence is mostly historical (Savage and Swetnam, 1990). During the period ‘before’ high livestock numbers masked out the relationships between burned area and other variables such as climate. In the period ‘after’, the release of livestock pressure has led to accumulation of fuels and consequent increase in burning. To some extent, the consumptive role of livestock was thus substituted by fire itself (Bond and Keeley, 2005). This is different from what was observed in other areas of the world, for example in the Sahel, where lands remained barren for 20 years after overgrazing (Sinclair and Fryxell, 1985).

However, the correlation between burned area and livestock numbers was not as strong as we expected, possibly due to possible threshold effect (Turner and Gardner, 1991; van de Koppel et al., 1997). The pattern of the relationship suggests the existence of a critical threshold in livestock numbers that was reached in early 1990s. After reaching this threshold, livestock density was no longer high enough to impact vegetation on a broad scale. The recovery of the vegetation was also coupled with successes in restoration activities and creation of nature reserve occupying a portion of the study area (Zonn, 1995b).

Vegetation

The decrease in livestock numbers was accompanied by an increase in vegetation productivity, especially in spring. Strong relationship between spring NDVI and burning supported the hypothesis that fuel amount before the fire season was significantly lower when livestock numbers were high and fire was not able to spread far. However, while NDVI was shown to be strongly related to primary productivity it is not a perfect measure of biomass (Box et al., 1989; Paruelo et al., 1997). The fact that we did not observe strong change in NDVI between 'before' and 'after' suggest that compositional and structural vegetation change might have occurred. This is consistent with long term plot-level studies of grass-dominated communities after release of livestock pressure that reported increasingly higher presence of perennial fire adapted grasses, such as *Stipa capillata* (Neronov, 1998). Such change has resulted in considerable increase in biomass, from 0.5 to around 1 ton/ha aboveground biomass of *Stipa* spp. in *Stipa* spp. dominated communities (Dzhapova, 2007). Increase in grassy fuels concomitant with an increase in burning is consistent with other parts of the world and represents a positive feedback loop (D'Antonio and Vitousek, 1992; Brooks et al., 2004).

Statistical evidence confirmed the hypothesis that burning was positively correlated with vegetation conditions during secondary vegetation season during previous year. Burning during the ‘before’ period was linked to previous year vegetation productivity of the secondary growing period (i.e., fall NDVI). Fall NDVI represents secondary growth that is not consumed by fire and which becomes fuel in the form of the dry litter in the subsequent year. The lack of a relationship in the ‘after’ period can be explained by the increase in burning itself, leaving less vegetation for the secondary growing season. Alternatively, we could have missed some of the previous year effects due to litter accumulation, because non-photosynthetically active vegetation is not captured by NDVI but can contribute greatly to fire spread.

Climate

Climate influenced burning both during the summer fire season and during spring green-up. During the fire season high precipitation and low temperature led to increased fuel moisture and inhibited burning in some years (e.g., in 2003). Strong relationship existed between late summer maximum temperature and burning overall. However, summer precipitation was significantly correlated only during period ‘after’. Comparison of change in climatic variables showed that the period ‘after’ experienced overall hotter summers.

High precipitation during green-up may have caused more fire due to more abundant fuel (Buermann et al., 2003). The amount of biomass in high precipitation years exceeds the average by 20-30% depending on vegetation type (Dzhapova, 2007). According to our correlation analyses we found support for our hypothesis that spring precipitation was significantly related to burning. However, prior-year precipitation was only marginally (r =

0.42) related to vegetation productivity in April. The lack of a relationship with prior-year precipitation suggested that after the decrease in livestock numbers, the amount of fuel increased to a point when fires occur in any year irrespective of additional precipitation during the previous year.

The representation of climate by station-based local climate measures could be unsatisfactory due to lack of stations and might be improved by using broad-scale climatic indices instead. Relationship between broad-scale climatic variables, such as previous year WRFall, and burned area yielded significant correlations. Low values of WR index indicate milder and wetter weather in the region, which in turn might have lead both to more fuel production and increased fuel moisture during that year. Particular importance of both indices during 'before' period showed that fall precipitation was a more important determinant of burning before 1996. Secondary growth during fall period of the previous year became dry litter to burn next year. The relationship between fall NDVI and climate variables is also supported by relatively high correlation between previous year NDVI9 and burning during 'before' period. However, though such explanations were plausible, we were not able to confirm them with our precipitation data. Different oscillation indices often showed high correlations without clear linkages to local climate. High correlation was often found for seasonal indices that were timed well before or after corresponding measures of local climate. We thus caution that broad-scale climate variables should be used with care. In our case study, mechanism of broad-scale climate influence on burning remained unknown, and this raised the question of how such indices were related to local climate in other studies, where the relationship with local climate variables was not explored, but oscillation indices were used to explain burning (Greenville et al., 2009).

Overall the effect of current climate was stronger than that of climate in the previous year. This was not consistent with the findings from the analogous ecosystems in the Great

Plains of the U.S., California chaparral, and arid spinifex grasslands of Australia where one- and two-years previous year climate variables were found to be better correlated with burning than current-year variables (Knapp, 1998; Keeley, 2004; Greenville et al., 2009). Our analysis of the relationship between climate and vegetation productivity showed that previous year climate had little effect on vegetation during period of intensive burning. However, correlations of burned area with a range of broad-scale climatic indices, especially during fall, and the results from the structural equation modeling suggested that both precipitation and temperature data might not be adequate for our study area, or that the effect of broad-scale changes on fires is transmitted through other parameters that we did not measure, such as evapotransipation.

In summary, we conclude that though climate was an important factor affecting burned area, especially in the 'after' period, it could not explain the drastic increase in burned area among the two periods by itself. Substantial change in burning regime was driven by change in vegetation, which recovered because of relief from livestock pressure. If the trend of rebounding livestock numbers since 2000 continues in the future, then we may expect a decline in burning in near future. In general, our results suggests that change in socio-economic conditions that lead to lower grazing pressure will likely lead to an increase in burning regardless of climate, and such declines in livestock grazing can be expected in particular after strong institutional changes.

Acknowledgements

We gratefully acknowledge support for this research by the NASA Land-Cover and Land-Use Change Program, a NASA Earth System Science Dissertation Fellowship for M. Dubinin (07-Earth07F-0053), and the National Geographic Society. We are thankful for J. Burton's help with structural equation modeling and N. Keulers assistance with statistics.

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Tables

Table 3. Explanatory variables that were significantly correlated with burned area in univariate correlations for the 'before' period (1983-1996), 'after' (1997-2006), and the entire time series. T-test shows p-value for significance of differences between before and after datasets. Significance levels: ** < 0.01, * < 0.05. Notation and abbreviations: º- variable lagged one year back, WR – West Russia, AO – Arctic Oscillation, NAO – North Atlantic Oscillation, tmx – maximum temperature, pre – precipitation, NDVI – Normalized Difference Vegetation Index. Numbers indicate period or specific month

variable	r-before	r-after	r-full	t-test
WRspring	0.13	0.54*	0.00	0.27
WRwinterº	0.11	0.46*	-0.18	0.57
WRfallº	0.45*	0.09	-0.62**	0.04
AOfallº	-0.43	0.83**	-0.05	0.11
NAOfallº	-0.32	0.70*	-0.13	0.12
tmx8	0.34	-0.09	0.48*	<0.01
pre4	0.38	0.42	0.57**	0.04
pre6-8	0.40	-0.65*	-0.21	0.99
NDVI4	0.14	0.64*	0.55*	0.04
NDVI5	0.24	0.89**	0.66**	0.06
NDVI9º	0.63*	0.14	0.06	0.18
livestockº	0.38	0.13	-0.52*	<0.01

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Table 4. Five best multivariate models explaining variability in log-transformed burned area a) with b) without broad climate variables (r denotes Pearson correlation between variable 1 and variable 2, significance levels: ** < 0.01, * < 0.05).

a)

response	variable 1	variable 2	interaction	adj.r^2	r
fire	AO°	NDVI5**	AO:NDVI5	0.73	0.23
fire	AOspring°	NDVI5**	AOspring°:NDVI5*	0.64	0.01
fire	livestock*	NDVI5**	livestock:NDVI5*	0.58	-0.37
fire	livestock°*	NDVI5*	livestock°:NDVI5*	0.58	-0.43
fire	AOspring°	pre4**	laospr:pre4*	0.57	0.14

b)

response	variable 1	variable 2	interaction	adj.r^2	r
fire	livestock*	NDVI5**	livestock: NDVI5*	0.58	-0.37
fire	livestock*	NDVI5*	livestock: NDVI5*	0.58	-0.43
fire	tmx8°*	NDVI5*	tmx8°: NDVI5*	0.56	-0.00
fire	NDVI5*	NDVI6	NDVI5: NDVI6	0.50	0.52
fire	NDVI5	NDVI9	NDVI5:NDVI6	0.48	0.00

Table 5. Correlations between fire, local climate measures and oscillation indices AO, NAO, EAWR (monthly values, summarized by season: W – winter, Sp – spring, Su – summer, F - fall and yearly average), number indicates month (significance levels: ** < 0.01, * < 0.05).

	tmx8	t3-6	t6-8	t9-11	t.sum	pre4	pre3-6	pre6-8	pre9-11	pre.sum
AO4	-0.20	0.05	-0.08	0.27	0.11	0.26	-0.17	0.26	-0.03	-0.01
AO8	-0.10	0.21	0.11	0.34	0.15	0.25	-0.30	-0.07	-0.22	-0.47
AO	0.15	0.17	-0.12	0.22	0.13	0.26	-0.37	0.43	0.10	0.17
AOwinter	-0.17	0.12	-0.21	-0.04	0.09	0.17	0.11	0.21	0.21	0.09
AOspring	-0.26	0.08	-0.30	0.31	0.08	-0.06	-0.33	0.34	0.04	0.15
AOsummer	0.07	0.24	0.10	**0.48***	0.25	-0.27	-0.28	0.08	-0.23	-0.16
AOfall	0.13	-0.02	-0.18	-0.02	-0.01	-0.29	-0.01	0.16	-0.14	0.34
NO4	**-0.48***	**-0.57***	**-0.63****	0.06	-0.41	-0.01	-0.01	0.19	0.15	0.39
NO8	**-0.44***	0.12	-0.16	0.27	0.18	0.14	-0.30	0.05	0.13	-0.42
NO	-0.21	-0.02	-0.36	0.02	0.01	-0.35	-0.17	0.08	0.04	0.21
NOwinter	0.07	0.41	0.05	0.15	0.33	0.23	-0.11	0.26	-0.07	-0.11
NOspring	-0.42	-0.34	**-0.51***	-0.17	-0.31	-0.30	-0.03	-0.13	0.15	0.19
NOsummer	-0.18	0.20	-0.15	**0.61****	0.28	**-0.58****	-0.33	0.25	0.06	-0.03
NOfall	0.04	0.11	-0.09	-0.24	0.13	-0.07	0.06	-0.11	-0.04	0.18
WR4	-0.24	**-0.51***	-0.25	-0.34	-0.43	0.06	0.19	-0.10	0.36	0.03
WR8	-0.19	0.37	0.16	-0.05	0.21	-0.26	-0.14	**-0.46***	-0.21	**-0.52***

WR	-0.31	-0.24	**-0.47***	-0.23	-0.40	-0.03	-0.14	-0.08	0.09	0.10
WRwinter	-0.08	-0.36	-0.25	-0.24	-0.37	**0.54***	-0.04	-0.02	0.10	0.18
WRspring	-0.14	-0.20	-0.26	-0.18	-0.07	-0.05	0.04	-0.05	**0.52***	0.15
WRsummer	-0.18	0.05	-0.30	0.00	-0.05	-0.15	-0.29	4	0.08	-0.07
WRfall	-0.27	0.00	-0.32	-0.29	-0.35	-0.18	0.05	-11	-0.28	0.07

Table 6. Relationships between a) April and b) May NDVI and other variables.

a)

	r-before	r-after	r-full
WRfall	-0.40	-0.68*	-0.62**
AOfall°	-0.45	0.79*	-0.05
NAOfall°	-0.26	0.69*	-0.13
pre4	0.54	-0.26	0.57*
pre.sum°	0.65*	0.39	0.42
livestock°	-0.53	0.04	-0.52*
NDVI10°	0.64*	0.25	0.08

b)

	r-before	r-after	r-full
WRwinter°	0.41	-0.67*	-0.18
WRfall°	-0.17	-0.44	-0.62**
AOwinter°	0.62*	-0.27	-0.21
AOfall°	-0.16	0.91**	-0.05
AO°	0.66*	-0.04	-0.38
NAOfall°	-0.02	0.75**	-0.13
pre4	0.87**	0.15	0.57**
pre3-6	0.61	-0.06	0.14
pre.sum	0.41	-0.82**	-0.04
pre3-6°	0.73**	0.23	0.17
pre.sum°	0.72**	0.58	0.42
livestock°	-0.23	0.05	-0.52**

NDVI4	0.71**	0.83**	0.54**

Table 7. Statistical significance of difference in slopes. Dummy – statistical significance for keeping 0/1 variable representing periods ‘before’ and ‘after’. Slopes – statistical significance for different slopes. Significance levels: ** < 0.01, * < 0.05

	dummy	slope	adj.r2
NDVI5		*	0.65
AOfall°	**	**	0.65
pre.sum	**	**	0.61
NDVI6°	**	**	0.60
NAOfall°	**	*	0.56
pre6-8	**	*	0.56
WRwinter°	**	*	0.56
WRspring	**	*	0.56
NAO	**	*	0.51

Table 8. Results of two-variable best-subset analysis; only variables which participated in more than 1 model are shown.

variable	number of models
NDVI5	12
WRfall°	7
pre4	3
AOspring°	2
AO°	2

Figures

Figure 11. Sheep population and burned area trends in the study area.

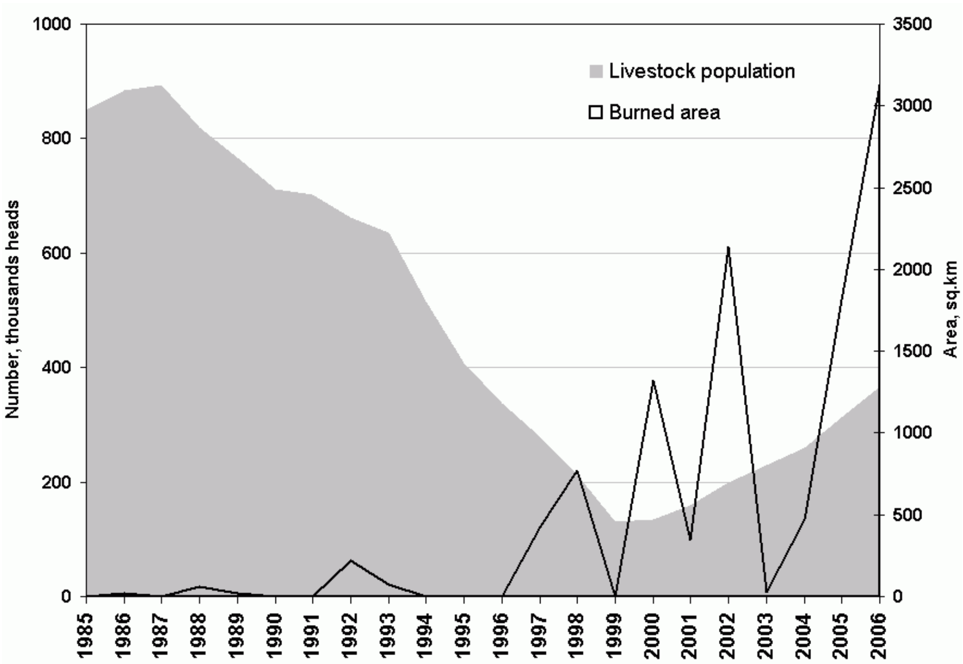


Figure 12. Study area boundary and location.

Figure 13. Change in (a) burned area, (b) maximum August temperatures, (c) livestock numbers, and (d) April NDVI between 'before' and 'after' periods.

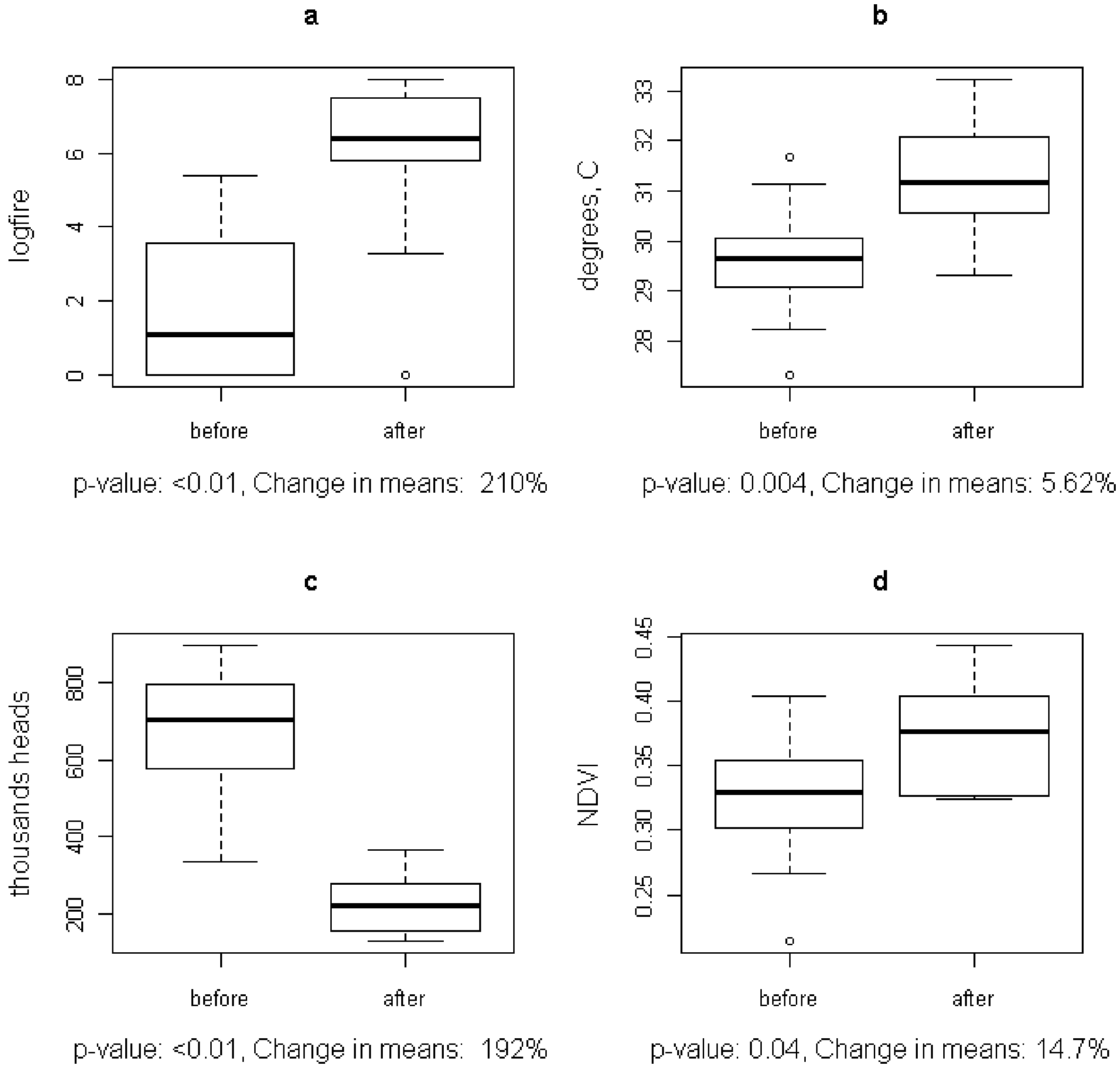


Figure 14. Correlation fields of broad-scale climatic oscillation patterns and local climate during fall months. Red indicates positive correlations, blue negative correlation, light colors - no statistical significance, and dark colors - statistical significance at 0.05 level, black – not analyzed.

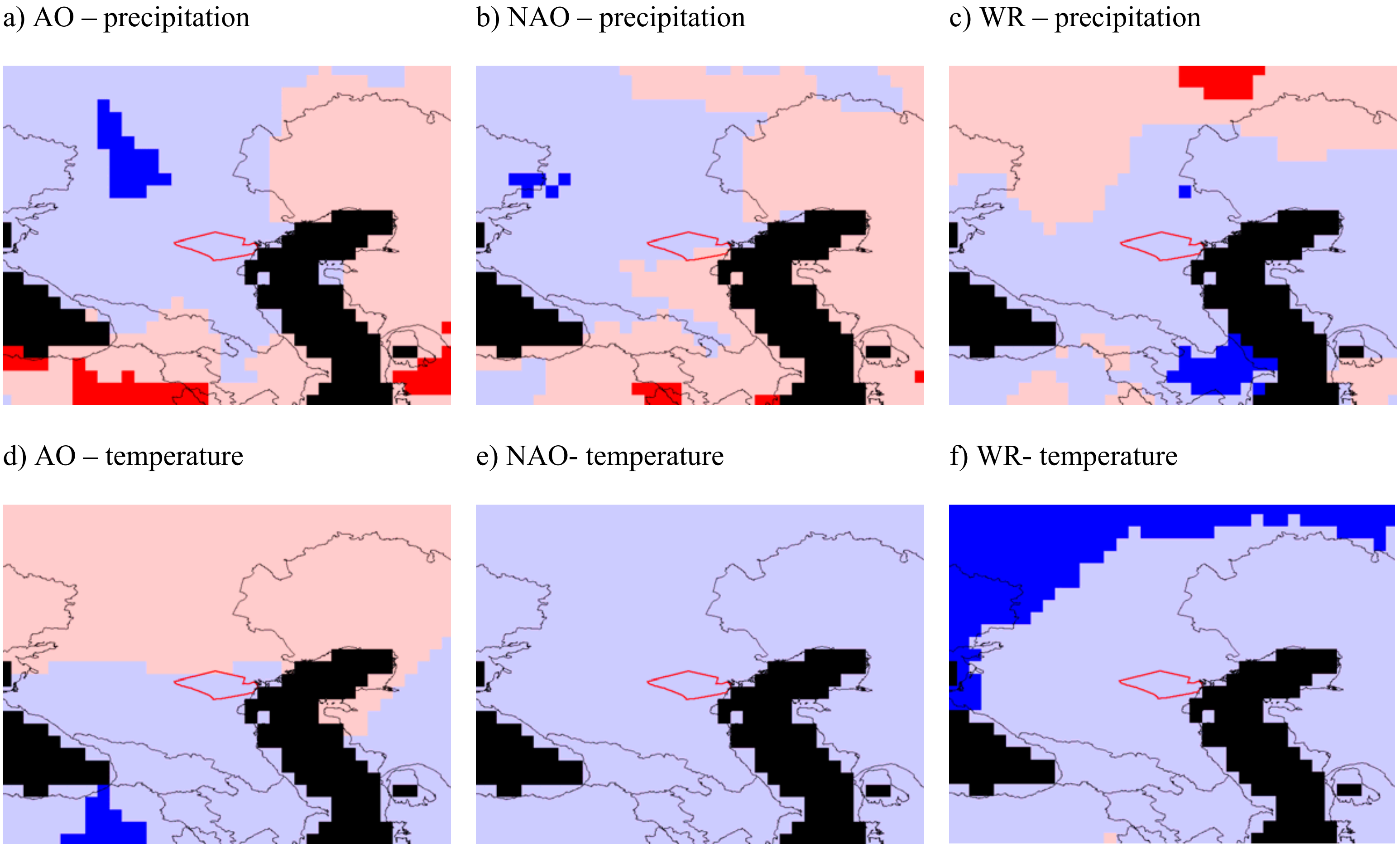
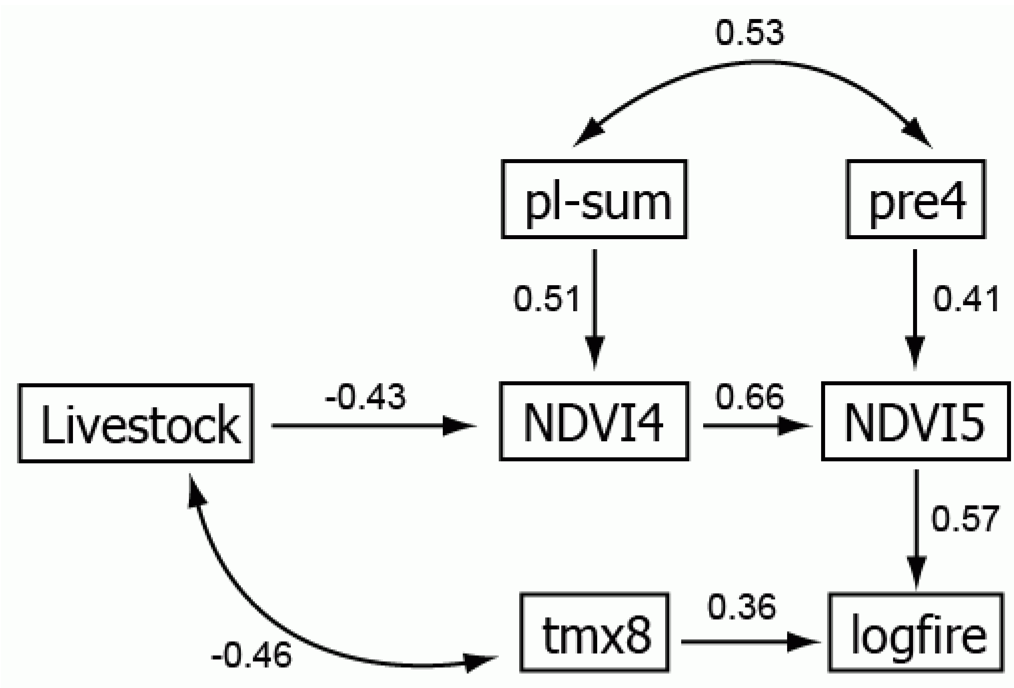
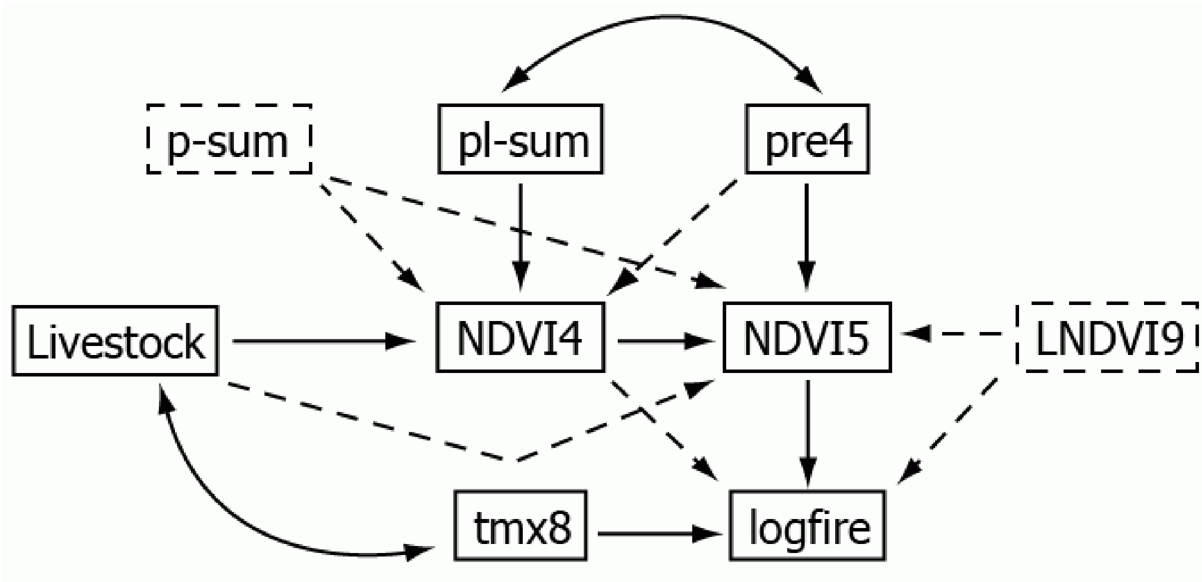


Figure 15. On top, (a) the final structural equation model, and below (b) full model with pathways and variables that were not retained (shown as dashed lines). Straight single-headed arrows represent significant effects of one variable on another ($\alpha = 0.05$), while curved double-headed arrows represent correlations between variables. The relative strength of the effect is indicated by a path coefficient. Only pathways that are significant at $\alpha = 0.05$ are included. See Table 1 for notations and abbreviations.

a)



b)


p-sum
pl-sum
pre4
Livestock
NDVI4
NDVI5
LNDVI9
tmx8
logfire

Chapter 3: Rapid vegetation change after land use changes and increasing wildfire activity in Southern Russian semi-deserts

Abstract

Vegetation composition is an important characteristic of ecosystems and is strongly affected by disturbance regimes. In semi-deserts, the ratio of shrubs to grasses in particular is closely linked to disturbance especially wildfire. Shrub-encroachment is a global phenomenon in semi-deserts, however shrublands may also be replaced by grasslands in response to increasing fire activity and – potentially – changes in land use. The effect of fires on vegetation in semi-deserts is well understood, but effects of land use are less clear. In particular, it is unknown how sharp declines in land use intensity such as grazing, and subsequent increases in fire activity due to increasing fuel loads, affect vegetation communities, because sharp declines in land use intensity are rare. Our question here was if a sharp decrease in land use intensity (i.e., a 90% decline in sheep numbers in Kalmykia, southern Russia, following the collapse of the Soviet Union), and a subsequent increase in burning has resulted in a widespread shift from shrublands to grasslands and if this vegetation shift is related to legacies of past land use. Our study area was located in the Precaspian plain in the Republic of Kalmykia in southern Russia and encompassed 18,500 km^2. We used satellite images, field data, and thematic maps to quantify change in vegetation communities, and to relate these changes to past and recent disturbances. Landsat TM satellite data for 1985, 1998, and 2007 were classified using Support Vector Machines and results were compared to the perimeters of burned areas and former open sands. We found a modest increase in the area covered by grasslands from 1985 to 1998, and a substantial increase from 1998 to 2007.

The shift from shrubs (*Artemisia* spp.) to grasses (*Stipa* spp.) was common in areas that had been plowed and then abandoned in the 1950s. However, these areas constituted less than 20 % and 10 % of the area that transitioned to grasslands by 1998 and 2007 respectively. Most of the cumulative burned areas of 1998-2006, were coincident with grasslands, especially in the southern part of our study area (on sandy soils). However, burned areas in the northern part did not convert to grasslands, and some areas in the southern part converted without evidence of fire. The observed vegetation changes in total represent considerable ecosystem changes in the Precaspian region of Russia. Decreasing grazing pressure, subsequent increases in burning, and long-term land use legacies resulted in a transition from shrub-dominated vegetation types to grass-dominated ones. Ultimately, these results highlight that the socio-economic changes that followed the collapse of the Soviet Union, resulting in strong declines in grazing pressure, caused a switch of ecosystem types due to the interaction of grazing, fire, and vegetation.

Introduction

In human-dominated landscapes, vegetation communities tend to homogenize and become less diverse because disturbance pressure is constant, and above natural limits (Olden et al. 2004). High disturbance pressure in particular may induce ecosystem shifts (Scheffer et al. 2001; van de Koppel et al. 1997). However, it is often unclear if ecosystems can recover if disturbance pressure subsides. The decline in species richness may hamper an ecosystem's ability to recover and to provide valuable ecosystem services: water availability, grazing capacity, and soil organic matter.

In semi-deserts, vegetation trends due to increasing disturbance (such as grazing) are well documented, but studies that show if and how semi-deserts recover when disturbances pressure is released are less common. The ratio of shrubs to grasses is an important indicator of the state of the vegetation in semi-deserts and changes in this ratio over time can elucidate ecosystem trends, and reflect the effects of historic and recent disturbances. Broadly speaking, there are two trajectories semi-desert ecosystems can follow. First, semi-deserts can undergo shrub encroachment. Currently, shrub-encroachment is a global phenomenon, occurring, for example, in the grasslands of Australia, Africa, USA, and India (Archer et al. 1995; Roques et al. 2001; Van Auken 2000). The increase in woody biomass has been attributed to increased CO2 concentrations and changes in rainfall patterns, but the most common causes are the suppression of fires, and/or increased land use intensity, particularly grazing, which allows shrubs to take over (Archer 1994; Schlesinger et al. 1990; Van Auken 2009).

The other broad trends is the replacement of shrublands by grasslands, which can be the result of frequent burning (Bond et al. 2005; Callaway and Davis 1993; Grigulis et al. 2005). Increases in grass cover in the semi-deserts of the Western hemisphere due to fires are particularly striking in the case of invasive annual grasses, such as cheatgrass (*Bromus* *tectorum*), and cheatgrass spread is both a drivers and a result of increase in fire frequency (Brooks et al. 2004; HilleRisLambers et al. 2010).

The relationship of fire and vegetation structure and composition might also be influenced by past land use. Long-term land-use legacy effects are well document in forests (Foster et al. 1998; Zarin et al. 2005). Semi-deserts might be as sensitive to prior land use. Indeed, which species are most abundant in semi-deserts is strongly related to the recent grazing history (Milchunas and Lauenroth 1993). However, effects of prior land use are often masked by gradual vegetation changes.

Vegetation change in arid ecosystems is usually studied either using historical approaches (Archer 1994; Mack 1981), via conceptual models (van de Koppel et al. 1997; van Langevelde et al. 2003), or with field plots and especially exclosures, which allows tracking of changes explicitly (Belsky 1992; Milchunas and Lauenroth 1993). In regard to land use, the dominant question has been how increases in land use intensity, and especially in grazing, affect vegetation. Landscape-scale studies focusing explicitly on the response of ecosystems to decreases in land use intensity over large areas and long (20+ year) time spans are relatively rare. That is unfortunate, because sheep numbers have declined rapidly in the last two decades across the globe (Figure 16), and since sheep are the main livestock species in semi-deserts, this may have resulted in substantial vegetation changes. However, the results of the decline in sheep numbers are unclear, and hard to predict, because there may be lag effects in the response of fire and vegetation to changes in land use (Dubinin et al. in review).

Russia represents a unique opportunity to study the effect of both decreasing land use intensity and long-term land use legacies on fire and ultimately vegetation communities. While livestock numbers, specifically sheep, have decreased throughout the world, former Soviet Union countries represent the case where the decline has been particularly rapid: the

Russian Federation and Kazakhstan occupy two out of three top spots among greatest ‘losers’ of sheep, and they lost most of their sheep in less than a decade (Figure 16). It is unknown how this decline translated into ecosystem changes, especially vegetation. The Republic of Kalmykia, located in the south of Russia is particularly interesting with regard to changes in livestock numbers because it is the biggest livestock producer together with the neighboring Republic of Dagestan (ROSSTAT 2006). In Kalmykia, environmental mismanagement started in 1960s, when increasing livestock numbers could not be supported by natural pastures alone, and substantial amounts of lands were plowed to grow forage crops (Zonn 1995a). Ultimately, these efforts failed, large areas became void of all vegetation, and forage production ceased almost completely by the 1980s. The land use legacy of the areas used in the past for forage crops is unknown though.

Similarly, it is unknown how decreasing land use intensity in Kalmykia affected the vegetation. Sheep density was as high as 130 head/ha in the 1970s and 1980s, but livestock numbers declined sharply after the collapse of the former USSR in 1990/1991, dropping from 3.2 to 0.6 million heads in 10 years (Zonn 1995b). The decline in grazing pressure resulted in higher vegetation productivity as measured by NDVI (Dubinin et al. 2010a), and also a very strong increase in the area burned each year (Dubinin et al. 2010b). There were very few fires before 1997, but the area burned annually increased almost exponentially thereafter, albeit with high variability in burned area from year to year (Dubinin et al. 2010b). Field studies suggested that there were an increase in grass cover, especially *Stipa spp.*, as well as *Bromus tectorum* (Neronov 1998; Shilova et al. 2007). However, prior studies of recent vegetation change in Kalmykia were small in geographic extent (Hoelzel et al. 2002; Neronov 1998), and provide an incomplete spatial picture of the vegetation changes that occurred.

Our overarching goal was to understand the effect of change in land use on semi-desert vegetation communities. Our specific objectives were: 1) to quantify change in different vegetation types for two periods: from 1985 to 1998 and from 1998 to 2007; 2) to relate the vegetation changes to spatial patterns of burning and to land use legacies, especially past attempts of row crop agriculture. Given the substantial declines in livestock and the subsequent increase in burning we hypothesized that: 1) the area dominated by grasses increased and the area dominated by shrubs declined; 2) the area of bare, open sands decreased; and 3) areas where grasses replaced shrubs were associated with both wildfires and formerly open sands.

Methods

Study area

Our study area was located in southern European Russia and occupied about 31,485 km^2 of the Republic of Kalmykia and Astrakhan Region (Figure 17). The study area is sparsely populated (population density 0.8 to 1.4 persons/km^2 (CIESIN and CIAT 2005). Contrary to expected worldwide trends in population growth in arid and semi-arid regions (Warren et al. 1996) no substantial population growth is expected in our study area (CIESIN et al. 2005).

The climate of the study area is arid, with hot, dry summers (mean daily temperature of +24°C in July; max +44°C, 280 clear days per year on average). Annual precipitation is 150 to 350 mm (with a mean of 286 mm for the study period of 1985 to 2007). We did not find decrease trend in spring and summer precipitation during the study period (Figure 18). Summer droughts are common; most of the precipitation falls in spring and fall, coinciding with the two major growing seasons (Walter and Box 1983). The topography is flat with a mean elevation of -15 m below sea level. Soils reflect past transgressions and regressions of the Caspian Sea, and are characterized by a gradient from sandy aeolian deposits and sandy loams in the southeast of the study area to clay loam in the northwest (Kroonenberg et al. 1997).

Vegetation is represented by the typical semi-desert/grassland associations of the northern Precaspian Plain. The main vegetation associations are shortgrass steppe (*Stipa* spp., *Festuca* spp., *Argopyron* spp., and other graminoids) and sage scrub (*Artemisia* spp., *Kochia prostrata*) (Golub 1994). Annual grasses, such as *Bromus tectorum* and *Poa bulbosa* are part of practically all vegetation communities but with varying cover. The presence of *Bromus tectorum* was already recorded in accounts of early explorers of the region (Kostenkov 1868; Krasnov 1886) and is presumable native to our study area. In the current landscape, *Bromus tectorum* is particularly common on recent burns, a pattern that has also been observed in the Western United States, where it is an invasive (Brooks et al. 2004). Shortgrass steppe is characterized by a short growing season in April and May and rapid senescence in the dry summer. The grasses are well adapted to fire due to the dense bunches which protect meristems and generates abundant fuels. Sagebrush (*Artemisia* spp.) dominated shrublands have less biomass than the grasslands, but a longer growing season, and sometimes exhibit a second vegetation peak in the fall and early winter (Kurinova and Belousova 1989). *Artemisia* spp. buds are situated above ground and can be killed or damaged by fires. The lack of fire tolerance by *Artemisia spp.* can lead to its gradual substitution by *Stipa* spp. and other graminoids (Neronov 1998). The primary human land use is grazing by domestic livestock, mainly sheep, and to a lesser extent cows and goats.

Study period

We studied changes in landcover from 1985 to 1998 and from 1998 to 2007 using Landsat satellite images. Our choice of study period was determined by the goal to include vegetation conditions before the collapse of the USSR, the desire to integrate our findings with information from previous research, and last but not least, the availability of remote sensing data for particular phenological periods.

Data

Remote sensing data and image selection

We used 30-m Landsat 5/7 TM/ETM+ data that correspond to World Reference System (WRS-2) path/row 170/28 for the landcover change classification. Terrain corrected (level 1T) images were acquired from USGS (Figure 24). No additional geometric or radiometric correction was applied. GIMMS AVHRR NDVI and MODIS MOD13A1 data were analyzed to obtain phenology information which aided Landsat image selection. To generate typical vegetation phenology curves, we used Normalized Difference Vegetation Index (NDVI) data from the MOD13A1 dataset, 250-m resolution, 16–day composites for 2007.

Landsat image selection was driven by the observation that the main vegetation communities had different annual phenology curves (Figure 20). Field data and literature indicated that there are at least two periods of spring greenness and these periods are specific for different vegetation types (Matyashenko 1985). Areas that showed the earliest green-up in spring were typically dominated by annual grasses, such as Bromus tectorum, and forbs. Areas that greened up later in the spring were typically dominated by perennial grasses, such as Stipa spp. Areas dominated by shrubs had a less pronounced spring green-up. For every study year, we derived a curve of the mean NDVI for the entire study area from either GIMMS (1985, 1998) or MOD13 (2007) data and every potential Landsat image candidate for classification was checked to see if it was right on, or close to, the greenness peak.

Ultimately we selected 2 images (early and late spring) for each study year. Optimal images were available for 1985 and 1998, but no cloud-free Landsat images were available for both phenological periods of 2007. In 2007 the only early spring image available was a suboptimal image because it was recorded in very early spring (March, 10) with portion of the data missing. Landsat 7 experienced difficulty since May, 2003 with its Scan Line Corrector (SLC), resulting in images with gaps that cover 22% of the entire scene (so called

SLC-off data) (Maxwell et al. 2007). However, we included this image since there was no alternative.

Fire data

Broad-scale burning in the region can be traced back at least as far as the 18th century (Pallas and Blagdon 1802). Though fire was rare in Soviet times, our previous remote sensing analysis showed a sharp increase in burning starting in 1997-1998 which continued until 2007 covering large portion of the study area (Dubinin et al. 2010b).

We used augmented burned areas database from Dubinin et al. 2010b to cover a larger area. Burned areas were delineated based on MODIS MOD02QKM Level 1B calibrated radiance data by visual image interpretation, which is at least as precise as automatic methods (Bowman et al. 2003). Delineating burned areas was straightforward due to distinct fire scar boundaries, often confined by linear features such as roads. The easy recognition of fires was further facilitated by the lack of other disturbances in the area that could have resulted in comparable patterns. Burned areas remained visible for 2 to 3 years after burning, unless a new fire overrode an older scar (Dubinin et al. 2010b).

Field data

Field data were collected during two field seasons in August of 2007, and in May and June of 2008. We collected information on vegetation and disturbances for 160 90-m transects. On each transect we registered plant species, and litter or bare ground at every 0.3 meter pin (270 hits total). Starting at the 12.5 m mark, we also took 6 breast height nadir photographs to assist later classifications of the vegetation community. We assigned each point to a broad vegetation group based on the phenology of dominating vegetation species. The first group represented shrub-dominated communities, which have typically a longer, but less pronounced vegetation peak, and included *Artemisia* species such as *Artemisia lercheana*, *A. austriaca*, *A. pausiflora*, *A. tcherneviana*, as well as *Kochia prostrata*. The second group included perennial grasses of both *Stipa* and *Argopyron* genus, such as: *S. capillata*, *S. sareptana*, *S. lessingiana, Agropyron desertorum*, *A. fragile* and others. The third group represented annual grasses, characterized by an early, and typically short, single vegetation peak and included *Bromus tectorum*, *Poa bulbosa*, *Carex arenaria* and others. While in the field, we also collected information on disturbances, especially burning, if we found burned woody plant parts above the root. We defined 'dominated' as at least 30% of the live vegetation cover represented by species from this community.

Legacy croplands data

In the 1980s, there were large areas of aeolian dunes which largely corresponded to areas that were either heavily overgrazed or plowed in 1960s and abandoned soon afterwards (Zonn 1995a). About 10% of Republic of Kalmykia was converted to true desert by the early 1980s (Dregne et al. 1991). We digitized the boundaries of permanent sands in 1983 based on natural forage grounds map which had been created in Soviet times using remote sensing data (Trofimov et al. 1983). After abandonment, these formerly plowed were fields first converted to open sands due to continuing overgrazing, but then started to recover in part due restoration activities initiated by the government and carried out using *Agriophyllum squarrosum*, *Leymus racemosus* as long as variety of annual grasses, and in part, because grazing pressure subsided.

Analysis

Classification

Previous studies have used multitemporal satellite imagery from specific phenological periods to map changes in vegetation composition (Singh and Glenn 2009), agricultural areas

(Lo et al. 1986; Prishchepov et al. 2010), and the phenological heterogeneity of grassland communities (Fisher et al. 2006). Vegetation change can be assessed either via vegetation indices, spectral mixture analysis (Asner and Heidebrecht 2002; Hostert et al. 2003; Kuemmerle et al. 2006), of the classification into different vegetation classes.

We classified the Landsat images into four classes: ‘grass’, ‘shrub’, ‘bare’, and ‘water’. Training data for a supervised classification were collected via a visual interpretation of the images. The ‘grass’ class was identified based on the greenness level, and had digital number values in band 4 that were higher than those in band 3. The ‘bare’ class was defined by high digital number values in all bands, ‘water’ by low digital numbers in all bands, and ‘shrub’ was defined as the remaining non-green areas. We collected at least 200 training points for each class per image.

Each image was classified separately using a Support Vector Machine (SVM). SVM is a machine-learning classification approach that is superior to other classifiers (Foody and Mathur 2004; Huang et al. 2002). SVM is a non-parametric classifier that achieves optimal class separation of classes by fitting hyperplane based on training data. SVM is well suited to map multimodal classes, which parametric-based classifiers (e.g., maximum likelihood classifier) have difficultly to classify accurately, due to violation of the normality assumption. We used the ImageSVM implementation of SVM (Rabe et al. 2009) and a “one-against-one” approach for multi-class SVM classifications to avoid unbalanced classifications that have been reported for the “one-against-all” approach (Melgani and Bruzzone 2004). We tested a range of parameter combinations (g from 0. 1 to 1000 and C from 0.1 to 1000) by fitting individual SVM to each parameter pair and comparing models based on three-fold cross-validation thus selecting least-error parameter combination for each image (Rabe et al. 2009). Areas covered with clouds in any Landsat image, and areas of no data due to SLC-off problems were masked out of the resulting classifications of all Landsat images.

The resulting classifications for early and late spring in a given year were combined together to form 9 new vegetation classes. Depending on whether grass-related greenness was mapped in the early or the late spring Landsat image, we defined classes that were dominated by either annuals, perennials, shrubs or permanent sands as well as their mixtures (Table 9). The resulting classes were: ‘annuals-perennials’ – mixed grasslands that include both annuals and perennials in proportions substantial enough to generate green response in early of late spring image, ‘shrub-perennials’ – vegetation communities co-dominated by shrubs and perennial grasses that have a single greenness peak during late spring, ‘perennials’ – vegetation communities characterized by bare ground response during early spring and covered by perennial grasses during late spring, ‘annuals-shrubs’ – communities that have early peak in greenness by lack one afterwards, ‘shrubs’ – areas that do not have neither greenness peaks, nor signs of bare ground, ‘degraded’ – early shrubs and late bare ground and vice versa, ‘annuals’ – early greenness followed by bare ground, and finally, ‘bare’ – bare ground during both periods and ‘water’ – water and any other class during any period.

Comparisons

We overlaid the resulting land cover classifications with the boundaries of bare ground from the Natural forage grounds map of 1983. We then calculated the percentage of correspondence between the bare ground class from the thematic map and the three perennial grasslands classes in 1998 and 2007.

We have compared the resulting land cover classifications with the boundaries of burned areas. We combined annual burned area maps from 1998 to 2007, calculated the total area burned, and mapped burn frequency, which ranged from once to seven times burned from 1998 to 2007. We then calculated: a) the percentage of the entire burned area that occurred in to each land cover class, and b) the percentage of the land cover class that had burned. We also calculated the proportion and area of each burned frequency class in each vegetation class.

Results

Greenness and landcover types

Field data collected in the spring of 2008 indicated that landcover types dominated by annuals and perennials had higher reflectance in band 4 and lower reflectance in band 3 compared to areas dominated by shrubs, which had less vegetative cover overall (two-tailed T-test, $N_{annuals} = 26$, $N_{perennials} = 98$, $N_{shrubs} = 51$, band3: shrubs vs annuals: p=<0.01, shrubs vs perennials: p=<0.01; band4: shrubs vs annual: p=0.03, shrubs vs perennials: p=<0.01, Figure 21). Field samples of perennials and annuals were not significantly different in terms of their band brightness (band3: p=0.89; band4: p=0.75, Figure 21).

Landcover changes

According to our analysis, the study area experienced a strong increase of grasslands, at the expense of permanent sands and shrubs (Figure 25). Areas covered by perennial grasses more than doubled from 1985 to 2007 and in the total area of bare ground in 2007 was only 20% of the area they covered in 1985. The increase in grasslands was moderate from 1985 to 1998, the majority of the grassland increase occurred from 1998 to 2007.

Comparisons of the classifications for 1985, 1998, and 2007 showed numerous vegetation changes. For example, only 32% of the area of 'annuals-perennials' class in 1985 stayed in that class by 1998, while 27% transitioned to the 'annuals-shrubs' class. Partly because they were so widespread, the shrub-dominated classes were the most stable (>40%). Transitions from 1985 to 1998 were dominated by transitions from 'shrub' class to 'annuals-shrub'. However, from 1998 to 2007, substantial parts of shrub-dominated classes (36% of ‘shrubs-perennials’ and 21% of ‘shrubs’) transitioned to grasses-dominated classes, especially perennials.

Vegetation changes exhibited strong spatial patterns. The most dramatic increase in grasslands coincided with the core area of the Black Lands (Chernye Zemli) where several protected areas have been established since 1991. The pastures in the northern part of the study area, still actively grazed, did not show as much of an increase in perennials, but changes from permanent sands to shrubs and annuals were still common.

Association with permanent sands and burning

Areas that changed from shrubs to grasslands were tightly associated with agricultural fields that were plowed in 1950s and ‘60s and later abandoned. About half of the areas (42%) mapped as permanent sand in 1985 were converted to one of the three grassland classes (annuals-perennials, shrubs-perennials, perennials) by 1998, and 74% by 2007 (Figure 23). The remaining part of permanent sands was covered by shrublands. However, new grasslands occurred not only on former permanent sands. In addition to perennial grasslands on former permanent sands (450 km^2 in 1985), there were 2491 km^2 areas covered by perennial grasslands in place of shrubs in 1998 and 3528 km^2 in 2007 (Table 12).

Areas of change to perennial grasslands were also associated with burning. However, our analysis could not distinguish if areas that burned were more likely to transition towards perennial grasslands, or if areas that had transitioned to grasslands were more likely to burn. From 1998 until 2007, 38% of the study area burned at least once. Of the burned area 49% was represented by the three perennials classes (which covered 32% of the study area) (

Table 10). At the same time 58% of total area of the three perennials classes in 2007 burned at least once between 1998 and 2007, compared to only 38% burned areas for the study area as a whole. Even more striking, the pure perennial class (grass in late spring, and bare ground in early spring), although occupying only a relatively small part of total area burned (7%), burned almost entirely (93%). The three classes dominated by perennial grasses were particularly common in areas where fire frequency was high (Table 11). 65% and more of the areas that burned 3 or more times had perennial grasslands (such areas represented 41% of total burned area and 16% of total study area). The proportion of the burned area represented by perennial grasslands classes increased with increased frequency of burning, with the biggest change between areas that burned twice (45% represented by perennial grasslands) versus 3 times (65% represented by perennial grasslands). On the opposite, classes dominated by shrubs (annuals-shrubs and shrubs classes combined) represented 60% of areas that burned only once, but only 25% of areas that burned twice and less then 10% of the areas that burned 3 times (Table 11).

Discussion

How human-induced disturbance, and especially grazing affects arid environments is hotly debated. Some studies reported sudden changes, and "catastrophic vegetation shifts" (Scheffer et al. 2001; van de Koppel et al. 1997) while other argued that recent grazing history has generally little effect on vegetation in arid environments (Milchunas and Lauenroth 1993) compared to other factors such as droughts, fires, insect outbreaks, and floods (Warren et al. 1996; Westoby et al. 1989). As a result, little is known about the ability of semi-desert vegetation to recover once disturbance subsides. Our results suggest that a strong decrease in grazing pressure can indeed lead to substantial changes in dominant species, including a shift from shrublands to grasslands, within a relatively short timeframe (9-10 years) due to changes in both, the direct impact of grazing and alteration of fire regimes.

The breakdown of the Soviet Union overturned dramatic forecasts for the Republic of Kalmykia which was predicted to become completely desertified in 20 years if trends of the 1980s would continue (Vinogradov 1995; Zonn 1995a). In stark contrast, the 90% decrease in livestock has lead to an almost complete re-vegetation of open sands by 1998 and a four-fold increase in the area covered by perennial grasslands at the expense of areas previously dominated by shrubs by 2007. This increase in perennial grasses at the expense of shrubs is supported by results from field plots conducted both locally (Neronov 1998) and in the other parts of the world (Grigulis et al. 2005; Hejcmanova et al.; Zhang et al. 2005).

Besides decrease in grazing, the increase of grasses could have been potentially caused by several other process, such as: 1) Increase in atmospheric CO2, 2) Climate change, 3) Change in burning regime. Increase in atmospheric CO2 is a global phenomenon, shown to be beneficial for at least several species of annual grasses (Smith et al. 2000). However, experimental studies highlight that the response to elevated CO_2 level is different for different species and overall response may be negligible (Grunzweig and Korner 2001). Besides grass species like *Bromus tectorum* (Smith et al. 2000), shrubs may also benefit from increase in CO_2, and global shrub encroachment is argued to caused by CO_2 increase as well (Archer et al. 1995). However, Smith at al. observed statistically significant differences between native grasses and *Larrea tridentate* and invasive *Bromus tectorum* at 500 ppm level, which is yet unreached. Since 1985, the CO_2 concentration in the atmosphere grew at about 1.5 – 2.0 ppm/year rate globally (345 ppm in 1985, and 380 in 2007) (Tans 2010). Given these relatively minor CO_2 concentration increases, and given that they may benefit both grasses and shrubs we suggest that they cannot explain the observed vegetation changes in our study area.

Climate change is another potential driver of vegetation change, and increased precipitation during local and global climatic events, such as El-Nino, can lead to amplified pulses of productivity, especially in annual grasses (Bradley and Mustard 2005; Holmgren et al. 2006). Given the limited availability of cloud-free satellite images, we were not able to fully control for precipitation when selecting study years. However, meteorological data from a station close to the center of the study region (Komsomolsk) did not show substantial differences among the years when our satellite images were taken (92, 125, 95 mm of rainfall total, for March through June of 1985, 1998, and 2007 respectively, Figure 18), and there was no systematic trend in precipitation over the last 25 years. Over next 50 year, a doubling of CO_2 may lead to increases in temperatures of 1-3 C°, and that would lead to higher potential evapotranspiration, which, assuming that rainfall will not change, will increase aridity. However, photosynthesis boost from increasing CO_2 would also increase water-use efficiency via the reduction of stomatal conductance making it difficult predicting net balance. In any event though, the observed recent vegetation changes cannot be attributed to climate change. Thus, changes in CO_2 concentration, temperature and precipitation may be by far outweighted by direct human impacts (Amthor 1995; LeHouerou 1996). Indeed, compared with slowly changing background factors, changes in burning and land use are more pronounced and rapid (Dubinin et al. 2010a).

The increase in grasslands may indicate a positive feedback loop given the substantial association between patterns of fire and shifts to grasslands in our study area. Grasslands provide more fine-fuel easily flammable material then shrub-dominated communities (D'Antonio and Vitousek 1992) . Furthermore, grasses can recover easier after fire, which thus creates a competitive advantage over shrubs (Grigulis et al. 2005).

It is not clear if the observed fire patterns, and hence the expansion of grasslands, are 'natural'. Historical literature provides mixed evidence on extent and frequency of fires in the North-West Caspian lowland in the 19th and 20th centuries. Some sources mention widespread grassland fires (Pallas and Blagdon 1802; Zwick and Schill 1831). However, reports from research expeditions, specifically aimed to study of natural resources and management do not mention any grasslands fires (Kostenkov 1868; Krasnov 1886), but this does not necessarily reflect their actual absence. Based on that we speculate that the release of livestock pressure in 1990s and restoration of fuel loads might have reverted vegetation communities to pre-Soviet conditions overall, potentially with grasses being more widespread. However, given the rebounding trend of livestock since 1999, we expect a potential decline in fires and a return of shrubs in the near future.

While fire is certainly the most important driver of vegetation change, it is not the only cause of the increase in grasslands. We found that a substantial proportion of the areas that shifted to grass-dominated communities had not burned. On the other hand, not all areas that did burn, converted to grasslands, suggesting interaction with soil type, fire return interval and exact timing of burning and land use legacies. In general, bare grounds undergoing first stages of succession were more likely to burn, as they are first populated by annuals. Additionally restoration efforts also led to establishment of species with high fuel load and contiguity.

Our satellite classifications did not provide species-level change information, and mapped only broad phenological groups. However, it is of interest to try to compare the role of annual grasses, especially *Bromus tectorum L.*, in the study area with its role in other areas of the world, where it represents highly invasive species and can lead to change in fire dynamics. Cheatgrass is a highly competitive pioneer species in both its native and foreign range (Mack 1981; Neronov 2006). While in invasive mode, cheatgrass can dominate vegetation communities forming monocultures in shrub or bunchgrass ecosystem (Mack 1981), but in its native range it is usually co-exist with perennial grasses and shrubs, such as of *Stipa* and *Artemisia*. Our results indeed showed common presence of an early green-up, which we interpreted as an annual grass signal in 'annuals-perennials', 'annuals-shrubs' classes, and we found very few areas that were pure 'annuals'. Similarly, our field data indicated that cheatgrass and other annuals were present in low abundances in most of the vegetation communities in our study area (results not shown).

Conclusions

The research that we present here provides evidence for substantial vegetation change in Kalmykia. These vegetation changes are a complex phenomenon, and were probably caused by a multitude of factors. The sharp increase in annual burned area in the semi-deserts of Southern Russia is most likely the main direct driving force of vegetation change in the region. However, burning in turn is affected by human activities such as livestock grazing (Dubinin et al. 2010a). Land use thus can impact the structure and composition of vegetation communities and outweighed in our study area the effects of increasing CO2 concentration and climatic change.

Acknowledgements

We gratefully acknowledge support for this research by the NASA Land-Cover and Land-Use Change Program (NNG05GA74G), by a NASA Earth System Science Dissertation Fellowship for M. Dubinin (07-Earth07F-0053), by the U.S. Fish and Wildlife Service, and by a grant from the National Geographic Society (8360-07). We are also grateful to field assistants A. Zakharov, S. Goryunova, E. Degtyarev, I. Glushkov.

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Tables

Table 9. The definition of nine vegetation associations and other land cover classes for each year based on the classification of four land cover classes for corresponding early and a late spring classifications.

Late spring classification		Early spring classification: grass	shrub	bare ground	water
	grass	annuals-perennials	shrubs-perennials	perennials	water
	shrub	annuals-shrubs	shrubs	degraded	water
	bare ground	annuals	degraded	permanent sand	water
	water	water	water	water	water

Table 10. The association of different land cover classes in 2007 with the cumulative burned areas (1998 – 2007). Subtotal for perennial grasslands is the sum of the three classes that had perennials (annuals-perennials, shrubs-perennials, perennials), the subtotal for shrubs is the sum of two shrub classes (annuals-shrubs, shrubs).

	Total area (km^2)	% of the total area	Burned area (km^2)	% of the total burned area	% of the total area in this vegetation class that burned
Annuals-perennials	4,320,863	21%	2,433,805	32%	56%
Shrubs-perennials	1,567,060	8%	754,644	10%	48%
Perennials	618,161	3%	572,526	7%	93%
Annuals-shrubs	4,979,386	24%	978,850	13%	20%
Shrubs	6,424,930	31%	1,989,557	26%	31%
Degraded	422,474	2%	216,439	3%	51%
Annuals	11,424	0%	1,244	0%	11%
Permanent sand	278,290	1%	60,991	1%	22%
Subtotal for perennials	6,506,084	32%	3760975	49%	58%
Subtotal for shrubs	11,404,316	55%	2968407	39%	26%

Table 11. The proportion of each land cover class that burned once, twice, etc. up to seven time.

	Number of times an area burned from 1998 to 2007						
	1	2	3	4	5	6	7
Annuals-perennials	19%	28%	40%	44%	53%	44%	49%
Shrubs-perennials	7%	10%	12%	12%	12%	17%	7%
Perennials	3%	7%	13%	12%	9%	9%	11%
Annuals-shrubs	20%	13%	8%	4%	3%	3%	2%
Shrubs	40%	30%	13%	9%	6%	9%	7%
Degraded	3%	3%	2%	3%	3%	2%	6%
Annuals	0%	0%	0%	0%	0%	0%	0%
Permanent sand	1%	1%	0%	0%	0%	0%	0%
Subtotal for perennials	30%	45%	65%	69%	74%	70%	67%
Subtotal for shrubs	60%	25%	10%	5%	2%	1%	0%
Total burned area, (km^2)	2628	1538	1221	931	482	189	29

Table 12. Land cover transitions for two periods: a) 1985 – 1998, b) 1998 – 2007, stable areas are shown in bold. Reported as the area (km^2) and the percentage in brackets.

a)		Land cover in 1998								
		Total area in 1985	annuals-perennials	shrubs-perennials	perennials	annuals-shrubs	shrubs	degraded	annuals	permanent sand
Land cover in 1985	annuals-perennials	887	**284 (32%)**	95 (10%)	5 (0%)	240 (27%)	156 (17%)	1 (0%)	0 (0%)	0 (0%)
	shrubs-perennials	1280	206 (16%)	**194 (15%)**	7 (0%)	256 (20%)	512 (40%)	2 (0%)	0 (0%)	5 (0%)
	perennials	126	5 (4%)	25 (20%)	**1 (1%)**	10 (8%)	68 (54%)	0 (0%)	0 (0%)	4 (3%)
	annuals-shrubs	2786	614 (22%)	132 (4%)	6 (0%)	**1290 (46%)**	611 (21%)	2 (0%)	0 (0%)	4 (0%)
	shrubs	10236	822 (8%)	644 (6%)	36 (0%)	3831 (37%)	**4263 (41%)**	28 (0%)	0 (0%)	73 (0%)
	degraded	1025	50 (4%)	152 (14%)	8 (0%)	153 (14%)	487 (47%)	**13 (1%)**	0 (0%)	52 (5%)
	annuals	1	0 (5%)	0 (7%)	0 (0%)	0 (16%)	0 (30%)	0 (1%)	**0 (0%)**	0 (9%)
	permanent sand	1061	18 (1%)	423 (39%)	9 (0%)	40 (3%)	326 (30%)	15 (1%)	0 (0%)	**137 (12%)**

b)		Land cover in 2007								
		Total area in 1985	annuals-perennials	shrubs-perennials	perennials	annuals-shrubs	shrubs	degraded	annuals	permanent sand
Land cover in 1998	annuals-perennials	2091	**526 (25%)**	196 (9%)	36 (1%)	651 (31%)	535 (25%)	7 (0%)	0 (0%)	1 (0%)
	shrubs-perennials	1732	629 (36%)	**294 (17%)**	110 (6%)	126 (7%)	356 (20%)	23 (1%)	0 (0%)	6 (0%)
	perennials	77	26 (34%)	22 (29%)	**4 (5%)**	2 (2%)	9 (12%)	2 (2%)	0 (0%)	1 (2%)
	annuals-shrubs	5966	1088 (18%)	261 (4%)	53 (0%)	**2609 (43%)**	1681 (28%)	32 (0%)	1 (0%)	8 (0%)
	shrubs	6549	1396 (21%)	484 (7%)	312 (4%)	849 (12%)	**2718 (41%)**	231 (3%)	2 (0%)	90 (1%)
	degraded	67	5 (8%)	4 (6%)	2 (3%)	3 (4%)	23 (35%)	**8 (12%)**	0 (0%)	11 (17%)
	annuals	2	0 (21%)	0 (4%)	0 (1%)	0 (18%)	0 (18%)	0 (3%)	**0 (4%)**	0 (6%)
	permanent sand	285	15 (5%)	18 (6%)	19 (6%)	2 (0%)	79 (27%)	35 (12%)	0 (0%)	**87 (30%)**

Figures

Figure 16. Average annual decline in total sheep from 1992 to 2008 for the ten countries with the strongest absolute declines. Number in brackets indicates time span in years between the lowest and the highest sheep number during this period (FAO 2010).

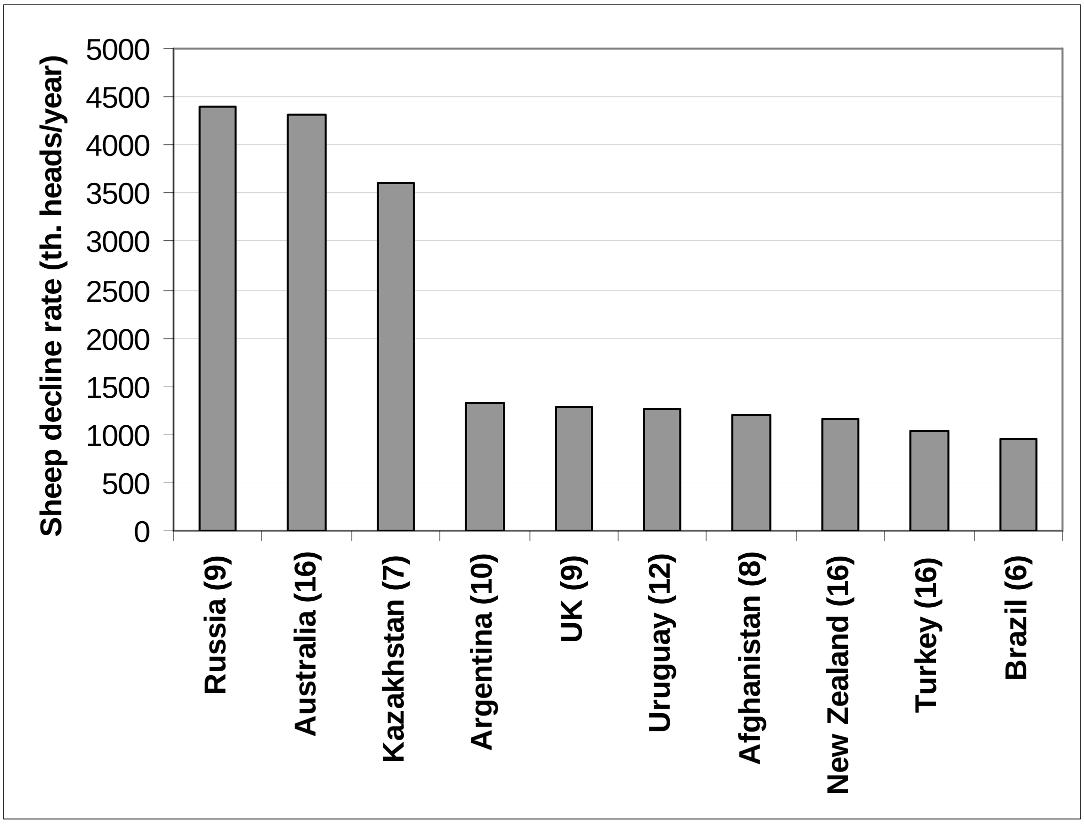


Figure 17. Study area location (hatched area with black outline) in southern Russia

Figure 18. Trends in spring NDVI and precipitation (sum of March-June period rainfall, mm) in the study area from 1982 to 2007 period. While there were strong increases in NDVI, there was no systematic trend in precipitation.

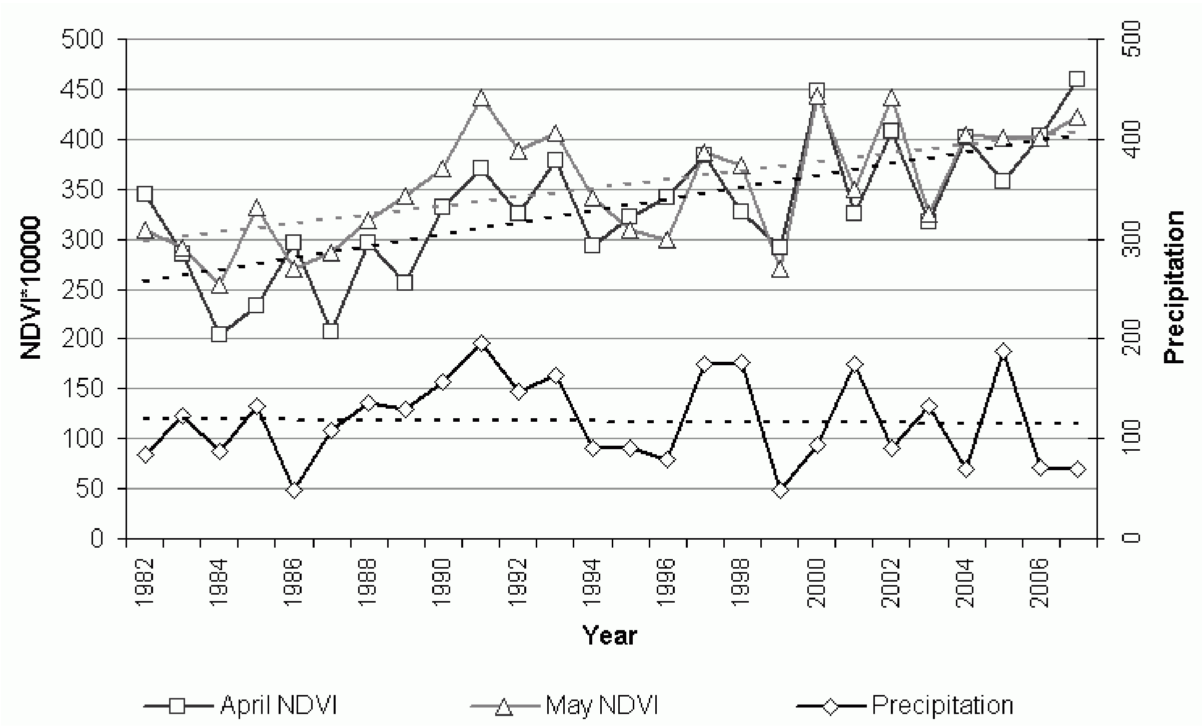


Figure 19. Examples of different vegetation types, from top left clockwise: *Stipa capillata* dominated community, *Artemisia lercheana* dominated community, community dominated by annual *Bromus tectorum* and perennial *Stipa sareptana*, permanent bare sands. Images taken by M. Dubinin in late spring.

Figure 20. Phenological differences for the three pure vegetation types based on field data and MOD13A1 NDVI in 2007, error bars indicate 1 standard deviation ($N_{perennials}$ = 6, N_{shrubs} = 6, $N_{annuals}$ = 6).

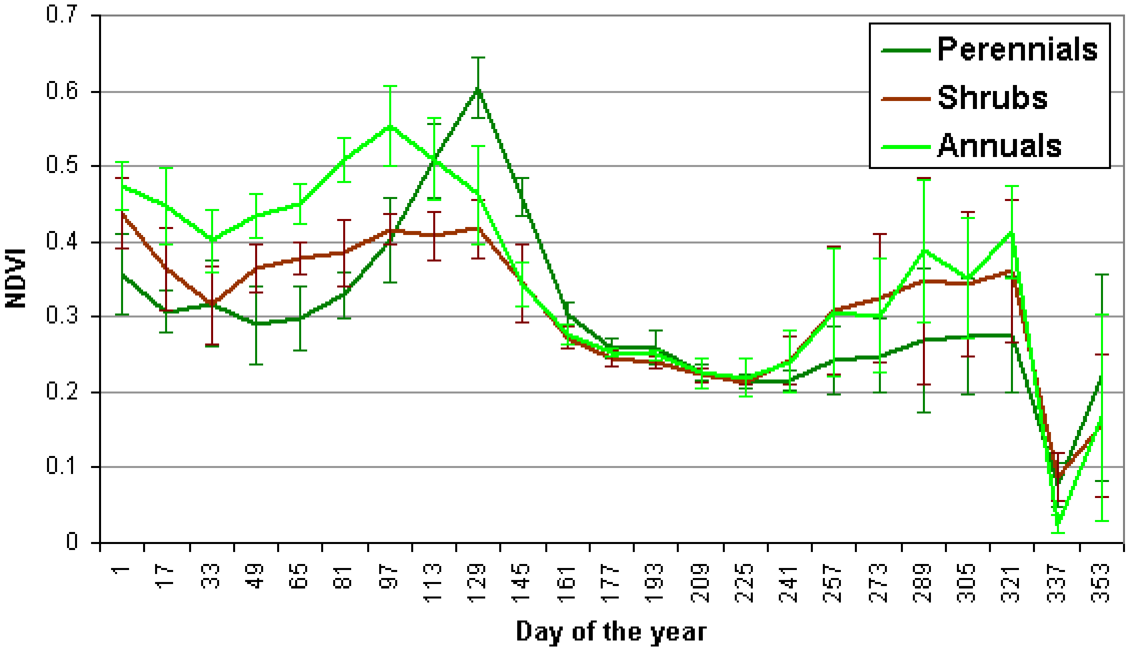
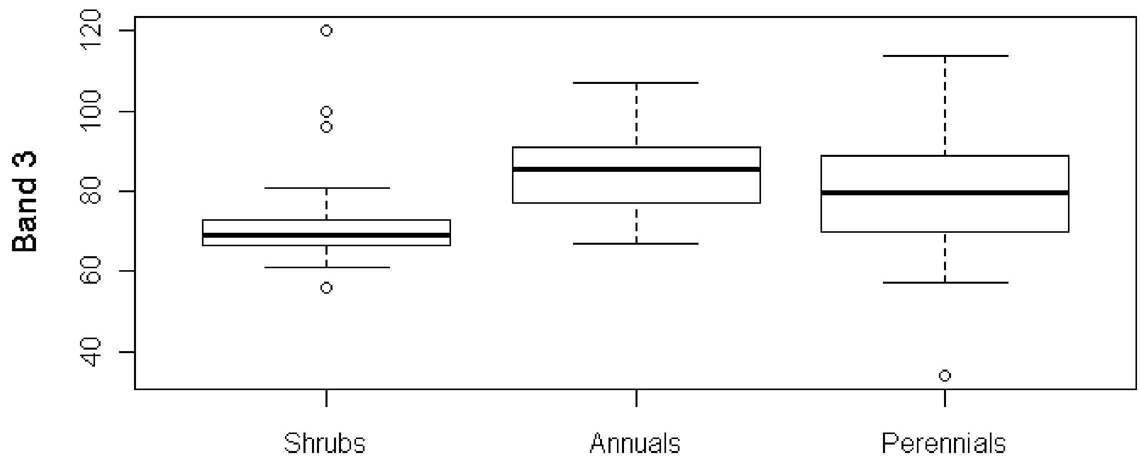


Figure 21. Differences in band brightness (0-255, Landsat 7/TM May 5, 2007) for different vegetation communities, $N_{shrubs} = 51$, $N_{annuals} = 26$, $N_{perennials} = 98$.



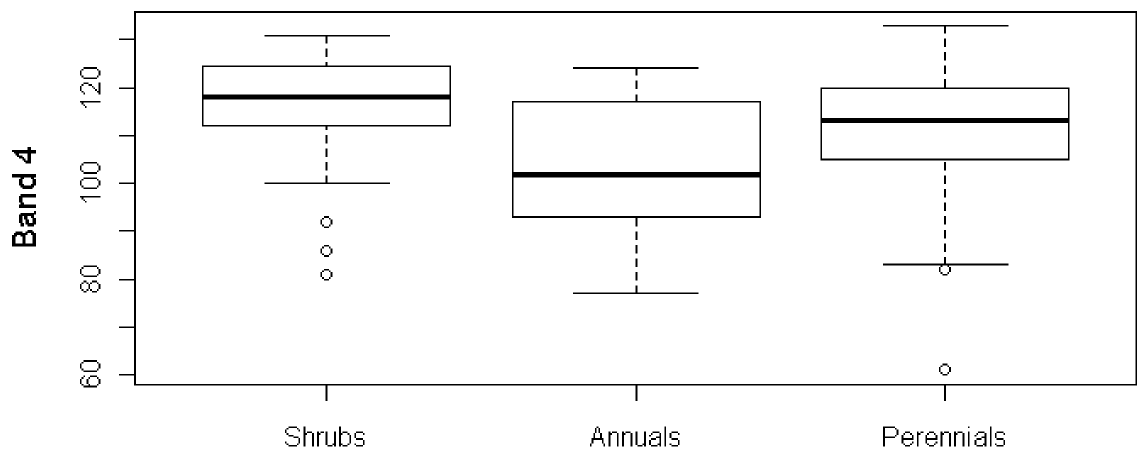


Figure 22. Total area in different land cover classes in 1985, 1998, and 2007.

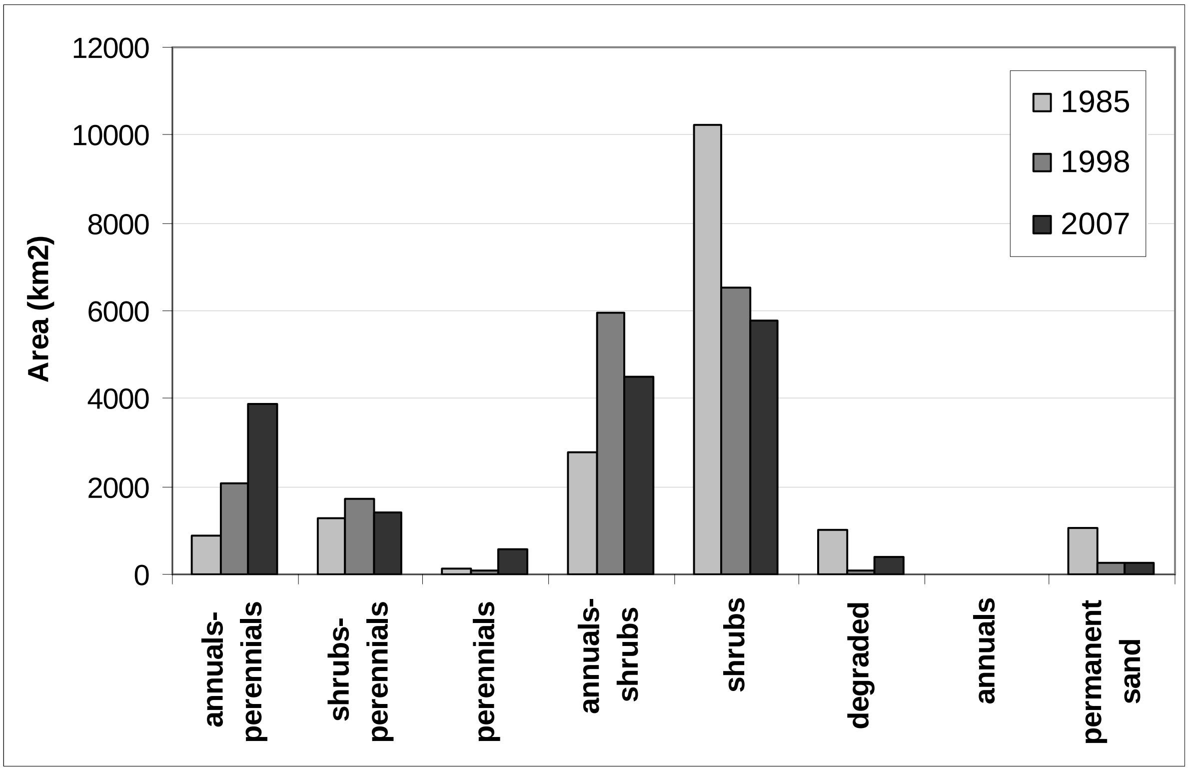


Figure 23. Association between former permanent sands (red polygons) and landcover (green – grasslands, grey – shrublands, yellow – permanent sands, blue - water) in late spring of 1998 (left, top) and late spring of 2007 (left, bottom). Also shown fragments of Landsat TM images, RGB combination: bands 5, 4, and 3 in red, green, and blue (right)

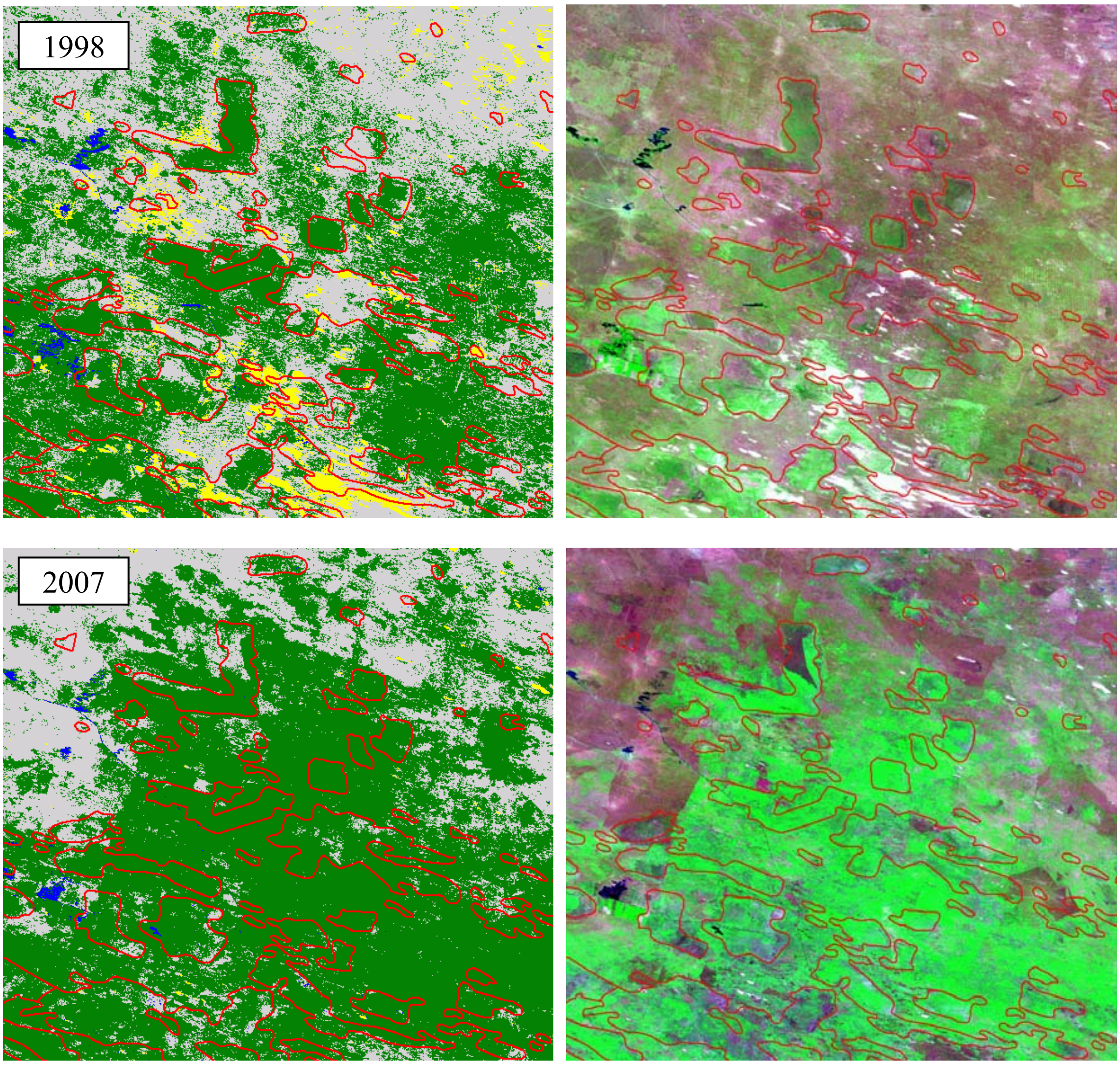
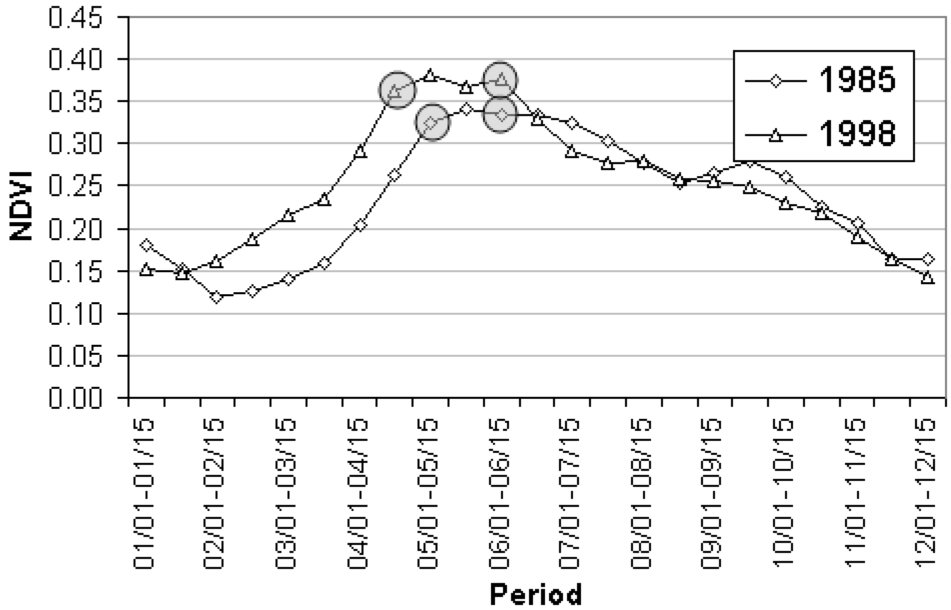


Figure 24. Annual NDVI trends for the years of selected satellite images in a) 1985, Landsat images recorded on May 8th and June 6th and 1998 April 26th and June 13th, and b) 2007 (March 10th and May 5th).

a)



b)

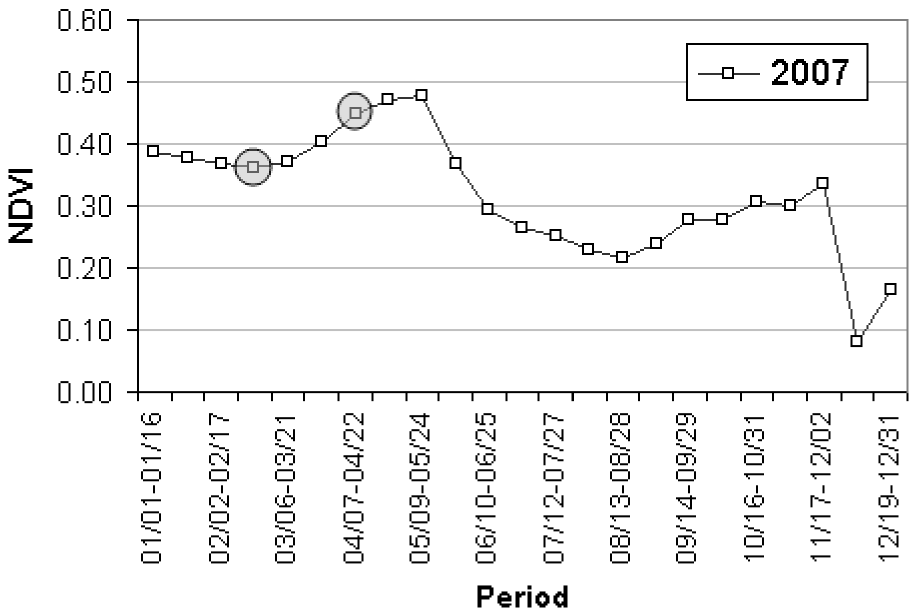
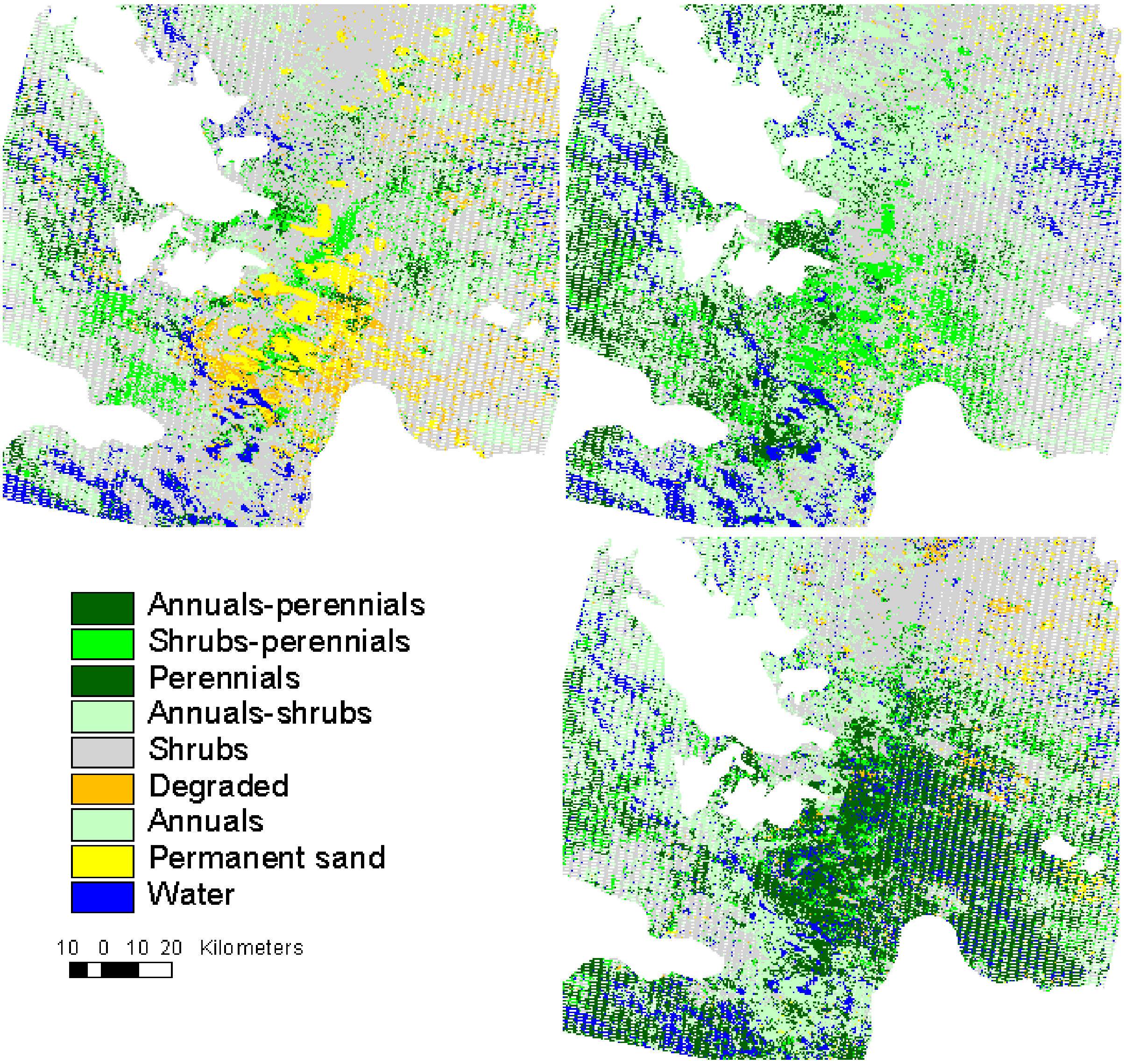


Figure 25. Land cover classifications for 1985 (top left), 1998 (top right), and 2007 (bottom right). White areas were masked out because of cloud contamination in at least one of the images. The apparent striping is causes by Landsat problems with the scan line correction procedures which caused stripes of missing data.



Chapter 4: Saiga habitat selection and habitat distribution in southern Russia

Abstract

Understanding the habitat preferences of endangered species is important for their long-term conservation. Saiga antelope (*Saiga tatarica tatarica L., 1766*) is the last free-roaming antelope in Europe, and is highly endangered. Saiga antelope numbers plummeted, first in the early 1980s, and again in the mid-1990s following the collapse of the Soviet Union. Though the decline of Saiga antelope is primarily attributed to poaching for its horns, substantial changes in vegetation and increasing fire frequency over the last 20 years may also have affected Saiga antelope habitat selection. Unfortunately, not much is known about Saiga antelope habitat and its selection in general and its change over time. The goal of this study was to understand seasonal Saiga antelope habitat selection and to map habitat changes between 2003 and 2007 in Kalmykia, southern Russia. We analyzed Saiga antelope presence-only data for 2003-2007, collected in two nature reserves in the study region. We used Maxent (Phillips, 2006) to explore the relative importance of a suite of environmental and human-disturbance variables for Saiga habitat selection, and to predict the distribution of Saiga habitat. We constructed a global model for the entire dataset, and models based on subset of our data representing different years and seasons. We hypothesized that vegetation composition is a key factor explaining Saiga presences, while fire-affected areas will be of less interest for Saiga antelope at least in particular seasons. Second, we hypothesized that habitat selection differs significantly among seasons, but is relatively stable among years.

To our surprise we found that vegetation patterns did not explain Saiga habitat selection well, but that Saiga antelopes selected areas that burned more often. Distance to water sources and proximity to farms were also important factors influencing Saiga antelope habitat selection. Variable importance did not vary substantially between years and seasons. Overall, suitable habitat occupied about 20% of the 2,500 km^2 study area, and the amount of suitable habitat did not change substantially during the time period studied. Our results indicate that suitable Saiga habitat was characterized by variables describing environmental change. The importance of burned areas indicates that Saiga antelope appear to select open areas, where poachers and predators can be more seen easily. Though vegetation variables were less important than burned areas and human disturbance variables in our models, the burning frequency variable may be a proxy for productivity of herbs and grasses, and thus forage availability. To ensure the conservation of this species the new understanding of Saiga habitat preferences our study provides should be incorporated into management and conservation practices.

Introduction

One of the essential questions for the conservation of large ungulates is whether their population declines are caused directly by humans, for example via poaching, or indirectly, for example via habitat loss caused by land-use or climate change (Ehrlich and Ehrlich 1981). Habitat loss and fragmentation are arguably the largest threat for populations of large mammals (Simberloff 1984). Poaching is a second major threat for ungulate populations, because these species are often hunted for their meat, medicinal properties, or trophies (Milner-Gulland and Bennett 2003; Morrison et al. 2007) . Poaching is especially widespread in the developing world, where poverty often forces people to exploit wildlife populations to subsidize their diets and incomes. Moreover, where collapsing states or warfare result in an eroding infrastructure for nature protection and a low level of control, poaching can become the main cause of species endangerment (Bhatnagar et al. 2009; Dudley et al. 2002) . Habitat change and poaching often co-occur (Sodhi et al. 2004; Wilkie et al. 2000), making it difficult to isolate their relative effect. Habitat selection studies in areas where poaching is restricted (i.e., in protected areas) can reveal how environmental change affects habitat availability, and ultimately may facilitate understanding of past population change.

The Saiga antelope (*Saiga tatarica spp. tatarica*) is a migratory species whose population once numbered in the millions and whose range stretched from the Sea of Azov in the West to Mongolia in the East. Currently, saiga antelope remain in substantial yet low numbers only in one part of Russia (Republic of Kalmykia) and in a few disjunct populations in Kazakhstan and Mongolia (Bekenov et al. 1998; Milner-Gulland et al. 2003). Saiga antelope populations precipitously declined throughout their range first in the beginning of 1980s, and again after the collapse of the Soviet Union (Figure 26) (Milner-Gulland et al. 2001). When the population decline started, its causes were unclear, with some studies highlighting poaching and others suggesting that skewed sex ratios and environmental degradation were the main cause (Chan et al. 1995; Milner-Gulland et al. 1995). The harsh economic crisis of 1998 resulted in a further decline of living conditions in rural Russia. The radical withdrawal of agricultural subsidies and a lack of alternative economic opportunities resulted in local people turning to poaching Saiga antelope for meat and horns (Robinson 2000), which is now seen as the predominant factor driving Saiga population collapse (Karimova 2002; Milner-Gulland et al. 2001). These drivers of population decline were also consistent among populations in Russia and Kazakhstan (Kuhl et al. 2009a).

The same socio-economic transformations and especially the breakdown of many state-farms in the transition period also led to rapid and widespread environmental changes across the range of Saiga antelope in Kalmykia, mainly spurred by a sharp decrease in numbers of livestock . These environmental changes, however, were lagged in time. Among the documented changes were a steady increase in fire frequency and burned areas since 1980s, with substantially more burning after 1997, covering up to 20% of the Saiga antelope's range in Russia (Dubinin et al. 2010c). Associated with changes in fire regimes, substantial vegetation changes occurred, resulting in an increase of areas covered by perennial grasses at the expanse of areas covered by shrubs (Dubinin et al. 2010b). These vegetation changes may have resulted in Saiga habitat loss, potentially contributing to the Saiga antelope decline in substantial ways (Abaturov 2007). Overall, potential Saiga habitat loss is of major concern for the conservation of this species, because it is even harder to control than poaching (Karimova 2002)

During the Soviet period, Saiga antelope were an important natural resource and the species was industrially hunted and therefore the subject of numerous studies. However, these studies were rarely spatially-explicit and usually provided only descriptive analysis of habitat use and diet (Bannikov 1967; Sokolov and Zhirnov 1998). When the population decline started, research almost exclusively focused on population parameters, such as sex ratios, survival, and fecundity (Kuhl et al. 2009b; Milner-Gulland et al. 2003). Though environmental conditions were not a research focus a few studies provide insight into Saiga habitat selection, for example showing that Saiga antelopes avoid perennial grasslands (Larionov et al. 2008). Current research was focused on calving site selection and migrations in Kazakhstan and Mongolia (Berger et al. 2008; Singh et al. 2010a, b). Overall though, what determines suitable Saiga habitat and how Saiga habitat has change recently remains almost unknown.

Our overarching goal was to analyze habitat selection of Saiga antelopes antelope in the Chernye Zemli region of the Republic of Kalmykia, to predict habitat suitability, and to assess how habitat preferences and habitat patterns change among years and seasons. Our specific objectives were: 1) to predict habitat suitability from 2003 until 2008; 2) to understand the role of different environmental and human factors in characterizing suitable habitat; and 3) to determine the inter-annual and seasonal variability of habitat availability and the driving factors of habitat selection. We hypothesized that: 1) vegetation is an important factor for habitat selection of Saiga, 2) there are no significant differences in habitat patterns between years, 3) differences between suitable habitats of different seasons are substantial; and 4) fire affected areas are avoided by Saiga antelope.

Methods

Study area

Our study area occupies 2,500 km^2 of the Republic of Kalmykia and Astrakhan Region in southern Russia (Figure 27). The study area includes two protected areas: "Stepnoy/Tinguta" Nature Preserve and a part of "Chernye Zemli" State Nature Reserve. Whereas the nature preserve is protected on a periodic watch basis, with one to two rangers teams with motorcycles patrolling the territory, the State Nature Reserve has permanent protection staff.

The climate of the study area is arid, with hot, dry summers. Mean daily temperature is +24°C in July with maximum temperatures of up to +44°C. Annual precipitation is 150 to 350 mm, with most of the precipitation in spring and fall (Walter and Box 1983). The topography is rolling with a mean elevation of -15 m below sea level.

Vegetation is typical for semi-deserts of the northern Precaspian Plains. Dominant vegetation associations are shortgrass steppe (*Stipa* spp., *Festuca* spp., *Argopyron* spp., and other graminoids) and dwarf sage shrublands (*Artemisia* spp., *Kochia prostrata*) with ubiquitous annuals such as *Bromus tectorum* and *Poa bulbosa* (Golub 1994). Shortgrass steppe has short growing season in April and May and rapid senescence in the dry summer. The grasses are protected from fire by dense bunches and resprout with the first rain after fire. Dwarf shrublands have less biomass than the grasslands, but a longer growing season, and sometimes exhibit a second vegetation peak in the fall and early winter (Kurinova and Belousova 1989). The primary human land use is grazing by domestic livestock, mainly sheep, and to a lesser extent cows and goats.

Data

Occurrence data

We studied changes in habitat selection from 2003 to 2007. Our choice of study period was determined by the availability of Saiga antelope presence data. Herd observation data were collected by the staff of the Chernye Zemli State Biosphere Reserve and Stepnoi Preserve under a protocol developed during an INTAS (International Association for the promotion of co-operation with scientists from the New Independent States of the former Soviet Union, EU) project. Saiga antelopes were observed from a vehicle or on foot while on patrol between 0200 hours and 2100 hours using binoculars. Data collection was opportunistic, with some areas being covered better and more often than others, thus introducing some sampling bias (Phillips et al 2009, see below). Different roads were used by rangers depending on road conditions and passability due to weather conditions.

A total of 1,979 direct observations of individuals and herds collected from September 2003 until December 2007 were analyzed. The maximum distance at which saiga were detected was about 600 meters, with a typical range of 250-600 (A. Khludnev, personal communication). To reduce positional uncertainty in our presence data, we therefore generalized the points to a 250-m (binary: presence / background) grid for total, yearly and seasonal models. A second reason for generalizing our dataset was to reduce spatial autocorrelation effects. Generalizing decreased the number of locations to about half of the full dataset (i.e., leaving 851 independent observations, Table 13).

Predictor variables

As a proxy for vegetation productivity, we calculated total, annual, and seasonal (across the entire period of study) mean and standard deviation of the Natural Difference Vegetation Index (NDVI) based on 250-m resolution MOD13A1 (23, 16-day composites per year) images. In addition, we used Landsat-based phenological vegetation classifications for 1998 and 2007 representing nine classes: 1 - mixed grasslands with both annuals and perennials, 2 - vegetation communities co-dominated by shrubs and perennial grasses, 3 – pure perennial grasses, 4 – annual grasses and shrubs, 5 – pure shrubs, 6 – degraded areas, 7 - vegetation communities dominated by annuals, 8 - bare ground and 9 – water (Dubinin et al. 2010a).

To capture burned area, we extended the burned area database from Dubinin et al. (2010b) to cover our entire study area. Burned areas were digitized from satellite data from 1998, when active burning started, until 2008 (Dubinin et al. 2010c) using the

RESURS/MSU-SK and TERRA/MODIS MOD02QKM Level 1B calibrated radiance images (Bowman et al. 2003). Delineating burned areas was straightforward due to distinct fire scar boundaries, often created by linear features such as roads. Furthermore, there are no other disturbances in the area that could have resulted in comparable patterns. Burned areas remained visible for 2 to 3 years after burning, unless a new fire overrode an older scar (Dubinin et al. 2010c). For each year of the study period we calculated two fire metrics for each 250-m pixel: 1) a binary classification of burned versus non-burned and 2) the number of times a particular area burned prior to the current year (i.e., in 2003 the variable described the number of times a pixel had burned from 1998 until 2002).

We used the Shuttle Radar Topography Mission (SRTM) elevation model (http://srtm.csi.cgiar.org) to obtain absolute elevation and a terrain ruggedness index (Riley et al. 1999). These two measures were used as proxies of disturbance, as Saiga antelope are assumed to select for higher and more flat terrain (A. Khludnev, personal communication). Human disturbance was measured as the distance to active farms. Initially, we also tested a distance-to-roads variable for inclusion in the models, but removed it because it interfered with our sampling bias in the occurrence data. Since most of the Saiga locations were obtained along roads, this variable caused areas near roads to be identified as optimal habitat, which was ecologically not meaningful.

Habitat suitability modelling

We used a maximum entropy modeling approach to analyze Saiga antelope suitable habitat. Maximum entropy modeling is a machine learning approach that uses species' presence data to predict their spatial distributions (Phillips et al., 2006). The true, but unknown, distribution of a species is approximated by deriving a probability distribution using constraints inferred from environmental variables associated with occurrence data points. The maximum entropy principle suggests that the distribution that approximates the 'true' distribution best given all current knowledge is the distribution with maximum entropy (Jaynes, 1957). To avoid over-fitted models, regularization parameters are used (Phillips et al., 2006; Phillips and Dudik, 2008). Maximum entropy models perform well with small sample sizes (Wisz et al., 2008) and frequently outperform classical statistical approaches in their predictive power (Elith et al. 2006).

To fit maximum entropy models, we used the software Maxent (version 3.3a, http://www.cs.princeton.edu/~schapire/maxent/). For all model runs we used a maximum number of 2,500 iterations with 5 replicate runs and default regularization parameters (Phillips and Dudik, 2008). We used 75% of the data to train models and 25% to test them. To estimate Maxent distribution, 10,000 random background points were collected. To account for sampling bias in our Saiga locations (because occurrence points were mostly collected along roads), we sampled background points only within 1-km buffers around roads (Phillips et al. 2009). The distance of 1 km was chosen because it represented the 50% percentile of the distribution of all presence point distances to roads.

Model validation was based on the area under the curve (AUC) of the receiver operating characteristics (ROC) curve (Fielding and Bell, 1997; Phillips et al., 2006; Wiley et al., 2003). A variable's importance was measured as: 1) the permutation importance which compares the drop in training AUC if training and background data are permuted, and 2) jackknife of AUC for the test data when using only a single-variable model. Habitat suitability maps were predicted by applying Maxent models to all cells in the study region, using a logistic link function to yield habitat suitability index values (HSI) between zero and one (Phillips and Dudik, 2008).

We used comparable sets of variables for our overall, annual (5), and seasonal (4) models, with the exception that the burned – non-burned area variable was excluded from the seasonal model because our burned area data did not have sufficient temporal resolution.

Results

Overall variable importance

Our overall habitat suitability model included ten predictors (cumulative burned area and burning frequency, distances to farms and water sources, vegetation in 1998 and 2007, mean and standard deviation NDVI and finally, absolute height and terrain ruggedness index). The overall model had a test AUC of 0.77 (range across the 5 replicate runs: 0.71–0.807) with a mean standard errors of 0.01. Habitat selection was mainly determined by five predictors: burning frequency, distance to farms, distance to water sources, standard deviation of NDVI, absolute height and total burned area, accounting for relative gain contributions of 36.6%, 24.4%, 18.6%, 5.9%, 3.5%, 3.4% respectively (combined, 92.4%) (Table 14).

Overall habitat distribution

Saiga antelope habitat suitability was highest in the central part of the study area on the border between Stepnoi Nature Preserve and Chernye Zemli Nature Reserve. Saiga antelope habitat with an HSI value larger 0.5 covered about 20% of the study area (~454 km^2), most of which occurred in one, contiguous patch. Habitat with an HSI value larger 0.6 covered only 7% of the study area (168 km^2), and areas with an HSI value larger 0.7 were small, occupying about 2% of the study area (Table 17). More suitable habitat (HSI > 0.6) formed three separate patches; two of these patches were located in the northern part of the study area, where spring calving grounds are located, and the remaining one was located in to the south of this area.

Relationships between variables and habitat suitability in the overall model

The overall model did not find substantial differences in habitat suitability between burned and unburned areas, and vegetation type in 1998 and 2007. For continuous variables, habitat suitability increased for areas that burned more often and areas with higher elevations. Saiga preferred also areas away from farms. Habitat suitability increased somewhat with increases in mean NDVI. Habitat suitability was negatively related to increasing distance to water sources. The standard deviation of NDVI, and terrain ruggedness had no apparent association with habitat suitability (Figure 31).

Annual and seasonal variability in variable importance

As in the global model, the most important variables in the yearly and seasonal models for 2003-2007 were distance to water sources and burning frequency. These two variables combined contributed 60% to the overall model. The order of variable importance differed slightly among our two different metrics of variable importance (AUC on test data). Burning frequency was always first, elevation second, and distance to water sources and farms were the third and fourth most important variables (Table 16). Generally, the impact of vegetation-related variables, both NDVI-based and those derived from vegetation classifications, was lower than burning and distance variables. The binary burned / not-burned variable also had very high permutation importance in 2004.

Annual and seasonal variability in habitat distribution

Spatially, we did not find substantial differences in suitable habitat distribution among years (Figure 29). The distribution of suitable habitat predicted by the global model was similar to the distribution predicted by the yearly models. The total area covered by suitable habitat (HSI> 0.5) was also comparable among years, with the exception of 2007, when it decreased from the average 20% of the total area to about 10% (Table 17). There were, however, some seasonal differences in the distributions of habitat suitability. In winter, suitable habitat was more likely to be found in the NW corner of the study area, whereas in spring and summer suitable habitat was concentrated on the calving grounds in the north. In fall, suitable habitat was more dispersed. Overall though, these seasonal changes were not marked, and a general NW-SE gradient of habitat suitability was retained throughout the year (Figure 30). Areas covered by habitat seasonally with HSI values larger than 0.5 and 0.6 were comparable to the annual distributions, though somewhat smaller. Areas with HSI larger than 0.5 covered the smallest area in spring (14% of the study area) and were most extensive in fall (20% of the study area) (Table 17).

Discussion

Our study represents one of the first attempts to understand habitat selection of Saiga antelope in the last remaining European population in Kalmykia (Russia) at the scale of the protected range. This is a constrained population that no longer migrates long distances. Our results indicate a higher importance of human-related factors, such as distance to water sources and farms, than natural ones, such as vegetation, in determining Saiga antelope distributions. Patterns of suitable habitat among years and seasons were rather consistent. Saiga antelopes aggregated in the vicinity of the calving grounds even outside of the calving season. Most probably, the higher presence of Saiga antelope in this specific location is explained by a combination of factors: presence of water sources, it is the traditional area of the rut, and of calving (Arylova 2008), the area was predominantly flat compared to other parts of the study area, systematic fires improve visibility and forage, and there is little disturbance from vehicles and livestock.

We found a surprisingly strong influence of burning frequency in explaining habitat selection by Saiga antelope. Frequently burned areas might be more beneficial for Saiga antelope from both a forage and disturbance standpoint. Firstly, the areas that burn more frequently might have a specific vegetation composition favored by Saiga antelope. Grasses resprout rapidly after fire, even if there no rain and ungulates are shown to react on it by higher densities after fire (Archibald and Bond 2004; Everson and Everson 1987). The effect of burning on diet quality can persist for several years and be different depending on the season (Hobbs and Spowart 1984). Secondly, burning is a good predictor of the openness of the territory - areas that burned more often are characterized by improved visibility and passability for Saiga antelope (A. Khludnev, personal communication). Presence of continuous unburned vegetation cover made it hard for Saiga antelope to gain speed, and introduced a high level of noise from wind moving through plants, decreasing their ability to see and hear predators and poachers. We caution however, that it is possible that there was a reverse cause-effect relationship in which there is more burning because of saiga presence (i.e. fires set by poachers to aid hunting), but there are as yet no data to confirm this and there are many fires in the areas where Saiga are not present as well.

The low importance of vegetation in characterizing Saiga habitat suitability was certainly surprising. Soviet era studies and current research indicates that diet preferences are strongly seasonal and Saiga selected vegetation types that had greener biomass (e.g., shrub-dominated communities in the fall and winter and grasslands during spring-summer period). Recent studies on Saiga antelope diet indicate an avoidance of *Stipa sp.* altogether (Larionov et al. 2008). This is of concern, because of the continuing transition of the region to a greater dominance of this vegetation community (Abaturov 2007; Dubinin et al. 2010a). Fine-scale studies of Saiga antelope habitat selection in Kazakhstan emphasize the importance of green biomass measured by NDVI, especially during migration and calving (Leimgruber et al. 2001; Mueller et al. 2008; Singh et al. 2010b). However, our results do not support these finding - according to our models, neither greenness as measured by NDVI nor vegetation types were particularly strong predictors of suitable habitat, compared to our other variables. The lack of importance of vegetation variables indicated that the selection for different vegetation communities might be happening at finer scales, which were not captured by either our occurrence data or environmental layers, especially after generalization to 250-m pixels.

Though our study area was dominated by protected areas, the effect of human presence was still apparent in our models via the decreasing habitat suitability closer to farms. While we do not suggest that farmers themselves poach, farms are connected to transportation infrastructure which may make it easier for poachers to enter areas where Saiga antelope are present. Unfortunately, the sampling design and the use of off-road motorcycles by poachers impede modeling poaching pressure more directly. Finally, the very strong effect of distance to water sources highlights the aridity of the climate in the study area and the dependence of Saiga antelope on available water during the summer and even in spring and fall. Currently the area of NBR "Chernie Zemli" does not have any artificial water sources and that may make the "Stepnoi" preserve more attractive to Saiga antelope. Habitat suitabilty in NBR "Chernie Zemli" may be increased through creation of more artificial water sources.

The occurrence dataset that we used is the only existing consistent dataset for our study area. However, this dataset suffered from a number of problems that could have potentially affected the results of our analysis. Firstly, the sampling effort does not follow specific protocol and oversampling in certain areas is likely a problem. Secondly, the locations do not represent the exact locations of the animals. This is why we aggregated the occurrence data (and all our predictor variables) to 250-m grid cells. Nevertheless, our training and test data were not truly independent and the resulting AUC of the models might be inflated due to autocorrelation in the residuals (Veloz 2009).

The clear patterns that we observed allowed us to make some management recommendations despite the limitations of the source data. We suggest that the key management implications for Saiga antelope resulting from this study are: habitat will be most effectively improved by optimizing the protected areas network, implementation of targeted anti-poaching activities, and enhanced monitoring. Because optimization of the network through removal of active farms form the management area (*Zakaznik*) is not feasible, we recommended building more water wells in the Nature Reserve where there are no farms, and no water sources either. Expanding the protected areas further to the north and center of our study area would also be helpful to fully capture currently unprotected portions of suitable habitat. Anti-poaching activities should be better focused on the area that we identified as suitable habitat. And last but not least, better wolf and saiga monitoring programs need to be launched to obtain more solid data on population dynamics and predation pressure. Given the ecological importance of fire, we do not advise fire suppression, but rather suggest use of prescribed burning to ensure that areas burn at least once every two years.

The dramatic decline of Saiga antelope since the 1980s was concomitant with strong environmental changes, and especially a rapid increase in the area burned each year, and changes in vegetation types. This led to the hypothesis that habitat changes were the primary reason for Saiga antelope population collapse, but our habitat selection results do not support this hypothesis. Currently, the avoidance of disturbed areas by Saiga antelope, represented by poaching and predation, appears to be more important for the population than the presence of good foraging grounds. One reason for the lack of a clear relationship between vegetation type and saiga habitat quality might be that overall low densities of Saiga places population numbers far below their carrying capacity, and forage quality may there fore not be a limiting factor. However, the ongoing change of the area to more grass-dominated communities with impaired palatability for Saiga antelope should be carefully monitored to ensure that saiga prosper here in the longer term.

Acknowledgements

We gratefully acknowledge support for this research by the NASA Land-Cover and Land-Use Change Program (NNG05GA74G), by a NASA Earth System Science Dissertation Fellowship for M. Dubinin (07-Earth07F-0053), by the U.S. Fish and Wildlife Service, by a grant from the National Geographic Society (8360-07) and by Fundamental Research Program "Russia's Biological Resources: assessment and monitoring fundamentals", (Project PII.10 by the Department of Biology Sciences, Russian Academy of Sciences). We are also grateful to the staff of Stepnoi Nature Preserve and Chernye Zemli Reserve for collecting occurence data. We thank T. Kuemmerle for help with Maxent and A. Estes for editing.

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Tables

Table 13. Predictors used to model suitable habitat and determine its drivers and variability in Southern Russia.

Variable	Type	Range	Source	Models
Burned area	Categorical	0-1	MODIS	annual, overall
Burned area frequency	Continuous	0-6	MODIS	overall, annual, seasonal
Vegetation in 2007	Categorical	1-9 (see text for classes description)	Landsat	overall, annual, seasonal
Vegetation in 1998	Categorical	1-9 (see text for classes description)	Landsat	overall, annual, seasonal
Elevation	Continuous	-37 – -9 (m)	SRTM	overall, annual, seasonal
Terrain ruggedness index	Continuous	0 – 20	SRTM	overall, annual, seasonal
Distance to farms	Continuous	0 – 25,500 (m)	Topographic maps	overall, annual, seasonal
Mean NDVI	Continuous	0 – 10,000	MOD13A1	overall, annual, seasonal
Standard deviation NDVI	Continuous	0 – 10,000	MOD13A1	overall, annual, seasonal
Distance to	Continuous	0 – 27,000 (m)	Topographic	overall, annual,

water sources	maps	seasonal

Table 14. Amount of data used for model construction and resulting performance of the models. SD AUC represents average standard deviation across 5 replications, while each SD AUC is calculated by folding and using remaining folds for cross-validation, repeating for every replication.

	Occurrence points	250-m grid occurrence points	Training data AUC	Test data AUC	SD AUC
2003	120	104	0.84	0.81	0.04
2004	260	133	0.81	0.86	0.03
2005	573	196	0.78	0.77	0.03
2006	428	251	0.77	0.71	0.03
2007	482	277	0.86	0.85	0.02
Winter	614	313	0.81	0.77	0.02
Spring	431	225	0.83	0.79	0.03
Summer	411	213	0.81	0.75	0.03
Fall	523	317	0.81	0.77	0.02
Overall	1979	851	0.77	0.77	0.01

Table 15. Variable contribution measured as permutation importance, percentages, averaged over 5 runs: a) overall and annual models, b) seasonal models

a)

	Overall	2003	2004	2005	2006	2007	sum	mean
burning frequency	27.8	35	14	35	36	24	143	29
burned-non burned	0	0	25	0	5	1	32	6
distance to water	35	21	24	13	15	51	123	25
distance to farm	12	6	12	6	10	8	42	8
elevation	7	8	8	27	5	6	53	11
terrain ruggedness	1	1	4	6	3	2	15	3
vegetation type in 1998	1	1	4	2	3	0	11	2
vegetation type in 2007	0	5	5	8	17	5	40	8
mean NDVI	4	13	1	1	6	0	21	4
standard deviation NDVI	10	9	3	2	2	3	19	4

b)

	Winter	Spring	Summer	Fall	sum	mean
burning frequency	44	29	20	24	116	29
distance to water	11	38	34	38	121	30
distance to farm	8	10	13	11	42	10
elevation	16	7	11	5	38	10
terrain ruggedness	4	2	5	1	13	3
vegetation type in 1998	1	2	2	2	7	2
vegetation type in 2007	2	6	13	3	24	6

burned-non burned	7	3	1	6	17	4
mean NDVI	8	3	2	10	23	6
standard deviation						
NDVI	44	29	20	24	116	29

Table 16. Results of jackknife test for variable importance as AUC on test data sorted by sum, as percentages, averaged over 5 runs: a) overall and annual models, b) seasonal models

a)

	Overall	2003	2004	2005	2006	2007	sum	mean
burning frequency	70	69	66	74	69	71	348	70
burned-non burned	63	50	70	48	66	53	288	58
distance to water	62	66	59	54	58	74	310	62
distance to farms	62	60	55	57	59	66	297	59
elevation	63	63	65	70	63	65	326	65
terrain ruggedness	53	51	49	52	50	52	254	51
vegetation type in 2007	62	50	54	53	55	58	270	54
vegetation type in 1998	54	58	65	59	64	63	310	62
mean NDVI	53	58	62	56	50	55	282	56
standard deviation NDVI	58	66	66	55	56	57	300	60

b)

	Winter	Spring	Summer	Fall	sum	mean
burning frequency	72	69	68	71	280	70
distance to water	54	56	57	55	222	56
distance to farms	64	64	60	62	251	63
elevation	58	64	60	60	242	61
terrain ruggedness	57	67	66	65	256	64
burned-non burned	71	64	62	64	260	65
vegetation type in 1998	52	51	48	51	201	50
vegetation type in 2007	58	47	51	57	213	53

mean NDVI	63	61	65	59	248	62
standard deviation NDVI	72	69	68	71	280	70

Table 17. Amount of suitable and highly suitable habitat and its change across years and seasons in percents of total study area.

	>50%	>60%	>70%
2003	18%	8%	2%
2004	16%	10%	3%
2005	22%	9%	2%
2006	23%	8%	2%
2007	10%	5%	2%
Winter	17%	7%	2%
Spring	14%	6%	2%
Summer	16%	6%	2%
Fall	20%	7%	2%
Overall	22%	8%	2%

Figures

Figure 26. Dynamics of livestock and Saiga antelope populations from 1952 to 2008 in Kalmykia.

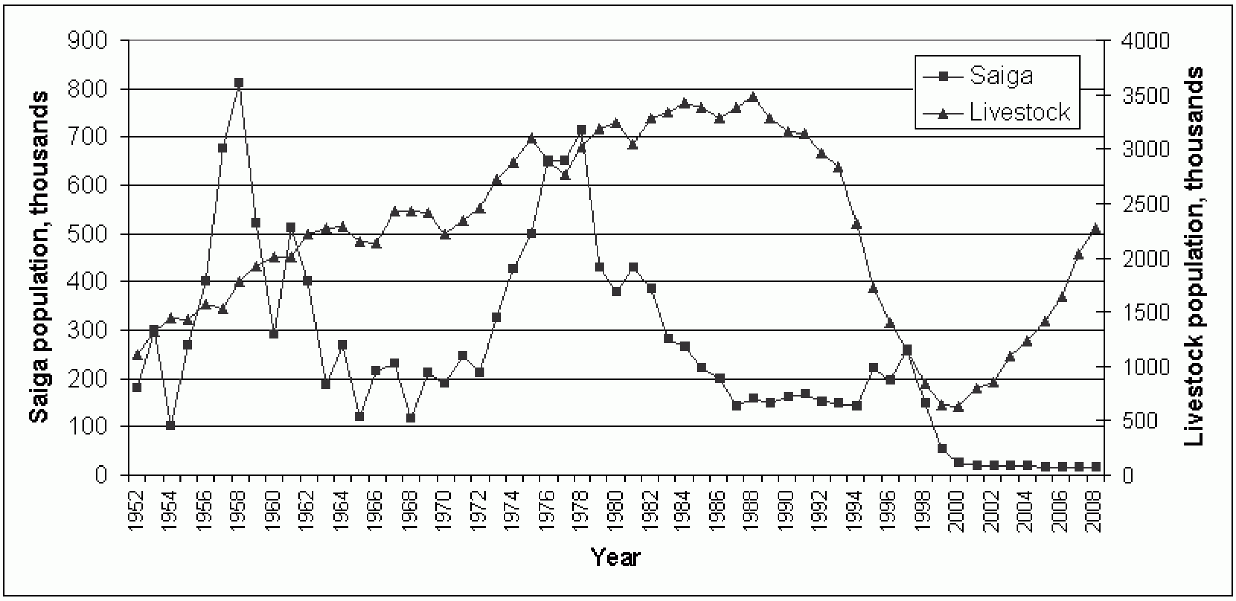

Figure 27. Study area location in southern Russia. Dashed lines in the study area represent protected areas (left - Stepnoi Nature Preserve and right - Chernye Zemli Nature Reserve), hatched area with black outline – study area, thick grey line – administrative boundary of Republic of Kalmykia.

Figure 28. Habitat suitability map using overall dataset showing the point-wise mean. Blue represents preferred habitat, brown non-preferred, white missing data. Violet dots represent Saiga antelope locations used as training points, green - locations used for validation. Dots represent one of the replications and provided as an example.

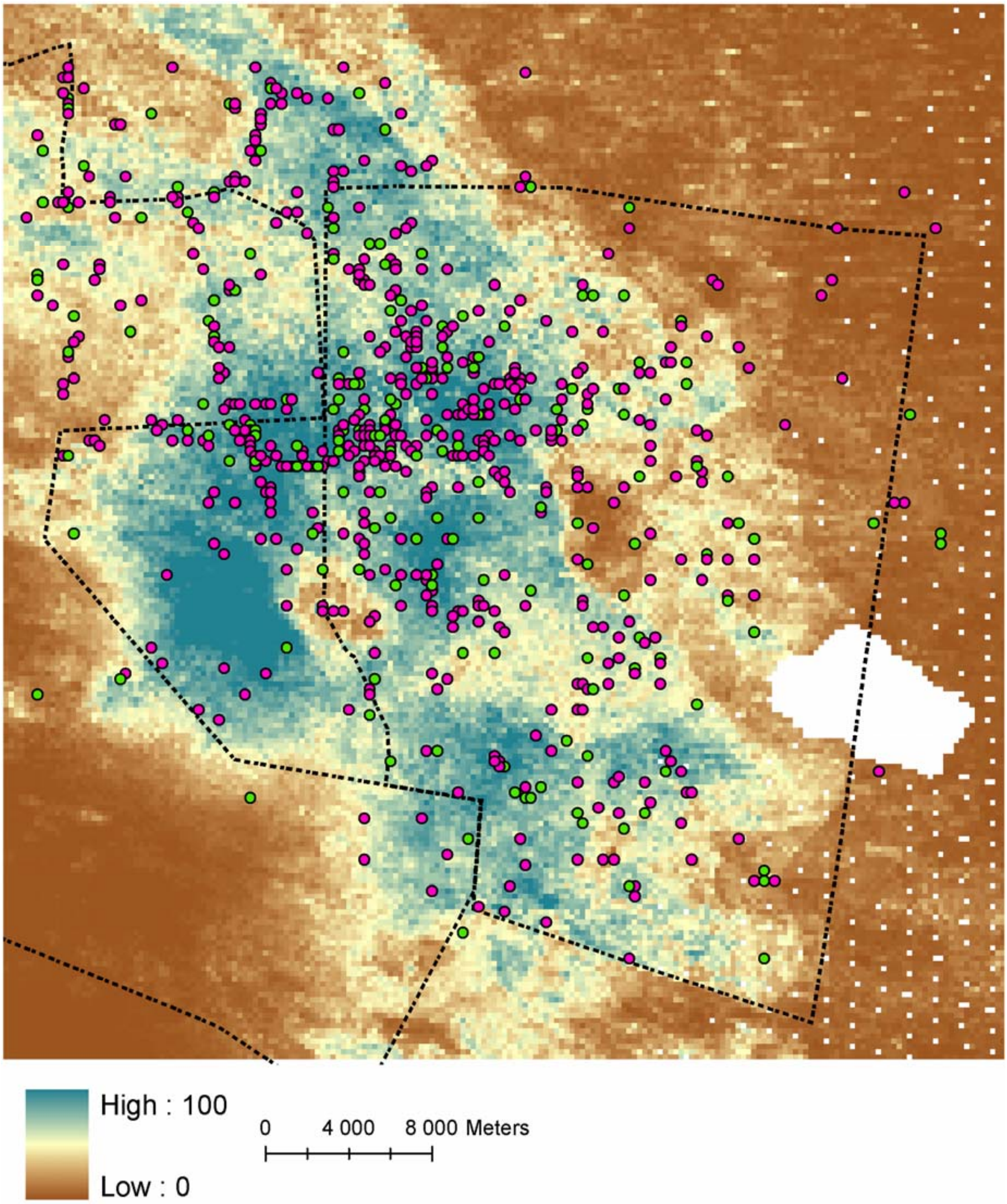

Figure 29. Annual distribution of suitable habitat, averages over 5 replications are shown.

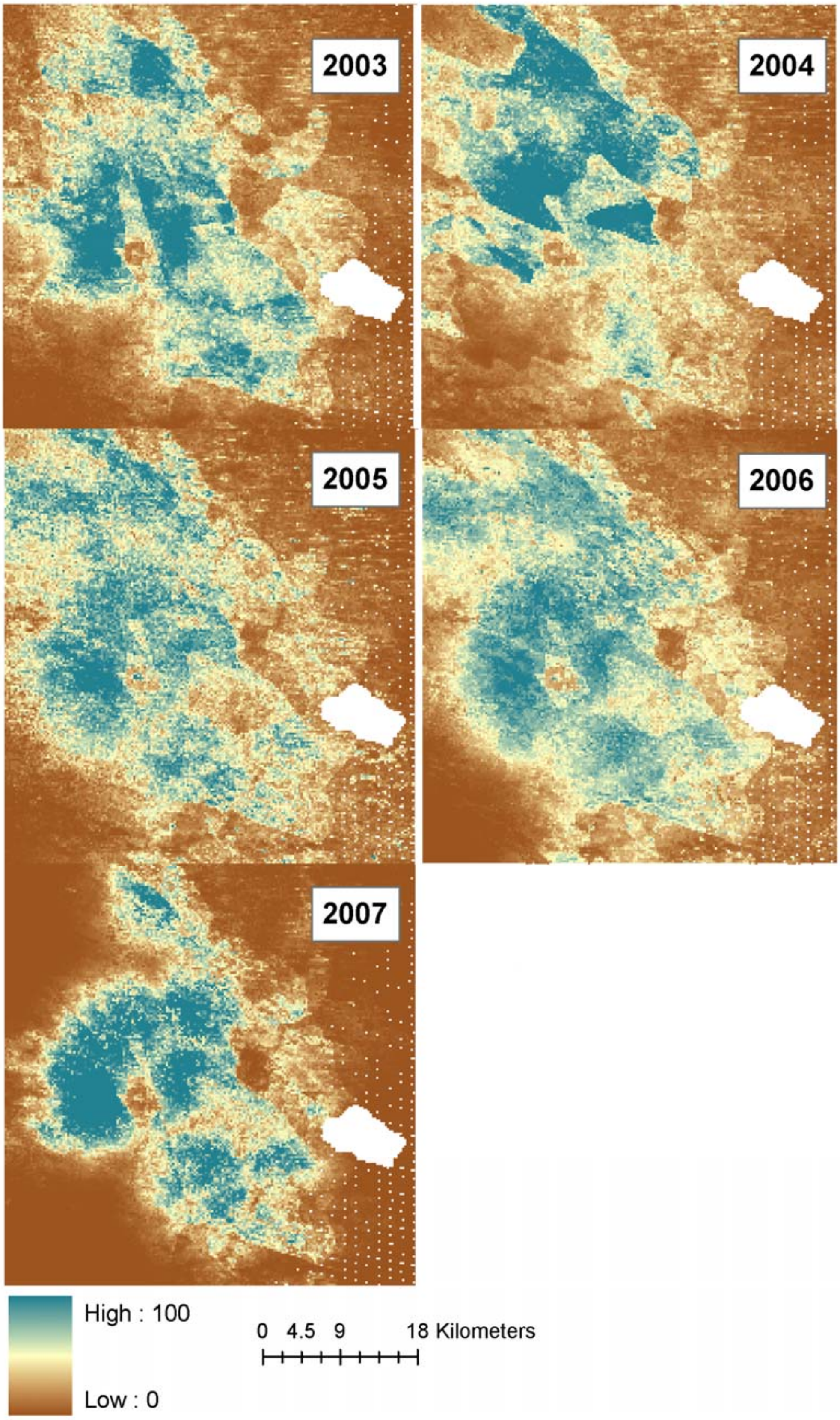

Figure 30. Seasonal distribution of suitable habitat, averages over 5 replications are shown.

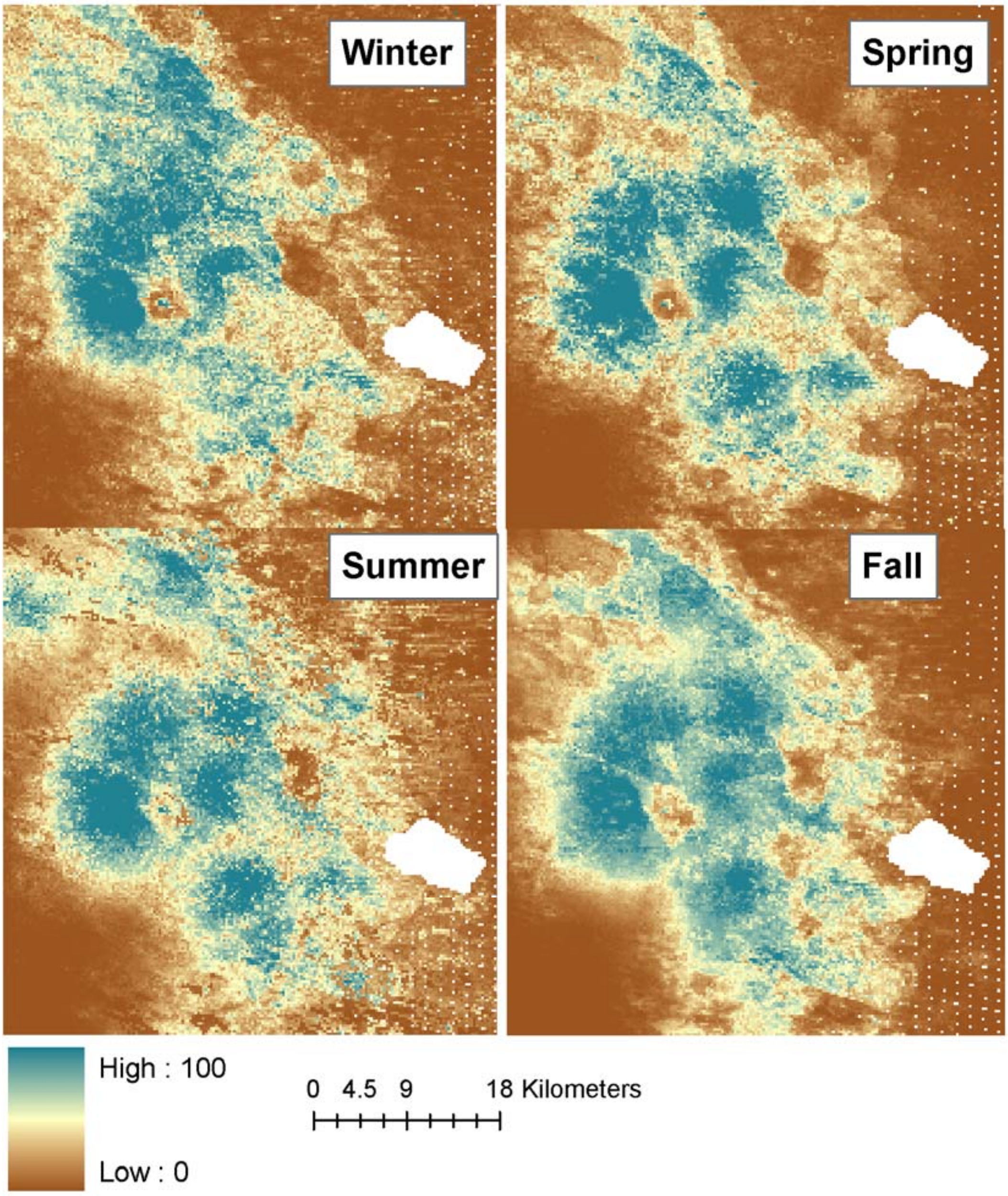

Figure 31. Response curves showing relationship between the particular variable and logistic output (probability of presence) according to the global model. Gray lines indicate minimum and maximum from 5 replications and thick black line represents a mean. See Table 1 for explanations of units of x-axis.