IMPACTS OF CATTLE GRAZING ON SPATIO-TEMPORAL VARIABILITY OF SOIL MOISTURE AND ABOVE-GROUND LIVE PLANT BIOMASS IN MIXED GRASSLANDS

By

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A thesis submitted to

The Faculty of Graduate and Postdoctoral Affairs

in partial fulfillment of the requirements for the degree of

Doctor of Philosophy

Department of Geography and Environmental Studies

Carleton University

Ottawa, Ontario

(November, 2012)

Your file Votre référence
ISBN: 978-0-494-94234-5

Our file Notre référence
ISBN: 978-0-494-94234-5

Abstract

Areas with relatively high spatial heterogeneity generally have more biodiversity than spatially homogeneous areas due to increased potential habitat. Management practices such as controlled grazing also affect the biodiversity in grasslands, but the nature of this impact is not well understood. Therefore this thesis studies the impacts of variation in grazing on soil moisture and biomass heterogeneity. These are not only important in terms of management of protected grasslands, but also for designing an effective grazing system from a livestock management point of view. This research is a part of the cattle grazing experiment underway in Grasslands National Park (GNP) of Canada since 2006, as part of the adaptive management process for restoring ecological integrity of the northern mixed-grass prairie region. An experimental approach using field measurements and remote sensing (Landsat) was combined with modelling (CENTURY) to examine and predict the impacts of grazing intensity on the spatial heterogeneity and patterns of above-ground live plant biomass (ALB) in experimental pastures in a mixed grassland ecosystem. The field-based research quantified the temporal patterns and spatial variability in both soil moisture (SM) and ALB, and the influence of local intra-seasonal weather variability and slope location on the spatio-temporal variability of SM and ALB at field plot scales. Significant impacts of intra-seasonal weather variability, slope position and grazing pressure on SM and ALB across a range of scales (plot and local (within pasture)) were found. Grazing intensity significantly affected the ALB even after controlling for the effect of slope position. Satellite-based analysis extended the scale of interest to full pastures and the surrounding region to assess the effects of grazing intensity on the spatio-temporal pattern of ALB in mixed grasslands. Overall, low to moderate grazing intensity

showed increase in ALB heterogeneity whereas no change in ALB heterogeneity over time was observed for heavy grazing intensity. All grazing intensities showed decrease in spatial range (patch size) over time indicating that grazing is a patchy process. The study demonstrates that cattle grazing with variable intensity can maintain and change the spatial patterns of vegetation in the studied region. Using a modelling approach, the relative degrees to which grazing intensity and soil properties affect grassland productivity and carbon dynamics at longer time-periods were investigated. Both grass productivity and carbon dynamics are sensitive to variability in soil texture and grazing intensity. Moderate grazing is predicted to be the best option in terms of maintaining sufficient heterogeneity to support species diversity, as well as for carbon management in the mixed grassland ecosystem.

Acknowledgements

I would like to extend sincere thanks to my supervisor, Dr. Scott Mitchell, for his guidance and valuable feedback throughout my PhD program. Special thanks go to my advisors Dr. Andrew Davidson and Dr. Elyn Humphreys for their time, helpful suggestions and comments.

This thesis was partially funded by a NSERC Discovery research grant awarded to Dr. Scott Mitchell, as well as in-kind contribution of Parks Canada; special thanks go to Parks Canada for providing logistics and data support. I also acknowledge the financial support in the form of "Torrance Graduate Scholarship 2008" from the Department of Geography and Environmental Studies, Carleton University.

I had substantial support throughout the duration of my PhD program and would like to make special note of all the people who made this journey, both comforting and manageable. I would especially like to acknowledge my friend Sheri Burke for reviewing my initial work and giving me valuable advices. I would also like to thank my field assistant Benjamin Deschamps for the long driving hours and assisting me with the collection of data throughout the field work conducted in summer 2008. I would also like to thank all the people, especially Pat Fargey and Rob Sissons at the Grasslands National Park (GNP) in Val Marie, Saskatchewan for providing resources, friendly smiles and enthusiasm towards my work; and Melody Mastad from Carleton University library for sharing her knowledge of grazing management in Saskatchewan and her support throughout my PhD program. From my colleagues and friend during my work at Agriculture Canada, I would like to thank Kevin Crichlow for providing help with ArcGIS. Special thanks go to Kristie for all her support especially at the end of my PhD journey. I am forever indebted.

I would like to thank my family for all their support and encouragement towards completion of my PhD. My deepest appreciation goes to my sweetest mom and best friend Ehab who have been incredibly supportive emotionally, financially and confident in my ability to accomplish what I started.

Finally, I would acknowledge the Carleton University and all the people who have touched my life in some way with their friendly smiles and inquisitive nature regarding my completion of thesis, in turning my dream into reality. Big thanks to all of you.

This thesis is solely dedicated to my mom, without her love and encouragement; it would have been an impossible task.

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"Nearly every acre of range has other uses and values besides forage production—to protect watersheds, give wildlife a home....these are the "other" values of the range. Each is important; on some ranges, indeed the demands of one or more may dominate or even exclude grazing. If grazing is properly managed, however the various uses are usually compatible with the use of forage by livestock."

Connaughton, C.A. (1948, p. 239)

1.0 Introduction

Grassland ecosystems were once mosaics of many different plant communities and animal species, mainly as a result of interactions of climate, fire and grazing. However, today little of this diverse ecosystem remains and existing natural grassland diversity is threatened (PCAP 2003). This is due to modifications such as controlled grazing from domesticated livestock (chiefly cattle and sheep) or complete eradication of grazing, fire suppression, introduction of exotic or alien plant species and overuse by human activities, such as conversion of grasslands into homogeneous croplands.

Grasslands play important roles in carbon storage, soil organic matter conservation, water cycle, and biodiversity at different scales. Therefore, management of these prized resources requires accurate information about their extent and spatial distribution, as well as the factors controlling their structure and functioning. Precipitation and temperature are particularly important climatic factors that influence grassland structure and functioning (Parton *et al.* 1994), since they largely determine the rate at which biological and chemical reactions occur. For example, processes involving production of new organic matter by plants and the decomposition of dead organic matter by microbes are temperature and moisture dependent

(Aber and Melillo 1991). In addition, climate determines what plants will grow and what kind of animals will inhabit the region.

A spatially heterogeneous area will have more biodiversity in terms of plants and animals compared to spatially homogeneous areas. This is because a heterogeneous area will provide habitat for species that prefer various configurations of grasses, forbs, and shrubs. For example, vegetation cover is a strong determinant of avian abundance during the breeding season, when individuals seek out appropriate breeding habitat (Fletcher and Koford 2002).

Vegetation indicators such as vegetation heterogeneity (refers to spatio-temporal variability in the structure and composition of plant communities), vegetation cover and plant productivity are important for monitoring biodiversity (Noss 1990). This is because a wider variety of vegetation heights (structure) and cover within a grassland ecosystem will provide a diverse suite of nesting and feeding habitats for grassland birds and animals. Therefore, it is important to acquire knowledge of the spatio-temporal patterns and variations in above-ground live plant biomass (ALB) and the factors responsible for such variability at multiple scales. This is because a multiple scale study will help take into consideration the spatial variability of vegetation, soils, and microclimates for future research and management plans. Additionally, it will also help track the vegetation response to year-to-year weather variation and management practices such as grazing. This is especially essential from a management perspective if the goal is to conserve and maintain the native grassland areas and the biodiversity within it.

Plant growth and survival is affected by the amount of plant available water in the soil, which is a function of soil properties such as texture (Breashears and Barnes 1999; Mendez-Barroso *et*

al. 2009). Since grazing can alter the soil hydraulic and mechanical properties, which in turn affect the plant available water and thus plant productivity, it is important to understand the interaction between soil properties and grazing for better understanding of plant processes (Krummelbein *et al.* 2009).

In this research, a combination of field data, satellite data and modelling is used to address both seasonal and inter-annual temporal effects, varying between three to twenty years, of different grazing intensities, and to assess the impacts of factors such as soil texture and weather variability on ALB and soil moisture variability in a mixed-grassland ecosystem at a range of scales. Among a variety of potential definitions, grazing intensity in this study refers to the cumulative effect grazing animals have on the land during a particular time period, expressed as percent utilization (Holechek *et al.* 2001). I only consider the aboveground biomass (field and satellite based analyses) in most of this study, because of the difficulty of collecting the below-ground biomass data and the difficulty in assessing it using remote sensing techniques (Lefsky *et al.* 2002; Patenaude *et al.* 2004; Naesset and Gobakken 2008). However, we know that grasses, particularly in semi-arid or arid conditions, invest a substantial amount of resources into below-ground productivity (Aber and Melillo 1991). Therefore, I employed a modelling approach to examine the impact of grazing intensity on predicted aboveground and belowground biomass data. Additionally, I collected multiple point field measurements at a plot scale (30 m x 30 m) to measure the spatial distribution of soil moisture and ALB. For larger extent (regional) mapping of soil moisture and ALB, use of in situ sensors is not practical and feasible. This is because a dense network of point observations would be required to accurately map the high spatial variability of soil moisture and ALB, and this would be very

expensive (Ujjwal Narayan *et al.* 2004). Therefore, soil moisture analysis in this study is limited to plot scale. Although microwave (radar) remote sensing could be used for the soil moisture mapping, it is constrained by spatial and temporal resolutions, cost and the interpretation of surface backscatter which is difficult because of the interactions between vegetation and underlying soils (Jackson and Le Vine 1996; Western *et al.* 1998). Satellite based ALB data was used to assess the heterogeneity in ALB at landscape scale.

1.1 Heterogeneity: Definition, importance and factors causing heterogeneity

Heterogeneity is defined as the complexity and/or variability of a system property, both in space and time, due to factors such as grazing, climate and land-use change, where a system property can be any factor, including plant biomass, soil moisture, and soil nutrients (Dutilleul and Legendre 1993; Li and Reynolds 1995). Since nature is intrinsically variable, variation exists everywhere and needs to be taken into account for good management practices. Soil, for example, is the product of highly variable natural processes including inputs, losses, transformations and translocations within the soil profile, therefore soil variability also factors into the diversity of both natural and managed environments. Diversity in the soil itself supports diverse landscapes, in addition to a variety of habitats for living organisms (Slater 2008). Highly variable soil conditions in combination with climatic factors such as precipitation also modify the availability of moisture for plants, which is essential for plant growth. In a grassland context, grazing associated activities such as trampling and wallowing can also modify the soil structure, inadvertently affecting the soil water holding capacity and plant available water, which are significant modifications to controls on plant growth in water-limited regions. Therefore, it is not surprising that both available soil moisture and primary productivity are

highly variable in both space and time. This creates additional challenges for accurate predictions and management of grassland ecosystems at a range of scales (local, regional and global).

Past research has shown that grazing has been a major driver in the evolution of the prairies by altering the spatial heterogeneity of vegetation due to selective grazing patterns, thus affecting the biodiversity of a region (Bock *et al.* 1993; Hobbs 1996; Collins *et al.* 1998; Dechant and Euliss 2001). For example, a study conducted by Truett *et al.* (2001) found that the presence of large herbivores such as bison increases the faunal diversity, especially among small birds and mammals that flourish in vegetation mosaics. As another example, moderate to heavily grazed grasslands with clumps or patches of woody vegetation provides the best habitat for both scaled (*Callipepla squamata*) and bobwhite quails (*Colinus virginianus*) (Saiwana *et al.* 1998). Furthermore, large wallows (depressions in the ground) generally created by large herbivores, such as bison, when abandoned, may seasonally hold water and support mesic and even aquatic vegetation (Knapp *et al.* 1999). In contrast, homogeneous tracts of lands will result in habitat suitable for only a subset of plants, birds and animals with preference to particular vegetation, such as Sprague's Pipit (*Anthus spragueii*) which is associated with medium-height grasses and moderate litter depth (Fuhlendorf and Engle 2001; Davis 2004).

Both plant and animal biodiversity within grassland ecosystems are not only dependent on the level of grazing (or grazing intensity), but are also affected by the timing of grazing and the animal species involved (Hulme *et al.* 1999; Humphrey and Patterson 2000). For example, overgrazing may often lead to land degradation by causing severe loss of soil fertility and the

loss of biodiversity, while too little grazing may lead to succession from grassland to woodland and the loss of grassland habitat (Smith *et al.*2000). Therefore it is important to explore the impacts of different grazing intensities on plant heterogeneity and spatial patterns, informing effective management of grasslands and grazing systems.

1.2 Importance of grazing-induced heterogeneity in Grasslands National Park, SK, Canada

Grasslands National Park was established in 1988 near the Saskatchewan-Montana border to preserve a representative portion of the remaining native northern mixed grass prairie ecosystem, which is rich in biodiversity and habitat to many rare and endangered species. Until recently, grazing exclusion from the park was a standard practice. Complete exclusion of grazing animals homogenizes the ecosystem; therefore some level of grazing disturbance is necessary to maintain ecological integrity in the grassland ecosystem (McCanny *et al.* 1996; Sutter 1997; Vermeire *et al.* 2004), habitat diversification, and to increase the number of species that can be supported (Saab *et al.* 1995). Therefore, for managing species-at-risk, as well as conservation of biological diversity throughout the Grasslands National Park area, Parks Canada initiated a biodiversity and grazing experiment with variable grazing intensities in 2006.

1.3 Why is Research Required?

When grazing is maintained properly, it can be an excellent management tool for maintaining primary production, biodiversity and habitat structure (Hobbs 1996; Collins *et al.* 1998). In contrast, improper grazing, such as overgrazing, can not only negatively affect productivity, but also cause severe loss of soil fertility (Lauenroth *et al.* 1999; Fuhlendorf and Engle 2004). For example, large numbers of livestock can reduce plant biomass and cover due to surface soil

compaction through trampling. All this can decrease the soil's water infiltration capacity, resulting in increased runoff and soil erosion, along with carbon losses from the soil (Sala and Paruelo 1997; Goudie 2000). A compact soil leads to lower moisture holding capacity and restricts plant root growth due to less total pore space and in particular, a reduced proportion of macropores (Kristoffersen and Riley 2005). Rhoades *et al.* (2003) and Kristoffersen and Riley (2005) examined the effects of soil compaction on plant growth and found that soil compaction affects plant growth, mainly through restricting root expansion and extension to depths of soil that could sustain plants during common short-term droughts. Management decisions regarding periods (length) of grazing and rest influence the soil water content, and soil water content can vary substantially as a result of animal impact and the duration of grazing, despite similar vegetation cover and soil type (Weber and Gokhale 2011). This is because grazers can change the soil structure through trampling, altering soil porosity and organic matter of the soils. Therefore, one needs to have an understanding of what level of grazing intensity is good for the sustainable management of an ecosystem and the biodiversity within.

In recent years, the study of soil water dynamics in grasslands has also become more important due to growing evidence that increased variability in amount and duration of precipitation during the growing season and soil textural differences affect both soil moisture variability and grassland productivity (Yang *et al.* 1998; Knapp *et al.* 2002; Nippert *et al.* 2006; Heathman *et al.* 2009). However, relative impacts of these causative factors in combination with grazing intensity to simultaneously study the variation in soil water content and plant productivity at a range of spatio-temporal scales are not well-understood. Some studies have documented the effect of grazing on plant diversity/productivity (Marriott *et al.* 2009; Cheng *et al.* 2011) or

evaluated the effect of grazing on various physical properties of soil (Augustine and Frank 2001; Jacobs *et al.* 2004; Zhao *et al.* 2010; Weber and Gokhale 2011). However, more information on how grazing intensity over time in combination with weather and slope location affects the variability in ALB, soil moisture and total soil and plant system carbon in a mixed grassland ecosystem is needed to comprehensively understand the impact of short-term grazing (within one growing season), mid-term grazing (inter-annual; varying between three and twenty years) and grazing termination on the plant processes.

1.4 Importance of Scale

For application of results, in particular, towards management decisions, it is critical to choose an appropriate spatial scale of study that is able to capture the spatial dynamics of the grassland processes that are of interest (Gordon *et al.* 1997). In grassland ecosystems where processes and the effect of disturbances such as grazing on the grassland processes vary across spatial scales, multiple scale studies are particularly valuable compared to single scale studies (Glenn *et al.* 1992; Fuhlendorf and Smeins 1999). This is because a multiple scale study has a greater potential to capture scale-dependent changes in relationships among ecosystem variables (Adler *et al.* 2001, Vallentine 2001). For example, the process of grazing can vary across spatial scales depending on factors such as water availability, forage depletion, plant phenology and diet selection. Laca and Ortega (1996), Bailey *et al.* (1996) and Vallentine (2001) conclude that cattle make foraging decisions at six spatial scales: (a) Home range, which represents landscape scale and is generally defined as a collection of camps (see below) demarcated by fences. (b) Camp, which represents a pasture scale and is a spatial foraging level defined as a set of feeding sites that share a common point for drinking water, resting and

seeking cover. (c) Feeding site, which represents a particular area within a pasture, foraged for a few hours. (d) Grazing patch, which represents plot scale and is generally defined as a collection of feeding stations. (e) Feeding station, which represents a group of plants that are within immediate reach of cattle without moving their front feet. (f) Bite, which represents plants ingested by cattle using gripping and severance motions (Laca and Ortega 1996, Bailey *et al.* 1996). Wallace *et al.* (1995) found that bison foraged more randomly within the patches, however they were more selective among the feeding sites within the landscape.

1.5 Research Objectives

The primary objective of this dissertation is to assess the spatio-temporal heterogeneity of soil moisture and ALB in an experimental area of Grasslands National Park at a range of scales (plot, pasture and larger regional extents) and under different grazing pressures. Using a modelling approach the relative degrees to which grazing intensity and soil properties affect grassland productivity and carbon dynamics at longer time-periods are investigated. These objective are achieved by investigating the following research questions. Is the spatio-temporal heterogeneity in soil moisture and ALB affected by grazing pressure, controlling for known factors such as weather variability and soil texture? If yes, then does the heterogeneity vary with site location within pastures due to other factors such as time and treatment? Are grassland productivity, total belowground soil carbon (SOMTC in g m^{-2}), and total plant system carbon (TOTSYC in g m^{-2}) influenced by variability in grazing intensity in continuously grazed pastures and soil texture? What happens to TOTSYC and SOMTC when grazing is terminated after 7 years of grazing at variable intensity?

It is important to understand these issues from a management perspective as grasslands play an important role in storing carbon both above and below ground and are mainly influenced by precipitation and herbivory, in addition to other factors (Frank and Groffman 1998; Flanagan *et al.* 2002; Knapp *et al.* 2002; Jones and Donnelly 2004). Additionally, it is more difficult to monitor parts of the carbon cycle belowground compared to the ALB using field experimentation alone. Hence, a modelling approach is suitable because it allows one to acquire knowledge of a given landscape's ecological issues under changing climate and land-use management as models can be used without any disturbance to the study area and can be used repeatedly.

Field experimentation (Cambardella *et al.* 1994; Shiyomi *et al.* 1998; Vieira and Gonzalez 2003; James *et al.* 2003; Zhao *et al.* 2010), remote sensing techniques (Reed *et al.* 1994; He *et al.* 2006; Tan 2007; Shen *et al.* 2008) and modelling (Riedo *et al.* 1998) have been used in grassland ecosystems to gain knowledge about the variability in plant productivity and soil properties such as soil moisture, soil texture and soil pH and the factors contributing to this variability (Cambardella *et al.* 1994; Jacobs *et al.* 2004). Literature shows that any or a combination of these can be used depending on the research question being considered. The novelty of this research is that it is using an experimental approach combined with remote sensing (Landsat) and modelling (CENTURY model) to examine and predict the impacts of grazing intensity on the spatial heterogeneity and patterns of ALB in mixed grassland ecosystem undergoing large extent (pasture size 300 ha) grazing experiment. Moreover, most grazing studies have been conducted within pastures smaller (< 100-ha) than most commercial pastures in southern Alberta and Saskatchewan, which have average sizes over 400 ha (Koper *et al.* 2008). Since

cattle grazing patterns (example, forage selection) differ among pasture sizes it is important to evaluate the ecosystem responses to grazing intensities at spatial scales relevant for range managers. It is hoped that the results of this research will help in the development of effective grazing system designs for management and conservation of grasslands. Another novel aspect of this research is that it examines the sensitivity of the model (CENTURY) predictions to changes in soil texture and grazing intensity. The knowledge of how much uncertainty is there within an important parameter (example, soil texture) will lead to better decisions in the long run than ones based on ignorance of uncertainty. This dissertation is divided into three parts: field based analysis (plot scale), satellite based analysis (pasture and regional scale), and modelling.

1.6 Dissertation Structure and Organization

This dissertation is organized into seven chapters. Chapter one is an introduction to the context of the research problem followed by the importance of heterogeneity in vegetation and scale of the study in grasslands, research rationale and justification based on the literature and research objectives. Chapter two provides a historical perspective of the grasslands in the North American Great Plains region. Dominant factors affecting the formation and maintenance of grasslands in North America are explained. Chapter three provides details on the overall research methods, including a description of the study area, measurement techniques and data analyses. Chapter four presents the results, discussion and conclusions for the field study examining the spatial heterogeneity of ALB and soil moisture in grazed and ungrazed pastures. Chapter five presents the results, discussion and conclusion for the study examining spatial heterogeneity in satellite-derived ALB at pasture and regional scales as a result of different

grazing intensities over time. Results, discussion and conclusions for experiments using the CENTURY model to look at the effects of variation in grazing intensity and soil texture on the grassland productivity and dynamics of carbon within a long term grazing pasture are presented in Chapter six. Effect of grazing termination after short-term grazing with variable intensity on the plant and soil carbon dynamics is also presented in Chapter six. Chapter seven discusses overall conclusions of this dissertation.

2.0 Historical Perspective on the Grasslands in North American Great Plains Region

Scientific definitions for grasslands vary; some studies classify grasslands by vegetation while others characterize them by climate, soils or human use of the ecosystem (White *et al.* 2000). Grasslands are complex ecosystems with climates intermediate between deserts and forests. In general, they can be defined as terrestrial ecosystems that are dominated by herbaceous and shrub vegetation and maintained by climate or ecological processes such as fire and grazing (Coupland 1991; Lauenroth *et al.* 1999; White *et al.* 2000). This broad definition of grasslands encompasses not only non-woody grasslands, but also savannas (open grasslands with dispersed trees), woodlands and shrublands.

Widely distributed on all the continents except Antarctica, grasslands account for 16% to 24% of the Earth's vegetation and cover more than 4.6 billion hectares of land (Whittaker and Likens 1975; Sims and Risser 2000). The variation in grassland estimation is mainly because of the difference between the potential grassland area (climatically determined grasslands) (24%), in the absence of human alterations, and the current distribution of grasslands (16%) that includes impact of human activities (Lauenroth *et al.* 1999).

2.1 North-American Great Plains Region

The Great Plains region in North America encompasses areas of grasslands stretching from southeastern Alberta, central Saskatchewan, and southwestern Manitoba to the highlands of central New-Mexico and from eastern Indiana to California (Sims and Risser 2000) (Figure 2.1).

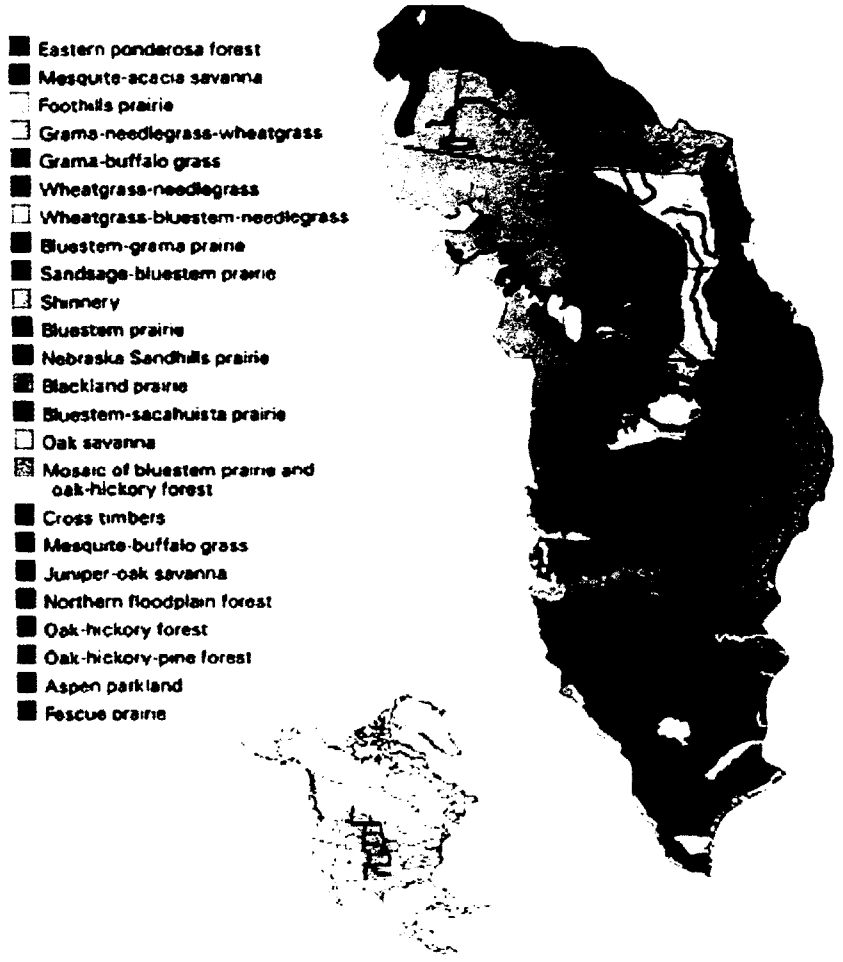


Figure 2.1 Major Vegetation types within Great Plains region (Sieg *et al.* 1999)

Factors such as regional temperatures, rainfall, soil conditions, fire, grazing, land-use and land management practices primarily determine the distribution and composition of North American grasslands (Sala *et al.* 1988). Although North American grasslands have been subdivided in various ways, most of the classifications distinguish at least six types: tall-grass prairie, mixed-grass prairie, short-grass prairie, desert grassland, California grasslands and Palouse prairie. The east-west precipitation gradient of central North America, along with the north-south temperature gradient give rise to diversity in soils (Sims and Risser 2000), which subsequently supports different plant communities.

North American grasslands are rich in biodiversity and are home to threatened species, such as sage grouse (*Oreoscoptes montanus*), henslow's sparrow (*Ammodramus henslowii*), mountain plover (*Charadrius montanus*), and lark bunting (*Calamospiza melanocorys*) (Ricketts *et al.* 1999; White *et al.* 2000; PCAP 2003). However, urbanization, change in fire regimes and grazing patterns (such as livestock ranching), conversion to crop fields, and invasion of woody and exotic species have all been implicated in the loss, fragmentation, and degradation of native grasslands, and loss of biodiversity (Davis 2004; Brennan and Kuvlesky 2005). Due to the fertile soils of the prairies, European settlers in the Great Plains region saw a great potential for agriculture, which resulted in conversion of prairies to farmlands (Riebsame 1990). Besides acting as a food source, the native grasslands also offered means of providing shelter for settlers, such as sod houses (Gauthier and Wiken 2003).

Policies such as the Homestead Act of 1862 (U.S.), the Dominion Lands Act passed in 1872 (Canada) and the Crowsnest Pass Act passed in 1897 (Canada) were mainly formed to support cultivation, which was also seen as a means to attract people to the region (Merchant 2005). The Crowsnest Pass Act allowed for subsidized freight rates to farmers for the transportation of grain, thus initiating settlement and development of agriculture in the region (PCAP 2003). The government also made certain provisions in the federal and provincial legislation that ensured that the settlers used the land only for cultivation purposes. If a settler used his land for any other purpose, such as pasture, his homestead rights were revoked or land taxes increased, as a result of violation of the provision prohibiting land to go wild (Merchant 2005). From 1979 to 1981, approximately 21,000 km^2 of grasslands were converted to cultivated land in the central and northern Plains, with grain production receiving increased government support until the

1990s (Riebsame 1990). Construction of the Canadian Pacific Railway (CPR) also played a leading role in defining the pattern of development in the prairies. Towns emerged along the railway line as collection points for grain and livestock exports and as distribution points for incoming supplies. All this led to further development of settlements and conversion of native grasslands into croplands.

In short, the implementation of concepts of private ownership of land, big consolidated farms, resource development (agriculture), economic development, and government policies resulted in substantial changes to grassland areas including habitat destruction for different species. All this might indicate development in terms of agriculture or economic production, since grasslands are known for their significance in terms of world grain production (Burke *et al.* 1989), nonetheless one also needs to consider sustainability for the protection of biodiversity (Hobbs 1996; Vermeire *et al.* 2004).

Biodiversity is defined as the "*...variety of life on earth at all levels, from genes to worldwide populations of the same species; from communities of species sharing the same small area of habitat to worldwide ecosystems*" (Secretariat of the Convention on Biological Diversity, Netherlands Commission for Environmental Assessment 2006). Some view biodiversity simply as a means for providing a product or a service (Callicott 1995), while others view nature as innately valuable and place the value of human beings as equal to that of all other species on the planet (Callicott 1995; Van de Veer and Pierce 1998). Despite the contradiction in the point of view (intrinsic or anthropocentric), maintaining biodiversity is crucial both in terms of aesthetic and cultural value, as well as for a variety of reasons including economic, recreational

and medicinal (Rolston 1994; Sieg *et al.* 1999). In grasslands, biodiversity allows the ecosystem to perform a variety of ecological services beyond the production of food and feed, including the recycling of nutrients, filtering non-point source pollution generated from activities such as farming, grazing and development, and sequestering atmospheric carbon (Shogren and Crocker 1995). Studies also show that species-rich, diverse grasslands allow for the production of high quality, animal products such as beef and milk (Smit *et al.* 2008; Fraser *et al.* 2009).

Grazing can promote the biodiversity of grassland ecosystems by changing the vegetation structure and height (Knapp *et al.* 1999; Truett *et al.* 2001). This is critical for numerous grassland birds and animal species that prefer specific sward structure for nesting, feeding and protection against predators. Previous studies have also shown that diverse mixtures of prairie plants produce more biomass and sequester more carbon compared to monocultures (Tilman *et al.* 2006). Therefore, for proper management of grassland ecosystems, it is essential to have an understanding of the factors affecting the biodiversity within the grasslands.

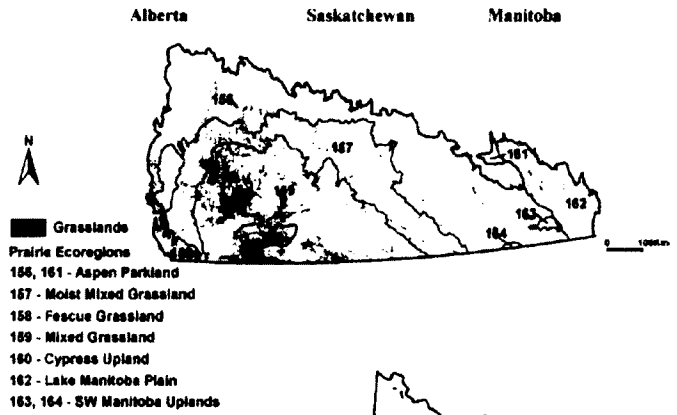
Despite covering lesser area compared to Eurasian grasslands, the Great Plains region in North America is unique in terms of political, topographical and climatic complexity (Lauenroth *et al.* 1999). For example, the Great Plains region crosses only one international boundary, between U.S. and Canada, compared to the Eurasian grassland ecoregion (stretching 9,000 km from Slovakia and Hungary on the west to China and Mongolia on the east) which crosses several international boundaries. The latter situation adds tremendous complexity to management, compounding the impact of human activities. Due to similarity in language and research applications, as well as availability of data from both U.S. and Canada, the political complexity

caused by human activities in terms of management of grasslands is quite low compared to Eurasian grasslands. Furthermore, the central North American grassland region has relatively little topographic variation compared to Eurasian grasslands, resulting in smoother gradients in climatic driving variables across the whole region (Lauenroth *et al.* 1999). As a result of these unique characteristics, the North American grasslands region is a major source of much of the world's knowledge about grasslands.

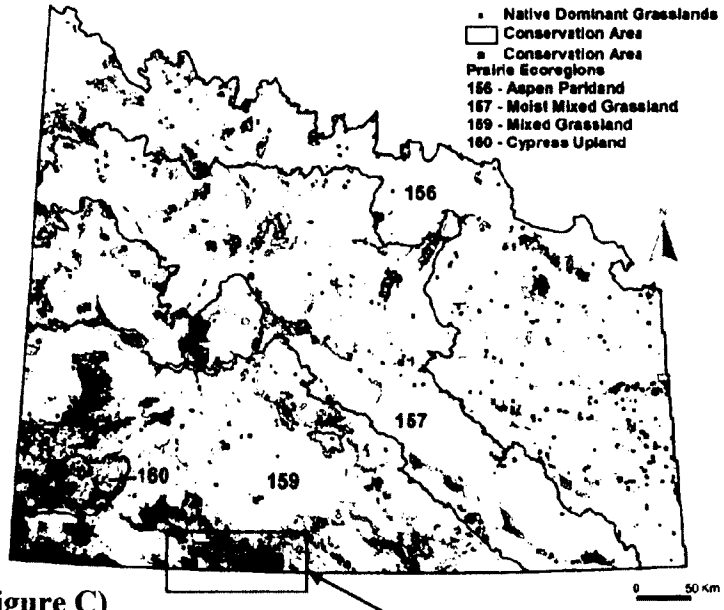
2.1.1 Grasslands in Saskatchewan

Alberta, Saskatchewan and Manitoba are known as the Prairie Provinces in Canada and cover 16% of the North American Great Plains region (Gauthier and Wiken 2003). Figure 2.2 shows the distribution of different eco-regions within the prairie region of Canada. Out of the three provinces, Saskatchewan includes the largest percentage of prairies in Canada, followed by Alberta.

(Figure A)



(Figure B)



(Figure C)

Grasslands National Park

Figure 2.2: (A): Ecozones of South-Central Canada illustrating the spatial significance of the Prairies (Source: Vaisey and Strankman 1999). (B): Distribution of grasslands in Prairie ecozone of Canada (Source: Gauthier and Wiken 2003). (C): Grasslands in Saskatchewan, showing the location of Grasslands National Park (Source: Gauthier and Wiken 2003).

Grasslands constitute about 24.4% of Saskatchewan's land area (Gauthier and Wiken 2003). Though they have long been manipulated, converted to other uses such as agriculture or degraded following European settlement during the 1800s, grasslands still support a large number of Saskatchewan's threatened and endangered animals (such as the black-tailed prairie dog (*Cynomys ludovicianus*), swift fox (*Vulpes velox*), prairie rattlesnake (*Crotalus viridis*) and eastern yellow-bellied racer (*Coluber constrictor flaviventris*)), birds (such as the Ferruginous hawk (*Buteo regalis*) and Piping Plover (*Charadrius melodus circumcinctus*)) and plants (such as Buffalograss (*Buchloe dactyloides*) and Hairy Prairie-clover (*Dalea villosa var. Villosa*)). Today, most of the remaining contiguous native grasslands found in mixed and moist mixed grassland eco-regions are under conservation or are protected areas because of their biodiversity and importance as habitat for rare and endangered species (Gauthier and Wiken 2003).

Grasslands National Park and the Cypress Hills in southwest Saskatchewan are conservation areas for native mixed grasslands. Agriculture is the dominant land use in the mixed grassland eco-region, with cereal cultivation being the main agricultural land use, followed by rangeland grazing (Padbury *et al.* 2002). In general, all of the public and private native grasslands are grazed by domestic livestock. Most of the federal and provincial community pastures in Saskatchewan practice conventional grazing which refers to grazing through a full growing season (June to October) of moderate intensity (50% utilization) (Adams *et al.* 2004). Spatio-temporal variations in land-use (agriculture, mining, forestry products and ranching) have been mainly driven by factors such as economic gains, governmental policies and weather cycles (Gauthier and Wiken 2003).

2.1.2 Grasslands National Park, Saskatchewan

Grasslands National Park was established in 1988 near the Saskatchewan-Montana border to preserve a representative portion of the remaining native northern mixed grass prairie ecosystem. Land acquisition for the park started in 1984, before the park establishment, and is still underway. The park is comprised of two areas, referred to as the East Block and the West Block, of relatively undisturbed mixed grass prairie.

The park currently occupies an approximate area of 906.5 km^2 near the northern edge of the Great Plains region of North America. The West Block is based in the Frenchman River Valley, while the East Block features the Killdeer badlands of the Rock Creek area and is also representative of the Wood Mountain uplands (Parks Canada 2002). The climate here is dry sub-humid to semi-arid and has long cold winters and short hot and dry summers (Davidson 2002).

During the summer, average temperatures range between 20 and low 30s ^{o}C. The mean annual precipitation is approximately 350 mm, with the potential annual evapo-transpiration being approximately 347 mm (Kottek *et al.* 2006). Approximately one-third of this total annual precipitation falls as snow during winter, whereas the rest of it falls as rain, mostly during the summer. Winds are strong and frequent, particularly in spring (Coupland 1991). The climatic conditions produce an environment that supports a unique flora and fauna, including rare plant species such as dwarf fleabane (*Conyza ramosissima)*, Bessey's locoweed (*Oxytropis besseyi*) squirrel tail grass (*Hordeum jubatum*) and Canada's only black-tailed prairie dogs (*Cynomys ludovicianus*). Sage, clubmoss (*Selaginella densa*), lichens and cacti (*Cactaceae*) also form a

significant part of the plant community in the drier locations. The park also supports pronghorn antelope, mule deer, elk, coyotes and numerous small mammals such as white-tailed jackrabbit (*Lepus townsendii*) and the Richardson's ground squirrel (*Urocitellus richardsonii*). It is also home to various species of birds (sage grouse (*Centrocercus urophasianus urophasianus*), prairie falcon (*Falco mexicanus*) and Sprague's pipits (*Anthus spragueii*)), reptiles and amphibians such as short-horned lizard (*Phrynosoma hernandesi*) and prairie rattlesnakes (*Crotalus viridis*) (Parks Canada 2002).

Until recently, grazing by large herbivores has been excluded from the Grasslands National Park, since Parks Canada started acquisition of land in 1984. Historically, bison were principal grazers in the North American grasslands, providing sustenance to First nations and Métis people, and staple food for early explorers, fur traders and European settlers (Boyd 2003). However, by the late 1800's most of the bison were decimated to near extinction (Isenberg 2000). Since grassland ecosystems were regulated by disturbances such as frequent and extensive fires and intensive grazing by bison, to maintain high species diversity in the remaining grassland areas, disturbances must now be provided through active management (Bock *et al.* 1993; McCanny *et al.* 1996; Sutter 1997; Vermeire *et al.* 2004).

In 2001, Parks Canada included reintroduction of bison in its management plan as a key action for emulating a pre-settlement grazing regime within the park, as well as for restoring the park's ecological integrity. In December 2005, Parks Canada re-introduced Plains bison (*Bison bison*) in the West Block of the Grasslands National Park (Parks Canada 2009, Figure 2.3).

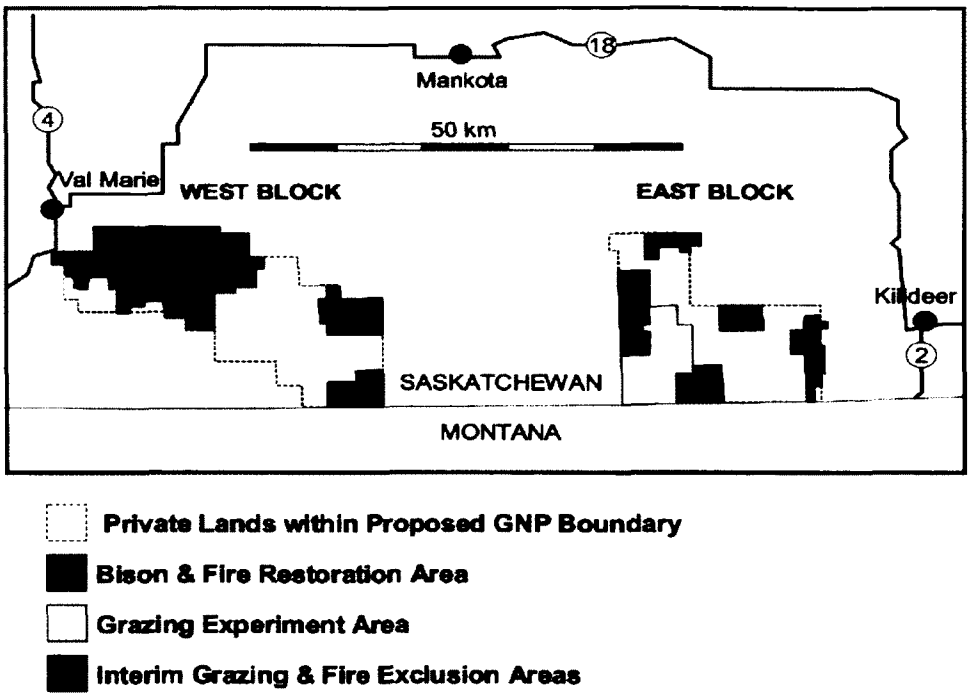


Figure 2.3 Grasslands National Park (GNP), Saskatchewan with both West and East Blocks (Source: Henderson (2006))

Initially Plains bison were released in a 16.2 ha holding facility to ensure their adaptability to new surroundings. Later, in 2006 they were released into the largest parcel (181 km^2) of the West block due to the block's size, availability of a natural water source and easy accessibility for park visitors (shown in green, Figure 2.3).

Grazing affects important habitat components such as grass height and the amount of litter (Hobbs 1996 and Truett 2003). Therefore, to maintain the full range of habitats required by the native vertebrates and invertebrates, a range of plant heights and litter is required. To determine impacts of before and after (BACI design) grazing on heterogeneity in the multi-scale structure and function of mixed-grass prairie communities, Parks Canada initiated a biodiversity and grazing experiment (BGE) (shown in yellow, Figure 2.3) in 2006 in the East Block of GNP. The experimental area occupies 26.5 km^2 in the East Block with a total of 9 pastures, and four

additional pastures in the adjacent Mankota Community Pasture. The experimental area except Mankota community pasture was not grazed from 1992 to May 2008. More details on this experiment are explained in section 3.1.

2.2 Dominant factors affecting the formation and maintenance of grasslands in North America

As explained in Chapter 1, heterogeneity is complexity and/or variability of a system property, both in space and time due to natural (climate) and/or adverse management factors (grazing and land-use change) (Dutilleul and Legendre 1993; Li and Reynolds 1995). Sections 2.2.1 – 2.2.5 outline the dominant factors affecting spatial heterogeneity in grassland ecosystems.

2.2 1 Climate

The climate of North America is extremely diverse and is affected by two geographical features: the western Cordillera, which is a series of north-south mountain ranges, and the Interior plains to the east. The former constitutes a major obstacle to the westerly and trade winds, while the latter provides an uninterrupted path for the flow of arctic and tropical air masses. The air mass approaching from the west over the Pacific Ocean is saturated with moisture when it reaches the North American continent. While moving inland, this air mass is obstructed by coastal mountain ranges followed by other successive tiers of mountains ending with the Rocky Mountains on the eastern extreme of the cordillera. During the journey eastward, some of the air masses are forced to move up and over the mountains resulting in precipitation on the windward side. Thus the west-facing windward slopes support the majority of the forests of the mountain ranges of western North America (Joern and Keeler 1995; Lauenroth *et al.* 1999). Following precipitation, this cool air descends down on the leeward side of the mountain and

reabsorbs or picks up surface moisture. This result in dry conditions on the leeward side of mountains in an area called the rain shadow. This is where the grasslands are found.

The second geographical feature is the broad, flat Interior plain that extends from the mountainous west across the central part of the continent. As this plain allows for unobstructed movement of the arctic and tropical air masses into the central region, variability in summer weather conditions is created with most of the rainfall occurring in convective thunderstorms (Joern and Keeler 1995). For example, proportion of mean annual precipitation that is received during summer time ranges between 30% and > 50% (Lauenroth *et al.* 1999).

Mean annual rainfall across the Great Plains grasslands ranges from 250 mm to 1000 mm with two-thirds of that occurring during the summer growing season, generally from April through September (Joern and Keeler 1995). Precipitation is the main limiting factor in terms of primary production in both arid and semi-arid zones (Noy-Meir 1973; Yang *et al.* 1998; Lauenroth *et al.* 1999). As a result succulents, shrubs and grasses can co-exist in semi-arid regions as these groups of species with different rooting depths use water stored in different soil layers. Shallow rooted succulents utilize the water from the uppermost soil layers, whereas grasses, generally with longer and finer roots, utilize water from deeper soil layers (Golluscio 1998).

Temperature is another important factor that explains the distribution of various types of grasslands. It plays an important role in determining the types of photosynthesis. The general temperature gradient dominating North America increases from northwest to southeast and generally affects the geographic distribution of plants (Joern and Keeler 1995). Cool-season grasses using the C_3 photosynthetic pathway are more efficient at photosynthesizing in cool

temperatures, which explains their dominance in northern regions. Conversely, warm-season (C_4 photosynthesis) species are more dominant in warmer, southern regions because of their photosynthetic efficiency in warmer climates (Aber and Melillo 1991).

To summarize, two great climatic gradients characterize the plains: a south-to-north decrease in mean temperatures and an east-to-west decrease in precipitation. These are fundamental in determining the different types of grasslands. Therefore, any change in climate may impact the structure and distribution of different grass species, nutrient cycling and plant productivity.

2.2.2 Water and Nutrient Availability

Water drives the composition and productivity of grasslands particularly in arid and semi-arid regions. It is required in much greater quantities than nutrients because nutrients allocated to the production of a given tissue tend to remain in that tissue until it is shed as litter (Aber and Melillo 1991). In contrast, water is continuously lost to the atmosphere through the process of transpiration. In most terrestrial ecosystems, the only major storage for water is in the soil where it is generally stored in the macro- and micro-pores of the soil structure. The macro-pores in general control a soil's permeability and aeration where water movement is generally accomplished due to gravitational forces. On the other hand, micro-pores are responsible for a soil's water holding capacity, where water movement is by capillary action to plant roots. Precipitation and soil water holding capacity are the two abiotic factors that determine the total amount and seasonal pattern of water availability for plants to carry out different functions. Soil moisture accumulation in the plant root zone influences the plant root growth, as well as ALB. This also determines how much water is available for transpiration during dry

periods, which in turn impacts the water stress in plants (Kleidon and Heimann 1998). For example, when water availability in the soil is lower than the evaporative demand in the atmosphere, plants undergo water stress (Schulze *et al.* 1972). To deal with this, plants generally close their stomata to minimize further water loss through transpiration. As a result intake of carbon dioxide is cut off too, thus, slowing down the process of photosynthesis and subsequently affecting plant growth.

Water deficits in the soil can also limit nutrient availability for plants as fewer nutrients are carried to the plants through the root system (Nye and Tinker 1977). Generally, plants require approximately fifteen elements for proper function and growth. These nutrients move cyclically, with uptake of water through the roots and transport to different tissues for growth. After plants die, the nutrient rich litter is acted upon by decomposers, which release nutrients back into the soil in an inorganic form through the process of mineralization, which allows plants to absorb them again. Lack of any one of these nutrients can limit the ability of a plant to carry out certain functions, affecting plant growth, survival and reproduction. For example, a plant's acclimation to high irradiance and photoinhibition is particularly influenced by availability of nitrogen. This is because lack of nitrogen in a plant can slow the turnover of proteins and thus the repair of damaged plant cells (Henley *et al.* 1991).

A common plant response to nutrient limitations is to increase the allocation of nutrients to roots and decrease the allocation of nutrients to leaves and stems (Porter and Nagel 2000). Root expansion enables the plant to find nutrients and water deeper in the soil column and capture nutrients necessary for continued leaf expansion. Over time, the amount of leaf area

per unit root mass is reduced compared to a plant of the same total size given free access to nutrients. Different plants have different adaptations to deal with the low-nutrient availability and water stress, such as leaf longevity, senescence, greater root mass and restriction of growth and reproductive activities (Aber and Melillo 1991).

In recent years, study of soil water dynamics in grasslands has become more important because there is growing evidence suggesting increased annual variability of precipitation and a higher frequency of climatic extremes as a result of global climate change (Lauenroth *et al.* 1999, Scott and Suffling 2000, Knapp *et al.* 2002). For example, Scott and Suffling (2000) suggested that prairie region parks are likely to be more susceptible to ecological shifts. In GNP, the climate change scenarios project increased temperatures year-round with less precipitation in summer and fall (Scott and Suffling 2000). This will also impact soil moisture and plant available water for growth, thereby affecting plant productivity and land-atmosphere interactions, which are of great importance in understanding the climate variability. Cunningham *et al.* (1979) found that enhanced soil moisture during the period of active growth (growing season) increased the total above-ground production. This is because more resources were allocated to vegetative growth as a result of favourable moisture conditions. On the other hand, enhanced soil moisture during periods of little or no plant growth (late fall and early winter) did not show any effect on the plant total above-ground production. This might be due to plants' allocation of resources towards accumulation of photosynthates to be used in the production of new vegetative and reproductive structure in the subsequent growing season.

2.2.3 Fire

Grassland fire prevents bush encroachment, removes dead herbaceous material and recycles nutrients. Without fire, organic matter and litter would accumulate which would increase soil moisture and lower soil temperature improving growing conditions suitable for increased shrub or tree growth. The timing, frequency and intensity of fires determines the effects of these events on the functioning of grassland ecosystems (White *et al.* 2000) with tall-grass prairies generally requiring fires at an interval of two to four years to remain vigorous (Aber and Melillo 1991).

Periodic fires also affect plant species composition, productivity and ecosystem nutrient cycling (Blair 1997). The cycling of nitrogen is affected greatly by fire frequency, since volatilization of nitrogen during the combustion of aboveground biomass and detritus is the major pathway of nitrogen loss in tall-grass prairie ecosystems (Ojima *et al.* 1990). Therefore, the degree to which plant productivity is limited by nitrogen availability in these grasslands varies substantially with fire frequency.

In grasslands, above-ground productivity is generally increased in the post-fire environment, especially in the wet years, due to the alteration of resource availability. Perennial grasses and forbs of prairie systems characteristically maintain large roots and rhizome systems from which leaves and stems reproduce following either fire or grazing (Aber and Melillo 1991). It has been found that many species such as purple needlegrass (*Nassella pulchra*) and bottlebrush squirreltail (*Elymus elymoides*) show very large increases in flowering and seed production following fires (Glen-Lewin *et al.* 1990). Frequent burning of grasslands in the spring season

increases the dominance of C_4 grasses (warm season grasses) and reduces the abundance of C_3 grasses (cool season species) (Steinauer and Collins 1996). This is because spring burning helps in removing canopies of dominant competitors and reduces accumulated litter for the growth of C_4 species. Conversely, summer fires reduce the abundance of warm season grasses by destroying the shoots during a time of normally vigorous growth and increase the abundance of cool season species.

Fire also affects soil structure and nutrient availability. For example, nutrients bound in litter are released during fire and are deposited as ash on the soil surface. When rainfall follows a burn, these mobile nutrients move down through the soil profile and tend to displace hydrogen ions from exchange sites in the soil, thus increasing pH (Aber and Melillo 1991). Following fire events, soil temperatures increase due to more solar energy being absorbed by the ash covered surface layer. Due to lower foliar biomass, as a result of the fire, evapotranspiration may decrease, resulting in higher water content in soils. When combined with high soil pH these factors tend to increase the microbial activity, resulting in more nutrient availability by increasing the rates of mineralization of the remaining soil organic matter. All these factors tend to increase plant production.

In terms of grazing, the grasses in recently burned areas are more palatable and nutritionally valuable to grazers because the burned areas have more nutrients that are released from the litter after the burn. A study conducted by Vermeire *et al.* (2004) in north-western Oklahoma on the Hal and Fern Cooper Wildlife Management Area found that the cattle were strongly attracted to burn sites, due to high quality of nutrients in the forage.

Though fire is an important factor that helps maintain grasslands, it also has detrimental effects. It can destroy habitats for animals, insects and birds, leading to loss of biodiversity. It also leads to the release of greenhouse gases, like carbon dioxide, into the atmosphere.

2.2.4 Introduction of exotic or non-native plant species

Introduction of exotic or non-native plant species in native grasslands can also lead to changes in plant community spatial heterogeneity. Between 1930 and 1970, about 800,000 ha of mixed grass prairie region in Canada was planted with *Agropyron cristatum* (crested wheatgrass), an introduced plant species (Wilson and Gerry 1995). Soon it became the most successful and widely used forage grass in Western Canada due to its drought and cold-resistant qualities, high productivity for forage and pasture, excellent palatability and nutritive qualities during spring and early summer (Rogler and Lorenz 1983). Despite its economic advantages, the introduction of the species meant replacement of the complex natural prairie ecosystems with a homogeneous system. Since many animal and bird species prefer certain types of plants for forage, nesting and breeding, any change in the native plant communities will impact their habitat. This presents challenges to species conservation. Introduced species also pose competition for native species. Native species must compete for a variety of resources including nutrients, light, and plant available water for growth.

2.2.5 Grazing

Since grasslands developed under the influence of grazing and fire, the proposition seems reasonable that both grazing and fire is required to maintain it (Walter *et al.* 2002). Due to human expansion into grassland areas, natural grazing has undergone change over time. The

introduction of wire fencing and replacement of wild herbivores such as bison and domesticated cattle has resulted in more controlled grazing (Pieper 2005).

Although cattle and bison share many characteristics, there are some important differences in their grazing patterns and behaviour (Lauenroth *et al.* 1994). In general, both cattle and bison prefer to graze in areas with proximity to water (< 1 – 2 km) and lower slopes (less than 10 – 20% slope gradients) (Pinchak *et al.* 1991; Fortin *et al.* 2003). This is because in such conditions grazers spend less time climbing hills and travelling between foraging patches and water sources. As a result, sites nearer to water and on gentler slopes are grazed more by cattle compared to sites with steep slopes and greater distance to water sources (Bailey *et al.* 1996). Bison diets consist of up to 90% graminoids, while cattle diets consist of 70% graminoids (Steinauer and Collins 1996). This selective grazing of graminoids by bison and cattle releases forbs from competitive pressure and increases plant diversity.

Large herbivores select grazing sites that vary seasonally based on their dietary requirements, forage quality and distance from water, and this selective behaviour can alter the abundance of plant species because of removal of preferred plant species (Steuter *et al.* 1995; Vermeire *et al.* 2004). Furthermore, consumption of all or part of a plant also affects plants according to the part of the plant that is consumed (Gurevitch *et al.* 2002). For example, removal of or damage to roots can reduce or prevent the plant's uptake of water and mineral nutrients. This can increase the plant's vulnerability to strong winds (uprooting), flooding, or soil erosion. If the grazers consume only plant leaves, just the photosynthetic surface of a plant is compromised, which may be more easily regenerated than if the plant's root system is damaged. The stage at

which the plant is damaged by herbivores is also important. For example, grazing on grasses that have just begun flowering can critically affect their ability to produce seeds, whereas similar grazing after plants have shed seeds may have less of an impact on plant population dynamics.

Grazers also supply fertilizer from their dung and discourage invasion by woody species of plants because they eat the young woody shoots. As herbivores eat forage with high nutrients, their excretory products are high in readily available nutrients (Steinauer and Collins 1996; Knapp *et al.* 1999; Truett *et al.* 2001). For example, bison urine patches increase local forage production, alter species composition and are more likely to be grazed than surrounding off-patch areas (Jaramillo and Detling 1992). These herbivores also transport nutrients across landscapes by differential rates of forage intake and excrement among various habitats (Steinauer and Collins 1996). Augustine and Frank (2001) compared soil and community characteristics at Yellowstone National Park, between ungrazed grasslands and grasslands grazed by large herbivores, such as elk, bison and pronghorn antelope. The results showed that species richness and diversity were greater in the grazed grasslands at a scale of 20 x 20 cm.

Studies have shown that grazing both increases and decreases heterogeneity in vegetation, depending on the intensity of grazing and level of plant productivity (Cid *et al.* 1991; Bakker *et al.* 2006). For example, in mixed-grass prairie, grazing at moderate intensities generally appears to increase plant species diversity by reducing the competitive advantage of dominant species (Hartnett *et al.* 1997; Harrison *et al.* 2003). Grazing also creates vegetation structural patches in the landscape because grazers are able to selectively forage on preferred species

(Hartnett *et al.* 1997). Through this behavioural mechanism, grazing promotes both vertical and horizontal heterogeneity in vegetation structure by reducing vegetation height, increasing basal cover of grass and cover of some forbs, and decreasing woody species (such as shrubs) in grazed patches (Stohlgren *et al.* 1999). However, if grazing intensity is high, then grazing may also act to homogenize the vegetation structure (McIntyre *et al.* 2003).

Factors such as precipitation and grazing strongly influence the average height of grasses in Great Plains region (Truett 2003). For example, grass height is directly correlated with precipitation and thus decline from east to west following the gradient of declining moisture (Lauenroth *et al.* 1999). In contrast, grass height is inversely correlated with grazing intensity (Hobbs 1996), such that the more intense the grazing pressure, the shorter the grass species in grazed areas compared to ungrazed sites. For example, frequent heavy grazing by cattle or bison can convert tallgrass prairie to mixed grass or mixed grass to short grass (Hartnett *et al.* 1996 and Gillen *et al.* 2000). Since most of the mixed, short and tallgrass prairies have evolved under the influence of variable grazing intensity, most dominant grasses have natural low growth forms with average height differing by only a few inches between grazed and ungrazed areas (Milchunas *et al.* 1988; Walter *et al.* 2002).

Golluscio *et al.* (2005) evaluated the impacts of grazing on the spatial heterogeneity in the plant biomass in Patagonian steppe. The results of the study showed higher internal heterogeneity (variability at a distance shorter than the minimum distance sampled) in the grazed sites compared to ungrazed sites. Hartnett *et al.* (1996) and Knapp *et al.* (1999) in their study of tallgrass prairie in Kansas found that selective grazing of big bluestem (*Andropogon gerardii*),

Indiangrass (*Sorghastrum nutans*) and other tall grasses by bison increased mid-size grasses, such as sideoats grama (*Bouteloua curtipendula*) and western wheatgrass (*Agropyron smithii*). Furthermore, grazing also increases the species richness. This is because grazing and trampling disturbs both the soil and plant canopy, thus encouraging the invasion of early successional forbs (Hartnett *et al.* 1996; Knapp *et al.* 1999). Additionally, grazing indirectly increases the amount of bare ground because of the associated trampling of vegetation and deposition of animal waste by grazers (Hartnett *et al.* 1997).

Some studies also suggest that grazing disturbance is necessary to maintain the ecological integrity in the grassland ecosystem (McCanny *et al.* 1996; Sutter 1997; Vermeire *et al.* 2004). Parks Canada (2002) reported five times more active Richardson's ground squirrel holes in grazed lands compared with ungrazed parkland. Since these ground squirrels are prey for endangered species such as ferruginous hawks, their presence is particularly important in maintaining the hawk species, which is a species of special concern within the parkland. Additionally, their burrows are also important in developing habitat and providing food for endangered species such as swift fox and burrowing owls (*Speotyto cunicularia*); since burrowing owls cannot dig burrows, they use abandoned burrows of Richardson's ground squirrels for nesting, while swift foxes prey on ground squirrels for food. Similarly, Klute *et al.* (1997) found that avian diversity was higher in moderately grazed pastures than in ungrazed Conservation Reserve Program fields.

Grazing can also alter the soil microclimate (temperature and moisture). For example grazing activity increases the radiant energy reaching the soil, leading to higher soil temperatures.

Additionally, grazing activity also reduces the transpirational surface area of the vegetation which reduces the rate of soil moisture loss (McNaughton 1985; Seastedt *et al.* 1988). Since soil carbon turnover rates are a function of a soil's microclimate, physical, chemical (texture, pH, bulk density) and biological (microbial biomass, composition and diversity) properties (Epstein *et al.* 2002), any change in soil properties as a result of grazing can affect the carbon cycling in grassland ecosystems.

Trampling by grazers can cause soil compaction (i.e. change in soil pore-size distribution) thus leading to alteration in soil hydraulic and mechanical properties (Greenwood *et al.* 1997; Richard *et al.* 2001; Pietola *et al.* 2005; Krummelbein *et al.* 2009). For example, a compact soil leads to lower moisture holding capacity and restricts plant root growth due to less total pore space and in particular, a reduced proportion of macropores (Kristoffersen and Riley 2005). Rhoades *et al.* (2003) and Kristoffersen and Riley (2005) examined the effects of soil compaction on plant growth and found that soil compaction affects plant growth, mainly through restricting root expansion and extension to depths of soil that could sustain plants during common short-term droughts.

In short, grazing affects biodiversity and also leads to landscape heterogeneity by creating a mosaic of vegetation and soil microclimates through differential grazing patterns (light to heavy) and preference, urine deposition and trampling within grasslands (Knapp *et al.* 1999; Truett *et al.* 2001). Therefore, it is useful to have an understanding of how much grazing is good for the sustainable management of an ecosystem and the biodiversity within.

3.0 Methodology

3.1 Study Area

The research was carried out in the East Block of GNP of Canada (Latitude: 49° 10′ N, Longitude: 107° 25′ W, Elevation: 800 m) and the adjacent Mankota community pasture (government-owned and –managed land used for communal grazing by local ranchers) located in southern Saskatchewan.

The East Block of the GNP has been ungrazed since its acquisition (Figure 3.1). The study area mainly consists of open, rolling upland prairie interspersed with riparian lowland and creeks. The vegetation is mainly characterized as northern mixed-grass prairie. Based on the vegetation survey conducted by Parks Canada (2005) common grasses within the block includes needlegrasses (*Stipa* spp.), blue grama (*Bouteloua gracilis*), western wheatgrass (*Pascopyrum smithii*), northern wheatgrass (*Elymus lanceolatus)*, and bluegrasses (*Poa* spp.). Salt grass (*Distichlis sricta*), sedges and reeds are more common in the lowland areas. In addition to short to medium grass species, the block also has forbs and shrubs which are scattered across the landscape. Sagebrush (*Artemisia cana*) is the most common shrub in upland areas; whereas Western snowberry (*Symphoricarpos occidentalis*) and greasewood (*Sarcobatus vermiculatus*) are more commonly found in lowland areas. There has been minimal invasion of exotic species such as crested wheatgrass, *Agropyron cristatum*, alfalfa, *Medicago sativa*, and leafy spurge, *Euphorbia esula* in the East Block (Bleho 2009).

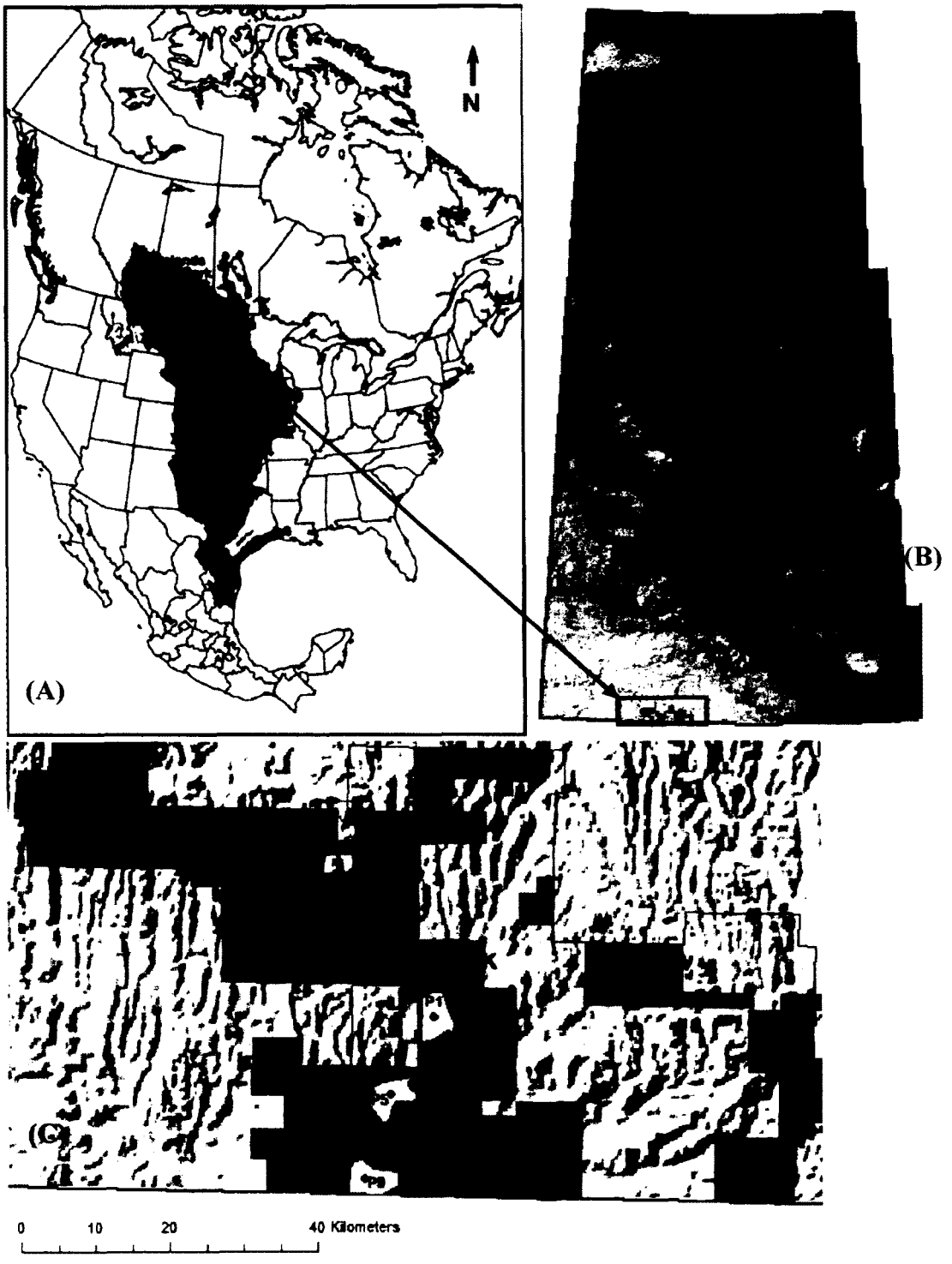
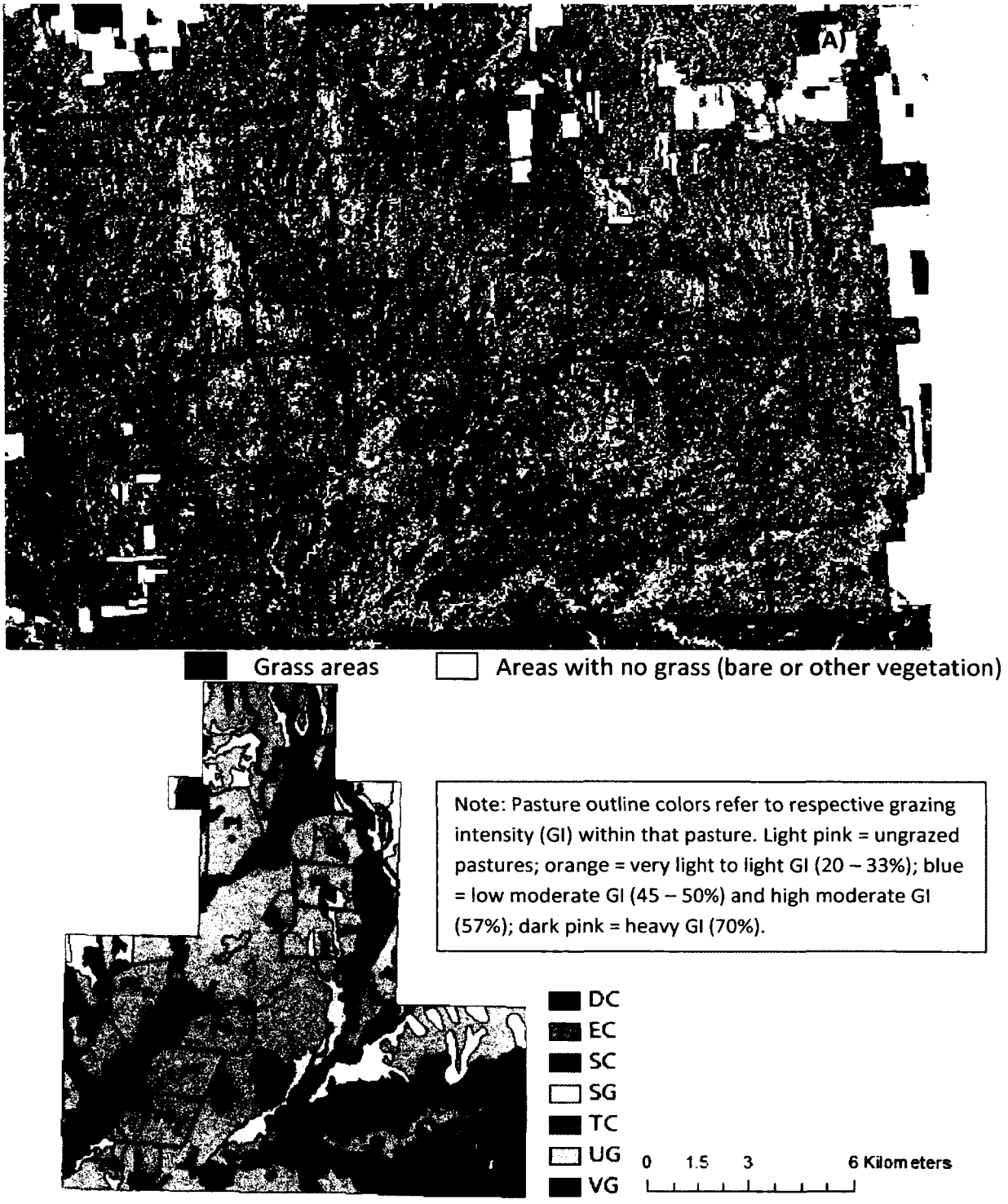


Figure 3.1: (A): Location of the Great Plains region and GNP in North America, (Parks Canada 2002). (B): Location of GNP and Mankota community pasture in Saskatchewan. (C): Location of research sites in the East Block, GNP (P1, P5, P9 = ungrazed pastures; P2 = 20% grazing intensity (GI), P6 = 33% GI; P7 = 45% GI; P3 = 57% GI, P4 and P8 = 70% GI) and Mankota community pasture (P10 to P13 with 50% grazing intensity). DEM source (Stafford 2002).

Cattle grazing with variable intensities were initiated in the East Block of GNP during June 2008 as a part of the biodiversity and grazing experiment (BGE) that started in 2006. In total, nine ~300 ha pastures (P1 to P9) with variable grazing intensities (0% ungrazed; 20% very light grazing intensity; 33% light grazing intensity; 45 – 50% low moderate grazing intensity; 57% high moderate grazing intensity and 70% heavy grazing intensity) were established in the experimental area (Figure 3.2). Grazing intensity in this study refers to the cumulative effect grazing animals have on the land during a particular time period, expressed as percent utilization. Here percent utilization is the percentage of the current year's primary production consumed or destroyed by livestock (Holechek *et al.* 2001). Yearling steers were introduced to six of the experimental pastures: 2, 3, 4, 6, 7 and 8. Pastures 1, 5 and 9 were ungrazed sites.

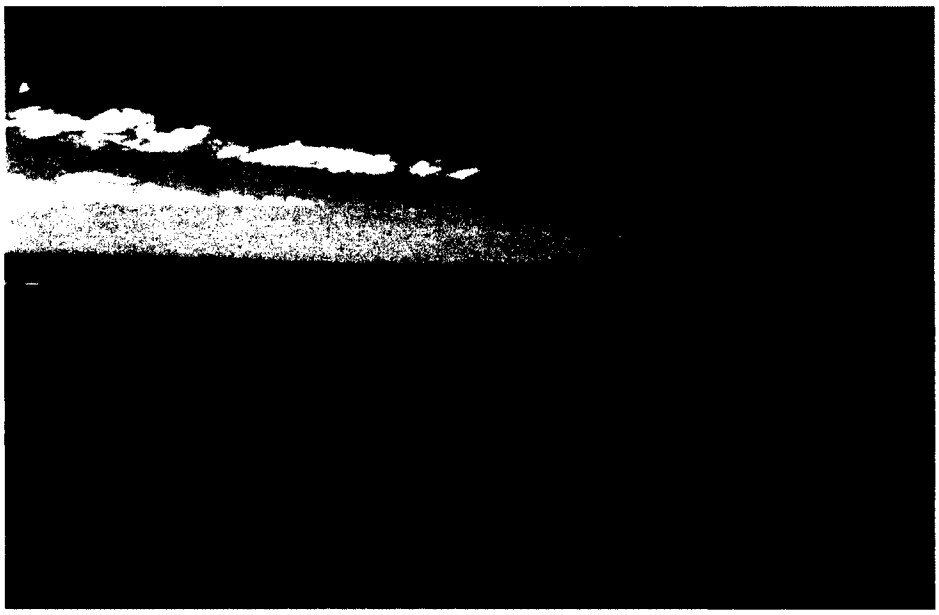
All the pastures were similar in shape and size, as well as proportion of lowland, riparian and upland habitats; location of water source and plant communities (Koper *et al.* 2008). All the East Block pastures contained several relatively large, permanent creeks, while most of the creeks in the Mankota grazed pastures were small and ephemeral. Additionally, to be consistent with the regional pasture management, all the experimental pastures also included an anthropogenic water source placed in the lowland areas. To restrict cattle movement between pastures, the experimental pastures were wire fenced (Figure 3.3B).



(DC = Disturbed Communities; EC = Eroded Communities; SC = Shrub communities; SG = Sloped Grasslands; TC = Treed Communities; UG = Upland Grasslands; and VG = Valley Grasslands)

Figure 3.2: Location of experimental pastures in the East Block, GNP and Mankota community pasture (Source: Fargey 2004) (A) with vegetation classification for the East Block, experimental pastures (B) (Source: Michalsky and Ellis 1994; Parks Canada 2005).

(A)



(B)

Figure 3.3 (A) Biodiversity and grazing experiment photographs for East Block, GNP. Cattle grazing in pasture 3 of East Block, GNP during June 2008. **(B)** Wire fence between pasture 8 (grazed) and 9 (ungrazed).

The Mankota community pasture is adjacent to the East Block of GNP. The community pasture is owned by the provincial government and is conventionally grazed annually following a season-long (June to October) grazing system. In a community pasture, patrons apply to bring their cattle in and are charged for the services such as grazing and breeding provided on the pastures. The annual stocking rates on the respective pasture is

established based on factors such as levels of subsoil moisture, water supplies, forage carry-over, range health and vigour (Mastad, M. 2010, personal communication).

As part of the BGE, controlled grazing treatments were initiated in the Mankota community pasture in 2008 (P10, 11, 12 and 13; Figure 3.2). These pastures were grazed annually from June to October at low moderate grazing intensity (50%). Additionally, these experimental pastures were fenced, resulting in controlled grazing as compared to pre-existing free range grazing until 2008.

3.2 Field Data, Experimental Design and Methods

Due to the varying landscape from gently rolling hills to badlands in the park, three broad vegetation landscape units dominate the experimental area: riparian shrublands, upland grasslands and valley grasslands. The field work was conducted mainly within the upland grasslands dominated by speargrass (*Stipa comata*), northern wheatgrass (*Elymus lanceolatus*), blue grama (*Bouteloua gracilis*), June grass (*Koeleria macrantha*), western wheatgrass (*Pascopyrum smithii*), as well as forbs such as fringed sagebrush (*Artemisia frigida*), moss phlox (*Phlox hoodii*), scarlet globemallow (*Sphaeralcea coccinea*) and clubmoss (*Selaginella densa*) (Henderson 2006). Field measurements of soil moisture and ALB were taken during May, June and August 2008 in the East Block. Descriptions of the experimental design, field data and data collection methods are provided below.

3.2.1 Field Experimental design

Based on a visual survey of pastures, 1, 6, 8 and 9 (two grazed, two ungrazed) were selected for experimental plot set-up (explained below) to collect soil moisture and ALB data (Figure 3.4).

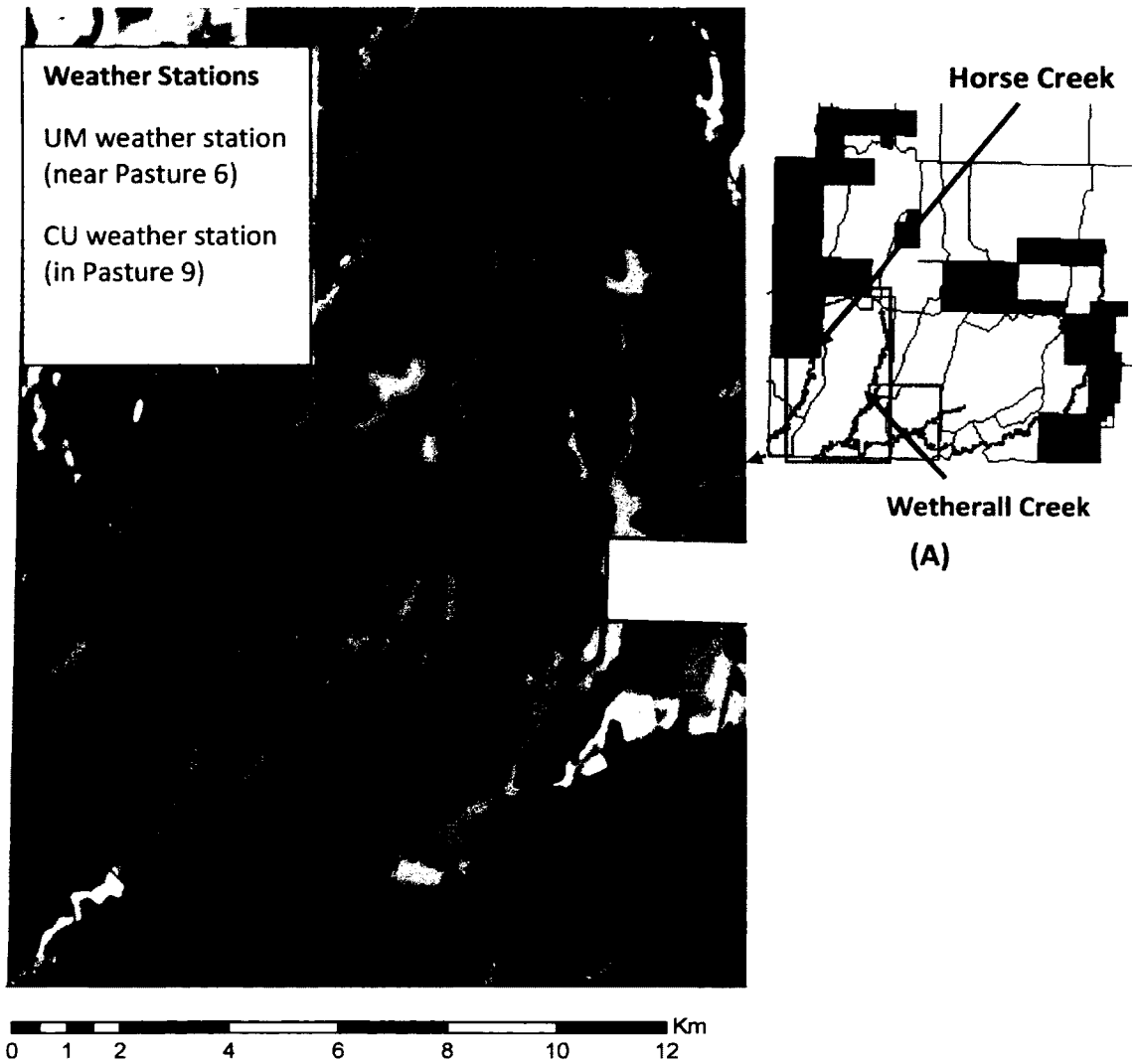


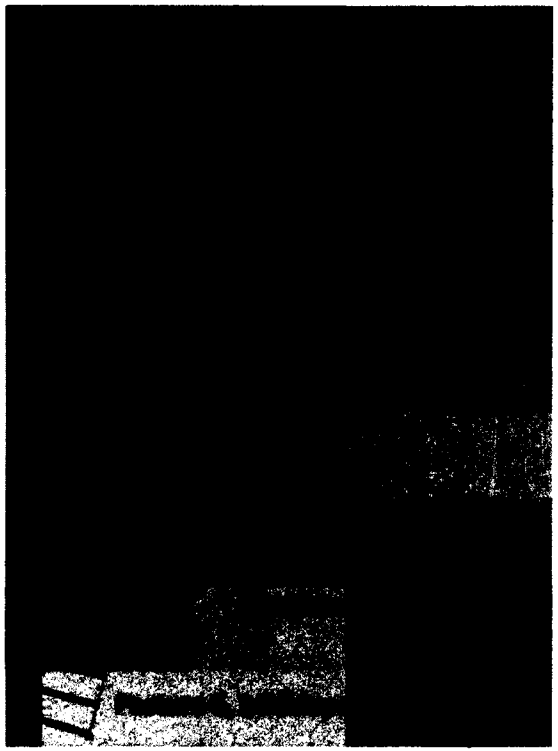
Figure 3.4: (A) East Block, Biodiversity and grazing experiment area (shown in yellow) (Parks Canada 2002); (B) Location of experimental plots (shown in purple) in Pasture 1, 6, 8 and 9 and location of two weather stations (UM and CU, approx. 4.2 km apart) in blue ovals.

Note: UM = University of Manitoba weather station and CU = Carleton University weather station. Source for East Block pastures, roads and DEM: Parks Canada 2008.

To capture likely grazing patterns with respect to proximity to water, the experimental plots were positioned between Horse Creek and Wetherall Creek to include upslope (U), midslope (M), and downslope (D) positions. Three experimental plots, each 30 m x 30 m in size, were placed within the selected pastures (Figure 3.4 and Figure 3.5). The locations of the plots were identified using a handheld Global Positioning System (GPS) with a horizontal accuracy of 2 m and were flagged with a pasture and plot number, using the labelling convention P9U, P9M and P9D (for Pasture 9 upslope, midslope, and downslope, respectively). Please note that due to spatial constraints there were two midslope plots placed in pasture 6 and no downslope plot, therefore to avoid confusion, the plots were referred as P6M (pasture 6, midslope plot 1) and P6M-2 (pasture 6, midslope plot 2).

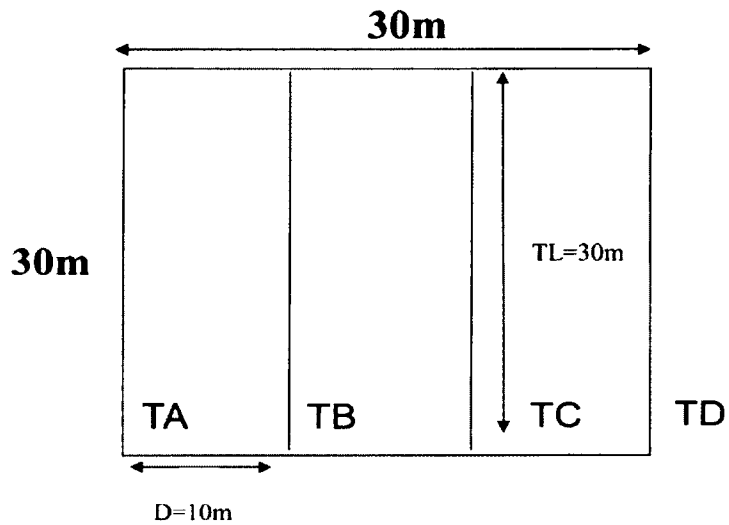
To examine the spatial variability in the vegetation between the experimental plots, long transects were also placed between the experimental plots. For example, a 180 m long transect was placed between P8U and P8M. The length of these long transects was dependent on the distance between the experimental plots.

Transect sampling following the method of Oliver and Webster (1986b) was used to collect the data from each experimental plot. In total four transects (*Transect A, B, C and D*) were placed within each experimental plot, where each transect was 30 m long (Figure 3.5 and Table 3.1). Flags were used to demarcate both experimental plots and transects within the plots.



(A): An example showing Pasture 6 experimental plots (30m x 30m) with transects placed inside and between the plots to capture the spatial variability in soil moisture and productivity.

(B)



(C)

Figure 3.5 Experimental Plot design for Soil Moisture Sampling and Productivity Measurements (Transect sampling). Example of Pasture 6 plots showing transect location (A). Photograph showing transects within the plot (B). Experimental plot showing all four transects placed within the plot (C).

Note: TA = Transect A, TB = Transect B, TC = Transect C, TD = Transect D; D = distance between adjacent transects and TL = Transect length

Table 3.1 Summary of number of experimental plots each pasture and total number of measurements taken per plot

Pasture Number	No. of Plots within each pasture	No. of Transects within each experimental plot	Total no. of measurements (SM and ALB) per plot	No. of Transects between experimental plots	Total no. of measurements (SM and ALB)
P1	3	4	124	0	0
P6	3	4	124	2 (90 m and 120 m)	210
P8	3	4	124	1 (180 m)	180
P9	3	4	124	1 (120 m)	120

Soil moisture and non-destructive ALB measurements were taken every meter from 0 m to 30 m for each transect within and between the experimental plots. The rationale behind the distance between two samples was to capture the local variability of both soil moisture and ALB (details of measurements follow). Equally important, however, the experimental plots were a reasonable size to be able to feasibly sample the pastures during the time frame of the study.

3.2.2 Soil Moisture: methods and data

A variety of destructive or non-destructive methodologies including gravimetric and electromagnetic (time-domain reflectometry, TDR) techniques exist to measure soil water content, with some being more accurate and better accepted than others (Gardner 1965; Topp *et al.* 1980; Zazueta and Xin 1994; Wilson *et al.* 2003; Rahman 2005; Carlyle 2006). As each method has its own advantages and disadvantages, one should be careful in selecting the appropriate method based on the objectives of the study. In addition, to the methodology selected for use when conducting a research, one must also develop the site specific calibration curves (Weber and Gokhale 2011).

In this research both destructive (gravimetric) and non-destructive (time-domain reflectometry) methods were used to measure the soil moisture in the field. In the gravimetric technique the

soil water content was measured through the difference in mass between a wet (field condition) sample and an oven dried sample. In comparison, the TDR method is a common electromagnetic method which measures the bulk dielectric constant of the soil by measuring the time it takes an electromagnetic pulse (wave) to propagate down and back up the insertion rods, when inserted in the sampling medium (Noborio 2001).

As the research permit limited how many destructive samples could be taken from the study area soil moisture measurements were taken using CS-620 Handheld Hydrosense moisture probes with a rod diameter of 5 mm and 32 mm of spacing between the probe rods (Campbell Scientific Incorporation 2001). Additionally, part of the research objective was to see the temporal variability in the soil moisture between and among the pastures. Therefore, CS-620 Handheld Hydrosense moisture probes were used which suited the sampling design, instead of gravimetric technique which does not allow repeat sampling at exactly the same location and is time consuming. Table A1.1 in Appendix 1 shows the date of soil moisture data acquisition from field plots along with averaged weather conditions at the time of data acquisition.

The probes estimate volumetric water content (VWC) integrated from 0 to 12 cm depth or 0 to 20 cm depth. The deeper placements (0 to 20 cm depth) were not possible in all plots across the study area due to stony soils; therefore soil moisture was measured at the 12 cm depth only. CS-620 probe rods are pushed into the ground and the probe generates high frequency electromagnetic energy to polarize water molecules, and it measures the dielectric permittivity to estimate volumetric water content (VWC, in %). VWC is the total amount of water held in a given soil volume at a given time and includes all water that may be present including

gravitational, available and unavailable water. The VWC is displayed on the attached display unit with an accuracy of +/- 3.0% with electrical conductivity of 2 dS m^{-1}. Three readings per point were taken and averaged to smooth instrument error caused by factors such as high salinity, stoniness or soil organic matter.

Sample sites with extremely rocky soil even at 12 cm depth were excluded from the analysis because probes could not be inserted properly. Soil moisture probe calibration was accomplished with simultaneous gravimetric and probe samples. In total, thirty soil samples (6 samples each from experimental pasture P6 and P9; and 9 samples each from pasture P1 and P8) at 12 cm depth over 4 different days were collected for probe calibration purposes. After taking 10 VWC measurements using the handheld hydrosense probe at the 12 cm depth, a soil corer was used to collect a soil sample (0 – 12 cm depth) at the same location. Once each sample was collected, it was placed in a small tin can, labelled and then sealed using electrical tape to retain the moisture. The samples were transported back to the research facility, where they were processed within 24 hrs of collection. The soil samples were weighed with the cans and then the weight of the can was subtracted to get the actual weight of the soil. Once weighed, the samples were dried in the oven at 105°C for 24 hrs or until a constant weight was recorded (c.f. Rode 1969). Using these data, soil bulk density (g cm^{-3}), water volume (ml) and volumetric water content (VWC m^3 water/ m^3 soil) were determined. VWC was regressed against raw probe output values to derive a line of best fit and a linear calibration function (R^2 = 0.64, $p < 0.05$). Percent VWC was determined by multiplying the calibrated VWC by 100. All soil moisture values are expressed as VWC (%).

3.2.3 Above-ground live plant Biomass: methods and data

Both destructive and non-destructive methods exist for the estimation of ALB (Sala and Austin 2000). In general, vegetation clipping used to estimate plant productivity is destructive and time-consuming (for example, sorting live from dead biomass). Additionally, destructive sampling can be problematic for field studies conducted in conserved or managed areas where destructive sampling is undesirable to minimize environmental disturbance. In such cases, non-destructive measurement techniques are preferred (Davidson 2002). Depending on the objective and scale of the research, both ground-based radiometers and satellite imagery have been widely utilized within grassland research to estimate plant biophysical parameters such as above-ground live biomass (Davidson and Csillag 2001; Moreau *et al.* 2003; Flombaum and Sala 2007; Miles 2009; Xie *et al.* 2009). For example, above-ground biomass is estimated using the vegetation indices by establishing an empirical relationship between the destructively measured biomass and the transformations of two or more remotely sensed spectral bands.

Vegetation indices are defined as "*dimensionless, radiometric measures that function as indicators of relative abundance and activity of green vegetation, often including leaf area index (LAI), percent green cover, chlorophyll content, green biomass and absorbed photosynthetically active radiation (APAR)*" (Jensen 2000). Over 40 different vegetation indices have been in use in the field of remote sensing applications, to qualitatively and quantitatively evaluate the percent vegetation ground cover (Bannari *et al.* 1995; Huete *et al.* 2002). Normalized difference vegetation index (NDVI) is one of the most widely used vegetation indices to assess vegetation phenology and estimate landscape patterns of primary productivity (Goward *et al.* 1985; Turner *et al.* 1992; Wylie *et al.* 1996; Yang *et al.* 1998; Davidson and Csillag 2001; Shen *et*

al. 2008). Davidson and Csillag (2001) examined the relationship between various spectral vegetation indices such as NDVI, Ratio Vegetation Index (RVI), Enhanced Vegetation index, (EVI), Renormalized Difference Vegetation Index (RDVI) and Soil Adjusted Vegetation Index (SAVI) and ALB in GNP, Saskatchewan and found that all the vegetation indices produced similar results. Flynn *et al.* (2008) concluded that the spatial variability of biomass in pastures and hayfields can be determined accurately using the NDVI measured from a ground-based sensor.

For this research, NDVI (Equation 3.1) as developed by Rouse *et al.* (1974) was used as it is easy to calculate and interpret, compared to other vegetation indices such as SAVI, atmosphere resistance vegetation index (ARVI) and EVI. Furthermore, NDVI, which is the most widely used index, compensates for varying view-angle illumination across a scene and topographic brightness variations in comparison to a simple vegetation index of Near-infra red (NIR) / Red (R) (Lillesand and Kiefer 2000). It is able to separate green vegetation from other surfaces because the chlorophyll of green vegetation absorbs red light for photosynthesis and reflects the NIR wavelengths due to the scattering caused by the internal structure of the leaf (Wilson and Sader 2002). As vegetation cover increases, *NIR* reflectance increases while *R* reflectance decreases, producing an enhanced difference in the numerator of the index. NDVI is highly correlated with vegetation parameters such as green leaf biomass and green leaf area (Tucker *et al.* 1981 and Roy 1993).

$$\text{NDVI} = \frac{\mathbf{NIR-Red}}{\mathbf{NIR+Red}} \qquad \text{Equation 3.1}$$

where NIR is the reflected energy in the near-infrared wavelength (0.76 – 0.90 μm) and Red is the reflected energy in the red wavelength (0.63 – 0.69 μm). Theoretically, values for NDVI vary

from -1 to +1, where negative values indicate non-vegetated areas such as water, soil, ice, snow, or clouds and positive values greater than zero indicate vegetated areas.

In this research, a CropScan MSR5 ground radiometer (CropScan Inc., USA) was used to acquire the spectral data corresponding to Landsat bands 1 to 5 every meter along the transects within an experimental plot at the same locations where soil moisture was measured. Table A1.2 in Appendix 1 shows the date of ALB data acquisition from field plots along with averaged weather conditions at the time of data acquisition. The radiometer was mounted 1.5 m above the ground and had an instantaneous field of view (IFOV) of 28°, giving a spatial resolution of approximately 0.75 m on the ground at nadir. Corrections for sun angle, irradiance and temperature were performed directly by the radiometer's software, pre-calibrated by the manufacturer. Radiometric calibration was conducted with white standard reference card using the White Standard Up and Down method (CropScan 1994).

While taking reflected irradiation measurements along transects, care was taken not to include the measuring tape within the view of the radiometer (Figure 3.6). All the radiometer readings were taken within 2 hours of local solar noon. Three radiometer readings per point were taken and averaged to smooth any instrument error. The final averaged measurements were used to calculate vegetation indices such as NDVI and EVI. Preliminary results showed high degree of correlation between NDVI and EVI ($R^2 = 0.94$, $p < 0.05$); therefore only NDVI was considered for further analysis. NDVI was converted to ALB using the equation 3.2 obtained through the radiometer calibration on page 53.

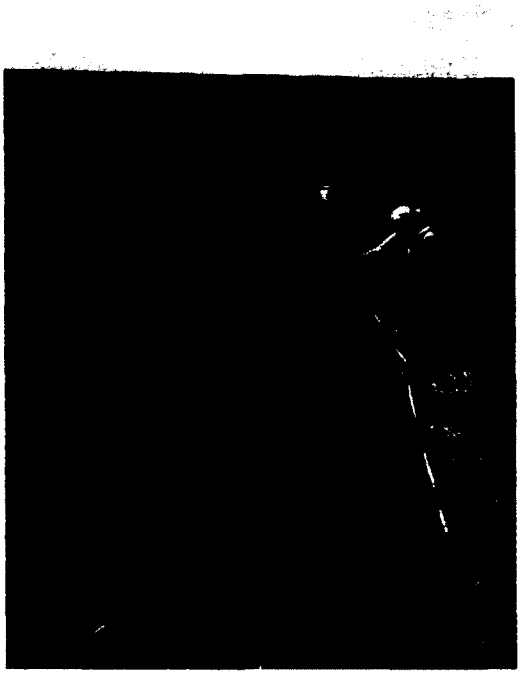


Figure 3.6 CropScan measurements in Pasture 9, upslope site (P9U) taken on May 16, 2008

Twelve additional biomass samples were collected for radiometer calibration purposes, at random locations near the experimental sites after taking the radiometer readings. This limited destructive sampling was done to ensure minimal disturbance to any sensitive natural or cultural heritage features of the park. Based on the criteria in the park research permit, the plot size for the biomass clipping was 50 x 10 cm. After setting up the plot, all above-ground vegetation was clipped using shears and stored in a sealed plastic bags. The vegetation was transported back to the research station and sorted within 36 hrs of collection into the following categories: grass, shrubs, forbs, litter and cryptogram (includes lichens, black algae and *Selaginella densa*). After sorting, the vegetation samples were weighed and then placed in paper bags and oven-dried at 60 °C for 24 hrs or until a constant weight was achieved. Dry weights were measured at the end of each drying cycle and biomass (g m^{-2}) was calculated by dividing the dry weight (g) by sample plot area. These data were combined with similar but

more extensive field data (vegetation type: grass, selaginella/lichen, forb/shrubs and juniper) collected by Davidson *et al.* (2006) and Miles (2009) to derive the calibration equation (Equation 3.2) used in this study ($R^2 = 0.61$, $p = 0.000$, $N = 130$, Figure 3.7). The correlations and slopes of this relationship did not change appreciably using different temporal subsets of the data. Flynn *et al.* (2008) also found a significant correlation ($R^2 = 0.64$) between the pasture biomass of tall fescue and NDVI obtained from the proximal active sensor, GreenSeeker (RT-500) with a spatial resolution of < 1m.

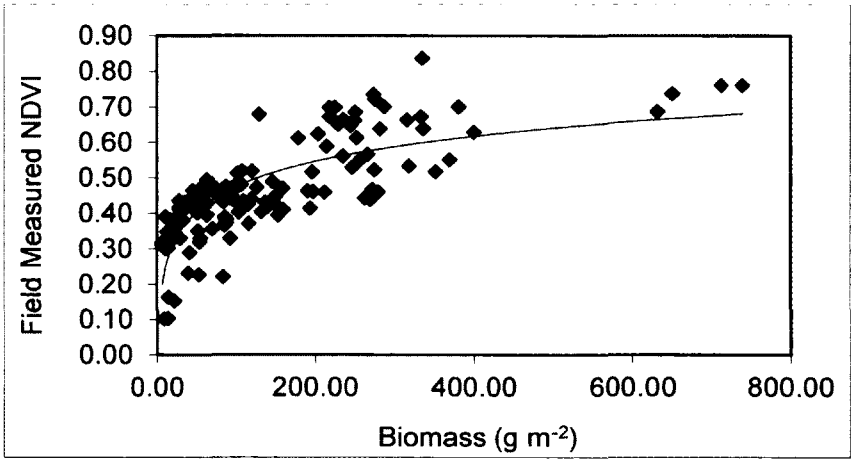


Figure 3.7 Calibration curve for converting field measured NDVI to area biomass (g m^{-2})

Inverted for the purposes of prediction, the regression equation for the above relationship is:

$$\text{Biomass} = (5.9803)e^{(5.873 * \text{Field measured NDVI})} \qquad \text{Equation 3.2}$$

To conclude, each experimental plot was sampled at least once for ALB and 2 to 3 times for soil moisture in the same month. For example, for pasture 6 plots, soil moisture data was collected on 17 May 2008 and 22 May 2008, whereas ALB data was collected only on 17 May 2008. The same strategy was used to collect data from other plots. Weather was a critical factor in terms of collection of biomass data. Due to highly variable conditions in summer 2008, at times it was not feasible to collect reflected radiation measurements using the radiometer along with the

soil moisture data. However, sampling was carried out during the entire growing season (May, June and August 2008, excluding July) thereby allowing temporal variations in surface soil moisture conditions and changes in plant phenology to be adequately sampled.

3.2.4 Weather Data

Soil water content and plant productivity are affected by various factors, including weather (Parton *et al.* 1994; Neufeld 2008). To better understand the interaction between site specific weather conditions and soil water content; and site specific weather conditions and ALB, an AE50 HOBO Weather Station (Onset Computer Corporation 2007), called the CU weather station (Figure 3.8), was installed in pasture 9. Beginning 15 May 2008, the weather station measured and recorded the precipitation (mm), temperature (^{o}C), photosynthetically active radiation ($\mu E\ m^{-2}\ s^{-1}$), relative humidity (%), wind direction (ϕ) and wind speed ($km\ h^{-1}$) until 21 August 2008. Five minute sample data was recorded to the on-board computer every 15 minutes, which was converted into daily format at the end of the field season in 2008.

In addition, a pit was dug (depth 65 cm) at the same site, to install soil moisture sensors (ECH_2O dielectric probes) at 5 different depths (5 cm, 10 cm, 0 - 20 cm, 26 cm and 50 cm) for continuous measurement of soil moisture data during the 2008 growing season (Figure 3.8B). One probe was inserted vertically into the soil to a depth of 4 – 6 cm and another at a depth of ~20 cm, while four others were inserted at angles to depth of 5, 10, 26 and 50 cm. The probe measured volumetric water content (VWC) (%) with an accuracy of +/- 0.041 $m^3\ m^{-3}$.

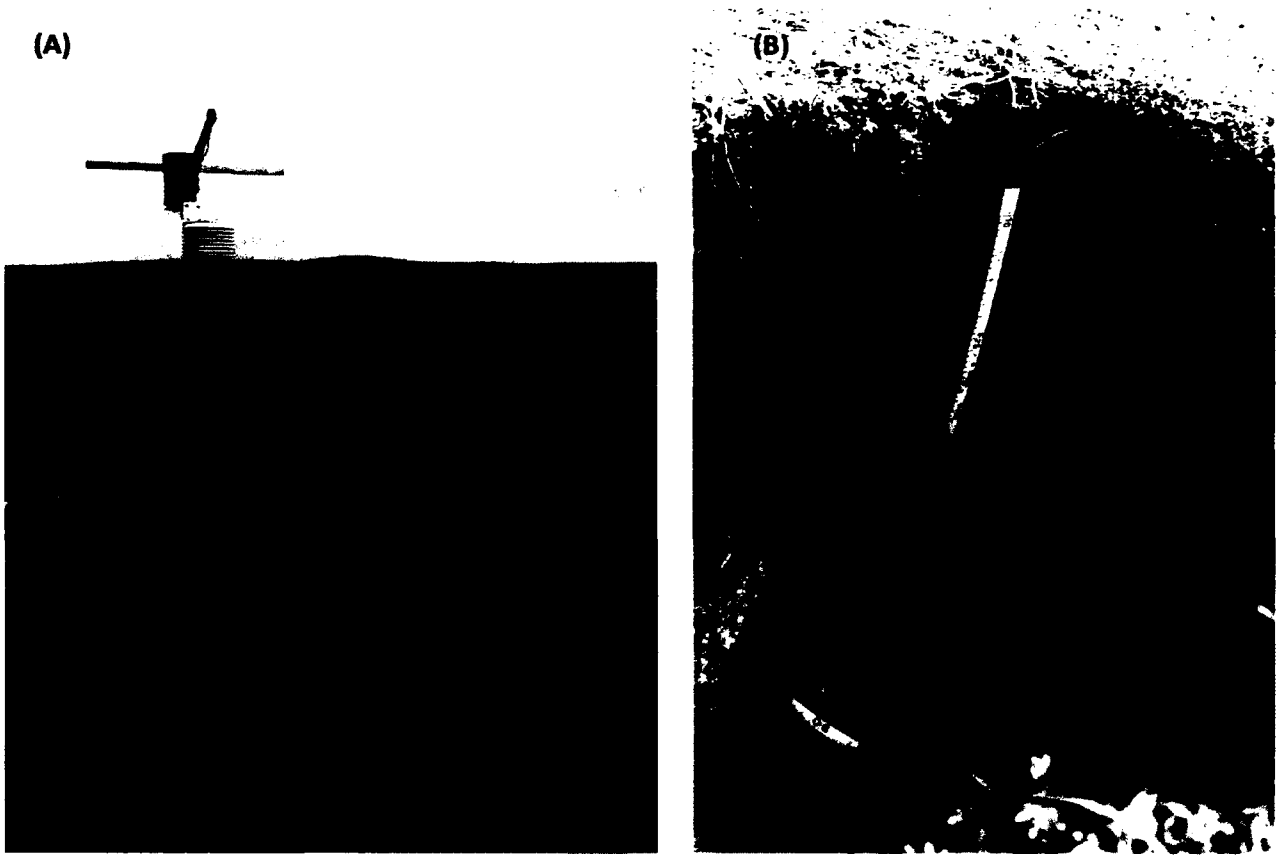


Figure 3.8: AE50 HOBO Weather Station (Onset Computer Corporation 2007) in pasture 9, East Block (A). 65cm pit for ECH_2O dielectric probes near weather station (B).

A second weather station was also installed by University of Manitoba researchers conducting research in the area, referred to here as UM weather station (see Figure 3.4). This weather station was installed near pasture 6 to measure precipitation (mm), temperature (°C), wind direction (ϕ) and wind speed (m s^{-1}). Data were recorded to the on-board computer every hour, and were converted into daily format at the end of the field season.

3.3 Satellite Data, processing, and sampling design

From previous studies in semi-arid region, it was found that Landsat TM is very well suited to estimate biomass and cover under different management practices, as well as appropriate for measuring spatial heterogeneity in grasslands (Guo *et al.* 2000; Zhang *et al.* 2003). He *et al.*

(2006) suggests ~35 m to be the minimum pixel size to capture the spatial variations in grasslands biophysical properties in this study area (i.e. GNP). Similarly, Davidson and Csillag (2001) determined that 10 - 50 m resolution has a potential for estimating C_4 species coverage in the GNP area. Therefore, Landsat TM was determined to be suitable for this study due to its resolution (pixel size approximately 30 m) and free availability from the United States Geologic Survey (USGS) Earth Resource Observation and Science Center (EROS). Despite better temporal resolution (8 day) than Landsat (16 days), coarser imagery such as Moderate Resolution Imaging Spectroradiometer (MODIS) (250 m and 500 m pixel size) and AVHRR (1.1 km pixel size) was not used, as it would not have been able to detect important spatial patterns below these scales in this area.

The satellite data used in this study consisted of five Landsat (TM) scenes (Path/Row: 36/26 and 37/26) with a pixel size of 30 m (Table 3.2). Images covering East Block, GNP and Mankota community pasture were acquired from the United States Geologic Survey (USGS) Earth Resource Observation and Science Center (EROS) (http://glovis.usgs.gov/).

Table 3.2 Landsat TM image acquisition information

Landsat Image Acquisition Date	Path/Row
30 June 2000	37/26
27 June 2007	36/26
29 June 2008	36/26
23 June 2009	37/26
26 June 2010	37/26

For image acquisition, less than 10% cloud cover was used as a selection criterion. All the acquired Landsat scenes from USGS EROS data center were geometrically and radiometrically corrected products (i.e. Level-1T, Standard Terrain Correction product) in Geographic tagged

image file format (GeoTIFF). The Landsat images were registered to a Universal Transverse Mercator (UTM) projection (Zone 13) and were corrected geometrically using 144 ground control points producing a root mean square (RMS) error of < 0.45 pixels (< 12m). The Level-1T products are free from distortions related to the satellite platform (such as altitude deviations from normal), the sensor (jitter and view angle effects), and global Earth characteristics (rotation, curvature and relief) (NASA 2009). The images were further checked for proper geometric alignment by overlaying vector data available for the region (example, road networks and park boundaries), and no problems were found (see, example, Figure 3.2, 3.4, and A2.1).

3.3.1 Image pre-processing

Due to molecular scattering and absorption the atmospheric conditions can significantly vary both spatially and temporally (Mathew *et al.* 2003; Davis 2006). Therefore it is important to perform atmospheric correction especially when establishing quantitative relationships with biophysical information, example ALB / LAI. A variety of models are available for atmospheric normalization or correction such as 6S (Second Simulation of the Satellite Signal in the Solar Spectrum), MODTRAN (Moderate Resolution Atmospheric Radiance and Transmittance Model), and image-based DOS (dark object subtraction) models, where each method has its own characteristics and requirements for the input parameters (Lu *et al.* 2002; Mahiny and Turner 2007; El-Hajj *et al.* 2008). Based on the research objective and data availability, one should choose appropriate models to perform atmospheric correction.

In this research, the Fast Line-of-sight Atmospheric Analysis of Spectral Hypercubes (FLAASH) algorithm based on MODTRAN 4 within ENVI image processing software (ENVI 2008) was used

for atmospheric correction on the Landsat scenes. FLAASH is developed collaboratively by Spectral Sciences, Inc. (SSI) and the U.S. Air Force Research Laboratory; with assistance from the Spectral Information Technical Applications Center (SITAC) (Adler-Goldon *et al.* 1999). This algorithm has been tested and shown to be accurate by Matthew *et al.* (2000), Bruce and Hilbert (2004) and Davis (2006). The algorithm has the benefit of not requiring any ancillary data other than solar zenith angle and visibility at the time of acquisition. Zenith angle was calculated by the software for each image based on data, time and scene location. Visibility estimates were retrieved for each image date from the closest Environment Canada weather station (Mankota). Since LANDSAT images do not have the appropriate bands (typically 1130 nm) to perform water retrieval, the amount of water vapour in the column is determined by the user-selected atmospheric model from a list of standard MODTRAN model atmospheres; mid-latitude continental atmosphere was chosen.

3.3.2 Identification of Grazed and Ungrazed Sites with variable grazing intensity (GI)

The acquired Landsat scenes covered part of Canada (southern Saskatchewan) and part of the USA (Montana), but only the portions over the study area were analysed. Vector files that included the East Block (GNP) and Mankota community pastures were used to clip the region of interest (ROI), pasture boundaries, from the Landsat scenes to ensure that other land-uses such as cultivated agriculture did not impact the analysis (Appendix A2.1).

3.3.3 Sampling design for satellite based data analyses

Each experimental pasture located in the East Block (P1 to P9) and Mankota (P10, 12 and 13) was analyzed to determine the Landsat NDVI based ALB variability within the pastures as a

result of variable grazing intensities. Additionally, transect sampling design was also used to see if slope position (upslope and downslope) had any impact on the ALB heterogeneity. Transect length was dependent on the size of the upslope and downslope area within each experimental pasture of the East Block (P1 to P9) and Mankota community pasture (P10, 12 and 13).

3.4 Data analysis and methods

Data were checked for normal distribution using the Kolmogorov-Smirnov test (p-value > 0.05), and were log-transformed where needed to satisfy assumptions of normality and homogeneity of variance. Additionally, data were tested for homoscedasticity using Levene's test (Levene 1960) and Bartlett's test (Snedecor and Cochran 1983).

A stepwise regression procedure similar to Gill (2007) was used to assess the role of summer weather (year 2008) on local ALB and soil moisture, where ALB and soil moisture were dependent variables and June precipitation, June temperature, June – August precipitation and mean June – August temperature were potential explanatory variables.

Mixed effect models (linear mixed effect models (LME) and generalized linear mixed effect models (GLMEs)) were (SPSS version 20.0, IBM Corporation, New York, USA) used to separate the fixed effects (i.e., where all levels of an effect are represented) of management (grazing treatment) and slope location from the random effects (i.e., where levels of an effect are assumed random and not fully represented; in this study, this included pasture as a random sampling variable) (Bartolome *et al.* 2004; Bell and Grunwald 2004; Johnson 2010; Mandle and Ticktin 2012). Year was treated as a fixed effect because treatment could have cumulative

effects over time since cattle remove vegetation every season, and the amount and distribution of remaining vegetation in year *n+1* depends on the amount of vegetation removed in year *n*. The rationale for using mixed effects models was their ability to analyze repeated measures data, allowing for sequential sampling from a single plot over multiple dates, and using both categorical and continuous effects (variables) simultaneously compared to traditional ANOVA approaches (Piepho *et al.* 2003; McCulley *et al.* 2005). Finally, including random effects in statistical analyses allows us to make inferences beyond the scope of study, compared to conclusions from fixed effects treatments that can only be applied to differences among those treatments addressed in the study (Sahai and Ageel 2000). The temporal data were analyzed using the repeated-measures ANOVA procedure of the SPSS general linear model to estimate the overall significance of treatment effects. When grazing treatment by year interaction or year effects were significant ($p \leq 0.05$), the Tukey-Kramer Honestly Significant Difference (Tukey's HSD) multiple comparisons test (Sall *et al.* 2005; Sasaki *et al.* 2009) and the Bonferroni corrected test were used to determine which treatment-year combinations and which years differed.

3.4.1 Geostatistical analysis using semivariograms

Geostatistics handles data sampled in space, allowing the exploration of variability with respect to distance. Most parametric statistics are inadequate to analyze spatially dependent variables due to the assumption that all the measured observations are independent (Cambardella *et al.* 1994). However, in geostatistics it is assumed that there is spatial autocorrelation (spatial dependence) in the variables, which can be measured and analyzed. Therefore semivariogram analysis was used in this study to detect the range and spatio-temporal variability in soil

moisture and ALB under ungrazed and grazed conditions. Several studies have successfully used semivariogram analysis to capture and estimate pattern and variation in soil properties (Oliver and Webster 1986; Oliver 1987; Cambardella *et al.* 1994; Davidson and Watson 1995; Western and Bloschl 1999; Vieira and Gonzalez 2003; Farkas *et al.* 2008; Krasilnikov 2008; Pan *et al.* 2008) and vegetation (Shiyomi *et al.* 1998; Flynn *et al.* 2008; Lin *et al.* 2010) using field and satellite data. For example, Western and Bloschl (1999) conducted a study in Tarrawara catchment area located in Melbourne, Australia and reported that semivariogram analysis was a useful technique in predicting the soil moisture patterns.

Semivariogram methods have also been successfully used to study the impacts of grazing intensity (low, moderate and heavy) on the spatial patterns of vegetation and soil fertility. For example, Lin *et al.* (2010) conducted a study in a desert steppe, which examined the impact of grazing intensity on the spatial patterns of vegetation, and soil at fine scale (0.1 – 2 m) and coarse scale (1 - 18.7 m). The results showed that grazing altered the spatial patterns of vegetation and soil fertility at the fine scale (< 2 m). The range of spatial autocorrelation of ALB decreased with increasing grazing intensity, indicating that vegetation patches were more fragmented under heavy grazing pressure. Similarly, the spatial heterogeneity of soil water content decreased with increasing grazing intensity at the fine scale. In comparison, spatial patterns of studied variables did not respond to grazing intensities at a coarse scale (1 – 18 m).

A semivariogram is a plot of semivariance (defined as half of the mean squared difference between two samples in a given direction and distance apart) against the lag distance, h , which is the distance between two sample points. The plot values should increase as distance

increases until they reach a plateau beyond which there is no spatial autocorrelation between sample points. This is because observations that are close together should be more similar than the points that are widely separated (also known as Tobler's law of geography) (Tobler 1979; Babish 2006). Besides separation distance, semivariance can depend upon the direction between sampling points. Semivariograms can be calculated either in a unique direction (anisotropic) or for all directions (isotropic) to see if there are any directional trends in the variable under investigation.

Semivariogram analysis consists of the experimental semivariogram, which is calculated from the data, and the semivariogram model fitted to the data. A semivariogram model (spherical, exponential, linear or Gaussian) is chosen from a set of mathematical functions that describe the spatial relationships of the data, as well as providing an estimation of variation between data points.

Semivariance is half of the average squared difference of all the pairs of points separated by a given distance (Equation 3.3, Babish 2006):

$$\gamma_i(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [z(x_i) - z(x_i + h)]^2 \qquad \text{Equation 3.3}$$

where $\gamma_i(h)$ is the semivariance at lag h or the spacing between the two points in the data; $z(x_i)$ is the value of a regionalized variable at location x_i; $z(x_i + h)$ is the value of regionalized variable at a location separated from $x_i$by lag h; and $N(h)$ is the number of sample pairs separated by lag h. The summation is over all pairs of points separated by distance h.

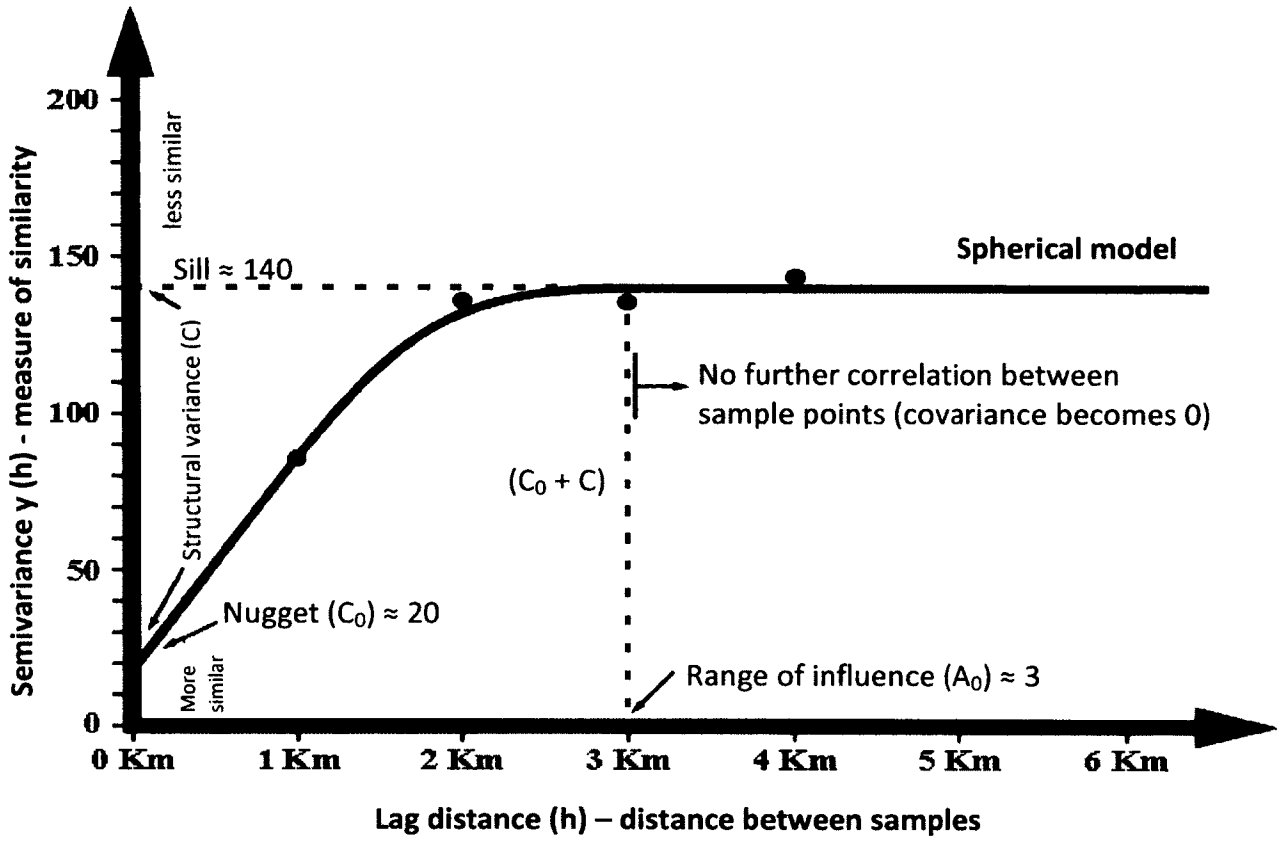


Figure 3.9 Example of an idealized semivariogram curve with spherical model fit (Source: Babish 2006). Note: nugget variance = C_0, range = A_0 and Sill variance = C_0+C; where C is structural variance

In an idealized semivariogram, the semivariance increases as lag increases up to a limit beyond which there is spatial independence. An example of an idealized semivariogram with a spherical model (for equation see Robertson 2008) fit is shown in Figure 3.9. It can be inferred from the figure that pairs of locations which are closer have smaller variance compared to pairs of locations further apart.

Three parameters define a semivariogram: the sill is defined as a maximum semivariance, the range is the maximum distance at which pairs of observations influence each other beyond which autocorrelation between sampling sites is negligible, and the nugget effect is the variance within the sampling units (Kitanidis 1997). The nugget effect in the semivariogram can

be attributed to measurement errors or spatial sources of variation at distances smaller than the sampling interval (or both). The measurement errors occur due to the error inherent in measuring devices. Since natural phenomena can vary spatially over a range of scales, variation at micro scales smaller than the sampling distance will also appear as part of the nugget effect in the model. If the nugget variance value equals the sill variance value, this means that there is no spatial autocorrelation between the sample points and the semivariogram represents pure noise.

Semivariogram analysis does have some limitations; therefore it should be used cautiously. Since the variogram is the average of all pairs of semivariances at each separation distance, it is dependent on the amount of data. Few data samples can result in erratic semivariogram results, especially if that data includes extreme values and outliers; 120 to 200 sample points in a region are good to estimate the semivariogram of that region with confidence (Webster and Oliver 1992). In this study, the n-number of measurements used to estimate a semivariogram were always greater than 120. Since semivariance is affected by outliers from skewed data, it is recommended that where applicable the data should be transformed to reduce the degree of skewness.

In this research, the spatial patterns of soil moisture and field and satellite-based ALB were investigated through their isotropic semivariograms. The fitted exponential model was selected principally on visual fit and coefficient of determination (R-square) with minimum error ranging from 0.0001 to 0.002. Both spherical and exponential models can be used to describe the variation in the soil properties (Cambardella *et al.* 1994; Webster and Oliver 2001; Clifford and

Valentine 2006). However, Webster and Oliver (2001) suggest that experimental variograms with exponential fitted models are expected where differences in soil textures contribute highly to variation in soil properties, as well as sites where the boundaries between textures occur randomly. Compared to spherical models, exponential models do not exhibit a finite range value but for practical purposes there is a point beyond which the semivariance stops increasing.

3.4.2 Moran's I

Moran's I was also used to assess any spatial autocorrelation among neighbouring observations of ALB, using the following equation (Moran 1948; Robertson 2008):

$$I = \frac{n}{S_0} \frac{\sum_i \sum_j w_{ij}(x_i - \bar{x})(x_j - \bar{x})}{\sum_i (x_i - \bar{x})^2} \qquad \text{Equation 3.4}$$

Where $\bar{x}$ is the mean of x variable, w_{ij} is the weight between observation i and j , and S_0 is the sum of all w_{ij} : $S_0 = \sum_i \sum_j w_{ij}$.

Moran's I values theoretically range from −1 to 1, where values of Moran's I near zero indicate randomness, or spatial homogeneity, while values significantly greater than zero (positive spatial autocorrelation) or less than zero (negative spatial autocorrelation) indicate spatial heterogeneity. Positive spatial autocorrelation occurs when similar values occur near one another, while negative spatial autocorrelation occurs when dissimilar values occur near one another. Following Lauzon *et al.* (2005), spatial autocorrelation was concluded significant when the absolute value of Moran's I was > 0.2.

Test of significance was conducted using the progressive Bonferroni correction (Legendre and Legendre 1998) because of its simplicity and robustness. It was calculated using the following equation:

$$\text{Progressive Bonferroni correction} = \alpha'(d) = \frac{\alpha}{d'}, \qquad \text{Equation 3.5}$$

Where d is the lag class (here d = 1 to 20) and α is the probability level of 0.05.

3.4.3 Measures of Heterogeneity

In addition to calculation of semivariograms to examine the spatial patterns and variability, four derivatives were also calculated: correlation ratio (CR), spatial dependence ratio (SDR) (or nugget %), magnitude of spatial heterogeneity (MSH) and relative heterogeneity (SH %).

Correlation ratio is the proportion of the nugget effect values to the sill, where values near zero indicate continuity in spatial dependence (Vieira and Gonzalez 2003). It was calculated as:

$$\textbf{Correlation ratio} = \frac{\textbf{Nugget effect}}{(\textbf{Nugget effect+Sill})} \qquad \text{Equation 3.6}$$

Spatial dependence ratio (SDR) or Nugget % was calculated based on Cambardella *et al.* 1994.

$$\text{SDR} = \left(\frac{\text{Nugget variance}}{\text{Total variance}}\right) * 100 \qquad \text{Equation 3.7}$$

This ratio was used to define the spatial dependency classes for the soil moisture and ALB. If SDR was:

(a) ≤ 25%, the variable was considered strongly spatial dependent

(b) Between 25% and 75%, the variable was considered moderately spatially dependent

(c) >75%, the variable was considered weakly spatially dependent

Based on Lin *et al.* (2010), MSH is measured as the proportion of total sample variation accounted for by spatially structured variation.

$$\boldsymbol{MSH} = \left(\frac{C}{[C_o + C]}\right) \qquad \text{Equation 3.8}$$

Spatial variance (C) can be calculated as follows:

$$\boldsymbol{C} = [\boldsymbol{C_0} + \boldsymbol{C}] - \boldsymbol{C_0} \qquad \text{Equation 3.9}$$

Here, C_0 is the nugget variance representing random variation (i.e. homogeneity); $C_0 + C$ is the sill representing maximum (or total) variation and C is spatial variance.

The MSH has been widely used to estimate the magnitude of spatial dependence for different soil variables within a site (Robertson *et al.* 1993; Boerner *et al.* 1998; Lin *et al.* 2010). Values for MSH range from 0 to 1, where a value of zero indicates no spatially structured heterogeneity (i.e. samples at all separation distances are independent from each other) and a value of 1 indicates high amount of spatially structured heterogeneity. Both MSH and spatial dependence are correlated, i.e. higher the MSH, the stronger the spatial dependence.

Based on Li and Reynolds (1995), the relative heterogeneity (SH %) represents the proportion of the auto-correlated spatial heterogeneity in the total variation, and is calculated from the nugget variance and sill:

$$\text{SH \%} = \left(\frac{\text{Sill variance} - \text{Nugget variance}}{\text{Sill variance}}\right) * 100 \qquad \text{Equation 3.10}$$

Therefore, auto-correlated variation (i.e. heterogeneity) can be calculated by subtracting the random variation (i.e. nugget) from the total variation (i.e. sill). MSH is sometimes also expressed as SH %,

$$\mathbf{SH\ \%} = \mathbf{MSH} * \mathbf{100} \qquad \text{Equation 3.11}$$

Similar to MSH, relative heterogeneity is also directly correlated with the spatial dependence. Therefore, the higher the relative heterogeneity, the stronger the spatial dependence. Similar to Western *et al.* (2004) study, correlations were also calculated between the averaged SM and semivariograms variables such as sill, range and MSH; as well as averaged ALB and semivariogram variables.

3.5 Modeling

Successful modelling of ecosystem biogeochemical cycles requires a good understanding of the primary controls on ecosystem processes. A wide range of modelling approaches to modelling vegetation productivity dynamics exists, from simple (for example, Monteith's (1977) light use efficiency model or ε-model) to complicated models (example, Regional Hydro-Ecological Simulation System (RHESSYS) (Band *et al.* 1991; Tague (1999)) and each has their own unique aims, key assumptions, spatial scales and shortcomings. For example, the RHESSYS model is relatively complex in that it uses many parameters. The key processes and the assumptions involved in this model need to be understood before it can be implemented in a landscape. Though it is well proven in forest ecosystems, more testing is required to determine whether or not this model approach works well in grassland environments. Despite modifications made to accommodate grasslands (example, Mitchell 2003), lack of validation of the parameters used in different sub-models in RHESSYS is an issue.

3.5.1 CENTURY Model

For this research, the CENTURY model (version 4.5) was used to model the effects of grazing intensity and soil texture on the grassland productivity and soil carbon dynamics within a long term grazing pasture. The rationale for using the CENTURY model was that it has been widely used and validated in the grassland ecosystem (see Burke *et al.* 1991; Holland *et al.* 1992; Gilmanov *et al.* 1997; Mikhailova *et al.* 2000; Mitchell and Csillag 2001; Ardo and Olsson 2003). In addition, the model also allows scheduling of events of interest to managers such as managed grazing, crop rotations or fires at specific times during the simulations to assess the impact of management options over long time-periods.

The CENTURY model was developed at National Resource Ecology Laboratory (NREL), Colorado State University, primarily to supply a tool for ecosystem analysis enabling the evaluation of changes in climate and human disturbance (Parton *et al.* 1987; Metherell *et al.* 1993). In general, the model simulates the carbon and nutrient dynamics, primary production and water balance on a monthly time step. The main driving variables for the model are: monthly precipitation; monthly average minimum and maximum temperatures; soil texture, plant nitrogen (N), sulphur (S) and phosphorus (P) content; lignin content of plant; atmospheric and soil N inputs; and initial soil C, N, P, and S levels (Metherell *et al.* 1993). The model also allows inclusion of the effects of fire, fertilization, irrigation, grazing, various cultivation and harvest methods, etc. in the simulations. The input variables are available for most of the natural and agricultural ecosystems, and therefore can generally be estimated from existing literature (Metherell *et al.* 1993).

CENTURY is composed of a soil organic matter/decomposition sub-model, a water budget sub-model, plant production sub-models (example grassland/crop production sub-model) and management and events scheduling functions. The soil organic matter/decomposition sub-model includes three soil organic matter pools (active, slow and passive) with different potential decomposition rates, above and below ground litter pools and a surface microbial pool which is associated with decomposing surface litter (Appendix 3, Figure A3.5.4). The sub-model accounts for the protection of soil organic carbon by including the soil texture and assumes that soil texture influences the decomposition rate of the active soil organic carbon (SOC) and the efficiency of stabilizing active SOC into slow SOC (Parton *et al.* 1987; 1993). The decay rate of active SOC decreases with increase in silt plus clay content of the soil. Furthermore, fraction of carbon lost as CO_2 when active SOC is decomposed and stabilized into slow SOC also decreases as the silt plus clay content increases. This results in an increase in the amount of carbon stabilized in slow SOC for the fine textured soils. Most of the soil organic matter sub-model initializations used empirical methods developed for the Great Plains region. For details see Parton *et al.* (1987). The simplified water budget model calculates monthly evaporation, transpiration, the water content of the soil layers based on soil texture, snow water content, and saturated flow of water between soil layers (Metherell *et al.* 1993).

3.5.2 Parameterization of the model and model set-up

The CENTURY model simulates carbon and nutrient dynamics, primary production and water balance using a monthly time-step. The main driving variables for the model include monthly precipitation, minimum and maximum temperature and soil texture, litter chemistry and management practices (grazing). Parameters used by Mitchell and Csillag (2001) for the GNP

area was used as the basis for this study (Appendix 3, A3.1). However, where possible, the parameters were modified based on field data collected for this thesis. Additionally, soil characteristics (sand, silt and clay fractions; soil pH and bulk density) in the model were defined using spot measurements taken at the field sites and the park's soil survey (Saskatchewan Institute of Pedology 1992). Since actual data on field capacity and wilting point were not available for the study area, soil water holding capacity and wilting point in the model were estimated using the equation of Rawls *et al.* (1982) (for details refer to Metherell *et al.* 1993). Vegetation mix parameters (88% C_3 and 12% C_4) in the study area were determined based on Davidson and Csillag (2001). The effect of temperature on grass biomass production was calculated using a vegetation mix with 88% C_3 and 12% C_4.

Daily meteorological data from 1970 – 2007 observed from an Environment Canada station at Mankota (Environment Canada 2012) were aggregated into monthly total precipitation and maximum, minimum temperatures (Table 3.3). This monthly climate data from 1970 – 2007 was the climate input for the CENTURY model simulations.

The model provides the user with an option of using long term average or stochastically generated precipitation and the mean temperature values. Standard deviation and skewness of monthly precipitation totals are needed, if the stochastic precipitation option is to be used to run the model. Therefore, the 30-year climate record data (1970 – 2007) was also used to generate precipitation means, standard deviations and skewness.

Table 3.3 Monthly summaries of daily data from Environment Canada climate station Mankota (1970 - 2007)

Month	Avg. total pptn (mm)	S.D. (avg. total pptn)	Avg. T_{mean} (°C)	S.D. (T_{mean})	Avg. T_{max} (°C)	Avg. T_{min} (°C)
January	18.44	19.62	-13.6	4.6	-7.3	-19.8
February	13.25	14.41	-9.8	4.9	-3.7	-15.9
March	16.61	18.35	-3.1	3.3	3.1	-9.4
April	22.12	16.32	4.7	2.2	12	-2.6
May	53.72	40.51	11	1.6	18.6	3.3
June	64.16	42.87	15.4	1.3	22.9	7.8
July	44.66	39.83	18.5	1.5	26.8	10.1
August	25.55	23.86	17.9	2.2	26.7	9.1
September	26.26	23.59	11.5	2.1	19.9	3.1
October	15.49	16.72	4.9	1.2	12.8	-3
November	13.84	14.49	-5.1	3.3	1.3	-11.3
December	14.92	11.68	-11.6	4.1	-5.3	-17.7
TOTAL	**329.02**					

Note: Avg. = Average; S.D. = standard deviation; pptn = precipitation; T_{min} = minimum temperature; T_{max} = maximum temperature; T_{mean} = average temperature

To accomplish the objectives of this study, the grassland/crop production sub-model (Appendix 3, A3.5.2) was chosen to simulate the long term impacts of light, moderate and heavy grazing intensity on the plant productivity and soil carbon dynamics in the mixed prairie region. It should be noted that this sub-model is linked to a common soil organic matter/decomposition sub-model. The grassland/crop production model assumes that the monthly maximum productivity is controlled by moisture and temperature, and that insufficient nutrient availability results in decreased productivity (Metherell *et al.* 1993). The model simulates the monthly dynamics of carbon and nitrogen in the live and dead above-ground plant material, live roots, and structural and metabolic surface and soil residue pools. For more details on this sub-model refer to Parton *et al.* (1987) and Metherell *et al.* (1993).

The effects of grazing on plant production are represented in the model by using data from Holland *et al.* (1992). Grazing removes live and dead vegetation, alters the root to shoot ratio, increases the N content of live shoots and roots and returns the nutrients to the soil (Holland *et al.* 1992). Version 4.5 of CENTURY has seven options (grazing effect (GRZEFF) = 0, 1, 2, 3, 4, 5 and 6) for dealing with the impact of grazing on the system. In the first option (GRZEFF = 0), there are no direct impacts of grazing on plant production except for the removal of vegetation and return of nutrients by the animals. Option 2 (GRZEFF = 1) is referred to as the lightly grazed effect by Holland *et al.* (1992) and includes a constant root to shoot ratio (not changing with grazing) and a linear decrease in potential plant production with increasing grazing intensity. Option 3 (GRZEFF = 2) is referred to as the heavy grazed (Holland *et al.* 1992) option and includes a complex grazing optimization curve where aboveground plant production is increased for moderate grazing and decreased sharply for heavy grazing levels (> 40% removed per month). The root to shoot ratio is constant for low to moderate grazing levels and decreases rapidly for heavy grazing levels. The fourth option, GRZEFF = 3, refers to a quadratic impact on root/shoot ratio, whereas GRZEFF = 4 refers to linear impact on root/shoot ratio. A complete description of the parameterization of the model for different plant systems and the use of the different management options is presented in the CENTURY User Manual (Metherell *et al.* 1993).

The light, moderate and heavy grazing intensity options were chosen for the simulation analysis. For the purpose of this research, light grazing in the CENTURY model was simulated using option 1 (GRZEFF = 0 and fraction of live shoots removed (*flgrem*) = 0.1); moderate grazing was simulated using option 2 (GRZEFF = 1 and *flgrem* = 0.2) and heavy grazing was

simulated using option 3 (GRZEFF = 2 and *flgrem* = 0.3). These grazing scenarios were designed to mimic the area's history with moderate grazing (defined in CENTURY as having a linear effect on productivity) from 1900 to 1990, no grazing from 1991 to 2005, and grazing with variable intensities from 2006 to 2020 (based on Parks Canada 2006). A sensitivity analysis was conducted to evaluate the impact of change in *flgrem* (model range 0 to 1) within a particular grazing intensity on the plant productivity and soil carbon dynamics.

Soil texture is important as it controls the water movement in the soil, and influences the chemical reactivity and nutrient availability to plants. Therefore, CENTURY was also used to examine the effect of variation within a soil textural class on the grassland productivity, total soil and plant system carbon. Site specific soil data based on the Park's soil survey (Saskatchewan Institute of Pedology 1992) was used for the model's soil parameterization, including soil texture (sand, silt and clay content), bulk density and field capacity. Based on the range of soils in the experimental pastures, according to the park's soil survey data, a total of six representative soil textures in the study area were chosen. % clay in each of the six soil textures was increased in ~5% increments to determine how CENTURY soil and plant system carbon and plant productivity estimates change due to changing soil texture inputs.

All the model runs, unless otherwise noted, were from the year 1 to 2020; the first 1900 years are "spin-up" time to ensure stability in the pools and fluxes, and output was only examined from 2006 to 2020 for the effects of variability in grazing intensity and soil texture on the ALB production and total soil and plant system carbon. Variability in annual net primary

productivity was examined from 1991 to 2020 to compare the effect of no grazing (1991 to 2005) and variable grazing intensities (year 2006 to 2020).

Since GNP is considering terminating grazing in the East Block by 2013, impact of grazing termination on annual net primary productivity and carbon dynamics was also tested. For this scenario, a schedule file was created with following options: no grazing (1991 to 2005); grazing with light, moderate and heavy intensity (2006 to 2012) and grazing termination (2013 to 2020).

3.5.3 Sensitivity Analyses: Grazing Intensity and Soil Texture

The purpose of performing a sensitivity analyses was to see how much uncertainty in the soil and grazing parameters can affect the overall model predictions. Model runs were performed with climate records from 1970 to 2007 and stochastically generated climate based on long-term averages for all other years. This was used to drive predictions for mixed-grassland vegetation (comprising both C_3 (88%) and C_4 (12%)) under the base climate and soil texture representative of the study area.

Based on the park's soil survey (Saskatchewan Institute of Pedology 1992) the study area had six distinct soil texture classes: loam (L), clay loam (CL), silt loam (SL), clay (C), sandy loam (SaL) and sandy clay loam (Scl). Each of these soil textures was associated with a wilting point and field capacity. The range of variation in the relative proportion of sand, silt and clay size particles for each of the six soil textures in the experimental pastures were determined using a Canadian soil texture triangle calculator (Saxton *et al.* 1986). Soil texture values used in the CENTURY model are averaged values, and thus do not include the entire range of values found

in a particular texture class (Updegraff *et al.* 2010). Analyses of the Canadian texture triangle (Saxton *et al.* 1986) and a study conducted by Soil Survey Division Staff (1993) demonstrated that proportion of each textural component can vary by as much as ~14.4% while still remaining in the same textural class. Therefore, during soil texture sensitivity analyses % clay and %sand in each of the six soil textures was changed in ~5% increments to assess the effect of this change on model's prediction of plant productivity and soil carbon.

To test the effect of grazing intensity, the model was run initially with no grazing, light grazing, moderate grazing and heavy grazing impact using the prescribed climate set-up on all the six soil textures found in the study area. For each grazing scenario, the grazing intensity was changed only from 2006 to 2020 to see the impact of variable grazing intensity (light, moderate and heavy) on the grassland productivity, total soil and plant system carbon. All the simulations were run using the 88% warm-grass (C_4 photosynthetic pathway) and 12% cool-grass (C_3 photosynthetic pathway) mixed-grassland vegetation parameterization.

Sensitivity analysis was performed to examine the effect of variations in the fraction of live shoots removed per month by grazing (*flgrem* variable in CENTURY). In these set of runs, the site-specific land management history was held constant while *flgrem* was varied from 0 to 1 in increments of 0.1.

3.6 Research Contributions

As part of the BGE initiated in 2006 in GNP, this research will help GNP evaluate how, when and where grazing-induced changes in ALB will occur. Since biomass is one of the key response variables of the experiment, the results will also help to evaluate how long-term ungrazed

landscapes remain different from grazed areas. The research will also test impacts of different grazing intensities (ranging from light (20 - 33%) to moderate (45 – 57%) and heavy (70%) grazing intensity) within the park on ALB at a range of spatial and temporal scales, which will be useful for adjusting grazer population sizes as well as deciding whether or not (or how) to continue future grazing treatments within the park to maintain plant heterogeneity.

This research also contributes towards understanding the impact of rainfall variability on the soil moisture and ALB variation in grazed and ungrazed areas. This kind of study is important both from rangeland and conservation perspectives. Livestock production is currently one of the biggest land uses in the North-American Great plains (Lueders *et al.* 2006). Therefore, composition and productivity responses to altered water regimes will be of particular interest to rangeland managers. This is because these responses influence the capacity of grasslands to support livestock production. On the other hand, from a conservation perspective, any alteration in the water patterns during the growing season may have important consequences for regional patterns of biological diversity in grasslands (Fay *et al.* 2003).

The field and satellite parts of this study also contribute to understanding of pattern (heterogeneity) and factors (grazing disturbance, weather and slope location) affecting the pattern in soil moisture and ALB at different spatial scales. The study also contributes to understanding of how scale can influence and alter the relationship between pattern and factors in the mixed-grassland ecosystem. The modelling part of the research contributes towards understanding of the effects of variability of soil characteristics on the biomass, soil moisture, plant productivity (both aboveground and belowground) and total soil and plant

system carbon. Additionally, the model results help to explore the long-term behaviour of the system under different scenarios of grazing management. From a management perspective it is important to understand how variability in a soil texture when combined with land use options such as grazing may impact the ecosystem processes. This is because spatial distribution of soil moisture is highly dependent on the soil texture, which in turn influences the plant root growth and the above ground biomass organization (Ursino 2009). All this affects the diversity of consumers ranging from insects to birds and mammals. Since one of the goals of Parks Canada is to maintain and sustain biodiversity within the GNP, understanding the impacts of variability will help in better decision-making and sustainable management of grassland areas.

4.0 Detection of Spatio-temporal Variation and Pattern in Field-based Soil Moisture (SM) and Above-ground live plant biomass (ALB): A case study of Experimental Pastures located in East Block, GNP, Saskatchewan

Semi-arid regions respond to both climatic variability and grazing pressure (Fuhlendorf *et al.* 2001; Cheng *et al.* 2011). Recent work by Fay *et al.* (2003), Swemmer *et al.* (2007) and Wu *et al.* (2010) suggests a link between amount and seasonality of water availability and reduction in biomass production. Therefore, field-based research was designed to quantify temporal patterns and spatial variability in SM and ALB, and determine the influence of local intra-seasonal weather variability and slope location on the spatio-temporal variability of SM and ALB across a range of scales (plot to pasture). In addition, the summer of 2008 was the first year that controlled grazing was reintroduced in the East Block of the park, providing an opportunity to document the baseline conditions and variation in the experimental pastures and to test to see if changes in heterogeneity caused by grazing could be detected within the first year. It was hypothesized that both ALB and SM would differ among plots and between pastures due to external factors such as weather, and that slope location (upslope, midslope and downslope) would also affect ALB and SM, with downslope plots showing more heterogeneity over time than upslope and midslope plots. This is because slope affects the surface run-off (upslope to downslope) and creates differential drainage; affecting the soil moisture available to vegetation and resulting in heterogeneity in plant cover (Bridge and Johnson 2000). It was also predicted that grazing disturbance would significantly affect the ALB and SM spatial patterns resulting in more heterogeneity in ALB and SM compared to non-grazed conditions. This is because grazers are selective in nature, resulting in heterogeneity in vegetation and potentially leading to altered soil properties due to trampling (Vermeire *et al.* 2004; Bakker *et al.* 2006).

To test these hypotheses, field work was carried out in summer 2008 at the East Block of the GNP located in southern Saskatchewan. The East Block of the GNP was selected for a number of reasons. There has been limited or no research on SM and ALB variation at a fine spatial scale (for example 1 m spacing) in this area. Fine resolution data is important particularly in semi-arid regions where spatial variability is very high. There were areas within the East Block that have not had any large mammal grazing since 1992, which helped to study the variability in ungrazed conditions. There was low or almost no disturbance from human activities, leaving the ecosystem in a relatively natural state. Good access and close proximity to the research station allowed the set-up of plots and acquisition of data during the 2008 growing season. In 2006 experimental pastures were established within the East Block of the park, and then from 2008 a range of grazing conditions was introduced on these same pastures. This provided an opportunity to see the short-term impacts of grazing on SM and ALB.

Four of the pastures, referred to as P1, P6, P8 and P9, were selected for this study in May 2008 for the collection of SM and ALB data. Within each pasture, three experimental plots (30 m X 30 m) were placed, where one plot each was placed in upslope, midslope and downslope areas of the pasture to assess the impact of slope location on the SM and ALB variability, as well as to capture baseline conditions prior to hypothesized differences in grazing patterns with respect to proximity to water. Details of the experimental design and plot-set up were provided in section 3.2, and the plot naming convention introduced there continues (enclosing pasture number combined with their slope location upslope (U), midslope (M) or downslope (D), for example, P1U, P1M, and P1D) throughout this chapter.

Grazing was introduced in late June 2008 in pastures P6 and P8, while pastures P1 and P9 were used as controls; sections 4.2.3, 4.2.4 and subsequent sub-sections provide analysis for SM and ALB before and after treatment and in section 4.2.3, the naming convention is modified as follows: pasture number will be combined with the respective treatment code: control (C), before grazing (BG) and after grazing (AG) (for example, $P1_C$, $P6_{BG}$, $P6_{AG}$, $P8_{BG}$, $P8_{AG}$, $P9_C$).

4.1 East Block experimental site characteristics

All the experimental pastures had a mix of shrub, grass and forbs; however the distribution of vegetation cover was highly variable in the experimental plots. For example, P9D had a high frequency of shrubs coinciding with the sample points for ALB measurement, resulting in high biomass values at those points. In contrast, vegetation cover in P1U was highly fragmented with more bare soil, resulting in very low biomass values at certain points.

In addition to grasses, some of the experimental plots also supported cactus (*Opuntia* spp.) and shrubs (*Artemisia cana, Atriplex* spp., *Chrysothamnus nauseosus, Sarcobatus vermiculatus*) along with the prairie rose (*Rosa acicularis*) (Figure 4.1). For example, P1U had sparser grass cover compared to P1M plot, and was dominated by cactus and clubmoss with patches of bare ground. In contrast, P1M had a mix of grass, forbs and shrubs including prairie rose.

(A) Pasture 1, upslope plot (P1U)

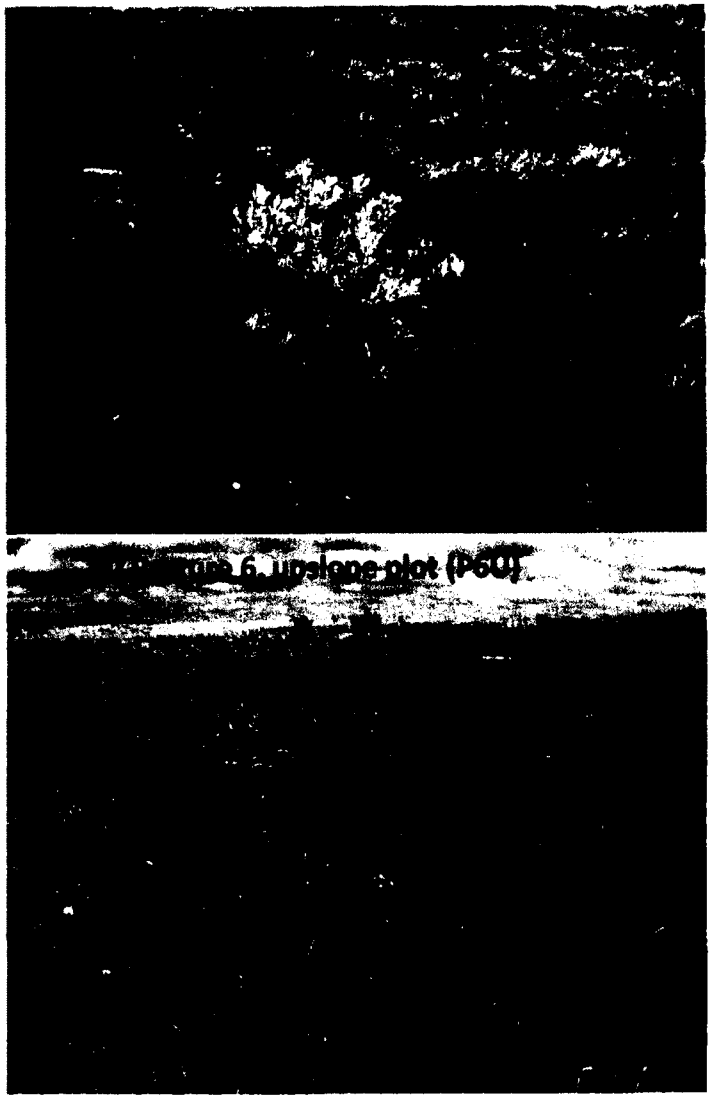


Figure 4.1 Photographs A, B, C, D and E showing vegetation cover in P1U, P1M, P9D and P6U plots.
Note: Photographs were taken during field season in summer 2008.

The experimental plots were also characterized by variability in physical characteristics (Table 4.1). Based on personal observations the slope ranged from gentle to moderate (3 – 8.5 degrees) in most of the experimental pastures. However, P9D was located in a region that had slightly higher slope compared to other P9 plots.

Soil salinity and pH values used in the research were based on the park's soil survey (Saskatchewan Institute of Pedology 1992). The majority of the experimental plots showed some degree of salinity (averaged electrical conductivity of 2 - 4 m S cm^{-1} observed at 0 - 60 cm depth). For example, 10 - 20% of P1D and P9D were affected by moderate degree of salinity. In contrast, 20 - 40% of P8D (lower half) was affected by strong salinity with most of the saline soils occurring in the depression at the end of transects A and B. Similarly, patches of white surface crust usually developed as a result of saline soils were also observed in P1U plot during the measurements. Salinity in soil can result in moisture stress and reduced plant growth, due to water soluble salts that inhibit plant uptake of moisture. In general, all the experimental plots had a soil pH varying between slightly acidic to slightly alkaline (pH range = 5.5 to 7.5).

Table 4.1 Generalized soil characteristics in experimental plots (extracted from Saskatchewan Institute of Pedology (1992))

Experimental plots	% Slope and description	Landform	Soil Texture	Stoniness	Surface pH* and % surface pH class (in brackets) description
P1U, P1M	(2 – 10%) Gentle to moderate	Undulating and dissected	$P1U_{UG}$ = Clay-loam clay; $P1M_{UG}$ = loamy sand to silt loam	Slightly stony. $P1U_{UG}$ more stony than $P1M_{UG}$	B5 (B^7A^3) = 70% area slightly acidic to neutral; and 30% area slightly acidic
P1D	(0.5 – 5%) Very gentle to gentle slopes	Undulating	Sandy clay to sandy clay loam	Non-stony	A4 ($A^3B^3C^3D^1$) = 30% slightly acidic; 30% slightly acidic to neutral; 30% neutral to slightly alkaline; and 10% alkaline
P6U, P6M, P6M-2	(2 – 5%) Gentle	Undulating and dissected	Clay to sandy clay loam	Very slightly stony	B2 (B^7C^3) = 70% slightly acidic to neutral and 30% neutral to slightly alkaline
P8U, long transect	(2 – 10%) Gentle to moderate	Undulating and dissected	$P8U_{G70}$ = Sandy clay to clay. Long transect= Loam	Slightly stony to very stony.	B3 (B^5C^5) = 50% slightly acidic to neutral and 50% neutral to slightly alkaline
P8M	(2 – 10%) Gentle to moderate	Undulating and dissected	clay loam	Slightly stony	B3 (B^5C^5) = 50% slightly acidic to neutral and 50% neutral to slightly alkaline
P8D (first half)	(2 – 5%) Gentle	Inclined	Loam-clay loam	Slightly stony	B5 (B^7A^3) = 70% area slightly acidic to neutral; and 30% area slightly acidic
P8D (second half)	(0.5 – 5%) Very gentle to gentle	Undulating	Loam-Sandy clay loam	Slightly stony	A3 ($A^3B^4C^3$) = 30% slightly acidic; 40% slightly acidic to neutral; and 30% neutral to slightly alkaline
P9U,P9M, long transect	(2 – 10%) Gentle to moderate	Undulating and dissected	Fine sandy loam to sandy clay loam soil	Moderately stony to very stony.	B2 (B^7C^3) = 70% slightly acidic to neutral and 30% neutral to slightly alkaline
P9D	(10 – 15%) Strong	Inclined and gullied	Clay loam-clay	Very stony at some places	B3 (B^5C^5) = 50% slightly acidic to neutral and 50% neutral to slightly alkaline

Note: *% surface pH class is provided in the brackets: pH class 'A' = pH range of 5.5 to 6.0 (slightly acidic), pH class 'B' = pH range of 6.1 to 6.7 (slightly acidic to neutral), class 'C' = pH range of 6.8 to 7.5 (neutral to slightly alkaline; and class 'D' = pH range greater than 7.5 (alkaline).

Amount of stoniness (visual estimate) varied from non-stony to very stony (stones cover 3 to 15% of the surface) between the plots, and this occasionally influenced the ability to take measurements. For example, transect A in both P9U and P9M had some sample points that were particularly stony at the surface, and the ends of transects A and B in P8U and P8M had a greater number of stones compared to rest of the plots. As a result it was not possible to insert probes at some of the sample points with high stoniness for SM measurement, resulting in no SM data at those points.

All the experimental plots showed some degree of variation in the soil texture. For example, based on the park's soil survey (Saskatchewan Institute of Pedology 1992) and field soil samples collected during 2008 within pasture 1, P1U had clay to loam-clay texture, P1M had texture varying between loamy sand and silt loam, and P1D was a mix of sandy clay, sandy clay loam and clay-loam (Table 4.1).

4.2. Results

4.2.1 Soil Moisture

4.2.1.1 Impact of local Intra-seasonal weather conditions

Rainfall data from the area illustrate the range of local variation in the 2008 growing season (May – August) (Figure 4.2). The University of Manitoba (UM) weather station (Figure 4.2B) installed near pasture 6 recorded more rainfall during the growing season (May – August 2008) compared to the Carleton University (CU) weather station (Figure 4.2A). Both weather stations (approx. 4.2 km apart) recorded a higher amount of rainfall during June (35.2 mm (CU) and 52.2

mm (UM)) compared to May, July and August, however it should be noted that in May and August, measurements were only made during a portion of the month.

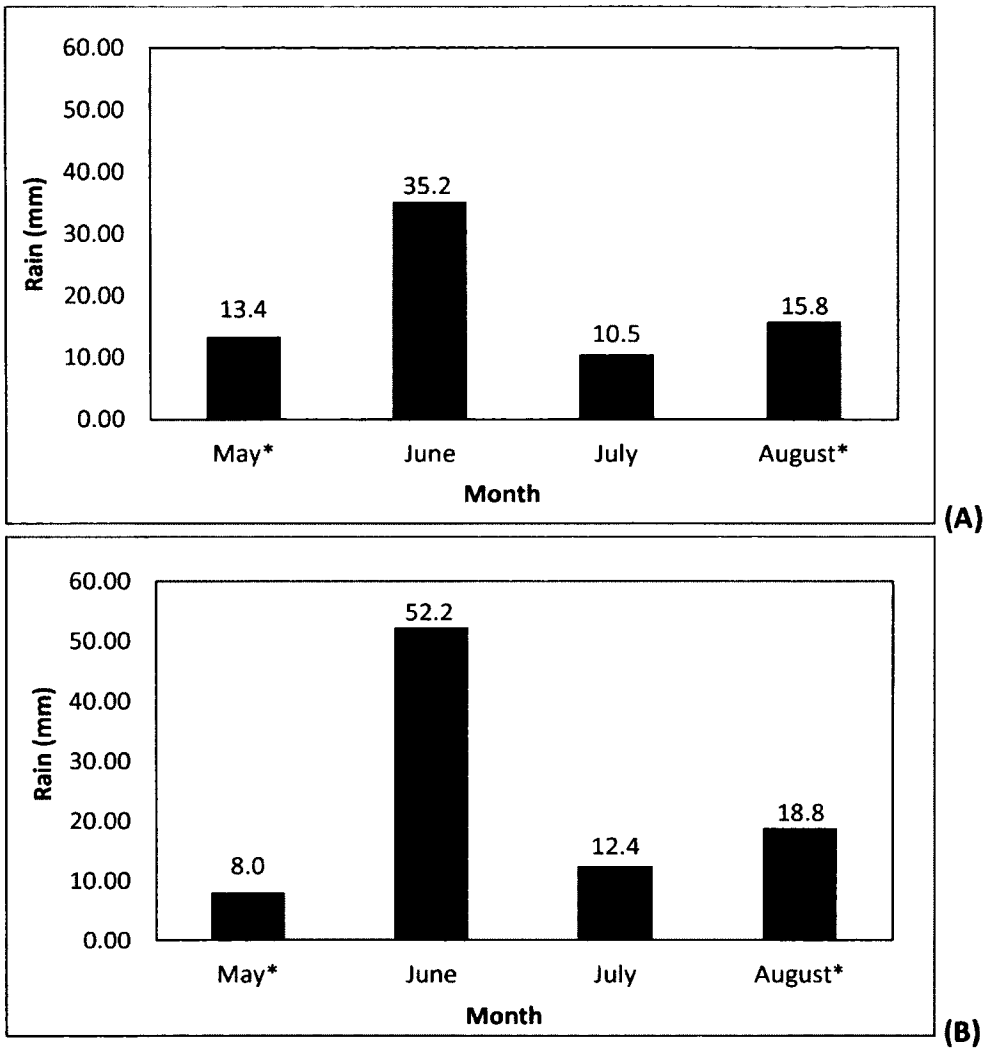


Figure 4.2 Variability in the local rainfall in the East Block, GNP for summer 2008. (A) Rainfall based on the CU weather station installed in the pasture 9, East Block of GNP. (B) Rainfall based on the UM weather station installed in the pasture 6, East Block of GNP.
Note:*To be consistent, due to data availability for both weather stations, May rainfall data is from May 15 to 31 and August rainfall data is from August 01 to August 21.

Most of the rainfall during June 2008 was received between June 2 and 12; however both the weather stations recorded variability in terms of amount and timing of rainfall events (Figure 4.3). For example, between June 10 and 12, UM recorded 29.8 mm of rain while during the

same 3 days, only 3.2 mm of rain was measured at CU. This difference in recorded rainfall is probably due to localized rainfall events over that particular weather station.

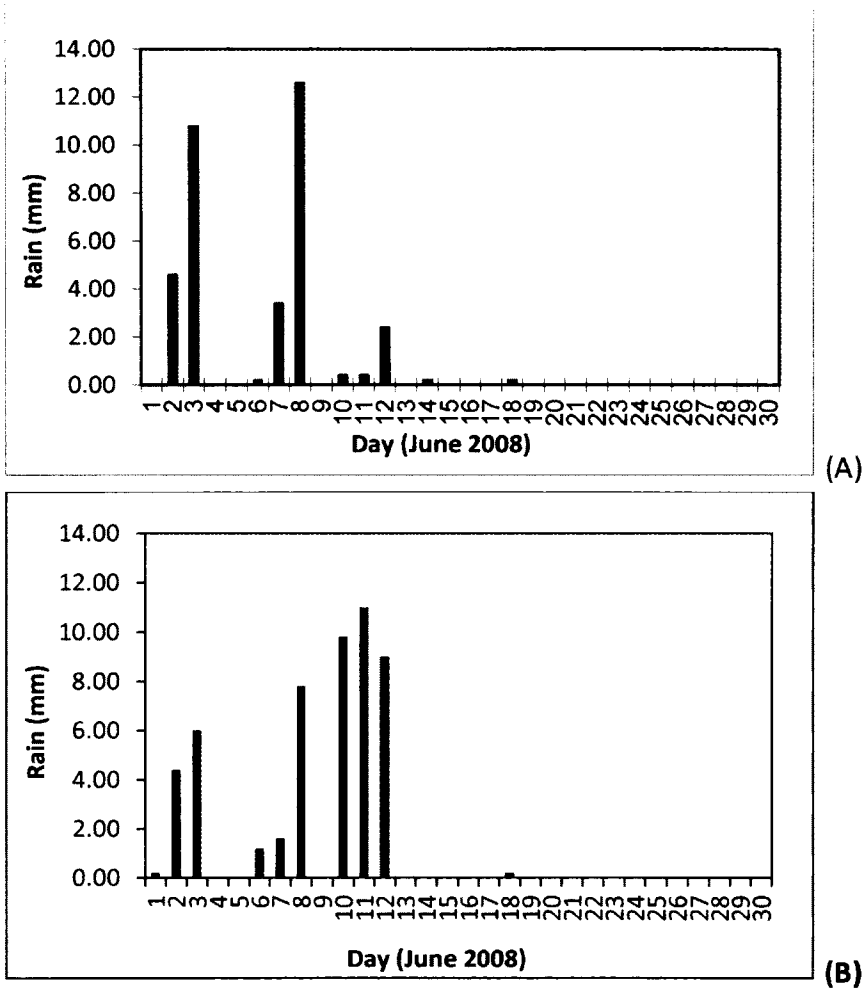


Figure 4.3 Comparison of rainfall events between the CU weather station (A) and the UM weather station (B) during June 2008.

Figure 4.4A shows the SM at 5 cm depth in response to rainfall during June 2008, while Figure 4.4B shows the air (T_{air} in °C) and soil temperature (T_{soil} in °C) at a 4 – 6 cm depth at the CU weather station. SM fluctuated between 0.13 $m^3 m^{-3}$ and 0.22 $m^3 m^{-3}$ from days 1 to 12 during June 2008 as a result of rainfall events wetting the soil and drying between events (Figure 4.4A).

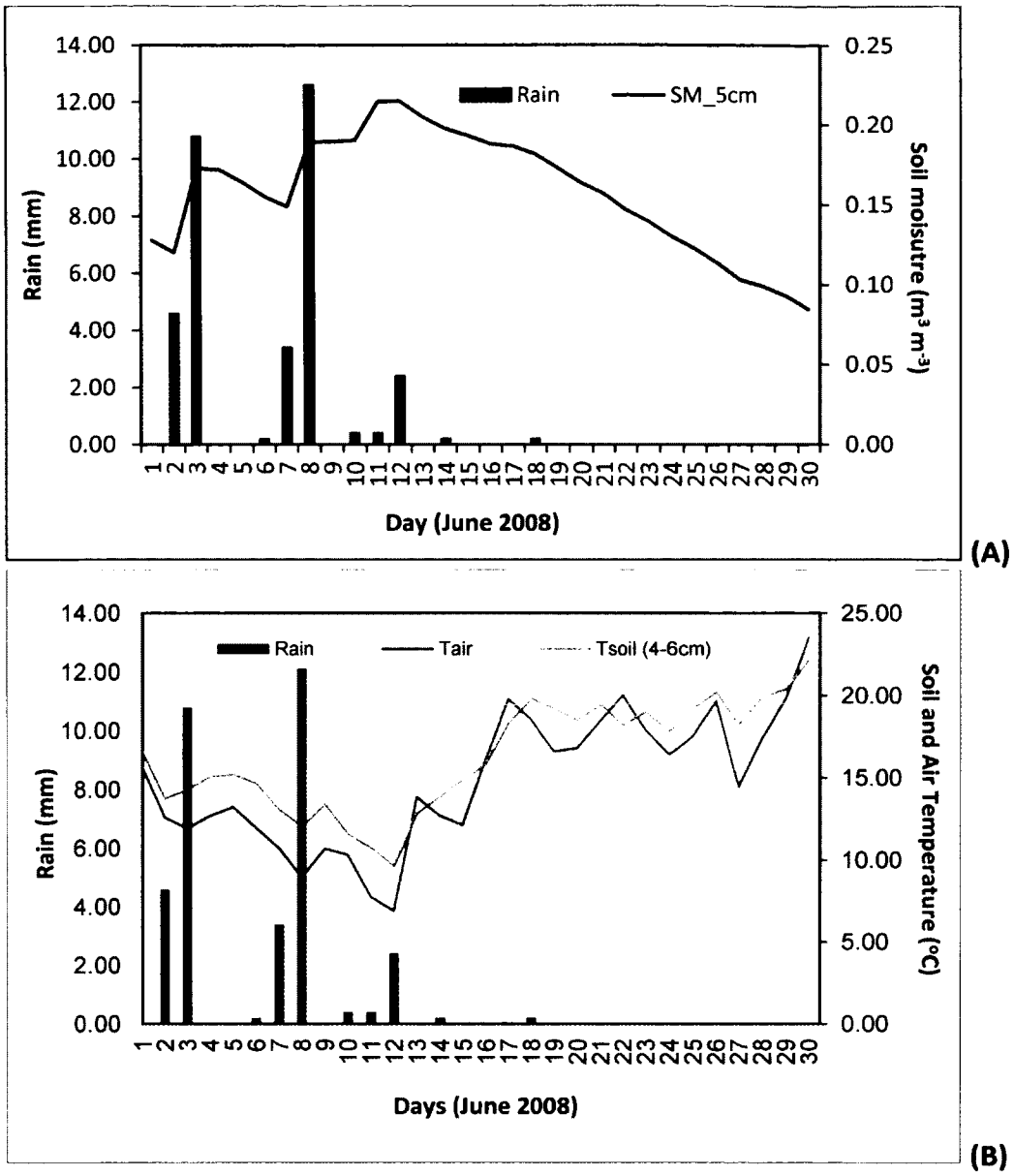


Figure 4.4 Local weather conditions during June 2008 for the East Block, GNP. (A) Soil moisture at 5 cm depth in response to rainfall. (B) Soil and air temperature conditions
Note: T_{air} = air temperature in °C, T_{soil} (4 - 6cm) = soil temperature in °C at 4 - 6cm depth and SM_5cm = Soil moisture at 5 cm depth (m^3m^{-3}). In both figures, x-axis depicts days in June.

In contrast, from day 13 onwards, SM showed constant gradual decrease and by the end of the June 2008 it had dropped to about 0.08 $m^3 m^{-3}$. This drop in SM from 0.20 to 0.08 $m^3 m^{-3}$ was due to an absence of rain (except for 0.4 mm of rain on days 14 and 18) and sunny or partly sunny weather with increasing air and soil temperatures with light to strong winds providing

the energy, atmospheric demand and turbulence needed to support high rates of evapotranspiration. Temporal variability in the SM was also examined using data from five SM probes installed in the ground at different depths: 5 cm, 10 cm, 0 - 20 cm (integrated), 26 cm and 50 cm, at the CU weather station (details provided in section 3.2.2). Figure 4.5 shows the temporal variation in SM in response to natural rainfall (sporadic to heavy rain).

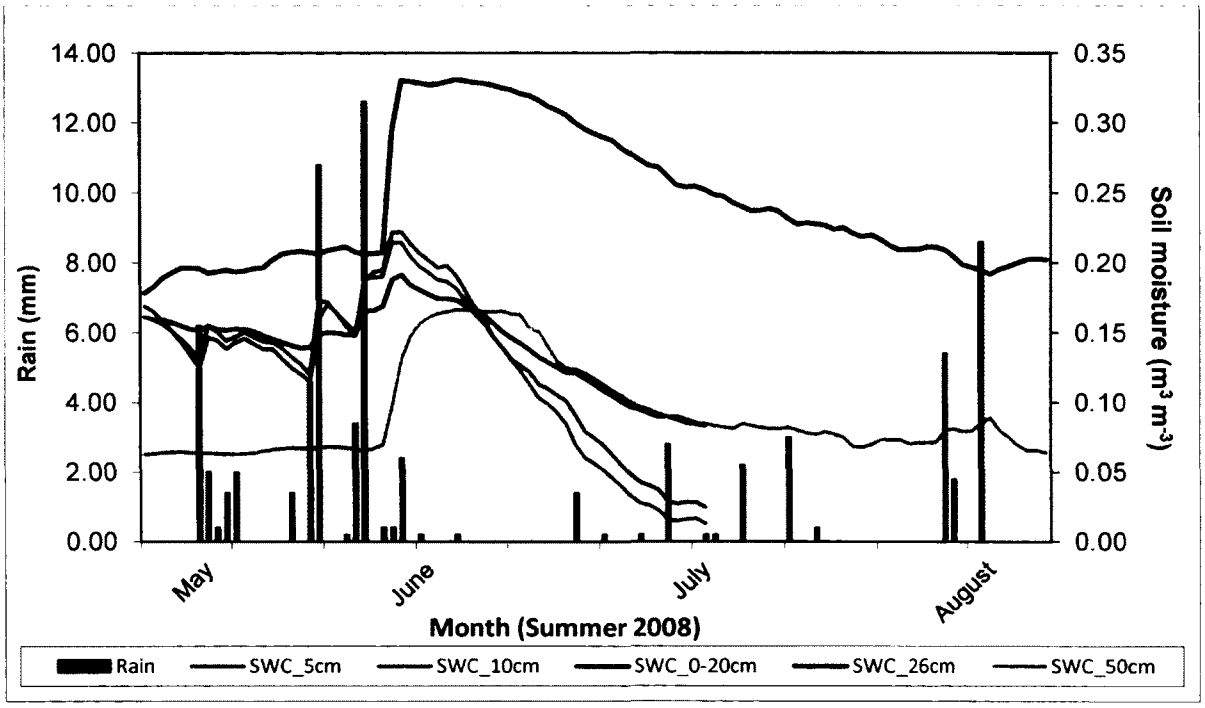


Figure 4.5 Temporal Variability in the amount of rainfall and SM at different depths, Pasture 9 East Block, GNP.
(Note: SWC = volumetric soil water content; SWC_5cm = SWC at 5 cm depth, SWC_10cm = SWC at 10 cm depth, SWC_0 - 20cm = SWC averaged at 0 – 20 cm depth, SWC_26cm = SWC at 26 cm depth and SWC_50cm = SWC at 50 cm depth.)

The permanent probes at 5 cm, 10 cm and 0 - 20 cm depths were disrupted due to animal activity in the second week of July, resulting in erroneous data, so these 3 probes were not used after July 10. No impact on probes at 26 cm and 50 cm was found at the time of final data acquisition during August 2008.

During sensor installation at the above-mentioned depths, the following characteristics were observed: (a) Soil from 0 – 30 cm was very moist and dark brown in color. (b) In general depths below 30 cm were rocky and dry. (c) A white layer of soil was also observed between 30 cm and 45 cm. In the absence of any chemical analyses, it is inconclusive to determine whether this was a salt layer (NaCl, $NaSO_4$, or $CaSO_4$) or a layer of calcium carbonate ($CaCO_3$). However, GNP is the Brown Soil Zone, so it can be assumed that the white layer is $CaCO_3$, which typically occurs within the upper 30 – 50 cm of the surface in the mid-slope landscape positions (Brierley, T. 2012, personal communication). (d) Soil was slightly moist at 45 – 50 cm depth, whereas 50 cm to 65 cm depths showed very dry soil. This suggested that most of the rainfall during 2008 growing season was used for recharging at this depth. However, the whitish layer acted as barrier (particle size discontinuity) for the downward flow of the SM to 50 cm depth. The upper 30 – 40 cm layers needed to be saturated before water could move beyond it. During sensor installation, roots were also observed at 60 cm depth. Since many rangeland species have deep fibrous root systems, it is entirely possible that this also contributed to the depletion of deep SM. All this explains the low amount of SM at 50 cm depth compared to 26 cm depth. Overall, a trend in SM with respect to factors such as precipitation is evident at 50 cm depth, but it needs further investigation.

The 5 and 10 cm depths showed similar SM pattern. These shallow layers show more rapid response to rainfall events due to high rates of surface evapotranspiration caused by variable soil and air temperatures, thus they dry/drain slightly faster than the 0 – 20 cm integrated volume.

T_{soil} at 4 – 6 cm depth showed a significant negative correlation with SM at 5 cm depth (R^2 = 0.61, p = 0.001) (Appendix 1, Figure A1.4). T_{air} showed a significant negative correlation with SM at 5cm depth (R^2 = 0.23, p = 0.043) and significant positive correlation with T_{soil} at 4 – 6 cm (R^2 = 0.74, p = 0.000) (Appendix 1, Figure A1.5 and Figure A1.6). When air temperatures increased, the surface soil temperatures also increased, whereas SM at 5 cm depth decreased.

Overall, seasonal changes in the SM at all depths were apparent. As expected, an increase and decreases in the SM corresponded to the higher and lower rainfall periods, and SM generally peaked after rainfall events. For example, SM at all depths reached a peak following a heavy rain event on 08 June 2008 and decreased thereafter. Although several small rain events occurred between 09 June and 09 August, they did not interrupt the trend in SM at 26 and 50 cm depths.

Significant correlation was also found between daily rainfall and SM (R = 0.83; $p < 0.05$). This correlation is based on the daily rainfall, and volumetric SM data measured at 12 cm depth with the Hydrosense probe during June 2008 from the plot located in pasture P9 near the weather station. SM also showed a lag effect following a rain event (Figure 4.5).

4.2.1.2 Effect of Slope Location and Time

Spatial variability in SM in pastures P1 and P6 is presented in Figure 4.6, showing that slope position has a significant effect on SM ($F(1, 743) = 51.55$, $p < 0.0001$, N = 744) tested using a mixed effects model with pasture as random effect. In particular, upslope plot in P1 (or P1U) showed higher SM compared to midslope and downslope plots. During the field measurements, accumulation of salt on the surface was observed in P1U. Since salinity in the

soil can affect the instrument measurements; this probably explains the high amount of variability in the P1U plot (Thompson *et al.* 2007). Also, although P1U was upslope according to the plot location methods used in this study, it was also in a local hollow with high surrounding contributing area, likely leading to high relative soil moisture.

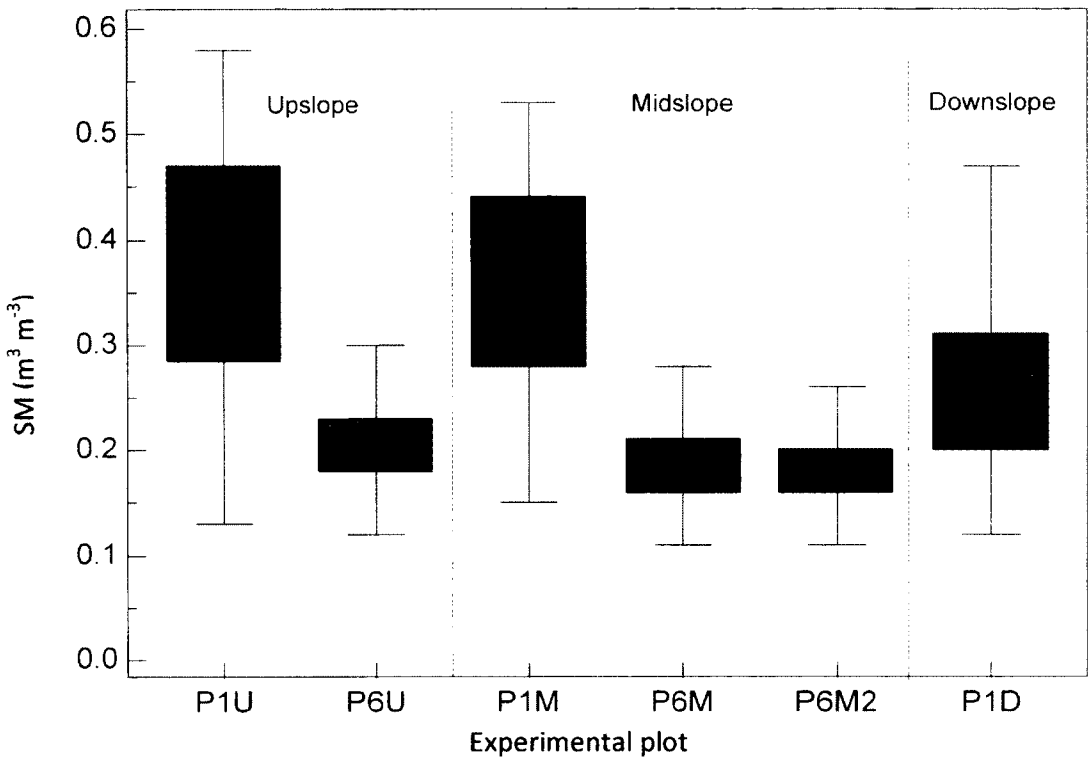


Figure 4.6 Boxplots showing the variability in SM measured on 22 May 2008 from upslope (U), midslope (M and M2) and downslope (D) plots in pasture P1 and P6.
Note: Black bars represent the median, and length of the box the inter-quartile range, i.e. the 25th and 75th quartile. Whiskers show the largest and lowest extremes if there are no outliers. Pasture 6 had two midslope plots (P6M and P6M2). Total no. of observations in each plot = 124.

May 22 SM measurements were selected to show the spatial variability as this was the only time when SM measurements were taken from all the plots located in two different pastures in a same day. Despite measurements being taken on the same day (May 22), SM in pastures P1 and P6 was significantly different overall ($F(2, 742) = 49.35$, $p < 0.0001$, $N = 744$). Overall, P1 plots show more variation compared to P6 plots (Figure 4.6). Since soil texture is highly

variable in pastures P1 and P6 (see Table 4.1), it is likely contributing to variability in SM in the two pastures.

Figure 4.7 shows the temporal variability in SM between all the pastures (P1, P6, P8 and P9) and within each respective pasture using the 3 to 4 measurements made when all plots experienced ungrazed conditions. Both slope location ($F(2, 5537) = 147.63$, $p < 0.0001$, $N = 5580$) and time (date) ($F(13, 5537) = 450.77$, $p < 0.0001$, $N = 5580$) significantly influenced the spatio-temporal variability in SM as tested using mixed effect model with statistically significant differences in the mean SM values based on dates and slope locations ($p < 0.05$).

There was no significant difference in mean SM in pasture 9 between May 16 and 25 (0.23 and 0.24 $m^3 m^{-3}$, respectively) ($p = 0.097$, Tukey's HSD). Overall, the difference in mean SM values over time among pastures is likely due to variability in local weather conditions before and on the date of SM data acquisition, which is confirmed by significant interaction between slope location and time ($F(26, 5537) = 21.13$, $p < 0.0001$, $N = 5580$). For example, during June, compared to other pastures, the mean SM values were higher in P1 plots (> 0.38 m^3m^{-3}, see Figure 4.7). The accumulated amount of rainfall from June 10 to 12 (29.8 mm) most likely contributed to high SM values.

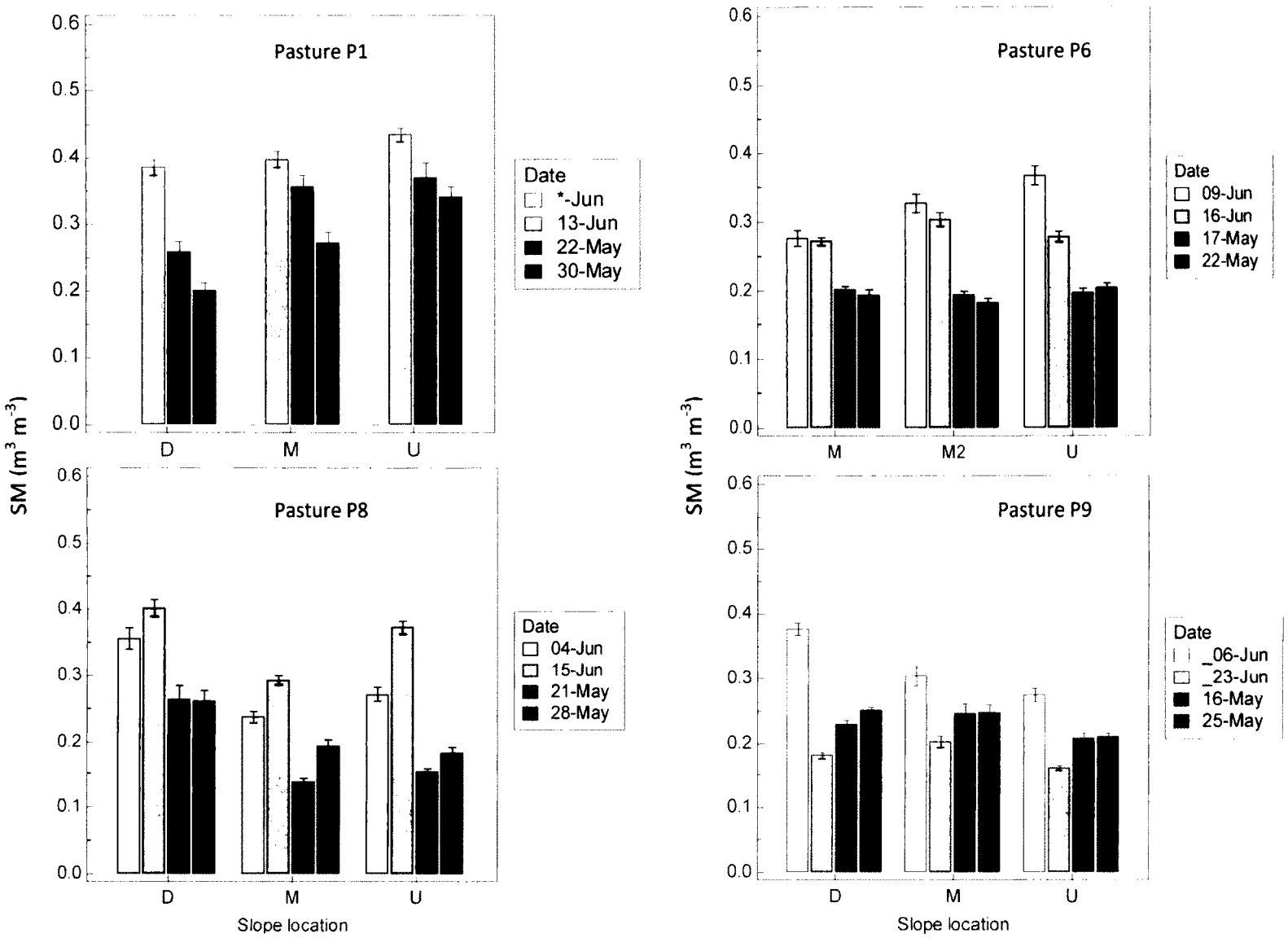


Figure 4.7 Temporal variability in SM between pastures P1, P6, P8 and P9 based on slope location. Here U = upslope, M and M2 = midslope and D = downslope. Error bars represent 95% CI for mean. In pasture P1, *-Jun = No data for early June.

The spatio-temporal variability of SM was assessed by semivariogram analysis of the upslope (U), midslope (M) and downslope (D) plots in pastures 1, 6, 8 and 9. Figure 4.8 present the examples for May and June semivariograms from different pastures graphically where symbols are the experimental semivariances and the solid lines show the fitted exponential model. Summary statistics describing the modelled semivariogram for all cases are presented in Appendix 1. Sill values were highly variable ranging between 0.014 and 0.204 between all the experimental plots during May and June. Since SM is affected by soil type, high semivariance values might be due to large differences in the SM between the two closest observation values.

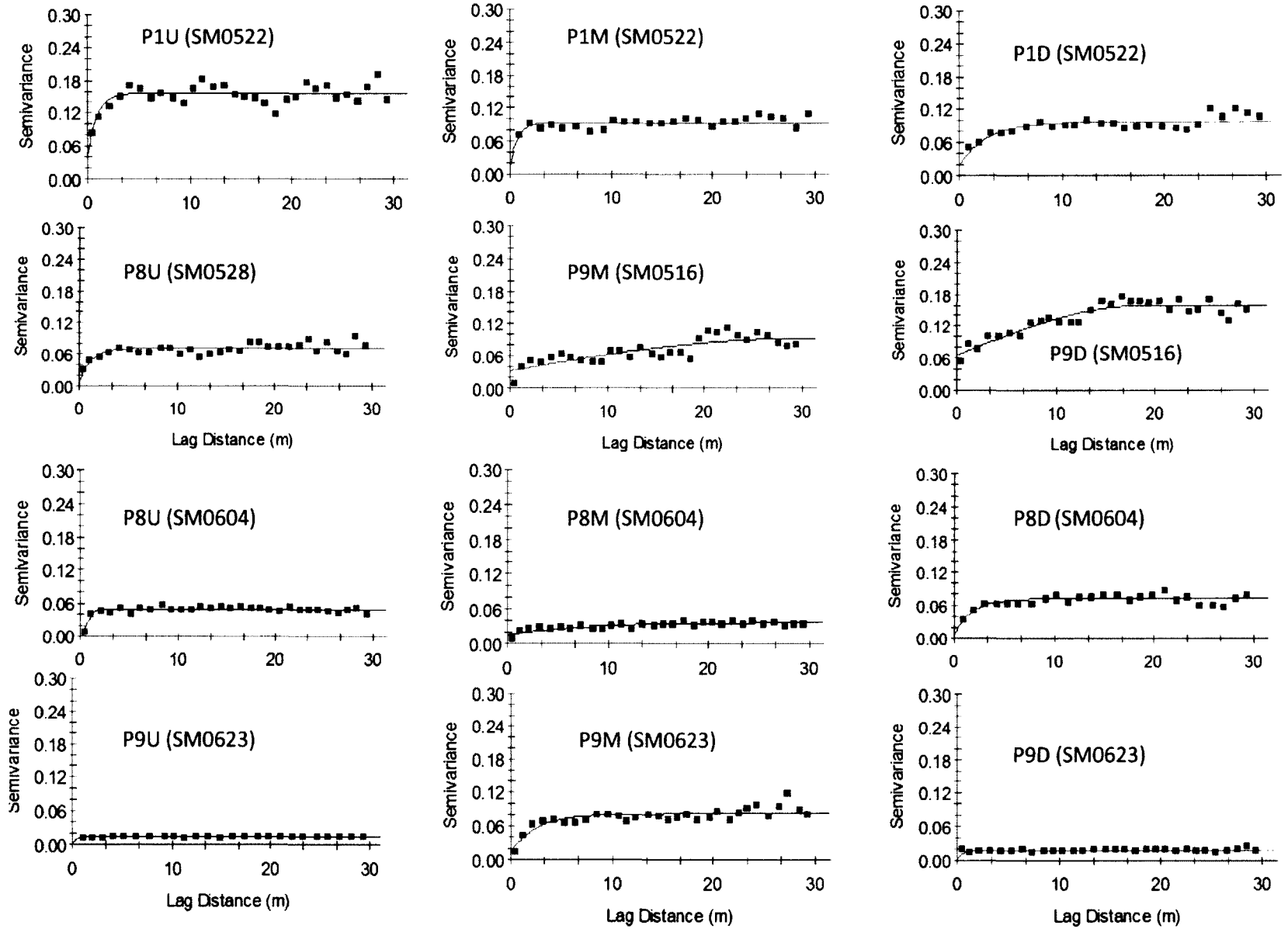


Figure 4.8 Example semivariograms for May and June ALB based on ungrazed conditions. Note: SM0522 = SM on May 22; SM0528 = SM on May 28; SM0604 = SM on June 04; SM0623 = SM on June 23. All soil moisture readings were taken at 12 cm depth under ungrazed conditions.

Small nugget effects (C_0) were also present in most of the experimental plots, suggesting some amount of random variation within the dataset. This can be attributed to measurement errors or variation at a scale smaller than the sample size (or both) (Kitanidis 1997). The exception was P9U, which showed a pure nugget effect (i.e. sill = nugget) for the SM measured on June 06. Thus, no spatially correlated variation in P9U was detected, likely attributable to the sampling interval being greater than the scale of spatial variation in the site.

Table 4.2 shows the averaged semivariogram variables for May and June SM from all the pasture plots. The ranges of influence (A_0) for both May and June SM semivariograms were highly variable in all the plots. In May, midslope plots generally had high SM ranges (between 15 and 38 m) compared to the upslope (3 to ~9 m) and downslope (~2.7 to ~19 m) plots. A notable exception was P1M, which showed a small range both on May 22 (2.91 m) and May 30 (3.54 m). In June, SM range was again highest, but by a narrower margin, followed by downslope and upslope plots.

Table 4.2 Summarized May and June 2008 SM semivariogram results

Month	Slope Location	Avg. Sill ($C+C_0$)	Avg. MSH	Avg. Range (A_0) (m)	Avg. SDR (%)	Avg. CR
May	Upslope (U)	0.06	0.72	5.53	23.49	0.18
	Midslope (M, M-2)	0.07	0.64	23.02	36.49	0.26
	Downslope (D)	0.10	0.79	9.71	23.06	0.18
June	Upslope (U)	0.03	0.76	6.77	28.25	0.17
	Midslope (M, M-2)	0.04	0.78	8.66	21.84	0.17
	Downslope (D)	0.04	0.87	6.99	13.37	0.11

Magnitude of spatial heterogeneity (MSH) was also variable between all the pasture experimental plots both spatially and temporally. For example, on May 22, P6U showed 51% heterogeneity in SM, whereas P1U showed 88% heterogeneity in SM. The spatial variability is

likely due to soil characteristics. P1U had more bare areas compared to other experimental plots. On May 22 and May 30, it was observed that some bare areas in P1U were very dry at the surface with cracks, causing difficulty in inserting the SM probes. In addition, the soil was very salty as was noticed based on the soil color (white) and had high amount of stoniness at certain points. All this likely contributed to the SM variation in this plot.

Similarly, P9U and P9M plots were characterized with fine sandy loam to sandy clay loam soil as based on the soil survey (Saskatchewan Institute of Pedology 1992). Both soil textural types have high sand content compared to silt and clay, resulting in low field capacity (between 0.11 and 0.27 $m^3 m^{-3}$). The Hydrosense probe measurements taken on June 6 in P9U and P9M plots showed low volumetric water content (0.17 – 0.24 $m^3 m^{-3}$) at certain sample points (such as 0 – 2 m in transect C of P9U and 18 – 20 m in transect A of P9M) after a heavy rainfall event on June 2 and 3 (15.4 mm rain). Therefore, it can be surmised that these measurements were taken in soil with high sand content (> 60% sand). Additionally, these plots were characterized as moderate to very stony, which could have affected the probe measurements at sample points coinciding with the stones. Within the same plots (P9U and P9M), certain sample points (such as 24 – 30 m of transect C in P9M) showed very high volumetric water content (~0.30 – 0.54 $m^3 m^{-3}$), indicating that the soil was highly saturated. One of the many possible reasons might be that the probe was placed in soil with a high clay content, which has high saturation capacity (between 0.51 and 0.57 $m^3 m^{-3}$) compared to sandy soils (between 0.36 and 0.39 $m^3 m^{-3}$).

In terms of temporal variability, P9M showed more heterogeneity in SM on May 16 (75%) compared to May 25 (55%). However, P9U and P9D plots showed less heterogeneity (59 and 69%, respectively) on May 16 compared to ~89% on May 25.

On average, downslope plots in all the pastures showed more heterogeneity (79% in May and 87% in June) compared to upslope (72% in May and 76% in June) and midslope (64% in May and 78% in June) plots (Table 4.2). Both sill ($F(1, 23) = 9.79$, $p = 0.005$, $N = 24$) and MSH ($F(1, 23) = 9.38$, $p = 0.006$, $N = 24$) were significantly affected by mean SM; however range (A_0) was not significantly affected by the mean SM ($F(1, 23) = 3.69$, $p = 0.068$, $N = 24$). Higher mean SM led to increases in the semivariograms' sill values (Figure 4.9). Overall, there was no relationship between mean SM and patch size (corresponds to the spatial range of the patches from semivariograms), but higher mean SM measurements were associated with higher variability.

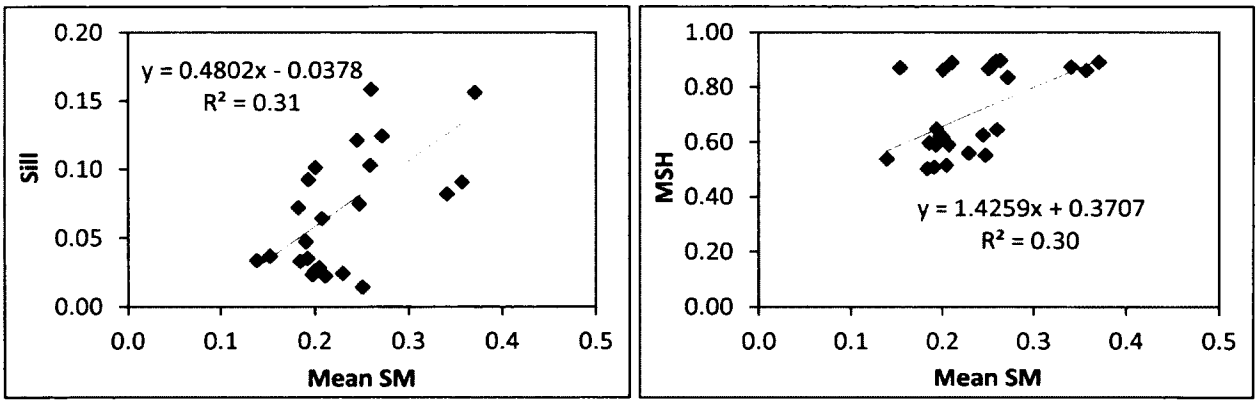


Figure 4.9 Mean SM and semivariogram parameters, Sill and MSH.

Spatial dependence ratio (SDR) (%) was calculated and interpreted based on Cambardella *et al.* (1994), where if SDR is ≤ 25%, the variable is considered strongly spatially dependent; if SDR is between 25 and 75%, the variable is considered moderately spatially dependent, and if SDR is

>75%, the variable is considered weakly spatially dependent. On average, semivariograms for downslope plots indicated strong spatial dependence for SM at short ranges (6 – 9 m). In comparison, upslope and midslope plots showed both strong and moderate spatial dependence at short - long distances. For example, P8U on May 21 showed strong spatial dependency for SM at a short range of 4.77 m compared to P8M, which showed moderate spatial dependency at a long range of 26.10 m. ANOVA test results showed no significant effect of slope location on SDR% over time ($F(2, 42) = 0.589$, $p = 0.56$, $N = 45$), but range was significantly affected by slope location ($F(2, 42) = 3.442$, $p = 0.04$, $N = 45$). A significant correlation was found between range and SDR% ($R = 0.770$, $p < 0.0001$, $N = 45$) where short ranges showed strong spatial dependence and long ranges showed moderate spatial dependency. This was further confirmed by ANOVA test which also showed significant effect of range on SDR% ($F(1, 43) = 62.69$, $p < 0.001$, $N = 45$). The strong to moderate spatial dependency may be controlled by intrinsic variations in the soil characteristics such as texture.

Local weather conditions were also highly variable during the field season in summer 2008, likely influencing the range of SM semivariograms in addition to other factors (Figure 4.2 and Figure 4.3). For example, P1 SM measurements on June 13 were preceded by 3 rainfall events that took place between June 6 and 8 (10.6 mm) and 3 rainfall events that took place between June 10 and 12 (total rainfall, 29.8 mm). As a result of these rainfall events, soils in P1 plots were heavily saturated resulting in very high mean SM values. Both P1M and P1D also showed puddles at certain points likely due to low infiltration rates, and thus these points were excluded from the analyses.

4.2.2 Aboveground live plant biomass

4.2.2.1 Impact of local Intra-seasonal weather conditions

Daily ALB data from the controlled plot placed in pasture 9 near the CU weather station, and the daily weather data collected from the CU weather station were used to determine the impact of local intra-seasonal weather conditions on ALB. Temporal variations in ALB showed a strong and significant correlation with rainfall ($R = 0.842$, $p < 0.05$, $N = 30$). No significant correlation was found between ALB and SM measured at 12 cm depth ($R = 0.197$, $p = 0.349$, $N = 124$; Appendix 1, Figure A1.7). The relatively low correlation indicates that ALB does not entirely depend on the water content at 12 cm; water from depths greater than 12 cm is also obtained by plants for growth. This is further confirmed from visual observation of fine roots up to a depth of 60 cm during installation of SM sensors at different depths near the weather station in P9.

4.2.2.3 Spatio-temporal variability in ALB between pastures and within pastures

Spatial variability in ALB in pasture P1 and P8 is presented in Figure 4.10. Despite measurements being taken on close to the same day (example, May 27 upslope and midslope plots; and May 29 downslope plots), plots in pastures P1 and P8 show considerable variability in ALB. This variability could be attributed to vegetation composition within each plot.

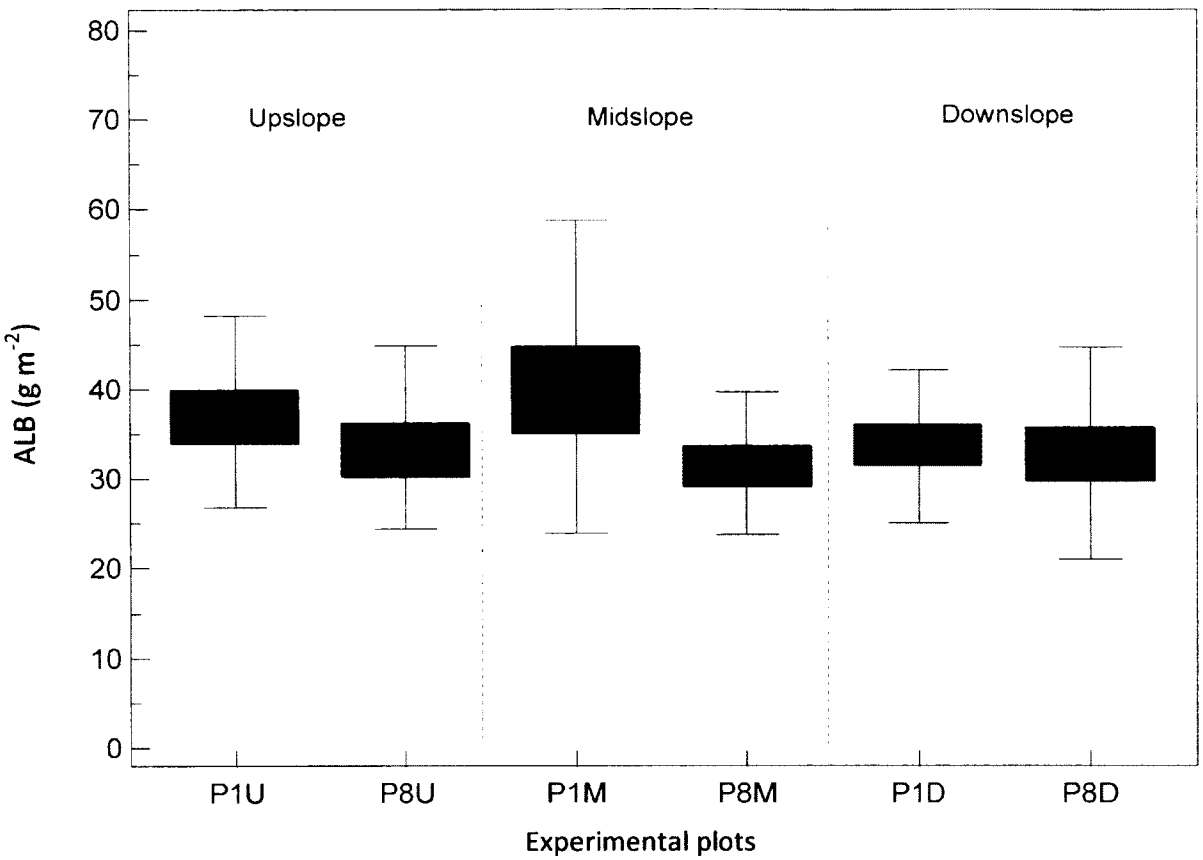


Figure 4.10 Boxplots showing the variability in ALB measured during May 2008 from plots in pasture P1 and P8.
Note: Black bars represent the median, and length of the box the inter-quartile range, i.e. the 25th and 75th quartile. Whiskers show the largest and lowest extremes if there are no outliers. Total no. of observations in each plot = 124.

Figure 4.11 shows the temporal variability in ALB between all the pastures (P1, P6, P8 and P9) and experimental plots in each pasture. Mean ALB is variable between pastures and within pastures. In general, mean ALB for June is higher in pastures P1 and P9 compared to P6 and P8. Also, the differences with respect to slope position are generally much greater in June than in May.

Both slope location ($F(2, 2968) = 17.15$, $p < 0.001$, $N = 2970$) and time (month) ($F(1, 2969) = 2677.25$, $p < 0.001$, $N = 2970$) significantly influenced the spatio-temporal variability in ALB for each pasture (P1, P6, P8 and P9). Subsequent Tukey's HSD, a post-hoc test, indicated that the

mean ALB values were significantly different between the slope locations for most of the pastures (example of within pasture variation, mean ALB in P1 plots during late June: upslope plots = 59.05 g m^{-2}; midslope plots = 82.64 g m^{-2}; downslope plots = 71.92 g m^{-2}; $p < 0.05$, N = 372). In terms of between pasture variations, mean ALB mostly showed significant differences as a result of slope location. Notable exceptions were P6U and P6M-2 plots, where Tukey's HSD test indicated no significant difference in the mean ALB (24.13 g m^{-2} and 23.58 g m^{-2} respectively); and P1D and P8D with mean ALB of 33.86 g m^{-2} and 33.11 g m^{-2} respectively for May data ($p > 0.05$).

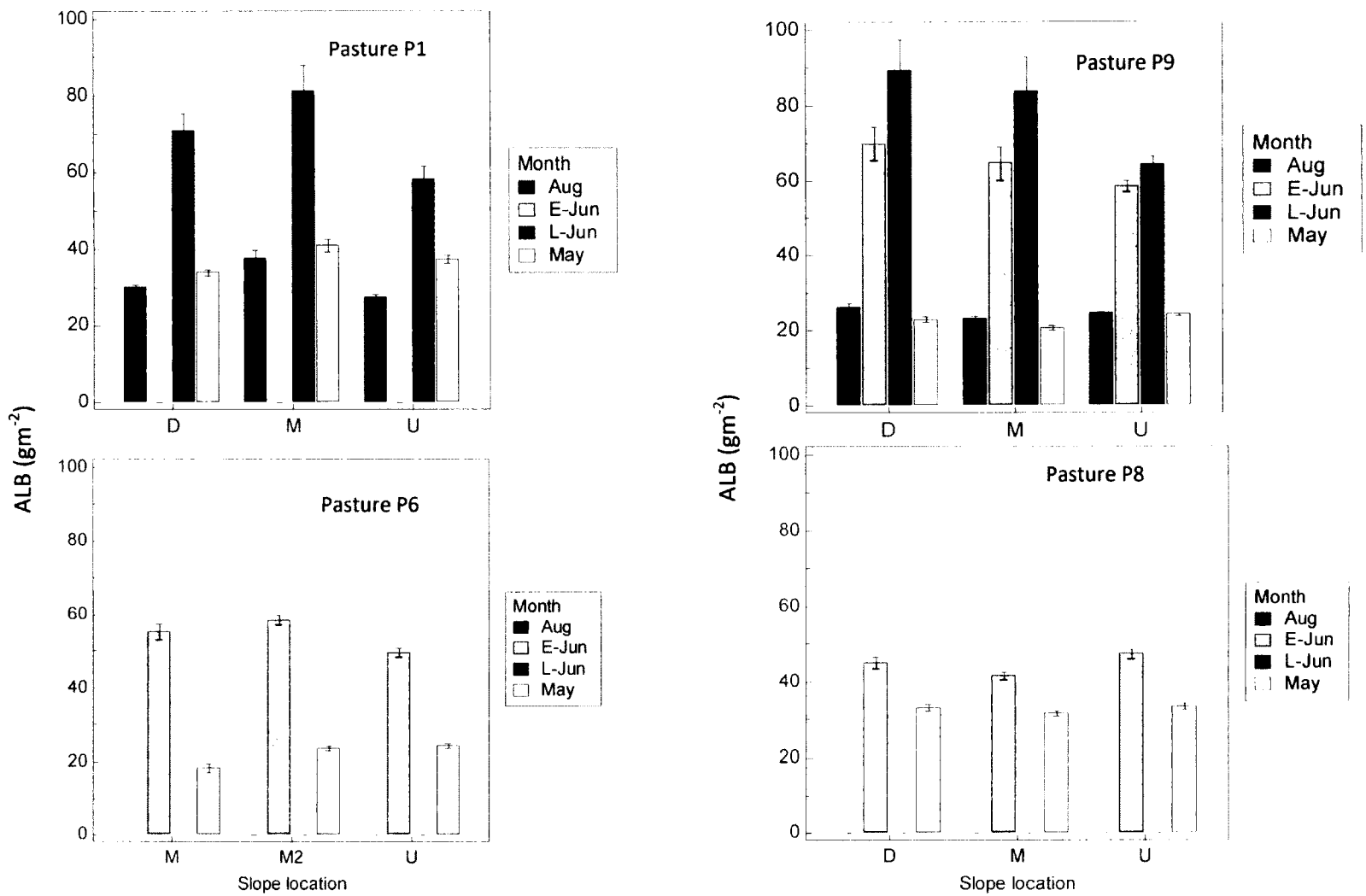


Figure 4.11 Temporal Variability in ALB between pastures P1, P9, P6 and P8 based on slope location.

Note: U = upslope, M and M2 = midslope and D = downslope. Error bars represent 95% CI for mean. E-Jun = Early June; L-Jun = Late June; No early June data available for P1; and no late June data available for P6 and P8; No August data available for P6 and P8 plots. P1 data: 31 May, 27June, 23 Aug; P9 data: 19May, 23 and 28 June, 20 Aug; P6 data: 17May and 16June; P8 data: 27May and 15June.

Figure 4.12 summarizes the spatio-temporal variability of ALB in all measured pastures P1, P6, P8 and P9 plots. Very low ALB values (between 8 and 9 g m^{-2}) were recorded on May 17 in transect C of P6M, thus impacting the semivariograms and resulting in a very low semivariance at short distances (0 to 10 m, see Figure 4.12) followed by a sharp rise in the variance. Overall sill and range values were highly variable between the experimental plots. For May ALB data, P6 and P9 plots showed high range values varying between 22 and 42 m beyond which there is no spatial auto-correlation compared to plots in pastures P1 and P8 showing range less than 20 m. ANOVA test results showed no significant effect of slope location on the semivariogram range ($F(2, 24) = 1.67$, $p = 0.21$, $N = 26$).

Both upslope and downslope plots located in P1 and P8 showed higher heterogeneity in ALB compared to upslope and downslope plots located in P6 and P9 (Appendix 1). In comparison, the midslope plots showed similar heterogeneity in ALB except P6M, which was highly heterogeneous (MSH = 0.99). P6M had especially high variability in biomass values; during ALB data acquisition it was observed that transect C in P6M had very sparse vegetation with ALB ranging between 8 and 9 g m^{-2} compared to transect A, B and D with ALB ranging between 19 and 30 g m^{-2}.

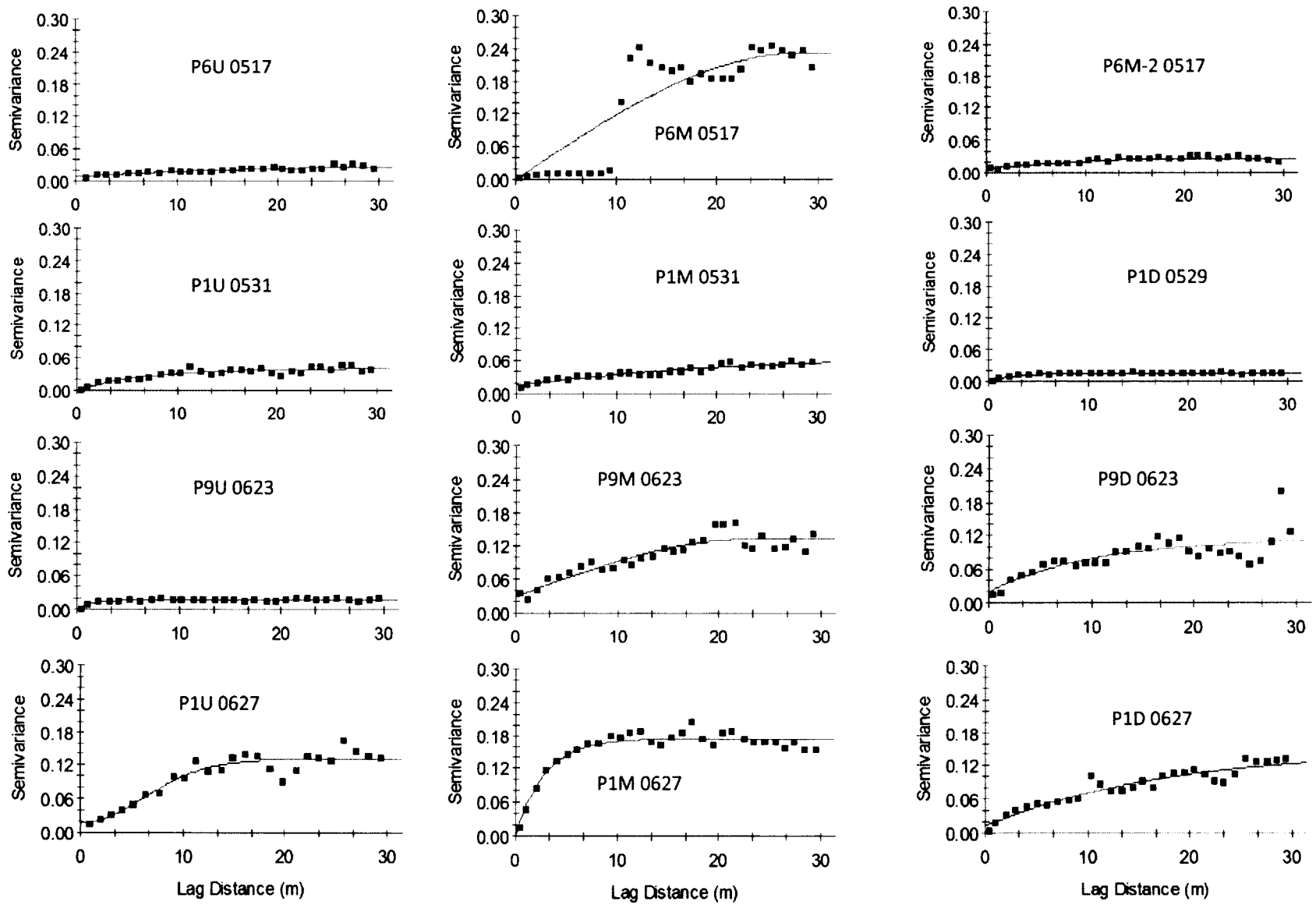


Figure 4.12 Example semivariograms for May and June ALB based on ungrazed conditions. Note: 0517 = ALB on May 17; 0529 = ALB on May 29; 0531 = ALB on May 31; 0623 = ALB on June 23; 0627 = ALB on June 27.

Table 4.3 shows the averaged semivariogram variables for May and June ALB from all the pasture plots. The range of influence for both May and June ALB semivariograms was highly variable in all the pasture plots. On average during May, midslope plots had the highest range whereas in June the downslope plots had the highest averaged range (Table 4.3). Therefore, no clear pattern with respect to patch size of ALB could be identified.

Table 4.3 Summarized May and June 2008 ALB semivariogram results

Month	Slope Location	Avg. Sill ($C+C_0$)	Avg. MSH	Avg. Range (A_0) (m)	Avg. SDR	Avg. CR
May	Upslope (U)	0.024	0.79	19.99	21.17	0.17
	Midslope (M, M-2)	0.076	0.78	29.17	20.94	0.17
	Downslope (D)	0.031	0.81	22.20	18.53	0.15
June	Upslope (U)	0.085	0.67	18.56	32.93	0.24
	Midslope (M, M-2)	0.115	0.81	17.52	18.51	0.15
	Downslope (D)	0.151	0.90	29.06	10.42	0.09

MSH was also variable between all the pasture experimental plots both spatially and temporally. In general, ALB heterogeneity increased from upslope to downslope during May. However, in P8 and P1, upslope plots showed the highest heterogeneity in ALB followed by downslope and midslope plots. Heterogeneity patterns were clearer in June. For all the experimental plots, heterogeneity in ALB increased from upslope to downslope plots.

Visual inspection of the vegetation at all the pasture plots indicated that it was not uniform at the field scale so it is likely that diversity in type of vegetation is causing this small-scale variation. For example, P8M also had some Shining arnica (*Arnica fulgens*) (~10%), Nuttall's yellow violet (*Viola nuttallii*) (~1%) and death camas (*Zigadenus venenosus*) (~2%), in addition to a mix of grass, shrubs and *Selaginella densa*. The transect also showed open vegetation with gaps partially occupied by *Selaginella densa* and lichens from 20 to 27 m and mix of grass and

shrub from 27 to 30 m. All this variability in type of vegetation also affected the amount of biomass in the plot because points coinciding with dense shrubs showed high ALB values in the same plot compared to areas with less grass cover. For example, P9M had thick shrubs from 18 – 20 m in transect B thus contributing to high ALB values ranging between 225.5 g m^{-2} and 325.8 g m^{-2}. Within the same plot, some points had primarily grass cover thus contributing to low ALB value, for example, the 30 m point in transect A had an ALB value of 35.1 g m^{-2}.

In addition to spatial variability, experimental plots in P1, P6, P8 and P9 also showed temporal heterogeneity in ALB. For example, on May 19, P9U showed ALB ranging from 13.6 g m^{-2} to 30.1 g m^{-2}. ALB in the same plot ranged from 36.4 g m^{-2} to 184.0 g m^{-2} on June 23 and from 33.1 g m^{-2} to 325.5 g m^{-2} on June 28. This temporal variability is a result of the state of vegetation development during the growing season.

Sill was significantly correlated with mean ALB ($F(1, 14) = 50.79$, $p < 0.001$, $N = 15$), however no significant correlation was found between the MSH and mean ALB ($F(1, 14) = 0.17$, $p = 0.69$, $N = 15$). The sill (variance) of the semivariogram follows the trend of mean ALB values with higher ALB leading to an increase in sill (Figure 4.13). Similar to MSH, range also showed no significant correlation with the averaged ALB ($F(1, 14) = 1.718$, $p = 0.213$, $N = 15$).

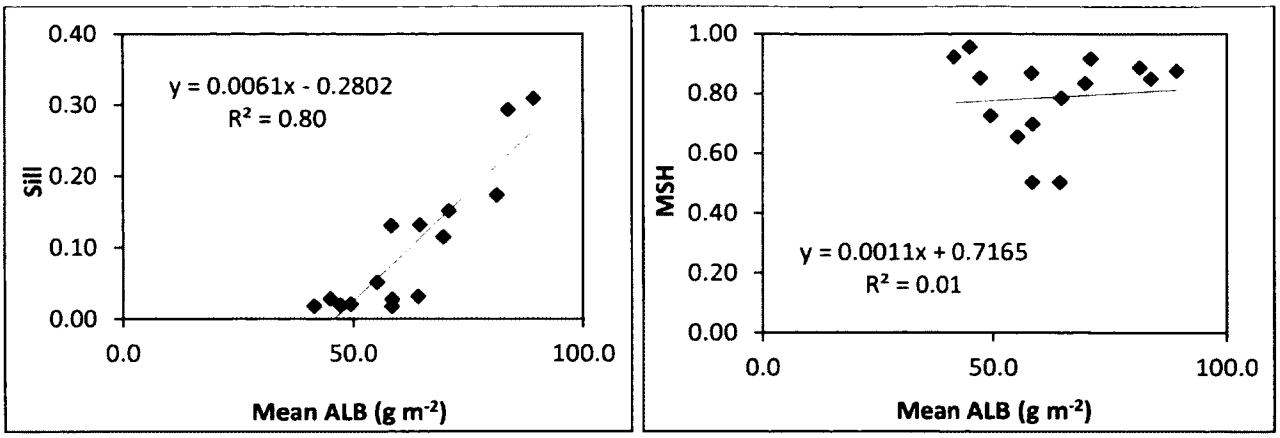


Figure 4.13 Mean ALB and semivariogram parameters, Sill and MSH.

On average during May, all slope locations showed strong spatial dependency (SDR% ≤ 25%). In comparison for June ALB, on average upslope plots showed moderate spatial dependency (SDR% between 25 % and 75 %), while both midslope and downslope plots showed strong spatial dependency (SDR% ≤ 25%). The ANOVA test results showed no significant effect of slope location on the ALB SDR% over time ($F(2, 24) = 0.070$, $p = 0.93$, $N = 27$). Also, no significant correlation was found between ALB range and SDR% ($R = 0.269$, $p = 0.174$, $N = 27$), further confirmed by ANOVA ($F(1, 25) = 1.959$, $p = 0.174$, $N = 27$).

4.2.3 Effect of Grazing on SM and ALB

This part of the field study is a part of the large-scale long-term BGE study at the East Block of GNP (see Henderson 2006 for details) following a modified Before-After-Control-Impact design ("Beyond BACI;" Underwood 1994). Grazing was introduced in mid-June 2008 in pastures P6 and P8, while P1 and P9 were used as control, therefore this section provides analysis for SM and ALB before and after treatment. From here on pasture number will be combined with their treatment code: control (C), before grazing (BG) and after grazing (AG) (for example, $P1_C$, $P6_{BG}$, $P6_{AG}$, $P8_{BG}$, $P8_{AG}$, $P9_C$).

4.2.3.1 Soil moisture and Grazing

Significant effects of grazing treatment ($F(1, 222) = 125.74$, $p < 0.001$, $N = 223$), time ($F(1, 222) = 727.06$, $p = 0.02$, $N = 223$) and their interaction ($F(1, 222) = 327.95$, $p < 0.05$, $N = 223$) on plot scale SM were detected. There were no significant pre-existing differences detected in the plot scale SM before cattle were allowed to graze ($p = 0.07$). For example, both control and before grazing pastures showed similar mean SM, 0.318 $m^3\ m^{-3}$ and 0.320 $m^3\ m^{-3}$, respectively (Figure 4.14). Tukey's HSD test showed significant drop in the mean SM values before (0.31 $m^3\ m^{-3}$) and after (0.19 $m^3\ m^{-3}$) grazing treatment (Figure 4.14).

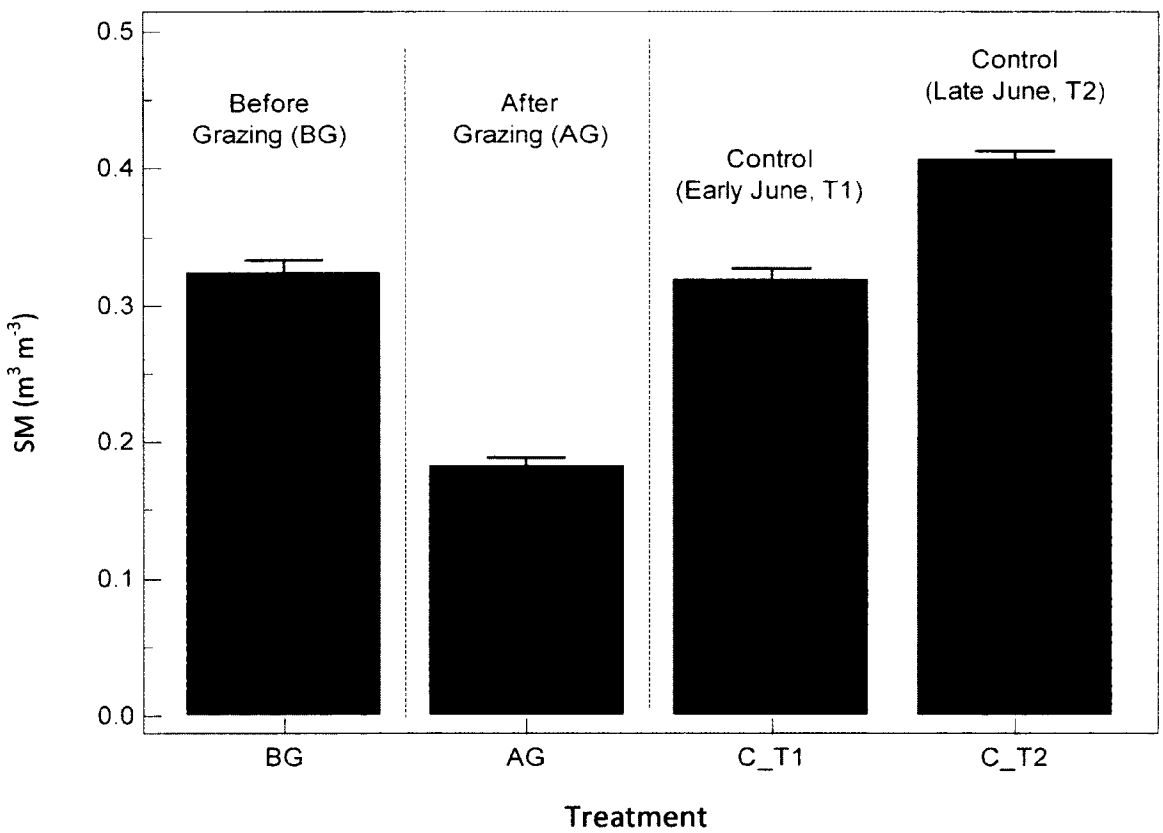
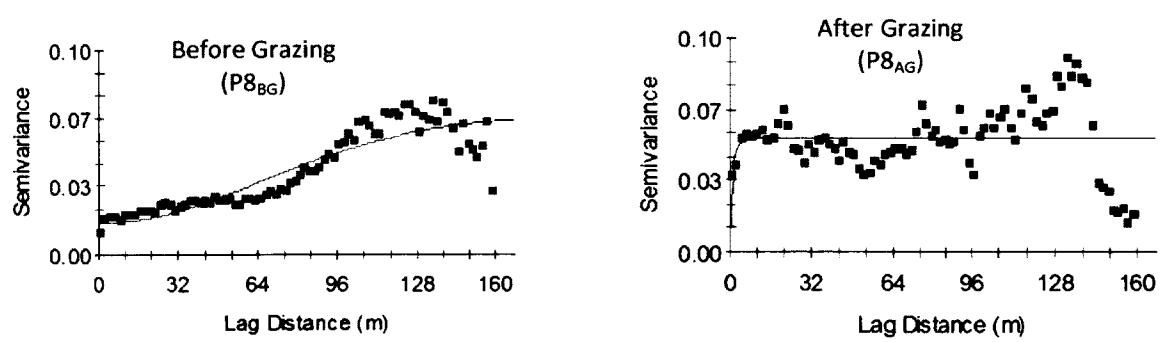


Figure 4.14 Grazing effects on SM. Error bars represent 95% CI for mean.
Note: Data is from pasture P6 and P8 before (first half of June 2008) and after grazing (second half of June 2008) period. Here AG = data from P6 and P8 after grazing; BG = data from P6 and P8 before grazing; C = data from control pasture P9. No SM data for T2 was available from control pasture P1, so it was not used in the analysis.

Semivariogram analysis results showed higher spatial heterogeneity and increase in spatial patchiness after grazing treatment than before grazing was introduced in pastures P6 and P8. For example, $P8_{AG}$ showed 89% heterogeneity in SM with a patch size of 5.4 m, whereas $P8_{BG}$ showed 81% heterogeneity in SM with a patch size of 18.4 m. No change in heterogeneity from time T1 (87.5% heterogeneous) to T2 (88% heterogeneous) was observed in control pasture $P9_c$. Similar to before grazed pastures, control pasture $P9_c$ also showed big patches (12 m).

4.2.3.2 ALB and Grazing

Significant effects of treatment ($F(1, 270) = 23.03$, $p = 0.002$, $N = 271$) and time ($F(1, 270) = 2982.35$, $p < 0.0001$, $N = 271$) on plot scale ALB were detected. ALB was lower in the $P6_{AG}$ and $P8_{AG}$ compared to before grazing treatment ($P6_{BG}$ and $P8_{BG}$) and control ($P9_c$). A significant treatment x time interaction effect ($F(1, 270) = 37.83$, $p < 0.0001$, $N = 271$) was also detected, which indicates the effect of grazing disturbance on the ALB, in addition to environmental factors and plant phenological change. There were no pre-existing differences detected in the ALB before cattle were allowed to graze (control ($P9_c$): Mean ALB 51.9 g m^{-2}; before grazing: Mean ALB 52.9 g m^{-2}; $p = 0.673$).



(Nugget = 0.015; Sill = 0.069; Range = 92.7 m; $R^2 = 0.83$; RSS = 0.0004; SH = 77.8%)

(Nugget = 0.0064; Sill = 0.053; Range = 4.2 m; $R^2 = 0.51$; RSS = 0.019; SH = 87.6%)

Figure 4.15 Semivariograms for before and after grazing treatment: An example of pasture P8 during summer 2008.

Semivariogram analysis results indicated higher spatial heterogeneity and increase in spatial patchiness after grazing treatment than before grazing activity introduced in pastures P6 and P8. For example, after cattle were introduced in P8, ALB heterogeneity increased to 87.6% from 77.8% with smaller patch size (4.2 m) than before grazing treatment (Figure 4.15). Similar to before grazed pastures, control pastures also showed bigger patch size (~40 m) with slight fluctuation in ALB heterogeneity from time T1 (78% heterogeneous) to T2 (76% heterogeneous).

4.2.4 Spatial patchiness as a result of grazing disturbance

Moran's I for SM and ALB was calculated at a 1 m lag interval to check for any spatial trends as a result of control and grazing treatment.

4.2.4.1 Soil Moisture

All the experimental plots showed some degree of positive significant autocorrelation at short distances (< 10 m). In fact, the strongest value of spatial autocorrelation (i.e. significant positive autocorrelation) was always within the first distance class supporting patchiness. It can also be inferred that with distance the spatial autocorrelation decreased and showed absence or non-detection of spatial pattern. Figure 4.16 provides an example of spatial patterns in SM at pastures $P1_C$ and $P6_{BG}$. The size of the patches (or range of influence) is indicated by the distance at which the first maximum negative autocorrelation is found. The first change of sign from positive to negative value occurred around 3 m in $P1U_C$ SMM22 correlogram; around 2 m in $P1M_C$ SMM22 correlogram and 9 m in $P1D_C$ SMM22 correlogram corresponding to the spatial range of the patches. In general, compared to $P6_{BG}$, $P1_C$ exhibited high degree of randomness

or homogeneity in SM spatial pattern with values oscillating along the zero value (i.e. absence of significant spatial autocorrelation). This could be attributed to similar SM values observed on May 22 in $P1_C$. However, significantly positive or negative values indicate some spatial heterogeneity (i.e. patchy distribution of the SM) in $P1_C$. This could be attributed to the variation in soil characteristics in $P1_C$.

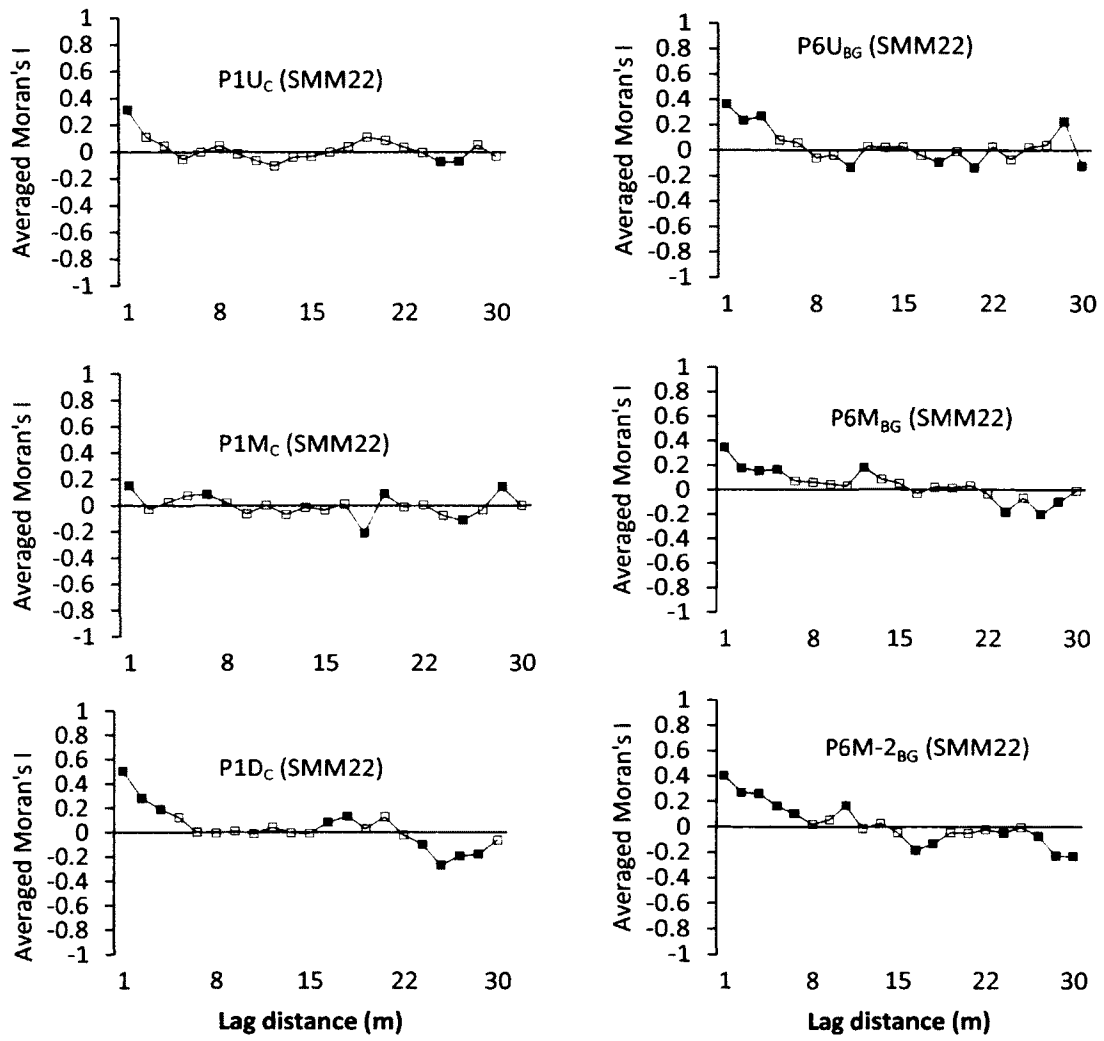


Figure 4.16 An example of spatial autocorrelation (Moran's I) in $P1_C$ and $P6_C$ for SM measured on May 22. Note: U = upslope, M = midslope and D = downslope. Solid squares indicate significant coefficient values at $\alpha = 0.05$; open squares indicate non-significant coefficient values after progressive Bonferroni correction.

Compared to $P1_C$, $P6_{BG}$ showed patchiness in SM with variable patch size (corresponds to the spatial range of the patches) and distance among patches with some degrees of randomness in the pattern. Since trampling by cattle can affect the soil structure, this might be affecting SM patterns. Overall, in both $P1_C$ and $P6_{BG}$, the correlograms did not detect any repetitive patterns except in $P1U_C$, due to variation in patch size and distance among patches.

Moran's *I* correlograms were also calculated using SM data from before and after grazing treatment from pasture 6 and 8. Overall, correlograms were globally significant; indicating that the overall spatial pattern of SM is not random so there is spatial heterogeneity in SM. In $P6_{AG}$ and $P8_{AG}$ a significant positive value of Moran's I ($p < 0.001$) was observed for the first distance class followed by significant negative and positive values for distance classes greater than 4 m, suggesting a certain degree of spatial periodicity in SM. Overall, SM patch sizes were smaller (5.4 m) during grazing activity than before (patch size 18.4 m) any grazing activity in the pastures P6 and P8, suggesting that the grazing disturbance created a more fragmented mosaic in terms of SM.

4.2.4.2 ALB

An example of auto-correlograms for ALB in pastures 6 and 8 before / after grazing is provided in Figure 4.17. Both $P6_{BG}$ and $P8_{BG}$ showed one big patch with first change of sign from positive to negative at distances smaller than 45 m. Overall, both pastures ($P6_{BG}$ and $P8_{BG}$) showed significant auto-correlation both at small and large distances, with a clear spatial pattern. In comparison, both pastures ($P6_{AG}$ and $P8_{AG}$) after the introduction of grazing disturbance

showed highly patchy spatial pattern (i.e. significant positive or negative values) with variable patch size and distance among the patches as a result of introduction of grazing disturbance.

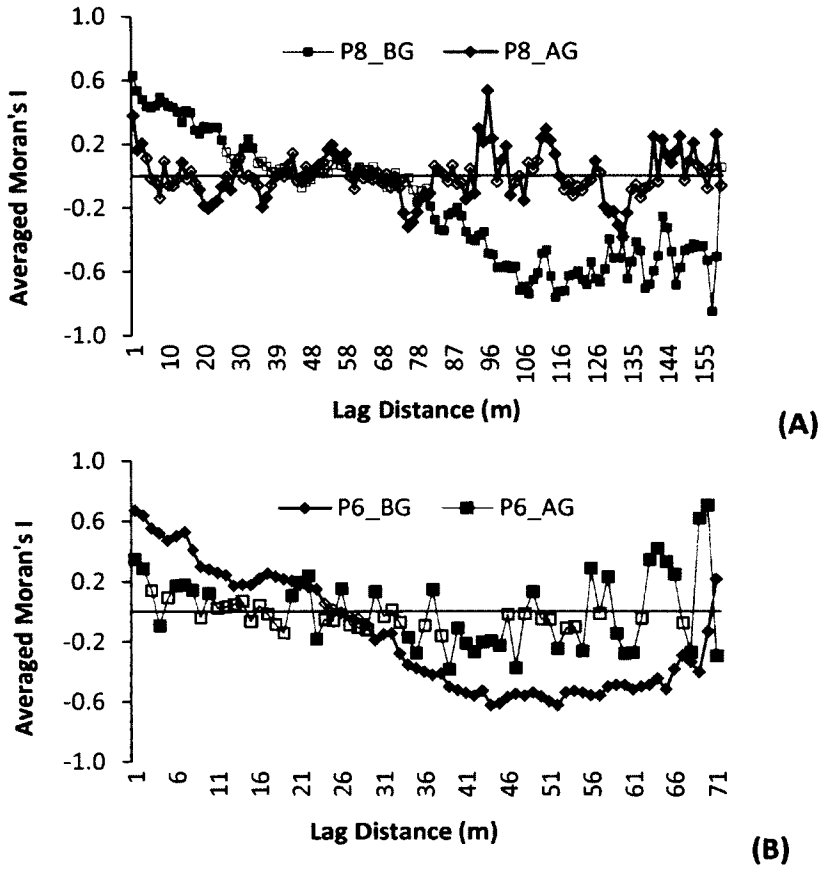


Figure 4.17 An example of spatial autocorrelation (Moran's I) of ALB in $P8_{BG}$ / $P8_{AG}$ (A) and $P6_{BG}$ / $P6_{AG}$ (B).

Note: Solid squares indicate significant coefficient values at $\alpha = 0.05$; open squares indicate non-significant coefficient values after progressive Bonferroni correction. BG = before grazing; AG = after grazing treatment. Data for P8 is based on a 180 m long transect going from upslope to downslope; whereas data for P6 is based on a 90 m long transect going from upslope to midslope.

4.3 Discussion

It was hypothesized that both ALB and SM would differ among plots and between pastures due to external factors such as weather, slope location and time. It was also predicted that grazing disturbance will significantly affect the spatial patterns in ALB and SM, and will result in more heterogeneity in ALB and SM than no grazing conditions. The following sections present

discussion of the field results and their applicability, certain limitations within the analyses, as well as recommendations for future research.

4.3.1 Spatio-temporal variability in SM and ALB

Spatio-temporal variability in SM at different depths was highly influenced by the local weather conditions. Both SM and ALB showed strong and significant correlations with rainfall. In general, SM diminished rapidly when there was a lack of rain. The results are similar to the study conducted by Fu *et al.* (2003), where rainfall together with topography, soil properties and land use contributed to the variability in the SM within 0 – 70 cm depth. A large proportion of the rainfall in arid and semiarid regions is produced in convective storms, resulting in spatial and temporal rainfall variability, one of the important factors affecting the regional productivity in arid and semi-arid regions (Noy-Meir 1973; Lauenroth 1979; Sala *et al.* 1988; Sivakumar and Hatfield 1990; Burke *et al.* 1991; Seastedt *et al.* 1994; Yang *et al.* 1998 and Truett 2003).

SM was highly variable both spatially and temporally between pastures and within the pasture plots. Since SM measurements were taken under variable weather conditions, where some measurements were taken after a series of heavy rainfall events spatially varying soil hydraulic properties can create differential infiltration rates during wet periods following rainfall, thus causing larger variation in SM. Studies such as Reynolds (1970) suggest that variability in SM should be lowest after a prolonged dry period and large immediately after a rain. This is probably because saturation after heavy rain would either lead to uniform conditions or the effect of soil pore size variations will be maximized (Reynolds 1970).

Mohanty and Skaggs (2001) suggested that at a particular point in time the SM content is highly influenced by factors such as: precipitation; slope, which affects the runoff and infiltration; and vegetation and land cover influencing evapotranspiration and deep percolation. Grayson and Western (1998) concludes that soil texture may also have a major influence on the soil water content in the semi-arid areas. This is because the relative proportion of sand, silt and clay affects the water retention capacity of a soil by determining the size and number of soil pores (Hawley *et al.* 1983 and Jacobs *et al.* 2004). Since the park's soil survey (Saskatchewan Institute of Pedology 1992) suggested high variability in soil texture in the pastures, which may have contributed to the difference in SM between and within the pastures, it is recommended that soil texture analysis should be included in future studies of SM dynamics for this study area. Vachaud *et al.* (1985) found a positive significant correlation between the soil water content and the amount of silt and clay present in the soil profile. Similarly, Cosby *et al.* 1984 also concluded that variation in soil texture contributes to SM variability.

ALB was also found to be variable both spatially and temporally, with significant differences between pastures and within the pasture plots with some exceptions (Figure 4.11). Based on personal observations the vegetation type (grass, shrubs, forbs and other) and cover (sparse, dense, bare or mixed) were highly variable within all plots, and between pastures. Ruiz-Sinoga *et al.* (2011) concluded that variability in SM also plays an important role in affecting vegetation cover in semi-arid regions. Since the amount of plant available water is dependent on the size, shape and arrangement of mineral particles, as well as the amount of organic matter present within a soil, this impacts the type of vegetation and plant processes such as transpiration and photosynthesis, which affects the plant productivity. Additionally, studies report that

vegetation type, density, uniformity, root characteristics, and litter depth can also influence the processes such as infiltration and evapotranspiration resulting in variation in SM (Reynolds 1970; Hawley *et al.* 1983; Trlica and Biondini 1990; Sala *et al.* 1992; Golluscio *et al.* 1998; Mohanty and Skaggs 2001). For example, grasses, shrubs and forbs present in the semi-arid region have different rooting depths, and thus exploit water stored in different soil layers (Trlica and Biondini 1990). Gómez-Plaza *et al.* (2001) found vegetation played a vital role in SM variability in vegetated zone while soil texture and slope explained a large part of SM distribution in non-vegetated zone.

Similar to Miles (2009) study in GNP, ALB showed no relationship with SM measured at 12 cm depth ($R = 0.197$; $p = 0.349$). The relatively low correlation indicates that ALB does not entirely depend on the water content of 12 cm layer; water content from depths greater than 12 cm is also obtained by plants for growth. Singh *et al.* (1998) showed a significant inverse relationship between above-ground net primary productivity (ANPP) and mean soil water content at different depths (30, 45 and 60 cm), where soil layers showed more water depletion with increase in ANPP during growing season.

Both slope location and time significantly ($p < 0.05$) influenced the spatio-temporal variability in SM and ALB. On average, downslope plots in all the pastures showed more heterogeneity in SM (Table 4.2) and ALB (Table 4.3) compared to upslope and midslope plots. Similar to Hawley *et al.* (1983), the results also showed significant differences in the SM from one sampling date to the next. Significant differences were also observed in May and June ALB data. This temporal variability is likely the impact of high amount of local weather variability during the

field season. Crave and Gascuel-odux (1997); Mohanty *et al.* (2000) and Qiu *et al.* (2001) showed that location on the slope is an important factor contributing to the SM variation. This is because the slope affects the surface runoff and infiltration processes, which in turn affects the plant available water for growth. For example, upslope areas due to less contributing area will have low SM compared to downslope areas with more contributing area, likely leading to high relative SM.

No significant effect of slope location was found on the semivariogram parameters SDR% and range over time for ALB ($p > 0.05$) suggesting that other factors such as vegetation type, vegetated or bare areas and management (grazed or ungrazed) may be affecting the semivariogram parameters. Similarly, no effect of slope location was found on the SM semivariogram parameter SDR% ($p > 0.05$). However, SM range was significantly influenced by the slope location. On average, SM range increased from upslope to midslope plots, but showed a decrease from midslope to downslope plots. Since slope in addition to soil characteristics affect the surface run-off (upslope to downslope) and moisture in the soil, this explains the significant influence of location on SM range. Lakhankar *et al.* (2010) suggest that soil type variation can have a significant influence on the semivariance values. It is stressed that these field based results assessing the effect of slope location on semivariogram parameters are applicable only to this study area and may not be suitable for other areas.

4.3.2 Short-term grazing disturbance on plot-scale SM and ALB

A decrease in SM and ALB was observed corresponding to grazing disturbance at the plot scale, which is consistent with results reported from similar studies of grazing effects on biomass and

soil water content (Frank and Groffman 1998; LeCain *et al.* 2000; Osem *et al.* 2002; Weber and Gokhale 2011). Weber and Gokhale (2011) in their study of the semi-arid rangelands of southeast Idaho reported that soil water content can vary substantially as a result of animal impact and the duration of grazing, despite similar vegetation cover and soil type. This is because grazers can change the soil structure through trampling, altering soil porosity and organic matter of the soils (Tollner *et al.* 1990). This altered soil structure can affect the soil water dynamics leading to variability in SM. Since plant available water is important for plant growth especially during the growing season, any amount of variability in it can affect the plant productivity. Therefore, lower ALB in grazed plots compared to ungrazed plots is most likely a result of decrease in SM, particularly since ALB was closely correlated with the rainfall during growing season. Grazing activity can reduce the accumulation of litter and standing dead through trampling, thus leading to increase in evaporation rates from the soil surface. This can cause decrease in SM (Frank and Groffman 1998).

Both treatment and time as main effects significantly ($p < 0.05$) affected SM. Another significant explanatory effect was attributable to the pasture variable ($p = 0.05$). This effect indicated that the treatment (grazing, no grazing) applied within each pasture (P6, P8, P9) accounts for some significant portion of the total variability seen in SM at this study area and, coupled with the pasture x time interaction, suggests that grazing treatment made changes to SM. Similarly, ALB results also showed significant combined effect of treatment and time. These changes are most likely a combined effect of both treatment and environmental factors such as precipitation (Lauenroth and Sala 1992; Knapp *et al.* 2001). However, it is difficult to separate these factors, especially in light of the likely errors in the measurements. Despite this,

the statistical results for treatment effect on SM and ALB are interesting and appear promising for future research which should be directed towards addressing this same question using a long term data.

This study also showed that ALB was more heterogeneous with smaller patch sizes under grazed treatment compared to ungrazed treatment. Selective grazing by cattle and slope location are contributing to ALB heterogeneity during the grazed treatment (Pinchak *et al.* 1991; Steinauer and Collins 1996; Vallentine 2001 and Fortin *et al.* 2003). The results are similar to Golluscio *et al.* (2005) which evaluated the impacts of grazing on the spatial heterogeneity in the plant biomass in Patagonian steppe. Golluscio *et al.* (2005) showed higher internal heterogeneity (variability at a distance shorter than the minimum distance sampled) in the grazed sites compared to the ungrazed sites.

4.3.3 Spatial pattern in SM and ALB

Most of the Moran's I correlograms were globally significant. The only ones that did not pass the test of significance were the SM correlograms from $P9U_C$ on June 06 and June 23. Both of them showed absence of significant autocorrelation at any distance class, indicating randomness or homogeneity (Pastor *et al.* 1998; Adler *et al.* 2001). For example, most of the SM values ranged between 14% and 16% for $P9U_C$. Similar to SM patterns, there was more randomness or homogeneity (i.e. Moran's I values were close to 0 and non-significant) in the ALB spatial patterns in the grazed treatments compared to ungrazed treatments. The results are similar to Adler *et al.* (2000) study which also showed more random distribution in the grazed treatments compared to the ungrazed treatments. Over time, the forage availability per

cattle is likely to decrease within the plot thus compelling them to feed on previously avoided less palatable vegetation (i.e. decreased selectivity) (Weber *et al.* 1998). Therefore, this would result in homogeneous pattern at plot scale. Bailey *et al.* (1996) and Vallentine (2001) suggests that factors such as plant intake rate and frequency of selection by cattle may affect the grazing spatial patterns at a plot scale.

The distance at which the correlogram first intercepts the abscissa can be used to estimate a patch size, as this corresponds to the shortest dimension of an irregularly shaped patch (Sokal 1979). In the present work, the patch sizes in the grazed treatments were about 4.2 and 5.4 m for ALB and SM, respectively. In comparison, some pastures showed bigger patch sizes for both SM (18.4 m) and ALB (~40 m) before any grazing disturbance. The patchy pattern in the grazed treatment of our study area is likely due to variation in vegetation cover and type within the plots, thus resulting in selective grazing by cattle. For example, pasture P8 had presence of death camas (*Zigadenus venenosus*) (~2%) besides other vegetation, which is avoided by cattle. Grassland studies such as Knapp *et al.* (1999) and Truett *et al.* (2001) report that grazing leads to landscape heterogeneity by creating a mosaic of vegetation and microclimates through selective grazing, urine deposition, and trampling. Similarly, Adler *et al.* (2001) also suggests that at field or landscape level grazing is highly influenced by resource availability (food and location of water), the interaction with plant community and management practices, thus resulting in patchy spatial patterns. Research conducted by Vallentine (2001) and Harrison *et al.* (2003) conclude that heterogeneity in the vegetation structure in the ungrazed treatments mainly exists due to various abiotic (slope and SM content) and biotic (small mammal disturbances and insect grazing) factors.

4.4 Applicability of results, limitations and research recommendations

This study demonstrates high spatial variability in the SM and ALB at plot and pasture scale as a result of local weather conditions, grazing disturbance and slope-position. This study also contributes to understanding of pattern (heterogeneity) and factors (grazing disturbance, weather and slope location) at different spatial scales and how scale can influence and alter the relationship between pattern and factors in the mixed-grassland ecosystem. These results provide insight into patterns and responses of SM and ALB to grazing disturbance and local weather variability during a single growing season in a mixed grassland ecosystem. The results also contribute to providing baseline conditions for an ecosystem model that can be used to acquire knowledge of a given landscape's ecological issues under changing climate and land use management.

Although this study was able to capture the local variability in SM as a result of local weather conditions and grazing disturbance, it is acknowledged that more research involving continuous SM at different depths would be required to gain better understanding of inter-relationships between spatial patterns of SM and phenomena such as plant water stress, evapotranspiration, and land management. One approach could be to install a long term, spatially extensive network of SM probes. Experiments could study SM spatial trends and patterns in response to long term grazing disturbance with variable intensities and local weather conditions. This would, however, require very expensive equipment and time commitments.

One of the most significant limitations of the field analyses was the failure of SM Hydrosense probes at the end of the season, causing a lack of SM data from the experimental pasture plots

during August 2008. Additionally, field measurements for SM at various depths at the weather station were also impeded by animal activity resulting in erroneous data for July and August. Therefore, most of the July and August SM data were also excluded from the final analysis. It is suggested to have redundant sets of SM sensors and that the sensors should be wire-fenced and cables should be run through a PVC pipe from just below the surface to the data logger to prevent any rodent damage. The sensors should also be checked frequently to avoid any interference by small animals causing erroneous or loss of data. Additionally, a more comprehensive soil survey should also be conducted to account for the variability in the soil structure. This information will further help in refining model parameters relevant to soil characteristics of the grassland productivity in the future.

The vegetation type (grass, shrub, forbs and other) at each sample point in all the experimental plots was determined by visual survey, adding some subjectivity. Therefore, it is recommended to use methods that can yield more accurate and true representation of the dominant vegetation at each point. This will certainly help in understanding the spatio-temporal variability in ALB. Additionally, combining this with SM samples at different depths will help in understanding how different vegetation types utilize water from different depths and the effect of plant available water on plant productivity.

4.5 Conclusions

The study was able to identify the spatio-temporal variability and pattern in SM and ALB between different pasture plots and within pastures at different times. Factors such as local weather conditions, slope position, time and grazing disturbance significantly influenced the

spatio-temporal heterogeneity in SM and ALB at plot scale. Overall, for both SM and ALB, on average downslope plots showed more heterogeneity compared to midslope and upslope plots. The SM and ALB semivariograms developed in this study were also used to estimate the range of influence (A_0), i.e. the maximum separation distance within which SM and ALB values appear to be related. This information is useful for determining sampling criteria in future studies at GNP.

Some of the observed differences in plot scale SM and ALB are a result of introduction of grazing disturbance in addition to other factors. However, to be conclusive about the observed differences within the pastures mainly due to treatment, a longer duration study is required.

5.0 Semivariogram Approach to determine Spatio-temporal Variability in Satellite-based Above-ground Live Plant Biomass (ALB): Case Study of Grazed and Ungrazed Experimental Pastures

Grazing can either increase or decrease the spatial heterogeneity of vegetation depending on the grazing intensity and level of plant productivity, thus affecting the biodiversity of a region (Bock *et al.* 1993; Hobbs 1996; Collins *et al.* 1998; Rietkerk *et al.* 2000; and Derner *et al.* 2009). Therefore, it is important to have an understanding of the effects of variation in grazing intensity (or utilization rates) on vegetation heterogeneity and spatial patterns. This will help in implementation of large-scale grazing management scenarios incorporating a wide range of grazing intensities for the sake of conservation of different plant and animal species within a mixed grassland ecosystem.

The analysis in the previous chapter concentrated mainly on characterizing the intra-seasonal spatio-temporal patterns of SM and ALB under ungrazed and grazed conditions across a range of scales (plot to pasture), in order to have a basis from which to quantify or enrich predictions of the potential impacts of grazing. Since the grazing experiment started in June 2008 and field data was collected only for the 2008 growing season (May – August 2008), the analyses were limited to looking at within-season variability in ALB and SM. In addition, it was only possible to gain a preliminary perspective on the effect of variable grazing intensities on ALB heterogeneity, since the experiment had only just begun its grazing phase, and there were limits to the range of observed conditions because only selected pastures could be measured. Therefore, this chapter uses satellite-based ALB estimates at pasture scales to

assess the effects of grazing intensity on the spatio-temporal pattern of ALB in mixed grasslands. It was hypothesized that:

(a) Spatial heterogeneity of ALB for grazed pastures will be higher compared to ungrazed pastures due to selective behaviour of cattle (example, Hartnett *et al.* 1997; Townsend and Fuhlendorf 2010) and will increase over time in response to grazing intensity, with heavy intensity leading to more heterogeneity in ALB compared to light-to-moderate grazing intensity.
(b) Grazing intensity in combination with slope influences the amount of ALB heterogeneity present within the region over time. Downslope areas will be more heterogeneous as a result of grazing intensity with high range of variation in ALB compared to upslope area, due to preferential grazing by cattle in areas near water and shallower slopes (example, Pinchak *et al.* 1991 and Fortin *et al.* 2003).

In this study, pastures P1, P5 and P9 were ungrazed. P2 was stocked with cattle for a target 20% utilization rate resulting in a very light (VL) grazing pressure in this pasture. Similarly, P6 was stocked with cattle for a target 33% utilization rate, thus resulting in light (L) grazing pressure in the pasture. P7 and P10 to P13 were stocked with cattle for a target 45 – 50% utilization rate resulting in low-moderate (LM) grazing pressure in these pastures. P3 was grazed with high-moderate (HM) (57%) grazing intensity, whereas P4 and P8 were very heavily (H) grazed with 70% grazing intensity. From here on grazed pastures will be referred with respective grazing intensities in subscript: $P2_{20}$, $P6_{33}$, $P7_{45}$, $P10_{50}$, $P11_{50}$, $P12_{50}$, $P13_{50}$, $P3_{57}$, $P4_{70}$ and $P8_{70}$. Ungrazed (UG) pastures will be referred to as $P1_{UG}$, $P5_{UG}$ and $P9_{UG}$.

5.1 Data Analyses

Following the methodology described in Chapter 3 temporal changes in ALB heterogeneity were quantified using semivariance analyses of five Landsat scenes taken June 2000 and June 2007 through 2010 (see section 3.3). Once experimental semivariograms were calculated, a model was fitted to the semivariogram to assess spatial correlation. Exponential models were used, as this form was found to provide the best fit, with minimum error, and low residual sums of squares (RSS) value (0.00004 to 0.0094). The exponential model is similar to the spherical model in that it approaches the sill gradually, but different from the spherical in the rate at which the sill is approached and in the fact that the model and the sill never actually converge. This model partitions variance according to the equation:

$$\gamma(h) = C_0 + C[1 - \exp(-h / A_0)] \qquad \text{Equation 5.1}$$

Where $\gamma(h)$ = semivariance for interval distance class h, h = lag interval, C_0 = nugget variance ≥ 0, C = structural variance $\geq C_0$, and A_0 = range parameter (Robertson 2008).

Once a variogram model was fit to the data, parameters such as range (A_0), sill ($C+C_0$) and nugget (C_0) were derived. In addition, magnitude of spatial heterogeneity (MSH) or relative heterogeneity (SH %), correlation ratio (CR) and spatial dependence ratio (SDR) were calculated.

5.2 Results

5.2.1 Local Weather Variability

Analysis was conducted to see the range of variation in weather conditions between different seasons as recorded at the Mankota weather station, thus affecting the SM and plant available

water for plant growth. Total monthly rainfall for the East Block, GNP illustrates the range of local variation in the 2000 and 2007 to 2010 growing seasons (Figure 5.1).

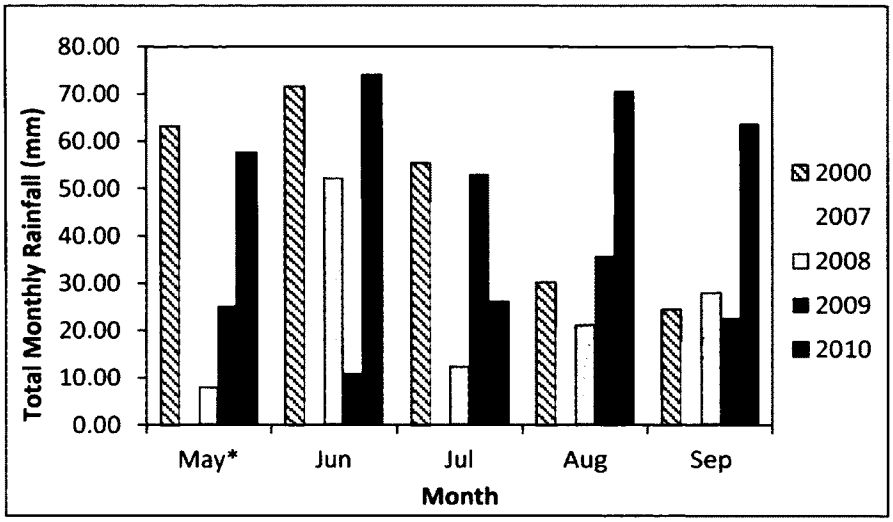


Figure 5.1 Total monthly rainfall (mm) for growing season in year 2000, 2007, 2008, 2009 and 2010.
* Since 2007 precipitation data for May was only available from day 19 onwards, in this graph only days 19 to 31 are presented

More rainfall was recorded during the 2010 growing season (292.04 mm) compared to all other years (244.80 mm, 174.60 mm, 121.80 mm and 147.38 mm). June usually received the most rainfall, except in 2009. Timing and amount of rainfall received were also highly variable between years and events.

Figure 5.2 shows variation in the average air temperatures for year 2000 and 2007 to 2010 in the study area. Year 2007 showed higher average air temperatures compared to other years.

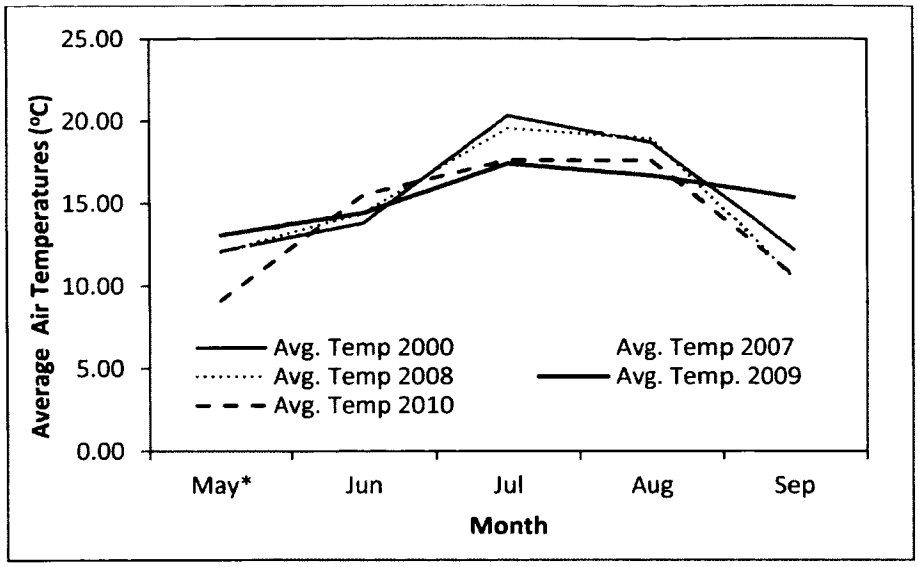


Figure 5.2 Average monthly air temperatures (°C) for year 2000, 2007, 2008, 2009 and 2010 in the study area.
*Since 2007 average air temperatures data for May were only available from day 19 onwards, the graph presents the May data for only days 19 to 31.

5.2.2 Effect of different grazing intensities on ALB spatio-temporal heterogeneity

The spatio-temporal variability in ALB as estimated using NDVI and equation 3.2 between different grazing intensity pastures is shown in Figure 5.3. There is a considerable variability in ALB between very light to light (VLL), low moderate (LM), high moderate (HM) and heavy (H) grazing intensities after 2 years of grazing. Overall, mean ALB increased with grazing intensity until LM, showing a quadratic effect (Figure 5.3). Pastures with LM grazing intensity showed higher ALB both in 2008 and 2010 compared to other grazing intensities.

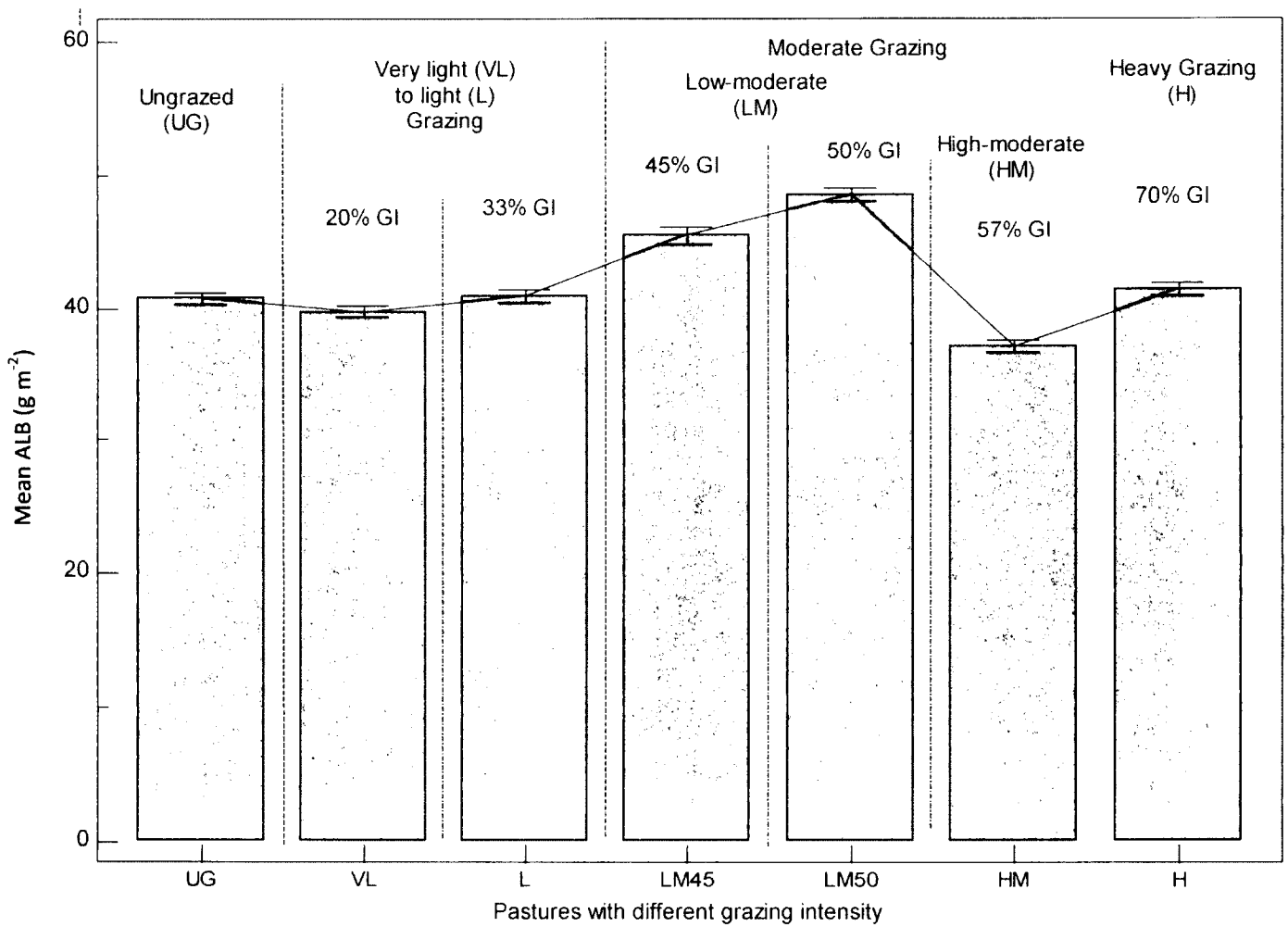


Figure 5.3 Spatial variability in mean ALB between pastures with variable grazing intensity. Mean ALB data are from 26 June 2010 (after 2 years of grazing)

Note: Error bars show 95% CI for mean. No. of observations for each pasture = 3,130.

To compare the ALB between different grazing intensities, a repeated –measures factorial ANOVA with grazing intensity (utilization rate) as the main factor and year as the repeated measure ($\alpha = 0.05$) was used. Mauchly's Test indicated that the assumption of sphericity had been violated, $p < 0.05$, therefore, degrees of freedom were corrected using Greenhouse-Geisser estimates of sphericity. Both grazing intensity (utilization rate) and year significantly affected the ALB (grazing intensity: F (2.2, 894) = 115.4, $p < 0.0001$; year: F (2.0, 799.6) = 877.9, $p < 0.0001$). Approximately 32.5% of the total variance in the ALB was accounted for by the variance in the year, while 68% of the total variance in the ALB was accounted for by the variance in the utilization rates. Main effects of grazing intensity and year were qualified by an interaction of grazing intensity and year (F (4.2, 1671) = 48.9, $p < 0.0001$) with 50.2% of total variance in ALB as a result of grazing intensity (utilization rate) and year. The significant interaction indicated that the magnitude of the grazing intensity effect varied with year. Tukey's HSD post-hoc test comparing all utilization rates (C, VL, L, LM, HM and H) after two years of grazing disturbance showed that mean ALB was significantly ($p < 0.0001$) different between VL and LM, VL and HM, VL and H, LM and HM, LM and H, HM and H indicating that variation in utilization rates affect the ALB. No significant difference in mean ALB was found between ungrazed (or C) (mean ALB 40.73 g m^{-2}) and VLL (mean ALB 40.43 g m^{-2}) grazing utilization rate. Significant difference in mean ALB was also found between before (i.e., year 2007) and after grazing (i.e., year 2008 to 2010) treatment ($p < 0.0001$). A significant year-by-pasture (with different utilization rates) interaction was also observed (F (1, 12) = 7.79; $p < 0.0001$). This secondary effect indicates that while ALB differs annually, it is differentially variable by pasture, suggesting both environmental and grazing intensity influence.

The effect of no grazing, VLL, LM, HM and H grazing on the spatio-temporal variability in ALB was also assessed by semivariogram analysis (Webster and Oliver 1992 and 2001). Semivariograms of the East Block pastures for years 2008 (start of grazing experiment), 2009 (1 year after grazing) and 2010 (2 years after grazing) are presented in Figure 5.4 and Figure 5.5. In general, all the pastures showed a moderate to strong spatial dependency in ALB varying from ~1% to ~42% according to the classification of spatial dependency by Cambardella *et al.* (1994). All experimental pastures (grazed and ungrazed) also showed nugget effect (C_0) suggesting presence of some local random variation in the dataset. This can be attributed to either the measurement errors or to variation prominent at spatial scale smaller than the pixel size (or both) (Kitanidis 1997; Tarnavsky 2008).

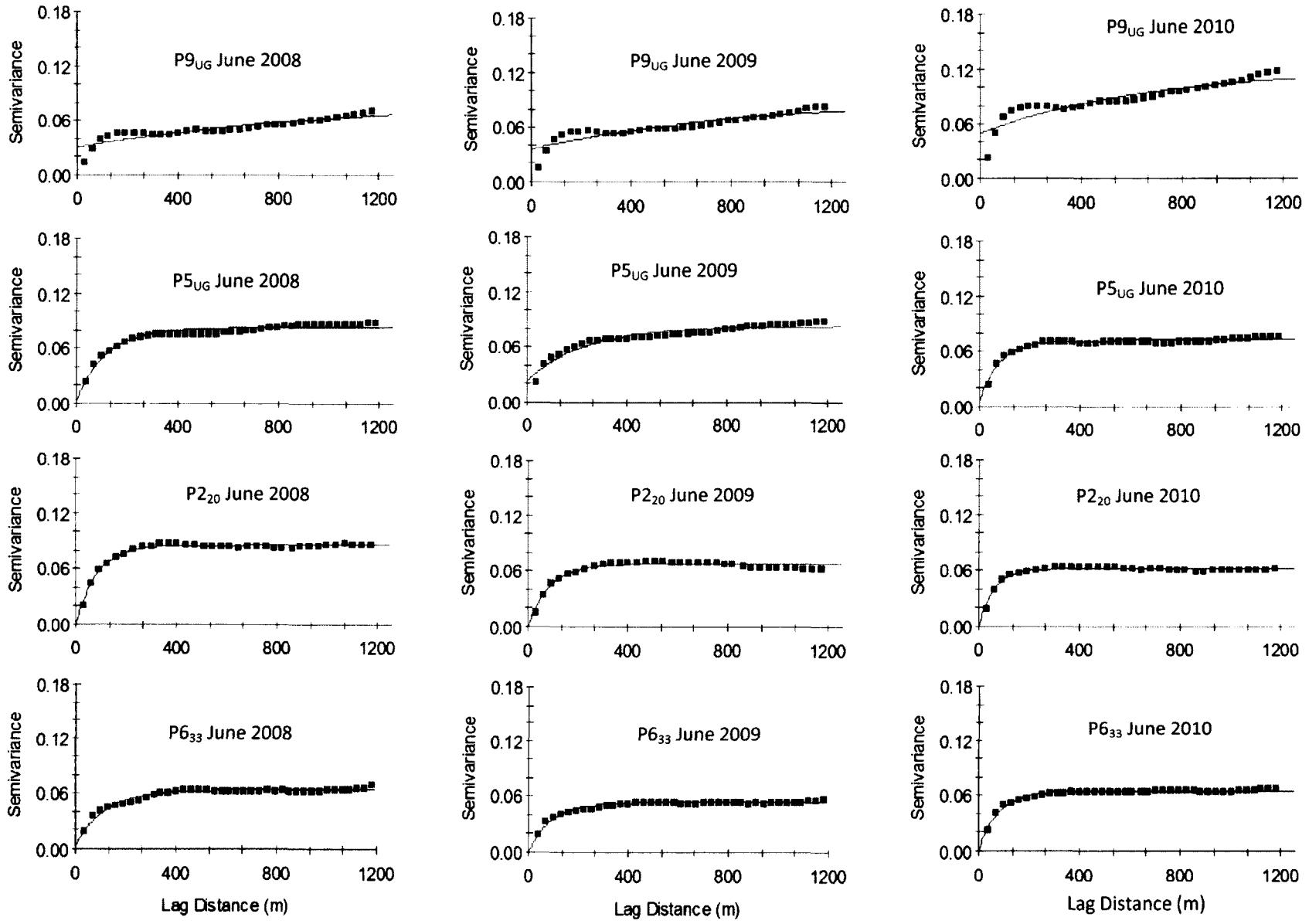


Figure 5.4 Semivariogram results for ALB in ungrazed ($P9_{UG}$ and $P5_{UG}$) and very light to light (20% - 33%) grazing intensity pastures. Note: $P2_{20}$ = pasture P2 with 20% grazing intensity; $P6_{33}$ = pasture P6 with 33% grazing intensity

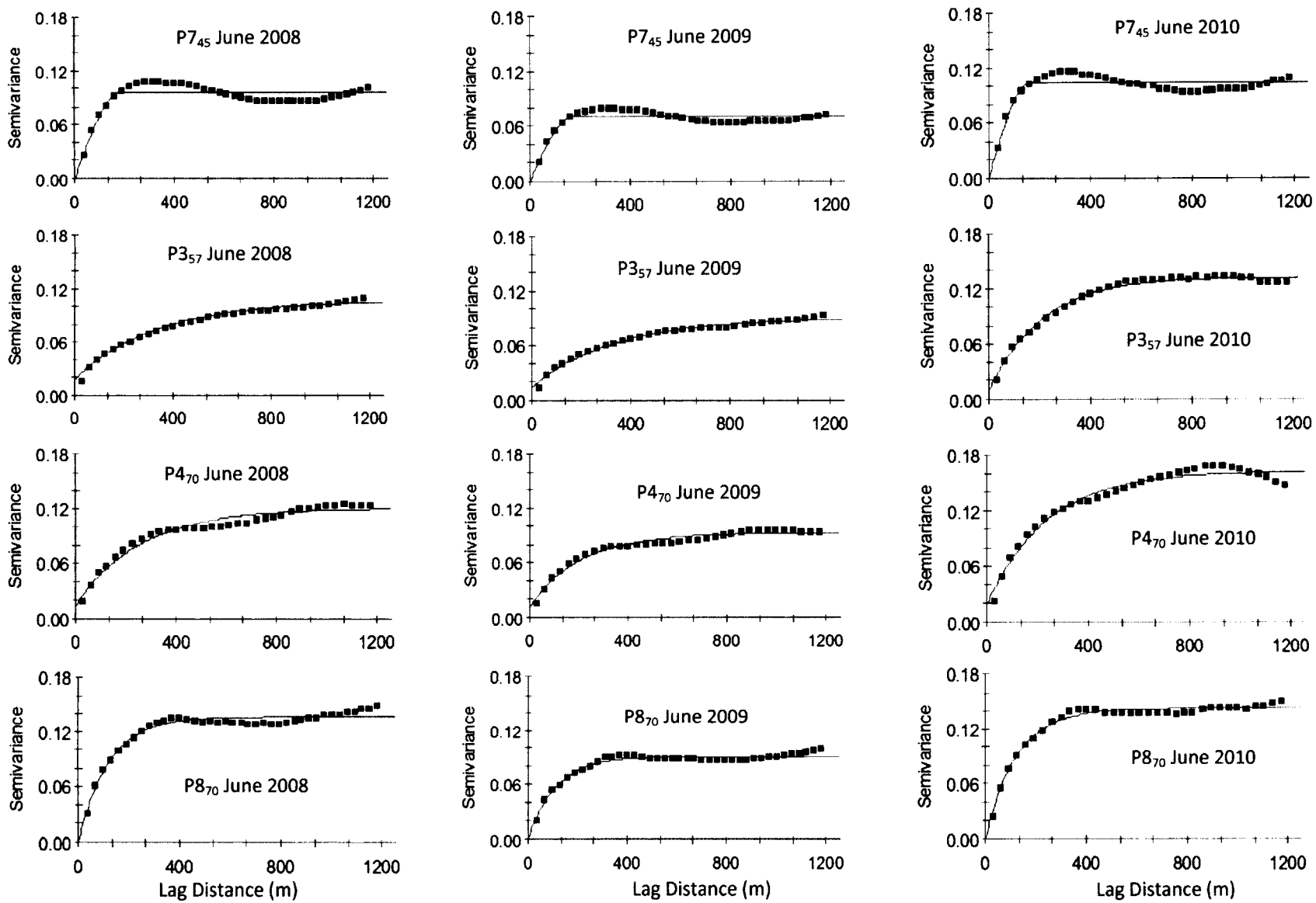


Figure 5.5 Semivariogram results for ALB in pasture P7 with low-moderate (45 – 50%) GI; pasture P3 with high-moderate (57%) GI and pastures P4 and P8 with heavy (70%) GI.

Semivariograms were calculated for the long-term grazed Mankota community pastures 10, 11, 12 and 13 (Figure 5.6). Pastures 10 ($P10_{50}$), 12 ($P12_{50}$) and 13 ($P13_{50}$) were grazed freely with moderate intensity until the start of biodiversity and grazing experiment in June 2008 (Bleho 2009). However, after the start of the grazing experiment in June 2008, the pastures had controlled grazing with 50% GI (low-moderate). Therefore from year 2008 to 2010 the results are based on controlled grazing within these pastures.

In general, after seven years of free range grazing (2000 to 2007), $P10_{50}$ showed an increase of ~18.2% auto-correlated heterogeneity, whereas $P13_{50}$ only showed an increase of ~4.2% auto-correlated ALB heterogeneity. In comparison, $P12_{50}$ showed a decrease of ~13% in relative heterogeneity. From 2008 to 2010 (controlled grazing), the heterogeneity in ALB increased by ~2.3% in $P10_{50}$; by ~16.2% in $P12_{50}$ and ~11.9% in $P13_{50}$ (Appendix 2). In comparison to these long-term grazed Mankota community pastures, the East Block pasture $P7_{45}$ with similar grazing intensity showed 5.6% increase in the ALB heterogeneity after two years of grazing.

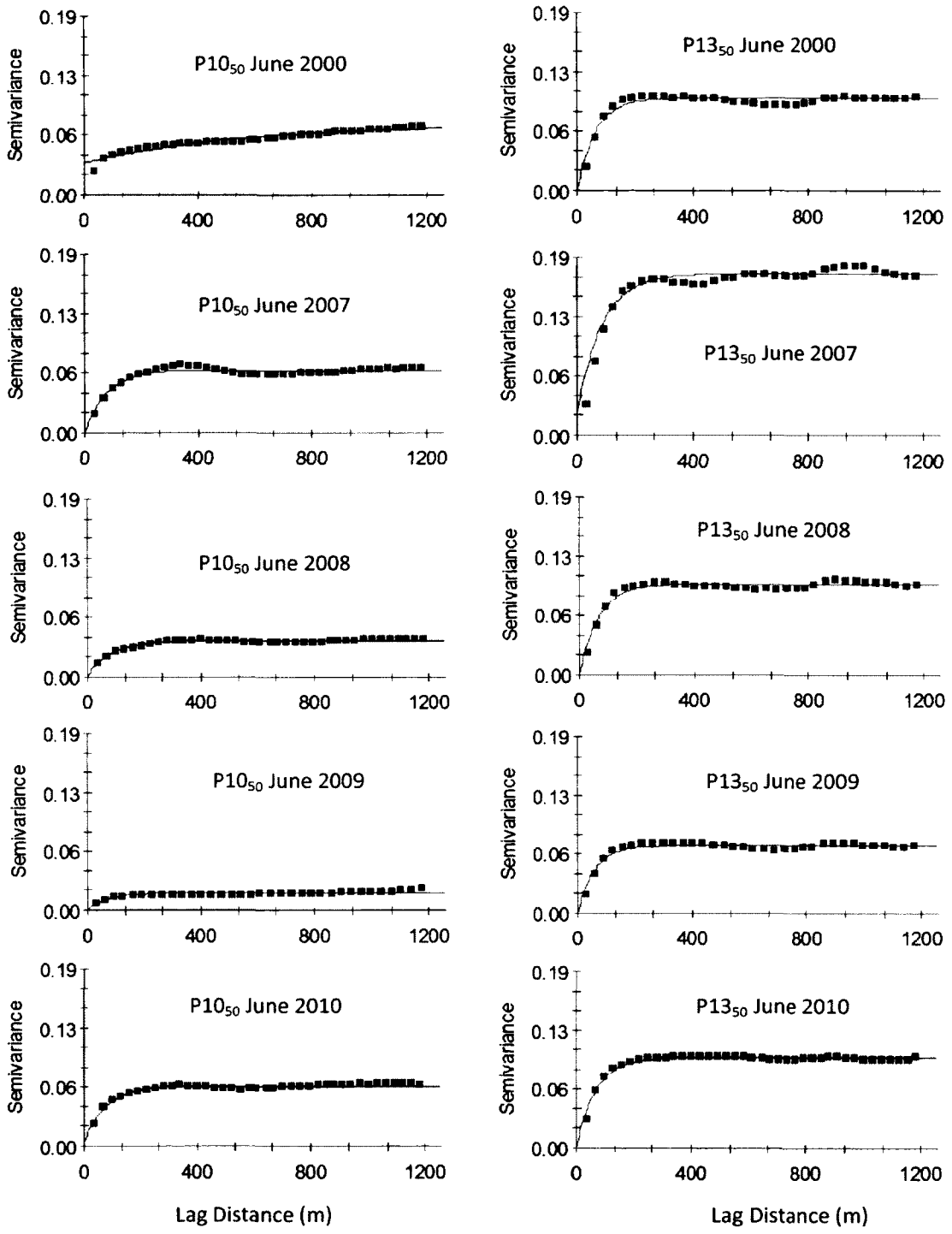


Figure 5.6 Semivariogram results for ALB in low-moderate grazing intensity (50%) pastures 10 ($P10_{50}$) and 13 ($P13_{50}$) located in Mankota community pasture. Note: Semivariograms for years 2000 and 2007 represent pastures with free range grazing, whereas semivariograms for years 2008 to 2010 represent pastures with controlled low-moderate grazing..

Table 5.1 provides the average sill, range, and magnitude of spatial heterogeneity (MSH) values during years 2007 (before grazing), 2008 (start of grazing) and 2010 (after 2 years of grazing) for the different grazing levels. Overall, ungrazed pastures showed the lowest heterogeneity in ALB with a large range of spatial auto-correlation compared to grazed pastures with variable grazing intensity.

Table 5.1 Summarized sill, range and MSH results for different grazing intensities for years 2007, 2008 and 2010

Year	GI	Sill ($C+C_0$)	Range (m)	MSH ($C / [C + C_o]$)
2007 (No grazing)	UG	0.180	1,348	0.690
	VLL_{BG}	0.143	324	0.970
	LM_{BG}	0.124	315.2	0.834
	HM_{BG}	0.212	257	0.894
	H_{BG}	0.239	172	0.938
2008 (Start of grazing)	UG	0.122	1,294	0.658
	VLL_{AG}	0.075	249	0.944
	LM_{AG}	0.075	293	0.887
	HM_{AG}	0.107	324	0.838
	H_{AG}	0.128	188	0.942
2010 (after 2 years of grazing)	UG	0.099	1,353	0.652
	VLL_{AG}	0.063	180	0.998
	LM_{AG}	0.095	236	0.957
	HM_{AG}	0.133	212	0.925
	H_{AG}	0.153	180	0.941

Note: GI = grazing intensity; BG = before grazing; AG = after grazing; UG = ungrazed; VLL = very light to light grazing (20 – 33% GI); LM = low-moderate grazing (45 – 50% GI); HM = high-moderate grazing (57% GI); and H = heavy grazing (70% GI); MSH = magnitude of spatial heterogeneity.

In general, grazed pastures showed 2.8 to 12.3% increase in heterogeneity (MSH) from 2007 to 2010 and 5.4 to 8.7% increase in heterogeneity (MSH) after two years of grazing (2008 – 2010). An exception was heavily grazed pastures, which showed no change in heterogeneity after two years of grazing period, suggesting that grazing maintained the existing heterogeneity over the

years. Grazed pastures with moderate grazing (low- and high-moderate) showed the most change in heterogeneity after grazing was introduced (2007 – 2010, change in MSH = 12.3% for LM and 7.1% for HM; 2008 – 2010, change in MSH = 7.0% for LM and 8.7% for HM). Although all the grazed pastures showed higher heterogeneity values for ALB compared to ungrazed, grazed pastures with very light to light grazing intensity had the highest MSH value after two years of grazing (0.998) followed by low-moderate (0.957). Overall, the range for ALB over time decreased for all the grazing intensities suggesting that vegetation patch size is negatively affected by grazing activity (Figure 5.7).

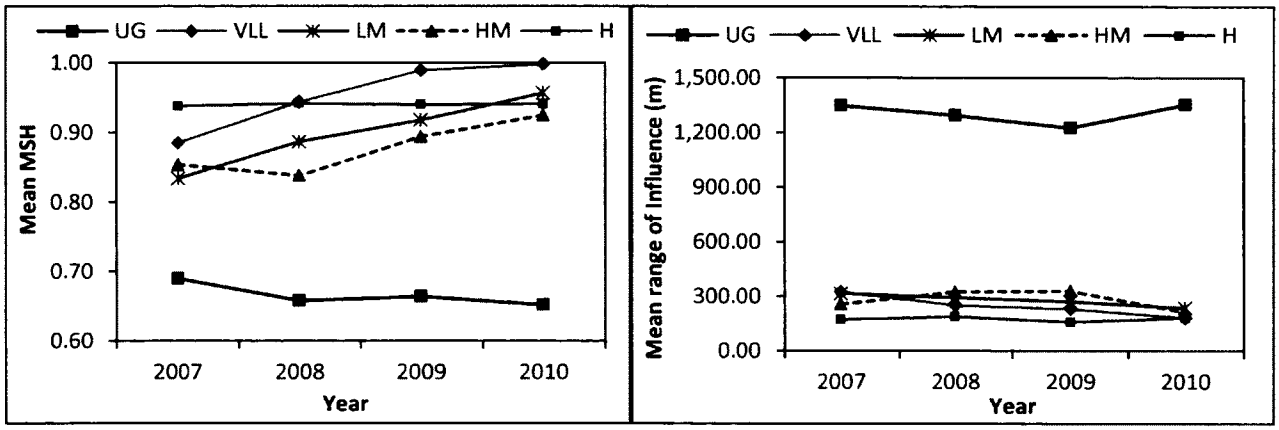


Figure 5.7 Comparison of mean MSH and mean range of influence between different grazing intensities for no grazing (year 2007), at the start of the grazing (year 2008), after one year of grazing (year 2009) and after two years of grazing (year 2010).

Similar to a study by Lin *et al.* (2010), range and MSH were compared among grazing intensities by one-way ANOVA and regression analysis to determine whether grazing intensity (utilization rate) was a significant predictor for the geostatistical metrics or not. Out of all the regression models, exponential and inverse models were found to be the best options representing the MSH and range data to show the trend over time. After cattle were allowed to graze for 2 years at different intensities, a decrease in patch size (range) for ALB was observed corresponding to

grazing intensity (exponential model, 2008: β = -0.959, t = -5.857, p = 0.009, R^2 = 0.92; 2009: β = -0.959, t = -5.901, p = 0.009, R^2 = 0.92; 2010: β = -0.959, t = -5.832, p = 0.01, R^2 = 0.92).

An increase in MSH for ALB was observed after two years of grazing corresponding to grazing intensity (inverse and exponential model, 2008: β = -0.922, t = -4.13, p = 0.026, R^2= 0.85; 2009: β = -0.952, t = -5.432, p = 0.01, R^2 = 0.90; 2010: β = -0.974, t = -7.45, p = 0.005, R^2 = 0.95). All the grazed pastures showed higher MSH than the ungrazed pastures. In other words, the proportion of total sample variation accounted for by spatially structured variation increased over time with the grazing pressure.

5.2.3 Spatial patterns of ALB

Spatial auto-correlation measured using Moran's I for ALB was calculated using two sampling designs, grid and transect, at a 30 m lag interval (Figure 5.8). The rationale was to see the effect of sampling design on the ALB spatial pattern detection. The number of observations per distance class and the maximum extent for interpretation of a correlogram varied with the spatial configuration of the two sampling designs. The maximum extent of interpretation ranged from 1,200 m for transect, and 1,200 and 2,400 m for grid design.

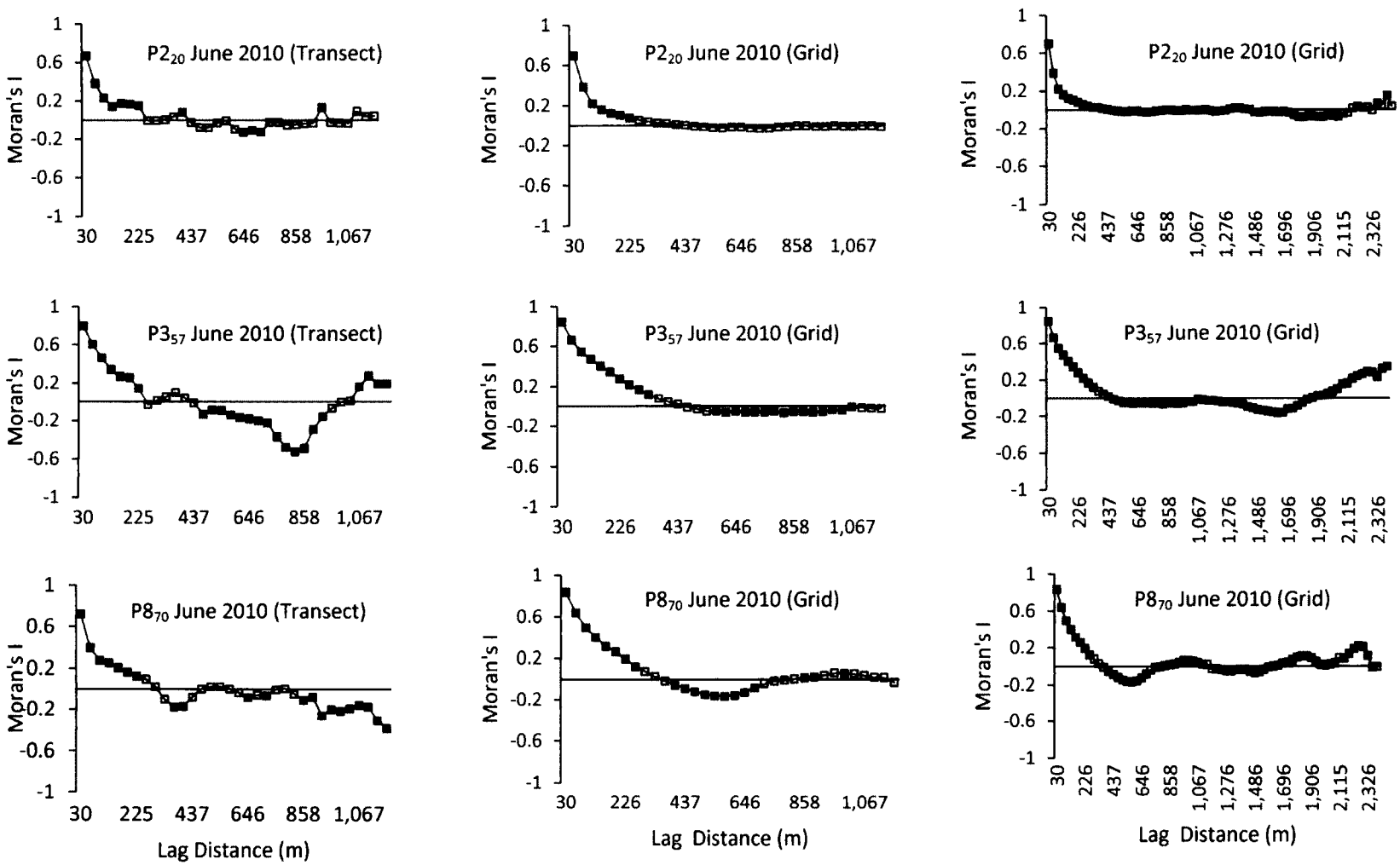


Figure 5.8 Effect of Grazing Intensity and sampling design (Grid VS transect) on the ALB spatial pattern (Moran's I): An example of VLL ($P2_{20}$), HM ($P3_{57}$) and heavy grazing ($P8_{70}$) is provided. (Lag distance = 1200 m for transect and Grid; 2400 m for Grid only, lag class distance interval = 30 m).

Note: $P2_{20}$ = pasture 2 with very light to light (20%) grazing intensity (GI); $P3_{57}$ = pasture 3 with high-moderate (57%) GI; $P8_{70}$ = pasture 8 with heavy (70%) GI. Solid squares indicate significant coefficient values at $\alpha = 0.05$; open squares indicate non-significant coefficient values after progressive Bonferroni correction.

The transect design resulted in a maximum of 3,800 observations per distance class, while grid design reached a minimum of 2,300 pairs per distance class. Transect sampling design (using sample points along a transect placed within a pasture) exhibited a wave-like pattern within the experimental pastures. For example, in the $P2_{20}$ correlogram, the first change of sign from positive to negative value occurred around 60 m, which corresponded to the spatial range of the patches. The correlogram showed some repetitive patterns of patches; however both the patch size and the distance among the patches were quite variable.

Similarly, other experimental pastures such as $P3_{57}$, $P8_{70}$ (Figure 5.8) and $P13_{50}$ also showed patchy spatial pattern with variable patch size and distance among the patches. In comparison, $P2_{20}$'s spatial correlogram based on the grid sampling design (1,200 m extent) showed a gradient spatial pattern with significant positive values at short distances to negative ones at large distances. However it levelled around zero, indicating absence or non-detection of spatial patterns at distances > 220 m. Similarly, $P3_{57}$, $P8_{70}$ and $P13_{50}$ showed positive values at short distances with significant spatial auto-correlation. $P3_{57}$ and $P8_{70}$ also showed alteration of values from positive to negative, thus indicating patchiness.

Similar to $P2_{20}$, the negative values for $P13_{50}$ were oscillating along the zero value suggesting absence of any significant spatial autocorrelation at large distances. The spatial range (zone of influence, patch size) for pasture $P13_{50}$ was around 360 m, a distance at which the sign of the values changed from positive to negative. Overall, correlograms based on either of two sampling designs were globally significant, indicating that the overall spatial pattern of ALB is not random. It is likely that there are spatial patterns existing at smaller distances than the

distance between sampling pixels, which could not be detected by the grid sampling design (Wiens 1989). Additionally, smooth curves displayed by the grid sampling design compared to transect design are from averaging over several directions. The patterns are clearer for grid sampling design when correlograms were calculated for 2,400 m extent.

Since the previous chapter (see section 4.2.2.3) showed that location within a pasture had significant impacts on ALB, a model using position (upslope/downslope) as a covariate was tested. A significant effect of grazing intensity on ALB was found even after controlling for the effect of position ($F(4, 943) = 12.74$, $p < 0.001$; $N = 953$). Post hoc test (Tukey's HSD) results indicated that mean ALB values were significantly different ($p < 0.001$) between the upslope and downslope locations for all the grazing intensities, where downslope areas showed higher mean ALB values than upslope areas.

5.3 Discussion

It was hypothesized that: (a) spatial heterogeneity in ALB in grazed pastures would be higher than ungrazed pastures and would increase over time in response to grazing intensity, with heavy intensity leading to more heterogeneity in ALB compared to light-to-moderate grazing intensity; and (b) grazing intensity in combination with slope would influence ALB heterogeneity within the region over time. Downslope areas would be more heterogeneous as a result of grazing intensity with high range of variation in ALB compared to upslope areas.

5.3.1 Spatial heterogeneity in grazed and ungrazed pastures

All the grazed pastures showed variability in ALB over the years (Figure 5.3). Despite the variability in ALB, overall grazed pastures showed higher ALB than ungrazed pastures, where

ALB was greatest in the moderately grazed pastures with 50% grazing intensity (Appendix 2, Figure A2.3). The findings were consistent with Holecheck *et al.* (2006) and Jamiyansharav *et al.* (2011) that also showed higher biomass production in the moderately grazed pastures than in ungrazed sites.

NDVI and therefore the estimated ALB values increased and decreased over the years within grazed and ungrazed pastures, which is common in semi-arid grasslands (Milchunas *et al.* 1994; Knapp *et al.* 2007). This is because temporal (seasonal and inter-annual) variability in plant processes is largely a function of changes in soil temperature and moisture over time (Keatley 2000; Epstein *et al.* 2002; Knapp *et al.* 2002a; Keatley and Fletcher 2003). As shown in Chapter 4, variation in daily ALB (June 2008) was strongly and significantly correlated with the natural rainfall ($R = 0.842$, $p < 0.001$, $N = 30$) and temperature ($R = 0.70$; $p < 0.0001$; $N = 120$). However, due to satellite data limitations and imagery quality at the desired scale, it was not possible to separate the effect of inter-annual variation on the mean ALB values in the grazed pastures. This is because desired image quality (< 10% cloud cover and shadows) for the study area limited the scene acquisition to single dates in one or two months of the growing season, for example 29 June 2008, 23 June 2009 and 26 June 2010. This resulted in insufficient data samples to run the correlation analysis to determine the impact of both rainfall and grazing intensity on ALB variability. Therefore, to differentiate rainfall-induced fluctuations from changes in vegetation dynamics caused by different grazing intensities, monitoring must include seasonal data and/or inter-annual data for long periods to have adequate sample size.

The study area showed variation in the weather conditions over a four year study period. Both timing and amount of rainfall received were highly variable between the years and events (Figure 5.1). Fay *et al.* (2000), in a mesic grassland ecosystem located in north-eastern Kansas, identified rainfall interval as the primary influence on the soil and plant responses, with increased intervals causing reduction in total aboveground net primary productivity (ANPP) and flowering duration. This is because increased intervals between rainfall events can create soil water deficits thus affecting the plant available water for growth.

A significant linear correlation was found between the amount of biomass available for grazing and grazing intensity ($R^2 = 0.862$, $p < 0.0001$), where in general ALB increased with grazing intensity. Wallace and Crosthwaite (2005) also reported a significant linear correlation between biomass and grazing intensity ($R^2 = 0.365$, $p < 0.0001$).

The semivariogram analysis showed higher spatial heterogeneity in ALB for grazed pastures with variable utilization rates than ungrazed pastures (Table 5.1). Johnson's (2010) study of grazing intensity effects on grassland birds also showed increased heterogeneity over time with increase in grazing intensity. In this study, heavily grazed pastures showed the highest ALB heterogeneity in 2007 (no grazing year) compared to other pastures. However, after the introduction of grazing in year 2008, higher heterogeneity in ALB over time was observed in VLL and LM pastures than heavily grazed pastures indicating a quadratic effect, reminiscent of the intermediate disturbance hypothesis (Figure 5.7). The results were similar to Lin *et al.* (2010), which also reported a quadratic relationship between ALB heterogeneity and stocking rates. Overall, grazed pastures showed increase in amount of heterogeneity after two years of grazing

with high-moderate grazing showing the most change in heterogeneity followed by low-moderate grazing intensity. An exception was heavily grazed pastures, which showed no change in heterogeneity after two years of grazing, suggesting that grazing maintained the heterogeneity over the years. It is likely that forage availability remained high enough even under the highest utilization rate that cattle did not have to utilize previously ungrazed patches during the grazing period, which most likely would had homogenized the vegetation structure to some degree. Similar studies of the effects of grazing intensity on spatial heterogeneity of vegetation have documented changes (increase or decrease) in spatial heterogeneity over time, where study results are likely affected by the vegetation utilization level, response variable, and spatial scale evaluated (Townsend and Fuhlendorf 2010). Since GNP's management goals include increasing heterogeneity in vegetation which is essential to maintain and sustain biodiversity within the GNP, 70% grazing intensity may be too high. However, other studies document the association of some rare native plants (example, blowout penstemon, *Penstemon haydenii*) and animals such as black-footed ferret and mountain plover with heavily grazed areas (Knowles *et al.* 1982; Klute *et al.* 1997; Stubbendieck *et al.* 1997). Therefore, if economically possible, a range of grazing intensities (low-moderate-heavy) will be more suitable in GNP in providing habitat for greater range of species preferring variable vegetation cover than incorporating limited range of grazing intensities (low and moderate) or a single grazing intensity.

Based on field analyses (Chapter 4) and personal observation during the field work conducted in summer 2008, soil characteristics, vegetation type (grass, shrubs, forbs and other) within the lowland, upland and riparian areas of each pasture and cover (sparse, dense, bare or mixed)

were highly variable between the experimental pastures. This is likely the cause of variation in addition to local weather variation within the ungrazed pastures (Vallentine 2001; Harrison *et al.* 2003). In comparison, grazing activity is an influencing factor for the variability in ALB in the grazed pastures. For example, compaction due to grazers can alter the soil structure which may change the soil aeration and moisture retention capacity, and thus affect the plant available water (Jacobs *et al.* 2004). Any change in plant available water will further impact the plant growth as well as plant productivity, thus contributing to more variability between the grazed and ungrazed pastures. In short, grazing disturbance can help create and maintain the heterogeneity in aboveground biomass which is crucial for the successful co-existence of many grassland species (Milchunas *et al.* 1998; Fuhlendorf and Engle 2001; Vermeire *et al.* 2004).

5.3.2 Effects of Grazing Intensity and slope location on ALB

Grazing intensity (utilization rates) and year significantly influenced the spatial variability in ALB ($p < 0.0001$) suggesting a combined effect of both treatment and environmental factors such as precipitation (Lauenroth and Sala 1992; Knapp *et al.* 2001). In Yang *et al.* (2012), precipitation explained approximately 12% of the variation in the relative production between the grazed and ungrazed sites. However, Lauenroth and Whitman (1977) indicate that other environmental factors such as air and soil temperatures may also contribute to the difference in production between different grazing utilization rates.

After two years of grazing disturbance, mean ALB was significantly ($p < 0.0001$) different between VL and LM, VL and HM, VL and H, LM and HM, LM and H, HM and H indicating that variation in utilization rates affect the ALB. However, no significant difference in mean ALB was

found between ungrazed (or controlled, C) (mean ALB 40.73 g m^{-2}) and VLL (mean ALB 40.43 g m^{-2}) grazing utilization rates. It might be possible that a 2 year time period is not long enough for changes in ALB to become apparent under VLL grazing. Additionally, the grazing intensity is very light to show any noticeable changes compared to ungrazed pastures.

A significant and higher correlation (R^2 = 0.61) was found between the field measured NDVI and above-ground live plant biomass in the study area compared to other studies such as Zhang *et al.* (2008) which showed a correlation coefficient of only 0.43. Standing dead material (SDM) is an issue in the grasslands that can make satellite based NDVI less efficient for quantifying production in grasslands. This is because SDM can decrease the contrast in the red and near-infrared wavelength regions between vegetation and background. Satellite based ALB was calculated using the regression equation based on field based NDVI and biomass, which adds some uncertainty to the analysis due to the lack of research on scaling effects between plot and pixel scales in this environment. Despite this, grazing intensity showed significant effect on mean ALB and it varied with utilization rates. The results are in agreement with other grazing studies such as Holecheck *et al.* (2006) and Jamiyansharav *et al.* (2011) conducted in grasslands. Though the magnitude of observed differences in mean ALB between different utilization rates was moderate and not very high, the results require some caution. Despite this, the statistical results for grazing intensity effect on ALB are interesting and appear promising for future research which should be directed towards addressing this same question using a long-term data.

Overall, the reduction of ALB by intensive grazing (high moderate to heavy) also led to the decline of range (used as a general index of average patch size (Dent and Grimm 1999)) for ALB, suggesting that vegetation patch size decreased with grazing pressure. Studies show that at pasture scale cattle mainly select feeding sites based on water availability, forage abundance, plant phenology and cover (Laca and Ortega 1996; Vallentine 2001). Thus, the results support a view of grazing as a characteristically patchy process (Adler *et al.* 2001), where patchiness could be due to plant defoliation, trampling and excretion during the grazing period (Damhoureyeh and Hartnett 1997). For example, grazers' excretory products are nutrient rich which creates patches with elevated nutrients readily available for plants. These nutrient rich patches generally have altered plant species composition (Steinauer and Collins, 1995). Also, some studies show that grazers often "patch graze" by preferentially grazing some areas repeatedly while other areas are left ungrazed until forage availability is low (Coghenour 1991; Cid and Brizuela 1998). As a result of this preferential grazing, patchiness is either maintained or enhanced in time.

In addition, field pictures from 2008 in the East Block of GNP showed a high amount of variability in the type of vegetation within all the experimental pastures. For example, $P8_{70}$ showed presence of poisonous grass death camas (*Zigadenus venenosus)* which is avoided by the grazers. This may have also contributed to patchy patterns within the experimental pasture. Additionally, vegetation patches with cow patties and urine are generally avoided by the cattle in the same year (Steinauer and Collins 2001). This could have also added to patchiness within the grazed pastures. Such patchy grazing tends to enhance biodiversity (Fuhlendorf and Engle 2001; Truett *et al.* 2001).

Cattle's selectivity based on plant palatability and nutritive quality is likely one of the contributing factors for the patchy vegetation patterns. Generally, the nutritive quality of forage declines as the growing season progresses, which affects the cattle foraging decisions. This is because cool-season grasses (C_3) have higher nutritive quality early in the season compared to warm-season grasses (C_4) that grow later in the season (Adams *et al.* 1996). Additionally, variation in grazing intensity with light grazing in some areas and heavy grazing in others also result in a mosaic of vegetation types, thereby influencing not only the plant community but diversity in animals and insects as well (Hartnett *et al.* 1996; Knapp *et al.* 1999).

There was a significant effect of grazing intensity on ALB even after controlling for the effect of slope position, and mean ALB values were significantly higher as one moves downslope. Plant community composition is notably different between the upslope and downslope areas, which may have led to some of the observed structural differences. Studies have shown that cattle generally forage in high moisture areas such as downslope or riparian areas due to availability of abundant and high quality forage (Senft *et al.* 1987; Phillips *et al.* 1999; Briske *et al.* 2008). However overgrazing can have negative impacts on certain species which prefer these areas for breeding or hunting, such as burrowing owls who hunt near creeks for mice, as well as aquatic species due to cattle excrement. This is a major concern for rangeland managers who are interested in wildlife habitat conservation. Artificial water supplies and salt cubes were provided in uplands to encourage forage utilization in these areas and to reduce cattle damage to riparian areas. This could have also influenced the grazing patterns in the pastures, because water availability is an important factor in cattle foraging decisions in addition to forage depletion (Willms 1990; Irving *et al.* 1995; Briske *et al.* 2008). Studies such as Adler *et al.*

(2001), Vallentine (2001), Fontaine *et al.* (2004) and Bradley and O'Sullivan (2011) concluded that factors such as slope, quality or desirability of forage, and distance to water influence the grazing distribution.

5.3.3 Spatial patterns of ALB over time

Two sampling designs, grid and transect, were used to determine the spatial patterns in ALB with specific focus on smoothness of the correlogram which is dependent on the number of pairs of observations per distance class and the maximum distance for interpretation, defined as half the maximum extent. In our study, both designs were able to detect the spatial patterns in ALB. However, a grid design provided smoother correlograms as a result of more number of pairs per distance class than transect sampling design for the same maximum distance.

Although CV measurements can provide an indication of the magnitude of variance, geostatistical analysis is needed to quantify different aspects of the spatial heterogeneity, including the degree and range of auto-correlation (Li and Reynolds 1995). All the spatial correlograms showed a strongest value of spatial auto-correlation within the first distance class and corresponded to the spatial range of the patches. For a separation distance > 440 m and < 1,200 m spatial autocorrelation of ALB was generally neutral to slightly negative. Spatial autocorrelation of ALB in high-moderate and heavily grazed pastures was consistently close to zero at intermediate separation distances indicating some random variation. For these, my sampling strategy failed to identify the spatial structure at intermediate distances due to lack of any spatial autocorrelation. On the contrary, the randomness as a result of lack of any

significant fluctuations in correlograms at intermediate distances might be indicative of random arrangement of patches created as a result of grazing disturbance (Pastor *et al.* 1998).

The spatial correlograms for the grazed experimental pastures based on transect sampling design exhibited a wave-like pattern compared to correlograms based on grid sampling design at the same maximum distance (1,200 m) which do not show this characteristic (periodicity) but display smooth curves from averaging over several directions. The patchy pattern observed in both sampling designs is most likely caused by selective grazing by cattle based on forage abundance, plant phenology and cover; trampling and waste deposition which provide sites for plant germination as a result of high nutrient availability (Sternberg *et al.* 2000). Overall, the correlograms from both sampling designs showed some repetitive pattern of patches; however both the patch size and the distance among the patches were quite variable.

5.4 Applicability of results, limitations and research recommendations

The results from this study contribute to a developing body of literature that suggests the effects of livestock grazing on the spatial heterogeneity of vegetation is variable depending on the grazing intensity, response variable, and spatial scale evaluated. The patterns observed in this study also support the notion that variability in grazing intensity can significantly affect the spatial patterns of ALB. For example, all five grazing intensities (VL, L, LM, HM and H) used in this study showed change (increase) in the spatial heterogeneity of ALB over time. Intensive grazing at 57% and 70% intensity showed decrease in amount of biomass compared to VL to LM (20 – 50%) grazing intensities. This means that with increase in cattle stocking rates more biomass was removed from the pasture, thus resulting in less energy available to other

consumers (Milchunas *et al.* 1998). In addition, removal of ALB by cattle can also influence the amount of vegetative cover and forage available to other grassland species for nesting, or escape from predators (Fuhlendorf and Engle 2004; Bock *et al.* 2006). This information is crucial for the development of a better grazing management plan for GNP with a goal to create a vegetation structure suitable for variety of birds and animal species.

Despite the dense set of sampling points presented in the previous chapter to capture the local variability in the SM across the landscape, the most significant limitation of this research was the lack of continuous temporal SM data from grazed pastures with variable intensity at a coarser scale (30 m) for the study area. As a result, it was not possible to see the impact of different grazing intensities on the SM variability. It is recommended that future study should use remote sensing techniques to acquire SM information at a landscape or regional scale, which will help in understanding the spatio-temporal patterns of SM at these scales. If cost is the issue, then it is suggested to place sensors randomly across the study area for continuous data.

It is also recommended that future studies should assess the impact of distance from watering points on the vegetation spatial patterns under different grazing intensities. This will help in better understanding of the impact of grazing intensity on plant heterogeneity. Future studies could also explore the relationship between the plant species richness and grazing intensity, as well as plant height and grazing intensity using long term grazing disturbance. This is because diversity in vegetation structure resulting from grazing can provide habitat opportunities for a variety of grassland birds and invertebrate species.

Since grasslands have developed under the influence of frequent and extensive fires, and intensive grazing, both these disturbances are required for proper maintenance of grasslands. It would be valuable to incorporate fire in additional studies to assess the combined effect of fire and grazing disturbance on ALB heterogeneity. Patch burning within a pasture is suggested as this will allow cattle to access both burned and unburned vegetation during the subsequent growing season.

Finally, the conclusions drawn from the spatial analyses in this study are limited by two aspects of the sampling design: extent of study area and the spatial resolution (30 m). As a result, it was not possible to detect patterns, if any, that existed at scales broader than the extent of the study area or finer than the distance between sampling points (Wiens 1989).

5.5. Conclusions

To conclude, this study showed that spatial characteristics of vegetation varied greatly by grazing intensity, time and slope position at the coarse scale (30 m) in the studied grassland ecosystem. This study also demonstrates that cattle grazing with variable intensity can generate, maintain and change the spatial patterns of vegetation in the studied semi-arid grassland ecosystem.

Finally, the results also show that heterogeneity increases (quadratic) and spatial range (patch size) decreases over time in response to grazing intensity and there is a significant effect of grazing intensity on ALB even after controlling for the effect of slope position.

6.0 Modelling productivity and soil carbon dynamics of a mixed grassland ecosystem under variable grazing intensities: A simulation analysis

6.1 Introduction

6.1.1 Grasslands and modeling

Ecologically, grasslands play an important role in storing carbon both above and below ground, and thus need to be maintained and protected. It is estimated that grasslands store approximately 34% of the global carbon stocks (White *et al.* 2000). Grasslands are mainly influenced by precipitation and herbivory, in addition to other factors (Frank and Groffman 1998; Flanagan *et al.* 2002; Knapp *et al.* 2002; Jones and Donnelly 2004), so it is important to develop an understanding of how grazing regimes influence the carbon dynamics in grasslands. There has been a surge of research activity directed at improving our understanding of the biogeochemical cycles in grasslands and the factors (climate and land use practices) affecting them (Prentice *et al.* 2001). However, it is difficult to predict effects of any change in management and climate on the ecosystem dynamics through field experimentation alone. This is because most field experiments are conducted for short time periods, and thus are not long enough to capture the dynamics of environmental processes (Thornley and Cannell 1997, Mitchell and Csillag 2001). Hence, a modelling approach is used to explore a given landscape's ecological issues under changing climate and land use management as models can be used without any disturbance to the study area and can be used repeatedly. The most important characteristic of models is that they can provide context and permit exploration of more combinations or a greater range of environmental and management conditions than we can assess in the field. In this study, it was not possible to assess the influence of grazing intensity

on the grassland carbon dynamics in the previous chapters as the field experiment was conducted for only one season which is not enough time to assess changes in belowground carbon stocks (Saha *et al.* 2010). The advantages of a modeling approach to explore potential impacts of grazing on carbon stocks at study area includes assessment at longer time periods, and exploration of more combinations or a greater range of environmental and management conditions than what can be assessed in the field.

6.1.2 The grazing history of the Study Area: Grasslands National Park, SK, Canada

The study area (Grasslands National Park, GNP) has never been fragmented by cultivation, nor heavily utilized by livestock in the time between homesteading in the early 1900s and purchase by Parks Canada. A few impoundments along Horse Creek and Weatherall Creek provided the stock water for the livestock. Starting in the 1930s, the land was summer grazed by cattle and in some cases a combination of horses and cattle were grazed year-round (Poirier 1993). This moderate grazing pressure continued until the complete livestock exclusion after purchase of the SW portion in 1990 and the NE portion in 1991 (Parks Canada 2006). However, in 2006 bison were released in the West Block of the park as a management tool to restore and preserve wildlife habitats, contributing to the maintenance of ecological integrity (Parks Canada 2009). Similarly, in 2008 cattle were released in the East Block of the park and adjoining Mankota community pasture as part of the Biodiversity and Grazing Experiment (BGE). The goal of the experiment was to see the effect of variable grazing intensities on the plant heterogeneity in the park. The modelling experiment reported here concentrates on the cattle-grazed area in the East Block of the park.

6.1.3 The CENTURY Model

The CENTURY model, described in detail earlier in this dissertation (section 3.5.1), was used to simulate vegetation, soil carbon and total plant system carbon responses to grazing and soil texture. This model has been widely used and validated for many different grassland sites in temperate and tropical regions (see Burke *et al.* 1991; Holland *et al.* 1992; Parton *et al.* 1993; Gilmanov *et al.* 1997; Mikhailova *et al.* 2000; Mitchell and Csillag 2001; Ardo and Olsson 2003). In addition, the option of scheduling events of interest to managers, such as managed grazing at specific times during the simulations, was crucial to the research as it helped in simulating the impact of variable grazing intensities (light, moderate and heavy) on plant productivity and grassland carbon dynamics over long time-periods.

6.1.4 Modeled variables of interest

From the analysis of data in the previous chapters of this dissertation, I concluded that grazing intensity and local weather were important factors causing heterogeneity in soil moisture and ALB. Therefore, these factors were considered in the modeling process. In addition, past research has shown that plant available water in a soil is affected by soil properties such as texture (Famiglietti *et al.* 1998; Wilson *et al.* 2005). For example, soils with high clay content have very large surface areas, resulting in the soil being able to hold more water and having a higher chemical reactivity (the ability to store more nutrients and supply them to plants, Cation Exchange Capacity). Despite the importance of soil texture, there is a lack of detailed regional soils data for the study area, especially with respect to the way it varies in space. Therefore, this factor was also considered in the model. Results from previous chapters provided the baseline conditions to run the model.

6.1.5 Modeling Goals and Hypotheses

The CENTURY model was used to investigate how grassland productivity, total soil carbon (SOMTC in g m^{-2}), and total plant system carbon (TOTSYC in g m^{-2}) are influenced by variability in grazing intensity. SOMTC is a total soil carbon including belowground structural and metabolic. Total plant system carbon includes carbon in above- and below-ground living biomass, carbon in soil organic matter, carbon in standing dead material and carbon in litter. Additionally, the model was used to assess how belowground processes varied in relation to soil texture while holding the climate and cover type constant. Modelled predictions were tested for sensitivity to % clay and % sand within a soil texture class and the fraction of live shoots removed during a grazing event. This helped to see how much uncertainty in soil parameters and fraction of live shoots removed per month by grazers would impact the carbon dynamics. By evaluating the sensitivity, one can not only evaluate the uncertainty in predictions but also get an idea of the range of possible outcomes that need to be incorporated when making decisions regarding grasslands management. It is expected that knowledge of how much uncertainty is there within an important parameter will lead to better decisions in the long run than ones based on ignorance of uncertainty. Additionally, sensitivity analysis will also help in gaining knowledge about important parameters that should be taken into consideration during the field experiments. It was hypothesized that:

(a) Model predictions for grassland productivity (above and below ground biomass) and total soil and plant system carbon are sensitive to the soil texture due to differences in water holding capacity and the resulting impacts on productivity and other carbon cycling processes.

(b) Aboveground net primary productivity increases with grazing intensity whereas belowground net primary productivity and soil carbon decrease with grazing intensity. This is expected because plants under grazing pressure will replace lost leaf area by allocating more resources to leaf versus root growth (as per Holland *et al.* 1992).

Since GNP is considering terminating grazing in the East Block by 2013, the impact of grazing termination on ALB and carbon dynamics was also tested. Snyman (1998) and Savory (1999) suggests that rangelands will respond in different ways to any changes implemented in a grazing system (example, rotational grazing VS continuous grazing) and the duration of grazing/rest period. Therefore, the objective of this simulation was to determine the degree to which land management decisions, for example, continuous grazing and grazing termination, affect the predicted productivity.

6.1.6 Addressing the Hypotheses: Two Modeling Scenarios

To address these hypotheses, two scenarios were run using the CENTURY model. Scenario 1 looked at impact of variable grazing intensities on annual plant productivity and dynamics of carbon, where the model was parameterized with the following grazing schedules: no grazing (1991 to 2005); grazing with light, moderate and heavy intensity (2006 to 2020). Results for scenario 1 are presented in section 6.2.2 and 6.2.3. Since GNP is considering terminating grazing in the East Block by 2013, the impact of grazing termination on ALB and carbon dynamics was also tested. For scenario 2, the model was run with following schedules: no grazing (1991 to 2005); grazing with light, moderate and heavy intensity (2006 to 2012) and grazing termination (2013 to 2020). Results for scenario 2 are presented in section 6.3. For

details on the grazing experiment refer to section 3.1 and on modeling parameterization and set-up refer to section 3.5.2.

6.1.7 Model Evaluation

The evaluation of the model included an empirical test of its main assumptions, a sensitivity analysis and the comparison of the model's output against field data. A set of simulation experiments using a common climate scenario were performed to evaluate the response of the system to different grazing intensity and soil texture scenarios.

6.2 Results

6.2.1 Validation of the CENTURY model

Field measurements for ALB taken during summer 2008 were used to evaluate the results of the model. Intra-seasonal patterns of ALB and responses to moisture availability showed some agreement with the field measurements and expected patterns in the study area (Table 6.1).

Table 6.1 Predicted and measured Above-ground live plant Biomass (ALB in g m^{-2}), Grasslands National Park

Date	CENTURY ALB (g m^{-2})	Measured ALB (g m^{-2}) (± 1 std. dev.)
March 2008	1.26	
April 2008	27.09	
May 2008	53.03	42.90 (±9.58)
June 2008	89.44	80.86 (±31.01)
July 2008	66.62	
August 2008	49.39	38.91 (±10.45)
June 2009*	55.26	40.75 (±7.39)
June 2010*	44.72	39.77 (±12.85)

Note: Predictions are CENTURY monthly predictions of ALB (g m^{-2}) for 2008 for Clay-loam soil texture. Measured ALB values are averaged field measured ALB values from 2008 growing season, where ALB is derived from NDVI measurements using Equation 3.2.
*Data from June 2009 and 2010 are satellite based ALB estimates.

Modelled values of ALB were slightly greater than observed values but were within +/- 1 SD of the observed ALB in each month. This deviation in values can be attributed to some uncertainty in both the (spatially variable) measured ALB and the parameterization used in the CENTURY model for ALB prediction, but the overall seasonal pattern was judged to be acceptable for the purposes of predicting future trends.

6.2.2 Model Predictions

6.2.2.1 Impact of soil texture on the ALB, SOMTC and TOTSYC

One-way ANOVA was conducted to confirm the importance of soil texture on predicted ALB and soil carbon dynamics. Factors examined in the model were: soil texture, ALB, SOMTC, TOTSYC and their interactions. Test concluded that soil texture significantly affects ALB ($F(4, 115) = 2.4$; $p = 0.049$), and SOMTC ($F(4, 115) = 49023$; $p < 0.05$), and TOTSYC ($F(4, 115) = 16309$; $p =< 0.05$. Among soil textures, ALB for clay loam (CL) soil (44.7 g m^{-2}) was slightly higher than for clay (C) (41.8 g m^{-2}) and loam (L) (41.6 g m^{-2}). Sandy loam (SaL) soil showed the lowest mean ALB (35.6 g m^{-2}) than all other textures. Post hoc tests confirmed a significant difference in the mean ALB for clay loam soil and sandy loam soil ($p < 0.05$). However, no significant difference in the mean ALB for clay and loam was observed ($p > 0.05$).

Figure 6.1 shows the predicted ALB under heavy grazing pressure for the six soil texture classes found in the park. In general, the peak in ALB typically occurred in June or July depending on the amount of growing season precipitation. For the 2008 growing season (April to August), both field and modelled ALB peaked in June.

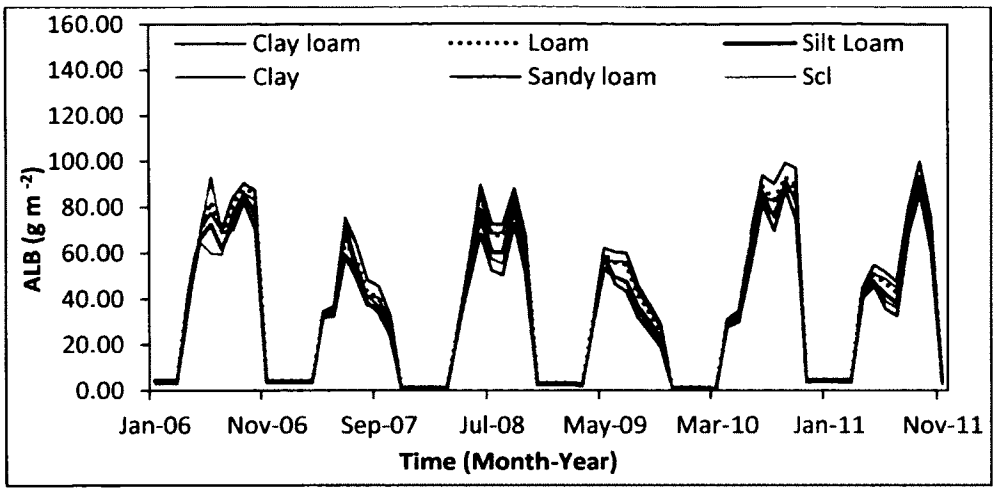


Figure 6.1 Effects of soil texture on ALB (g m^{-2}): An example of variability under heavy grazing (*flgrem* value = 0.6). Note: Scl = Sandy clay loam.

Table 6.2 shows effect of soil texture on ALB for light, moderate and heavy grazing pressure using an example from June 2008. In general, out of all the soil textures, ALB was predicted to be higher on clay loam soil for all the grazing intensities (light, moderate and heavy), whereas sandy loam was predicted to lead to the lowest ALB. However, an exception was early summer 2006 (Figure 6.1) when CL did not have the highest ALB. Additionally, in some years the differences in predicted ALB were also greater than others. This fluctuation could be attributed to stochastically generated precipitation and the mean temperature values used to run the model simulations and the effect of precipitation on the amount of plant available water for growth.

Table 6.2 Effects of soil texture on ALB (g m^{-2}): An example of June 2008 ALB under light, moderate and heavy grazing

	ALB (g m^{-2}) during June 2008		
Soil Texture	**Light grazing**	**Moderate grazing**	**Heavy grazing**
Clay loam (CL)	71.5	76.7	89.6
Clay (C)	69.4	74.2	85.7
Loam (L)	67.9	72.8	85.6
Silt Loam (SL)	63.1	67.7	78.6
Sandy clay loam (Scl)	58.3	65.4	73.9
Sandy Loam (SaL)	53.7	60.1	68.1

The post hoc test results also indicated that mean SOMTC and TOTSYC values were significantly different between the soil textures ($p < 0.05$). Clay soil lead to the highest mean SOMTC and TOTSYC values (9.9 kg m^{-2} and 10.8 kg m^{-2} respectively) followed by clay loam, loam, silt loam, sandy clay loam and sandy loam soil. Sandy loam soil caused the lowest predicted SOMTC and TOTSYC values compared to other soil textures in the study area.

Both SOMTC (Pearson's r (1810) = 0.9, $p < 0.01$) and TOTSYC (Pearson's r (1810) = 0.9, $p < 0.01$) also showed a very strong and significant correlation with the % clay in a soil texture. In comparison, ALB showed a weak but significant correlation with the soil texture (Pearson's r (1810) = 0.06, $p = 0.02$). Soil texture is related to important factors for plant growth such as plant available water and soil fertility (Lauenroth *et al.* 1999). However, the weak correlation between ALB and soil texture is most likely a result of additional factors influencing ALB such as the change in management conditions (light to heavy grazing), where grazers alter the soil structure by trampling, thus affecting the soil water holding capacity, as well as, plant available water over time.

6.2.2.2 Impact of grazing intensity on the grassland productivity and total soil and plant system carbon (Scenario 1)

(A) Results for predicted SOMTC and TOTSYC

A MANOVA test was conducted to see if treatment had any effect on the grassland soil carbon (SOMTC) and total plant system carbon (TOTSYC) dynamics. Treatment significantly affects the SOMTC ($F (3, 115) = 18.7$; $p < 0.05$) and TOTSYC ($F (3, 115) = 25.9$, $p < 0.05$). After 14-yr of grazing treatment, both mean SOMTC and TOTSYC values were found to be significantly different between the light, moderate and heavy grazing intensity ($p < 0.05$).

Figure 6.2 and Figure 6.3 show the results from the grazing period (2006 to 2020) for the predicted SOMTC and TOTSYC. Light and moderate grazing intensity showed higher total belowground soil carbon and total plant system carbon compared to heavy grazing intensity.

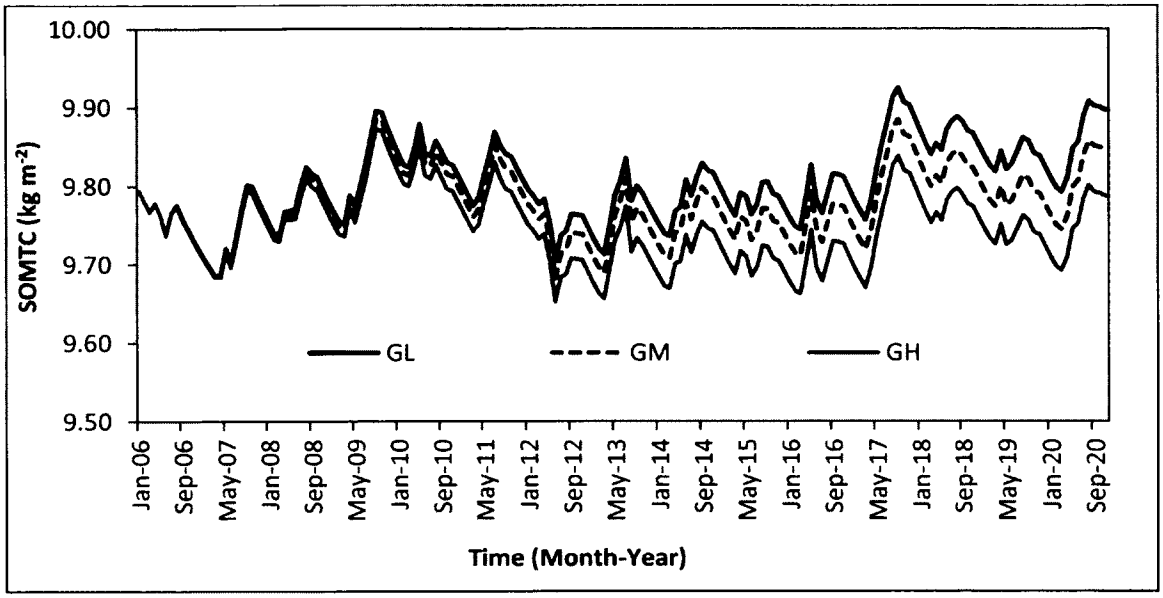


Figure 6.2 Effect of light, moderate and heavy grazing intensity on predicted SOMTC (includes belowground structural and metabolic) from 2006 to 2020.
Note: GL = light grazing; GM = moderate grazing; GH = heavy grazing.

Both SOMTC (Pearson's r (120) = -0.46, $p < 0.0001$) and TOTSYC (Pearson's r (120) = -0.50, $p < 0.0001$) also showed a significant correlation with the treatment (grazing intensity). In general, model predictions for SOMTC (Figure 6.2) and TOTSYC (Figure 6.3) showed a decreasing linear trend with increase in grazing intensity. Light grazing had more dead organic matter and plant litter compared to heavy grazing scenario.

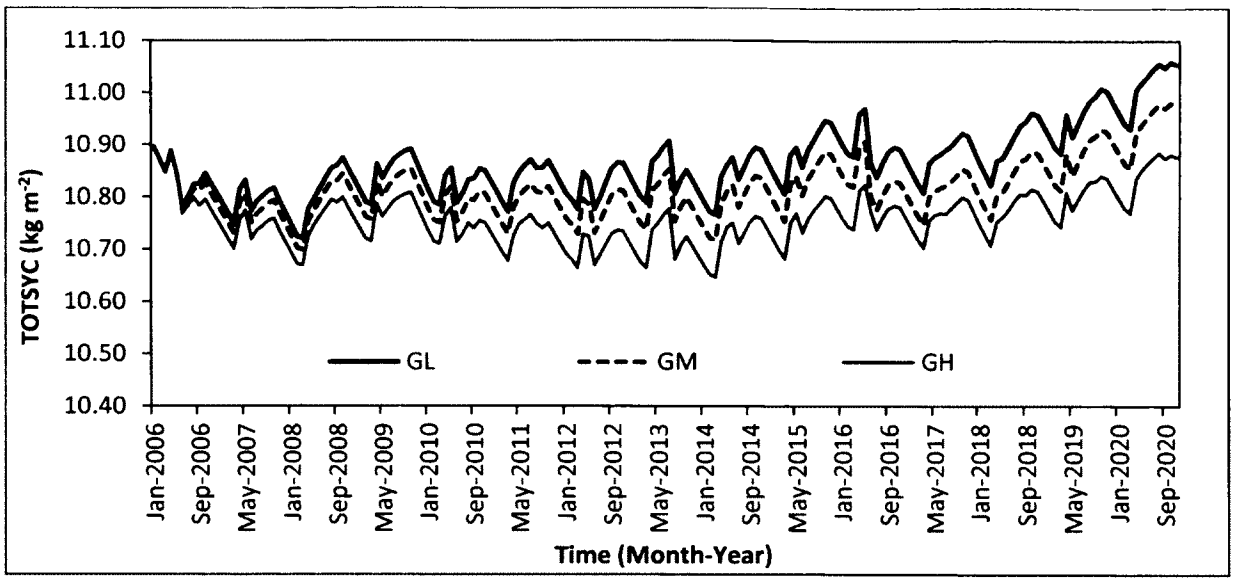


Figure 6.3 Effect of light, moderate and heavy grazing intensity on predicted TOTSYC from 2006 to 2020. Note: GL = light grazing; GM = moderate grazing; GH = heavy grazing.

(B) Results for predicted annual net primary productivity (above- and below-ground)

Figure 6.4 presents the CENTURY predicted annual net primary productivity (NPP) for each grazing regime. In this figure, years 1991 to 2005 show the effects of no grazing on annual NPP, whereas years 2006 to 2020 shows the effects of light, moderate and heavy grazing intensities on the annual NPP.

To compare the above- and below-ground NPP between different grazing intensities, a repeated–measures factorial ANOVA with treatment (grazing intensity) as the main factor and time (year) as the repeated measure ($\alpha = 0.05$) was used. Simulation with no grazing treatment from year 1991 to 2020 was used as a control. Mauchly's Test indicated that the assumption of sphericity had been violated, $p < 0.05$, therefore, degrees of freedom were corrected using Greenhouse-Geisser estimates of sphericity.

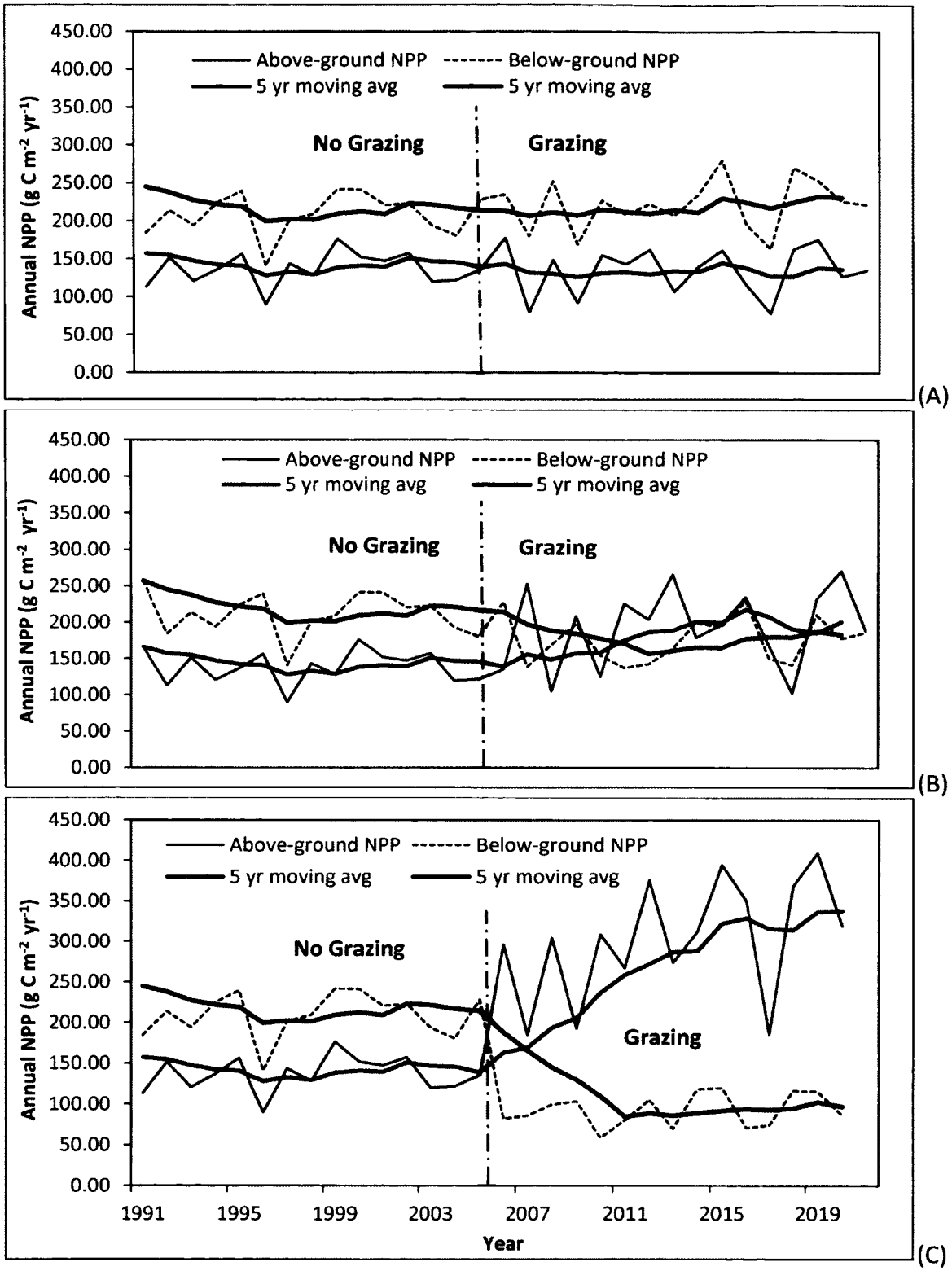


Figure 6.4 Effect of ungrazed (1991 – 2005) and grazed (2006 - 2020) conditions on annual NPP (CENTURY predicted NPP) variability. Light (A), Moderate (B) and Heavy (C) grazing.
No Grazing = year 1991 to 2005 and Variable Grazing Intensity = year 2006 to 2020. Red and Green lines are 5 year moving averages of predicted above-ground and below-ground NPP respectively on clay (C) soil.

Both treatment and time significantly affected the annual above- and below-ground NPP (treatment: above-ground NPP, $F(3.5, 115) = 18.4$, $p < 0.0001$, and below-ground NPP, $F(3.5, 115) = 16.5$, $p < 0.05$; time: above-ground NPP, $F(1, 115) = 28.8$, $p < 0.001$, and annual below-ground NPP, $F(1, 115) = 5.4$, $p < 0.001$). Approximately 25.5% of the total variance in the above-ground NPP and 20.1% in the below-ground NPP were accounted for by the variance in the time, while 72% of the total variance in the above-ground NPP and 79.8% in the below-ground NPP were accounted for by the variance in the treatment. Main effects of treatment and time were qualified by an interaction of treatment and time ($F(4.2, 230) = 36.8$, $p < 0.0001$) with 69.8% of total variance in above- and below-ground NPP as a result of treatment and time. The significant interaction indicated that the magnitude of the grazing intensity effect varied with year. There was significant difference in the mean annual above- and below-ground NPP between the control, light, moderate and heavy grazing intensity (Tukey's HSD test, $p < 0.05$) indicating that variation in treatment affects both above- and below-ground NPP.

Overall, out of light, moderate and heavy grazing treatment, mean annual above-ground NPP was highest for the heavy grazing intensity (220.7 g C m^{-2} yr^{-1}) followed by moderate grazing (166.2 g C m^{-2} yr^{-1}) and light grazing (136.6 g C m^{-2} yr^{-1}). In terms of below-ground NPP, the light grazing scenario showed the highest mean (216.1 g C m^{-2} yr^{-1}) than moderate (193.2 g C m^{-2} yr^{-1}) and heavy grazing (151.6 g C m^{-2} yr^{-1}) scenarios. Simulations based on light, moderate and heavy grazing intensity showed variability in the annual above-ground and below-ground NPP (g C m^{-2} yr^{-1}) from 2006 to 2020.

Table 6.3 shows the minimum, maximum, range of variability, 25th percentile, 75th percentile and the inter-quartile range for above-ground- and below-ground NPP over a 14 year time period (2006 to 2020). The results indicate high range of variability in the above-ground NPP with increase in the grazing intensity. For example, heavy grazing showed the highest range of variability whereas the light grazing intensity showed the lowest range of variability. To ensure that the extreme values do not impact the range of variability, 25th and 75th percentile were also calculated to determine the inter-quartile range. Similarly, below-ground NPP also showed high range of variation as a result of grazing intensity. However, with increase in the fraction of live shoots removed by the grazers (heavy grazing), the range of variability decreased for the below-ground productivity.

Table 6.3 Range of variability in above-ground- and below-ground NPP (g C $m^{-2} yr^{-1}$) for low, moderate and heavy grazing intensity (2006 to 2020)

	Light Grazing		**Moderate Grazing**		**Heavy Grazing**	
	Above-ground NPP	**Below-ground NPP**	**Above-ground NPP**	**Below-ground NPP**	**Above-ground NPP**	**Below-ground NPP**
Min value	77.8	162.8	102.7	137.1	184.6	58.7
Max value	177.7	279.4	270.7	229.8	409.3	120.0
Range	99.8	116.6	168.0	92.7	224.7	61.3
Q1	110.5	200.6	151.7	146.4	270.4	77.0
Q3	161.5	243.3	233.4	198.8	359.3	110.3
Inter-quartile Range	51.0	42.7	81.7	52.4	88.9	33.4

Note: Q1 = 25th percentile; Q3 = 75th percentile.

Mean annual above- and below-ground NPP was also significantly different between ungrazed (year 1991 to 2005) and grazed (year 2006 to 2020) period ($p < 0.0001$). In the light grazing intensity scenario, the mean annual above-ground NPP was slightly lower, while mean annual below-ground NPP was higher during the grazing period (2006 to 2020) compared to ungrazed

period (1991 to 2005). In comparison, above-ground productivity showed an increasing trend as a result of moderate grazing intensity from 2006 to 2020. Similarly, the heavy grazing scenario showed an increasing trend in the above-ground productivity and a decreasing trend in the below-ground productivity during grazing compared to the no-grazing period (Figure 6.4C).

6.2.2.3 Effect of light, moderate and heavy grazing intensity on the model predictions for ALB

One-way ANOVA was conducted to confirm the importance of treatment on predicted ALB. Treatment significantly affected the monthly averaged ALB ($F(3, 416) = 34.0$; $p < 0.05$, $N = 420$), where grass production decreased with increase in grazing intensity. Significant differences in the predicted ALB values were found between light and moderate grazing ($p = 0.023$); and light and heavy grazing ($p = 0.0001$). However, there was no significant difference in the mean ALB values between moderate and heavy grazing ($p = 0.864$) (Figure 6.5).

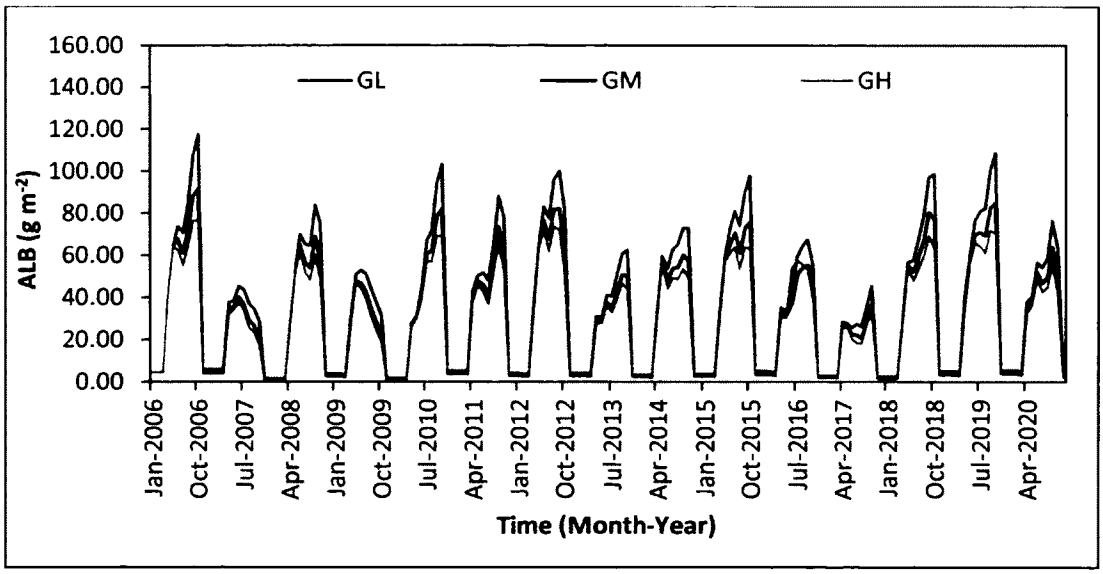


Figure 6.5 Effect of light, moderate and heavy grazing intensity on the variability in predicted ALB (g m^{-2}) from 2006 to 2020.
Note: GL = light grazing; GM = moderate grazing; GH = heavy grazing.

Grazing intensity was not able to explain a large amount of the seasonal ALB variance in the model. This is likely due to the many other factors that influence ALB in the model such as weather and soil moisture.

6.2.3 Sensitivity analyses

The following sections specifically look at the sensitivity of the model predictions to changes in soil texture parameterizations within a soil texture class and changes in fraction of live shoots removed during a grazing event. The goal was to see which one of these two input model parameters was more important from rangeland management perspective, as well as to get an idea of the range of possible outcomes that need to be incorporated by managers in decision-making process.

6.2.3.1 Sensitivity to change in soil texture parameterizations

The sensitivity model simulations with change in % sand, % silt and % clay proportions in 5% increments (explained earlier in Chapter 3's modeling section) within a soil texture class (example, clay-loam) show a clear trend in ALB despite marginal changes (Figure 6.6). Overall, predicted ALB based on sensitivity analysis increase when there is more % sand and less % silt within a same soil texture class (Figure 6.6C and Figure 6.6D). For example, S1 showed mean ALB of 44.7 g m^{-2} compared to S2 with 46.5 g m^{-2} and S3 with 46.9 g m^{-2} during ungrazed period (1991 – 2005). In addition, the effect of weather was also observed on the model predictions. For example, during growing season of year 1991, S1 showed a mean ALB of 57.2 g m^{-2}, compared to S2 with 58.5 g m^{-2} and S3 with 59.0 g m^{-2}.

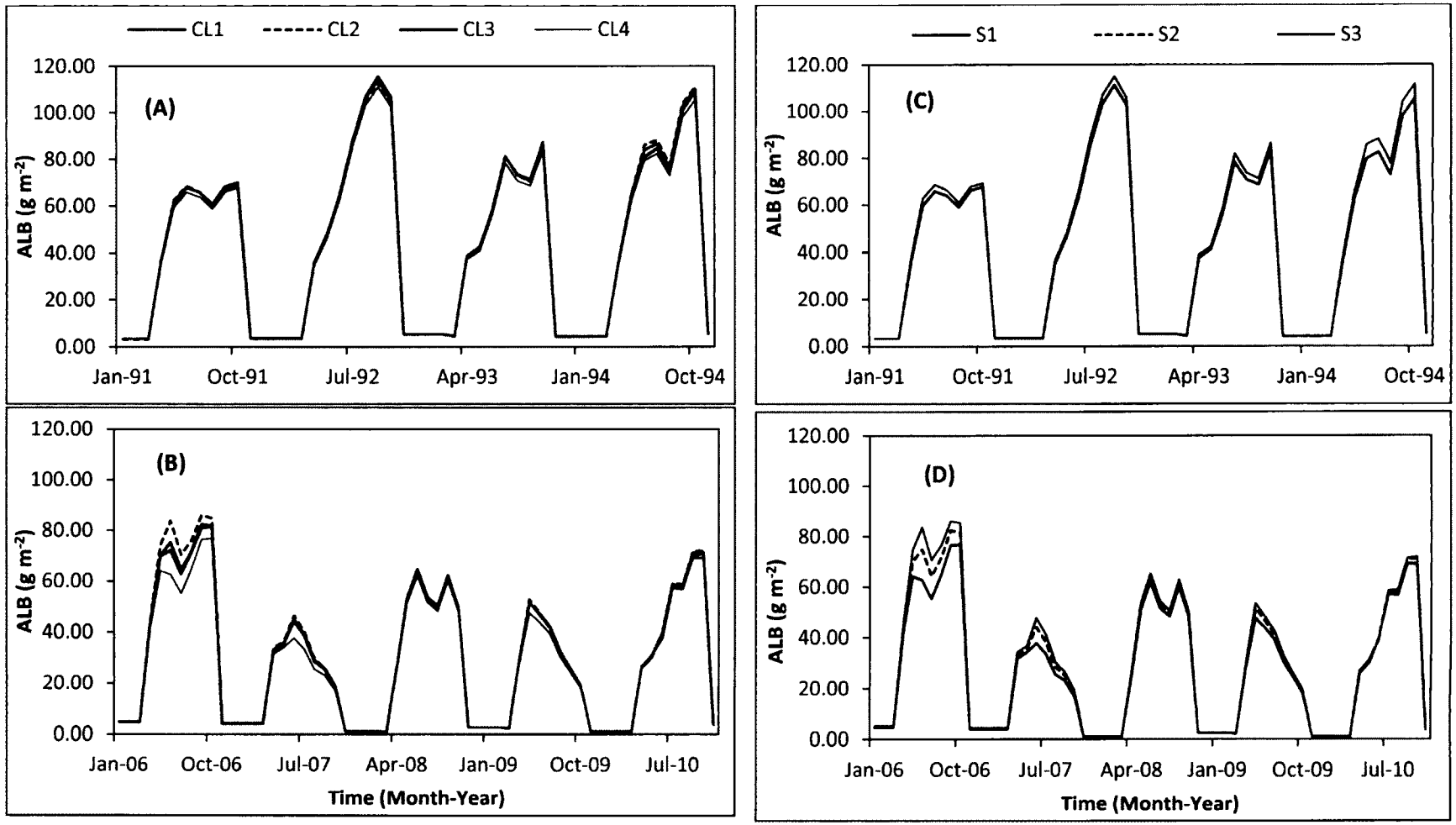


Figure 6.6 ALB sensitivity to variability in sand, silt and clay proportions within a clay-loam soil texture under ungrazed (A and C) and grazed (B and D) conditions. Here, ungrazed period = 1991 – 1994 and grazed period with heavy grazing = 2006 – 2010

Figures A and B: CL = clay-Loam; CL1 = sand 40%, silt 24.06%, clay 35.94%; CL2 = sand 44%, silt 16.31%, clay 39.69%; CL3 = sand 35%, silt 35%, clay 30%; CL4 = sand 22%, silt 42.06%, clay 35.94%. In this simulation % clay was changed in increments of 5 % to see the effect of change in % clay within a soil texture on the model predictions.

Figures C and D: S1 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29; S2 = sand 39%, silt 25.06%, clay 35.94%, BD = 1.32; S3 = sand 44%, silt 20.06%, clay 35.94%, BD = 1.34. In this simulation % clay was held constant to see the effect of change in % sand and % silt within a soil texture on the model predictions.

In contrast, during 1994 growing season, S1 showed a mean ALB of 66.7 g m^{-2}, compared to S2 with 69.6 g m^{-2} and S3 with 71.1 g m^{-2}. Noticeable differences in ALB were observed during 2006 and 2007. This could be a combined effect of grazing disturbance which started in year 2006 and weather. For example, more rainfall was recorded during year 2006 (151.9 mm) and 2007 (174.6 mm) growing season compared to year 2008 (121.8 mm) and 2009 (147.4 mm).

Data were analysed using a mixed-design ANOVA with a within-subjects factor of time (ungrazed and grazed) and texture (CL1, CL2, CL3, CL4). A simulation with no grazing treatment from year 1991 to 2020 was used as a control. Mauchly's Test indicated that the assumption of sphericity had been violated, χ^2 (5) = 194.5, $p < 0.001$, and, therefore, degrees of freedom were corrected using Greenhouse-Geisser estimates of sphericity (ε = 0.70). There was a significant effect of time (ungrazed or grazed) (F (1, 104) = 55.2, $p < 0.05$) and texture on ALB (F (2, 217) = 53.0, $p < 0.05$). Main effects of time and texture were qualified by an interaction between time and texture (F (1.8, 195) = 3.0, $p = 0.05$). This implies that variability in ALB is a result of the interaction of grazing disturbance introduced over time with texture rather than effect of texture alone. For example, mean ALB for CL1 during ungrazed period was 46.7 g m^{-2} compared to 29.3 g m^{-2} during grazed period. Furthermore, when grazing disturbance was increased by 30 %, the mean ALB also showed increase from 29.3 to 43.1 g m^{-2} during the grazing period. This is due to the effect of grazers on the soil structure as a result of trampling. Tukey's HSD test also indicated significant differences in the mean ALB between the ungrazed period (1991 to 2005) and grazed period (2006 to 2020). Overall, no significant difference in the mean ALB was determined for textures CL1, CL2 and CL3 ($p > 0.05$). However, mean ALB was significantly different between CL1 and CL4; CL2 and CL4; CL3 and CL4 ($p < 0.05$) both during grazed and

ungrazed period suggesting importance of proportion of % sand and % silt within a texture (Table 6.4).

Table 6.4 ALB sensitivity to variability within clay loam soil texture during grazed and ungrazed period

	Mean ALB in CL1	Mean ALB in CL2	Mean ALB in CL3	Mean ALB in CL4
Ungrazed period (1991 – 2005)	46.7 g m^{-2}	46.1 g m^{-2}	46.8 g m^{-2}	44.7 g m^{-2}
Ungrazed period (2006 – 2020)	44.8 g m^{-2}	45.1 g m^{-2}	45.2 g m^{-2}	43.0 g m^{-2}
Grazed period (2006 – 2020)	29.3 g m^{-2}	29.6 g m^{-2}	29.7 g m^{-2}	28.6 g m^{-2}

Note: CL1 = sand 40%, silt 24.06%, clay 35.94%, BD = 1.32; CL2 = sand 44%, silt 16.31%, clay 39.69%, BD = 1.31; CL3 = sand 35%, silt 35%, clay 30%, BD = 1.34; CL4 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29

Figure 6.7 shows comparison for ALB (g m^{-2}) between two simulations where Scenario 1 was run with an option of no grazing from 2006 – 2020 and Scenario 2 was run with an option of grazing with heavy intensity from 2006 – 2020. In this figure effect of grazing on ALB is clearly visible, where introduction of grazing in Scenario 2 shows reduction in ALB compared to Scenario 1 with no grazing disturbance.

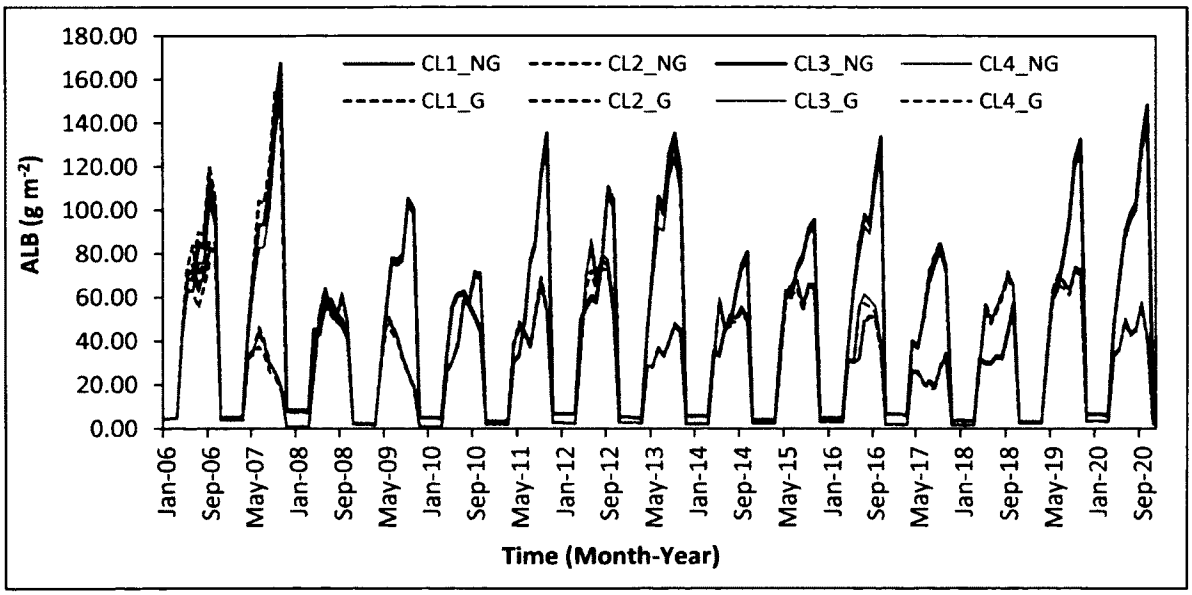


Figure 6.7 Effect of variability within Clay loam soil texture class on predicted ALB (g m^{-2}).
Note: CL = clay loam; NG = Not Grazed, G = Grazed; CL1 = sand 40%, silt 24.06%, clay 35.94%, BD = 1.32; CL2 = sand 44%, silt 16.31%, clay 39.69%, BD = 1.31; CL3 = sand 35%, silt 35%, clay 30%, BD = 1.34; CL4 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29. The graph shows comparison between two simulations where Scenario 1: 2006 – 2020 no grazing; Scenario 2: 2006 – 2020 grazed with heavy intensity.

The soil texture (sand, silt and clay fractions) and bulk density (g cm^{-3}) values provided for the study area in the CENTURY model were used to calculate the wilting point, field capacity and plant available water using equations developed by Rawls *et al.* (1982). Any change in sand and silt proportions within a soil texture will also affect its bulk density resulting in change in wilting point, field capacity and amount of water available to plants for growth; this explains the variability in the ALB patterns during Scenario 1, example CL1 and CL4, as predicted by the model. Since grazers affect the soil structure due to trampling, this will affect the bulk density of the soil, as well as amount of water available to plants for growth. This explains the variability in ALB patterns from Scenario 1 to Scenario 2.

Similar to ALB, Mauchly's Test indicated that the assumption of sphericity had been violated for SOMTC, χ^2 (5) = 160.3, $p < 0.05$ and TOTSYC; χ^2 (5) = 132.8, $p < 0.05$, therefore, degrees of freedom were corrected using Greenhouse-Geisser estimates of sphericity (ε = 0.56 for SOMTC; and ε = 0.63 for TOTSYC). Even with the adjustments, the within-subject effects of time-period was not significant for predicted SOMTC (F (1, 167) = 1.07, p = 0.303) (Figure 6.8). This means that the predicted SOMTC of the ungrazed and grazed periods were not significantly different. This was further shown by a very low Partial Eta-squared value, 0.006, which indicates that < 1% of the total variance in the dependent variable SOMTC is accounted for by the variance in the independent variable, time.

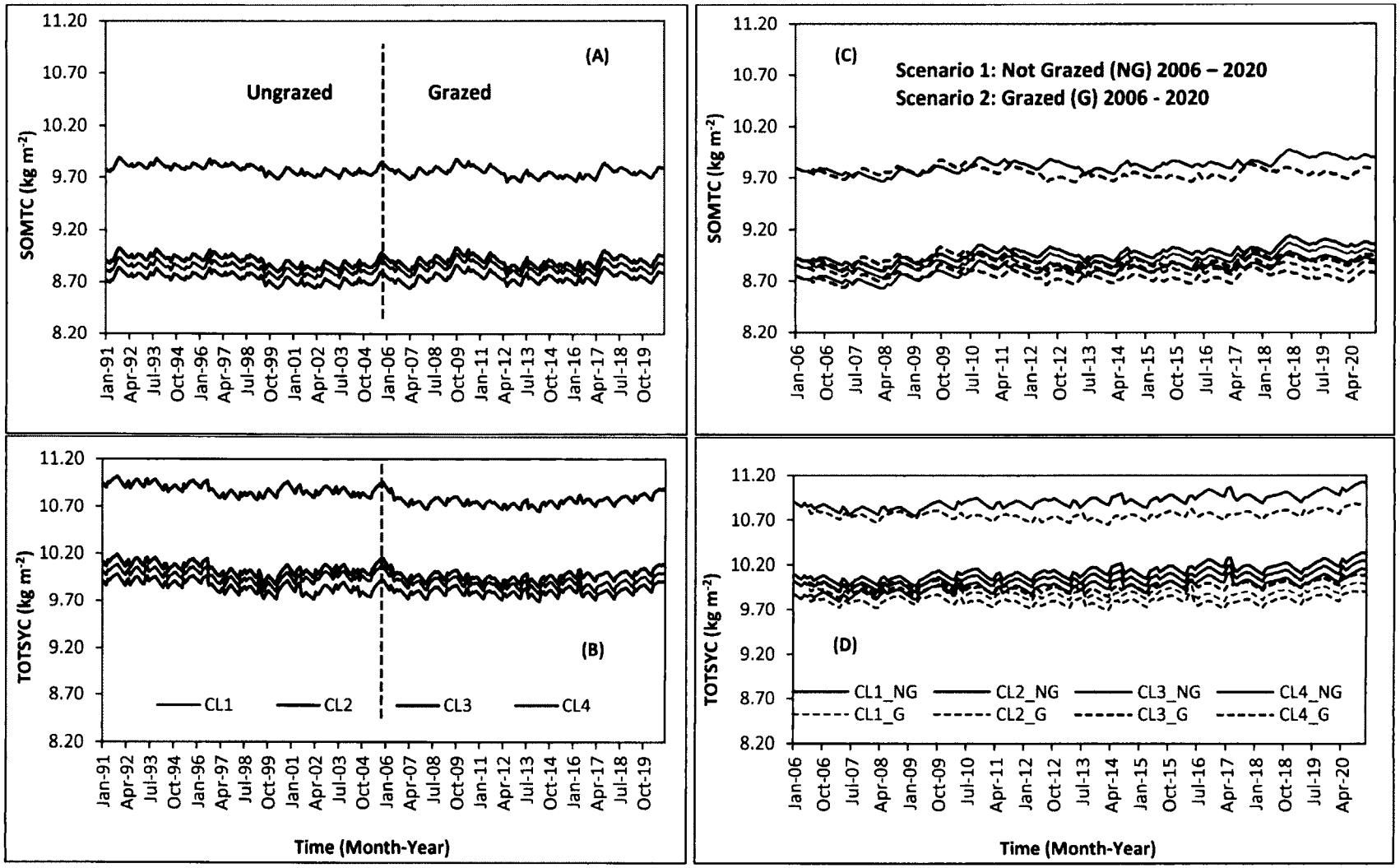


Figure 6.8 Effect of variability within Clay loam soil texture class on predicted SOMTC (A, C) and TOTSYC (B, D).

Note: CL = clay loam; CL1 = sand 40%, silt 24.06%, clay 35.94%, BD = 1.32; CL2 = sand 44%, silt 16.31%, clay 39.69%, BD = 1.31; CL3 = sand 35%, silt 35%, clay 30%, BD = 1.34; CL4 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29. In Figures C and D, NG = Not Grazed, G = Grazed.

Texture significantly affected the predicted SOMTC (F (1.1, 198.4) = 377.5, $p < 0.05$). Approximately 99% of the total variance in the SOMTC was accounted for by the variance in the texture. Tukey's HSD test indicated significant difference in the mean SOMTC between CL1, CL2, CL3 and CL4 indicating that sand, silt and clay proportion affects the predicted SOMTC. CL4 showed the highest mean SOMTC (9.8 kg m^{-2}) than CL3 (8.9 kg m^{-2}), CL2 (8.7 kg m^{-2}) and CL1 (8.8 kg m^{-2}).

Compared to predicted SOMTC, both time (F (1, 179) = 362.3, $p < 0.05$) and texture significantly affected the predicted TOTSYC (F (1.8, 333) = 295.6, $p < 0.05$). Approximately 10.8% of the total variance in the TOTSYC was accounted for by the variance in the time (grazed and ungrazed), while ~90% of the total variance in the TOTSYC was accounted for by the variance in the texture. Main effects of time and variation within texture were qualified by an interaction between time and texture (F (2.1, 379.1) = 392.6, $p < 0.05$) with 88.4% of total variance in TOTSYC as a result of time and texture. Tukey's HSD test indicated that mean TOTSYC values were significantly ($p < 0.05$) different between CL1, CL2, CL3 and CL4 indicating that variation in sand, silt and clay proportion affects the predicted TOTSYC. Similar to SOMTC results, CL4 showed the highest predicted TOTSYC values (10.8 kg m^{-2}) compared to CL1, CL2 and CL3 (Figure 6.8). Mean TOTSYC for CL2 (9.8 kg m^{-2}) was significantly lower than CL1 (9.9 kg m^{-2}).

6.2.3.2 Sensitivity to change in fraction of live shoots (*flgrem*) per month during a grazing event

ALB, SOMTC and TOTSYC sensitivity to change in fraction of live shoots (*flgrem*) removed per month during a grazing event was also tested. An example of effect of change in *flgrem* value on ALB is provided in Figure 6.9. The results are based on a simulation run with heavy grazing

option (grzeff = 2 with quadratic impact on aboveground production and root/shoot ratio) for clay-loam soil texture. The heavy grazing intensity with changed fraction of live shoots (*flgrem*) value (FLGREM0.4 to 0.6) lead to higher ALB values than the regular heavy grazing parameterization with *flgrem* value of 0.3. This means that the predicted ALB is sensitive to the parameter *flgrem* in the model.

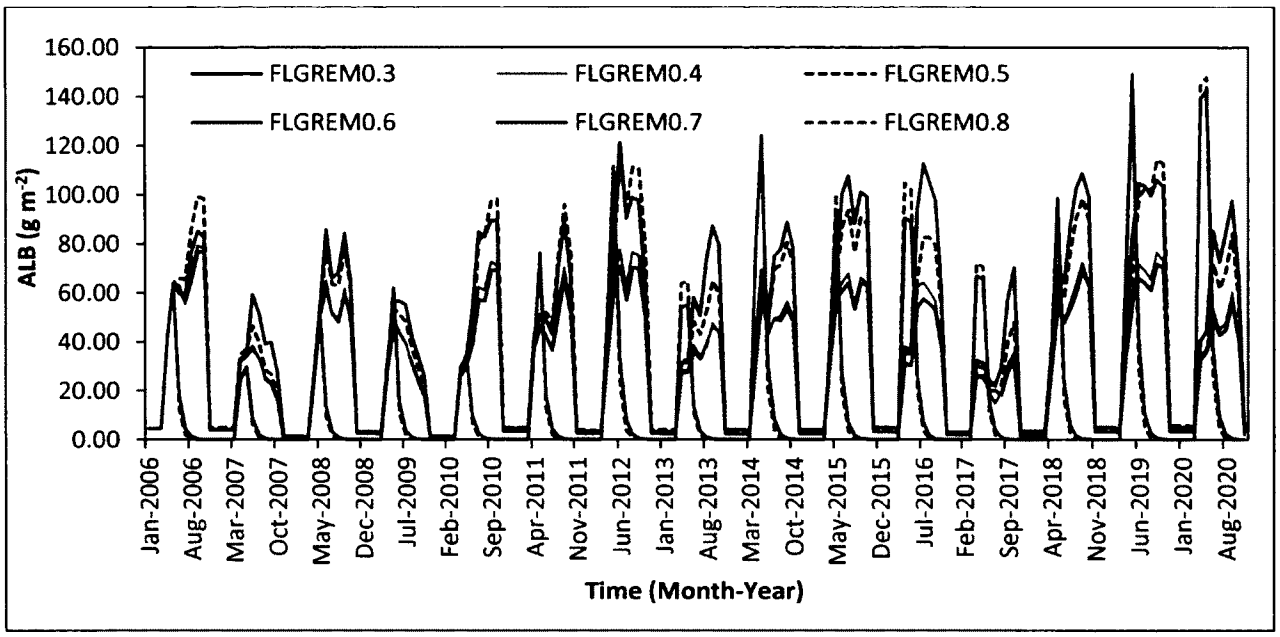


Figure 6.9 Effect of changed fraction of live shoots (*flgrem*) removed per month for heavy grazing option on predicted ALB.

Note: FLGREM = fraction of live shoots removed during a heavy grazing option; FLGREM0.3 = fraction of live shoots removed with *flgrem* value of 0.3; FLGREM0.4 = fraction of live shoots removed with *flgrem* value of 0.4; FLGREM0.5 = fraction of live shoots removed with *flgrem* value of 0.5; FLGREM0.6 = fraction of live shoots removed with *flgrem* value of 0.6; FLGREM0.7 = fraction of live shoots removed with *flgrem* value of 0.7; FLGREM0.8 = fraction of live shoots removed with *flgrem* value of 0.8.

Furthermore, when *flgrem* value of 0.7 and 0.8 was used, the model predictions showed a different pattern for ALB. The model with FLGREM0.7 and FLGREM0.8 predicted the peak rate of vegetation growth around end of May, exerting greatest demand on soil water during this and the following period.

Data were analysed using a mixed-design ANOVA with a within-subjects factor of time (ungrazed and grazed) and change in *flgrem* value (0.3, 0.4, 0.5, 0.6, 0.7, 0.8) on ALB, SOMTC and TOTSYC. Mauchly's statistic was significant for factor, changed *flgrem*, (χ^2 (14) = 42.6, $p < 0.05$) thus indicating that the data violate the sphericity assumption of the univariate approach to repeated-measures ANOVA, and, therefore, degrees of freedom were corrected using Greenhouse-Geisser estimates of sphericity (ε = 0.22). Even with the adjustments, the within-subject effects of time-period are significant (F (1, 104) = 95.4, $p < 0.001$). This means that the ALB of the ungrazed and grazed period is significantly different. Approximately 28.5% of the total variance in the dependent variable ALB is accounted for by the variance in the independent variable, time.

Similarly, changed *flgrem* significantly affected the amount of ALB (F (1.08, 112.2) = 60.3, $p < 0.05$). Approximately 72.8% of the total variance in the dependent variable ALB is accounted for by the variance in the independent variable, changed *flgrem*. Main effects of time and changed *flgrem* were qualified by an interaction between time and changed *flgrem* (F (1.08, 112.3) = 59.3, $p < 0.05$). Post hoc test (Tukey's HSD) indicated that there was a significant difference in the mean ALB between the ungrazed period (1991 to 2005) and grazed period (2006 to 2020) ($p < 0.05$). An examination of the means showed significantly higher ALB for ungrazed period (M = 73.86) than grazed period (M = 45.03).

Predicted mean ALB values were also significantly different between the *flgrem* values (FLGREM0.3, FLGREM0.4, FLGREM0.5, FLGREM0.6, FLGREM0.7) ($p < 0.05$). An examination of the means showed significantly higher ALB for FLGREM0.6 (70.2 g m^{-2}) than FLGREM0.5 (68.0 g

m^{-2}), FLGREM0.4 (61.0 g m^{-2}) and FLGREM0.3 (60.2 g m^{-2}). Both FLGREM0.7 and FLGREM0.8 showed significantly lower ALB (48.8 g m^{-2} and 48.4 g m^{-2} respectively) than FLGREM0.3, FLGREM 0.4, FLGREM0.5 and FLGREM0.6.

Both SOMTC (F (1, 104) = 121.8; $p < 0.05$) and TOTSYC (F (1, 107) = 199.9; $p < 0.05$) were also significantly affected by the change in *flgrem* per month during grazing period (2006 – 2020). Based on SOMTC test results, the Partial Eta-squared was 0.80 for time and 0.74 for changed *flgrem*. This means that ~ 74 - 80% of the total variance in the dependent variable SOMTC is accounted for by the variance in the independent variables, time and changed *flgrem*. Tukey's HSD test results showed significant difference in the predicted mean SOMTC and TOTSYC between all the *flgrem* values ($p < 0.05$). Mean SOMTC values were higher during the ungrazed period (10.0 kg m^{-2}) compared to grazed period (9.5 kg m^{-2}) and significantly decreased with increase in the *flgrem* value (Figure 6.10). An examination of the means showed significantly higher SOMTC for FLGREM0.3 (9.8 kg m^{-2}) than FLGREM0.4 (9.7 kg m^{-2}), FLGREM0.5 (9.6 kg m^{-2}), FLGREM0.6 (9.5 kg m^{-2}), FLGREM0.7 (9.2 kg m^{-2}) and FLGREM0.8 (9.2 kg m^{-2}). Overall, modelled results for FLGREM0.7 and FLGREM0.8 showed lower SOMTC values compared to FLGREM0.3, 0.4, 0.5 and 0.6.

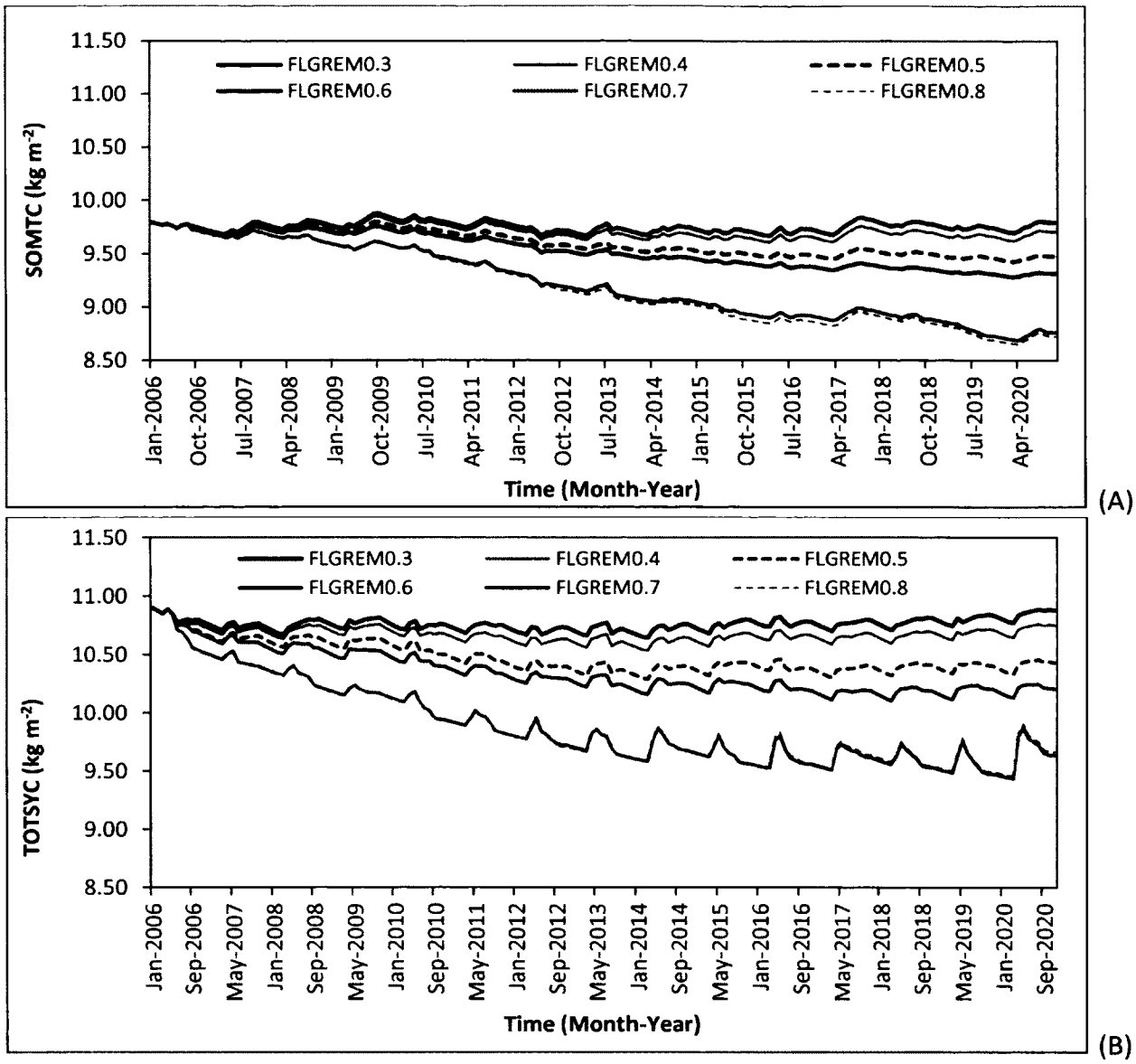


Figure 6.10 Effect of changed fraction of live shoots (*flgrem*) value for heavy grazing option on predicted SOMTC (A) and TOTSYC (B).

Note: FLGREM = fraction of live shoots removed during a heavy grazing option; FLGREM0.3 = fraction of live shoots removed with *flgrem* value of 0.3; FLGREM0.4 = fraction of live shoots removed with *flgrem* value of 0.4; FLGREM0.5 = fraction of live shoots removed with *flgrem* value of 0.5; FLGREM0.6 = fraction of live shoots removed with *flgrem* value of 0.6; FLGREM0.7 = fraction of live shoots removed with *flgrem* value of 0.7; FLGREM0.8 = fraction of live shoots removed with *flgrem* value of 0.8.

Figure 6.10A shows a large drop in total belowground soil carbon (SOMTC) from FLGREM0.6 to FLGREM0.7. The effects of grazing on plant production are represented in the model by using data from Holland *et al.* (1992), where grazing removes vegetation, returns nutrients to the soil,

alters the root to shoot ratio, and increases the nitrogen content of live shoots and roots. In all the three grazing options (light, moderate and heavy) used in the model, the nutrient content of the new shoot increases in relation to the residual biomass (Metherell *et al.* 1993). The decrease in the predicted SOMTC is most probably due to allocation of more carbon to the above-ground live biomass during the growing season. As the grazers eat the leaves, grasses respond by increasing the photosynthetic rate in remaining tissue, stimulating new growth, and reallocating nutrients and photosynthates from one part of the plant to another, especially from roots to stem (Smith and Smith 1998).

For TOTSYC test results, the Partial Eta-squared was 0.94 for time and 0.86 for changed *flgrem*. This means that 86 - 94% of the total variance in the dependent variable TOTSYC is accounted for by the variance in the independent variables, time and changed *flgrem*. Tukey's HSD test results showed significant difference in the predicted mean TOTSYC between all the *flgrem* values and time ($p < 0.05$). Similar to mean SOMTC, mean TOTSYC values were also higher during the ungrazed period (11.1 kg m^{-2}) compared to grazed period (10.6 kg m^{-2}) and significantly decreased with increase in the *flgrem* value (Figure 6.10). Overall, predicted TOTSYC was higher for FLGREM0.3, FLGREM0.4, FLGREM0.5 and FLGREM0.6 compared to FLGREM0.7 and FLGREM0.8.

6.3 Effect of grazing termination (Scenario 2)

A repeated–measures factorial ANOVA with treatment as the main factor and time as the repeated measure ($\alpha = 0.05$) was used to compare the ALB between different scenarios. Simulation with no grazing treatment from year 1991 to 2020 was used as a control. Treatment

(No grazing, grazing with variable intensity and grazing termination, Figure 6.11) had a significant effect on ALB ($F(4, 536) = 5.3$, $p < 0.0001$). Overall, predicted biomass was significantly higher during no grazing period (1990 – 2005) than grazed period (2006 – 2012) ($p < 0.05$).

Mean ALB was significantly different between scenario 1 (2013 – 2020: grazing with variable intensity) and scenario 2 (2013 – 2020: grazing termination) (grazing with variable intensity VS grazing termination: $\beta = -15.14$, SE = 4.43, $t = -3.41$, $p = 0.001$) (Table 6.5). No significant differences were found between the mean ALB predicted for no grazing period (1990 – 2005) and grazing termination period (2013 – 2020, Scenario 2) (no grazing VS grazing termination: $\beta = 4.26$, SE 4.38, $t = 0.97$, $p = 0.577$).

Table 6.5 Effects of grazing intensity and grazing termination on predicted ALB (g m^{-2}) during 2013 – 2020

	ALB Light Grazing (mean ± SE)	**ALB Moderate Grazing (mean ± SE)**	**ALB Heavy Grazing (mean ± SE)**	**p -value**
Scenario 1 (Grazing)	39.71 ± 3.95	34.09 ± 3.35	30.99 ± 3.03	< 0.0001*
Scenario 2 (Grazing termination)	44.67 ± 4.46	44.62 ± 4.47	44.64 ± 4.48	< 0.0001*

* The mean difference is significant at the 0.05 level. Scenario 1 = year 2013 – 2020 with option of variation in grazing intensity; Scenario 2 = year 2013 – 2020 with grazing termination option

Overall, the model predicted increase in ALB as a result of grazing termination compared to ALB during the grazing period (Figure 6.11). There were no significant differences in the mean ALB for light, moderate and heavy grazing intensities during the grazing termination period.

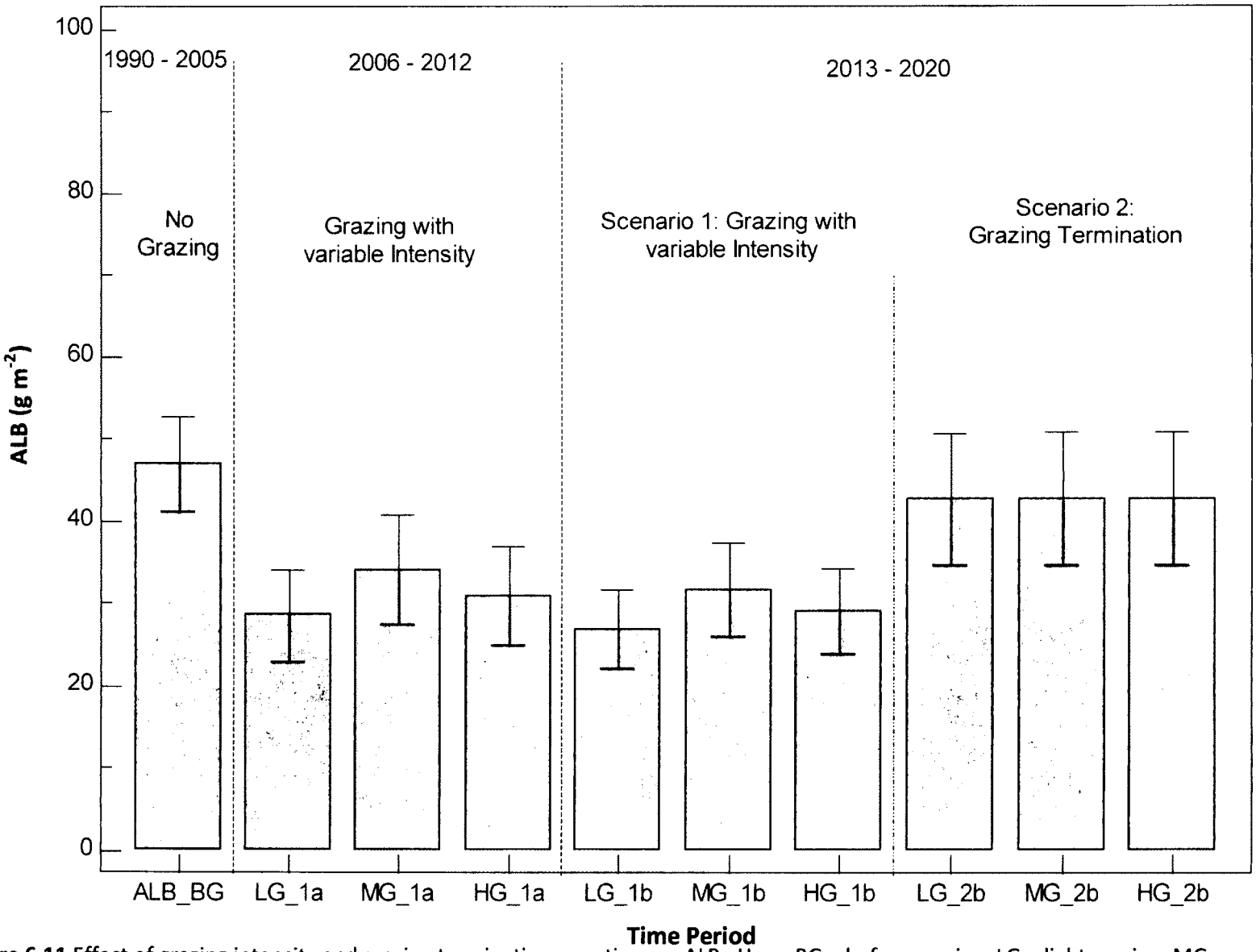


Figure 6.11 Effect of grazing intensity and grazing termination over time on ALB. Here, BG = before grazing; LG = light grazing; MG = moderate grazing; HG = heavy grazing; 1a = Scenario 1 & 2 with variable intensity from year 2006 to 2012; 1b = Scenario 1 with light, moderate and heavy grazing intensity from year 2013 to 2020; 2b = Scenario 2 with grazing termination from year 2013 to 2020. Error bars show 95% CI for mean.

Figure 6.12 and 6.13 shows the effect of grazing intensity and grazing termination over time on the total soil and plant system carbon. Treatment significantly affected the SOMTC ($F(4, 536) = 6.8$, $p < 0.0001$) and TOTSYC ($F(4, 536) = 95.1$, $p < 0.0001$). Tukey's HSD test demonstrated that mean SOMTC and TOTSYC were significantly different between the grazing with variable intensity (scenario 1: 2013 to 2020) and grazing termination (scenario 2: 2013 – 2020) period ($p < 0.0001$) (Table 6.6).

Table 6.6 Effects of grazing intensity and grazing termination on predicted SOMTC (kg m^{-2}) and TOTSYC (kg m^{-2}) for years 2013 to 2020

	ALB Light Grazing (mean ± SE)	**ALB Moderate Grazing (mean ± SE)**	**ALB Heavy Grazing (mean ± SE)**	**p -value**
SOMTC				
Scenario 1 (grazing)	8.93 ± 6.1	8.92 ± 5.8	8.91 ± 5.5	< 0.0001*
Scenario 2 (grazing termination)	8.99 ± 6.3	8.96 ± 6.4	8.92 ± 6.4	< 0.0001*
TOTSYC				
Scenario 1 (grazing)	10.04 ± 4.2	10.00 ± 4.3	9.96 ± 5.1	< 0.0001*
Scenario 2 (grazing termination)	10.18 ± 8.6	10.14 ± 9.3	10.10 ± 10.4	< 0.0001*

* The mean difference is significant at the 0.05 level. Scenario 1 = year 2013 – 2020 with option of variation in grazing intensity; Scenario 2 = year 2013 – 2020 with grazing termination option

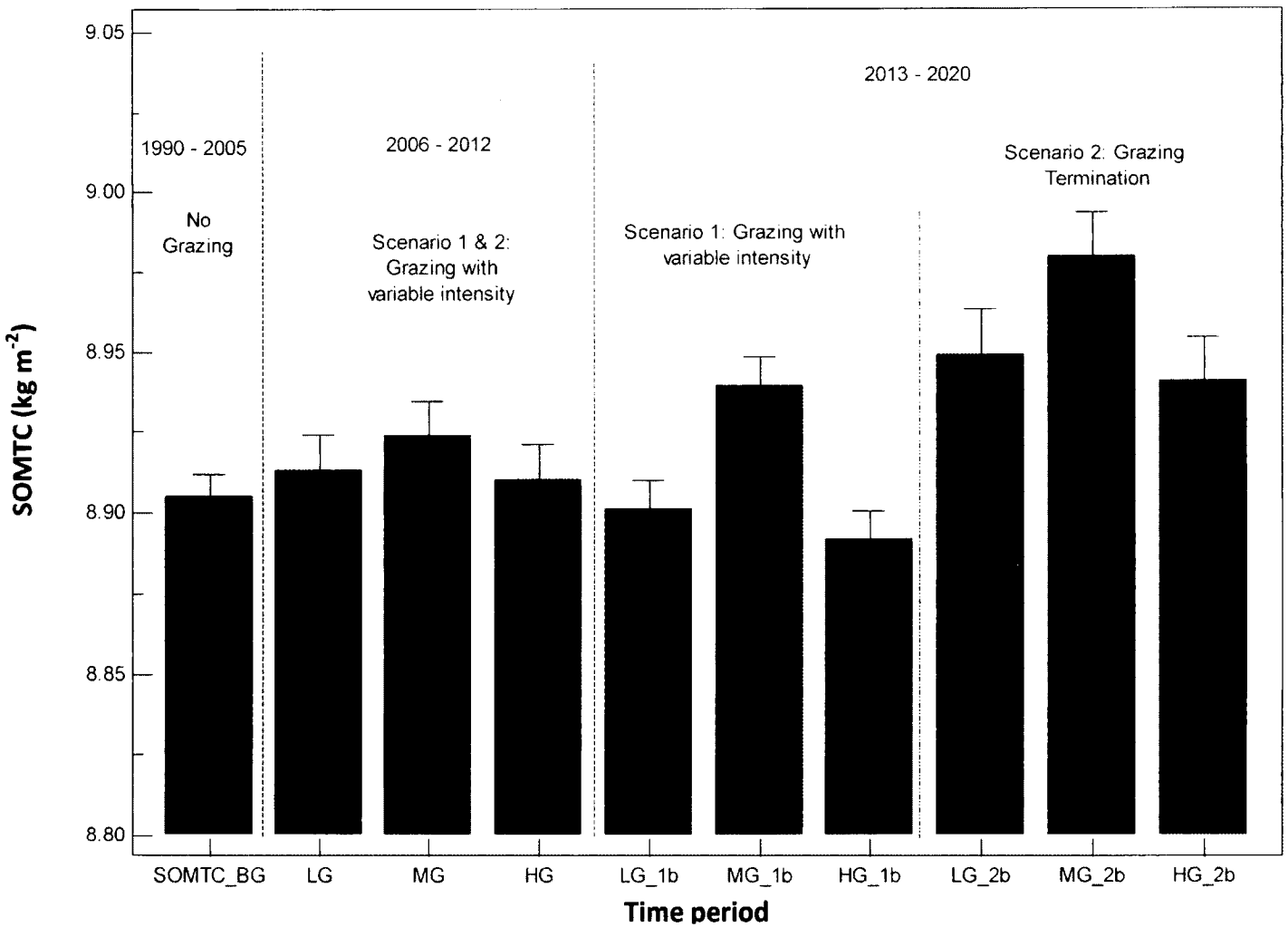


Figure 6.12 Effect of grazing intensity and grazing termination over time on SOMTC. Here, BG = before grazing; LG = light grazing; MG = moderate grazing; HG = heavy grazing; 1b = Scenario 1 with light, moderate and heavy grazing intensity from year 2013 to 2020; 2b = Scenario 2 with grazing termination from year 2013 to 2020. Error bars show 95% CI for mean.

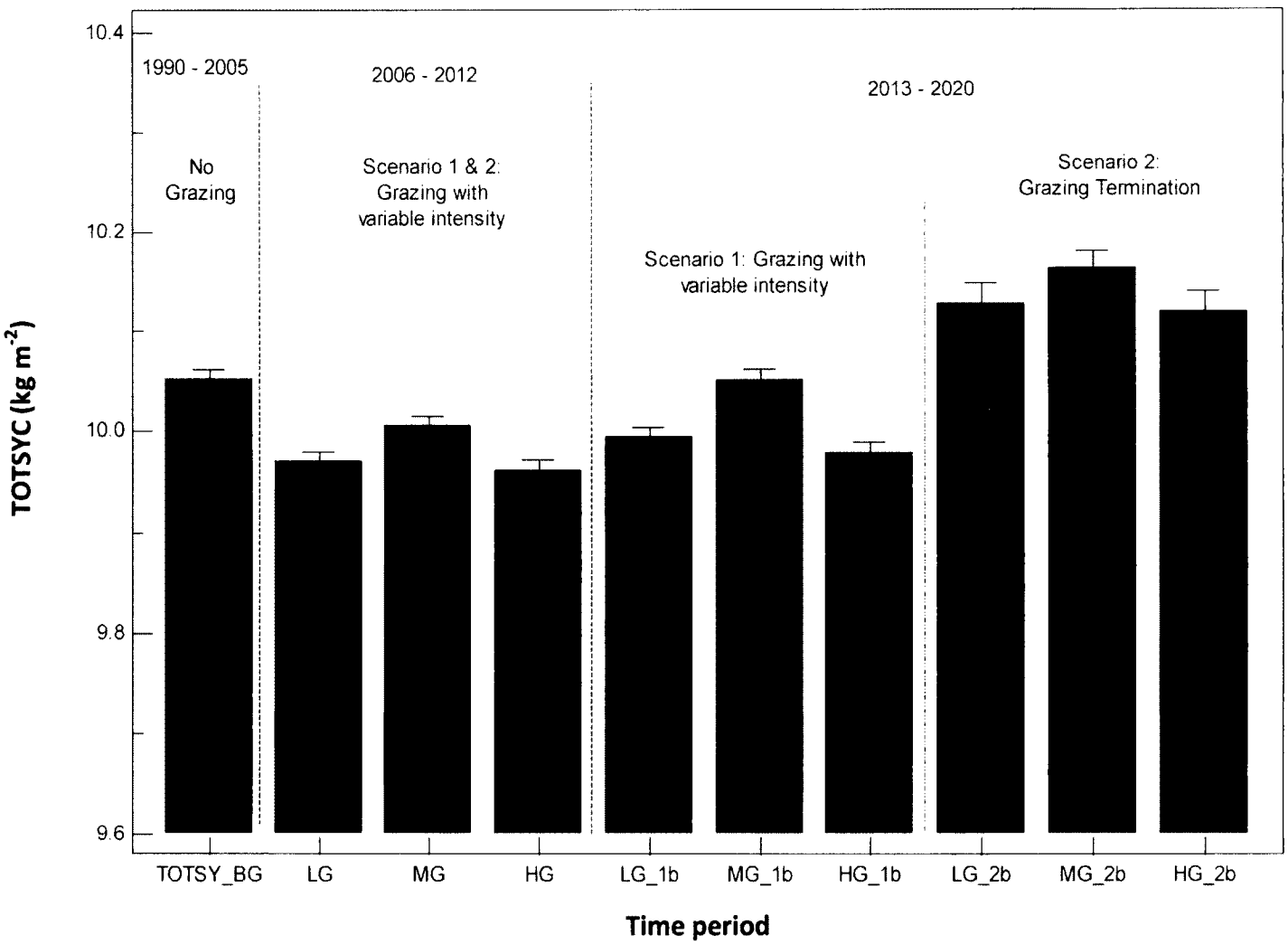


Figure 6.13 Effect of grazing intensity and grazing termination over time on TOTSYC. Here, BG = before grazing; LG = light grazing; MG = moderate grazing; HG = heavy grazing; 1b = Scenario 1 with light, moderate and heavy grazing intensity from year 2013 to 2020; 2b = Scenario 2 with grazing termination from year 2013 to 2020. Error bars show 95% CI for mean.

Significant differences in the predicted mean SOMTC and TOTSYC were found between no grazing period (1990 – 2005) and grazing termination period (2013 – 2020, Scenario 2) ($p < 0.05$, Appendix 3, A3.7). After 6 years of grazing with variable intensity, an increase was observed in predicted SOMTC and TOTSYC values. Overall, moderate grazing showed the highest carbon pools compared to light and heavy grazing.

6.4 Discussion

6.4.1 Effect of soil texture and grazing intensity on model predictions

In general, the overall seasonal pattern for ALB predicted by the CENTURY model in the study area was judged to be acceptable for the purposes of predicting future trends. The model predictions showed some agreement with the field measurements which also showed higher ALB during June and expected patterns in the study area (see Table 6.1). Although modelled values of ALB were slightly greater than observed values, they were within +/- 1 SD of the observed ALB in each month. Similarly, the Mitchell and Csillag (2001) study in GNP also showed that CENTURY predictions for annual NPP compared well with the estimates based on the field observations and with general trends in Great Plains region as reported by Tieszen *et al.* (1997).

The CENTURY model predicted greater soil carbon and total plant system carbon in clays compared to loams or sands similar to Cerri *et al.* (2004) study, which also showed greater soil carbon in soils with more clay content than sand. This is because more passive soil organic matter is formed in sites with high clay content, reflecting the role of increasing soil surface area on carbon protection. Additionally, in the CENTURY model the efficiency of carbon

transfers is also affected by the texture with more CO_2 lost during transformations between pools in soils with high sand proportion (Metherell *et al.* 1993). Since no measured data for total soil and plant system carbon from the study area was available to verify the simulations, it cannot be confirmed if the model is under- or over-estimating the carbon storage in the respective soils of the study area. However, studies such as Foereid and Hogh-Jensen (2004) and Bricklemyer *et al.* (2007) show good agreement between the CENTURY predicted soil organic carbon and measured carbon. For future studies it is recommended to collect the soil samples from each treatment (ungrazed and grazed with variable intensity plots in the study area) and determine the carbon in the soil. This will help in adjusting the model parameters, as well as verifying the model predictions for soil and plant system carbon in response to grazing disturbance.

The grazing scenario in the model was designed to mimic the study area's land-use management history (Appendix 3, A3.4). SOMTC and TOTSYC values decreased with increase in grazing intensity (Figure 6.2 and Figure 6.3). However, it was noticed that short-time periods (single season to 3 years) were not enough to capture the effect of grazing on SOMTC and TOTSYC and required predictions over longer time-period to see any significant changes in the SOMTC and TOTSYC as a result of grazing disturbance. Significant differences were also observed for predicted SOMTC and TOTSYC between the treatments (light, moderate and heavy). These results were similar to those given by Updegraff *et al.* (2010) study, who also showed that grazing is an important factor affecting short- to medium-term carbon fluxes. Studies using CENTURY model to predict soil organic carbon in different ecosystems have also shown that the model reproduces trends in both SOC and above-ground plant production very

well (Parton *et al.* 1993; Parton *et al.* 1994; Alvarez 2001). Parton *et al.* (1987) also concluded that soil carbon is sensitive to the grazing intensity, with soil carbon decreasing with increased grazing rates. They also acknowledged some amount of uncertainty inherent in the model predictions of soil carbon due to limited and uncertainty in the data available on pre-settlement grazing.

Similar to Schimel *et al.* (1997) and Wang (2008), the NPP (aboveground and belowground) and carbon storage predictions in this study showed sensitivity to grazing disturbance. The grassland/crop production sub-model used to run the simulation for light, moderate and heavy grazing intensity effects assumes that the monthly maximum productivity is controlled by moisture and temperature, and that insufficient nutrient availability results in decreased productivity. This explains the low monthly productivity predicted by the model in certain years (Figure 6.5). Studies conducted in other ecosystems have reported that herbivores can increase both soil moisture and temperature by removing the transpirational, shade-casting plant surface area (McNaughton 1985, 1993; Seastedt *et al.* 1988), which should lead to higher decomposition rates in grazed compared to ungrazed grassland. Studies have documented that grazing activity removes live and dead vegetation, alters the root to shoot ratio, increases the nitrogen content of live shoots and roots and returns nutrients to the soil (Holland *et al.* 1992).

The model results also showed sensitivity to changes in the allocation patterns. For example, the model results (see Figure 6.4) for the heavy grazing intensity show that aboveground net primary productivity increased at the expense of belowground net primary productivity. Since the model for all grazing options (light, moderate and heavy) increase the nutrient content of

new shoots in relation to the residual biomass, this explains the variation in aboveground and belowground net primary productivity (Holland *et al.* 1992). In the real world, plants respond to herbivory by switching the allocation of nutrients towards the growth of new shoots and vegetative part to compensate for the losses by grazing disturbance (Jaramillo and Detling 1988; Painter and Belsky 1993; Smith and Smith 1998).

Modelling results were similar to field results, which showed higher ALB in ungrazed pastures than grazed pastures. Chapter 5 showed that ALB is significantly affected by the grazing intensity, where pastures with light and moderate grazing intensity showed higher ALB values compared to high-moderate and heavy grazing intensity pastures (Lin *et al.* 2010). Similarly, modelling results also predicted higher ALB in low and moderate grazing intensity pastures compared to heavy pastures.

The model also predicted decrease in total plant system carbon with increase in grazing intensity as plants utilize more carbon towards the growth of shoots than roots in response to grazing disturbance. The results could not be validated due to lack of field data for belowground soil and total plant system carbon. However, it has been documented in grassland studies that increased number of grazers can decrease the plant root production due to defoliation, which can result in decreased soil carbon stocks (Holland *et al.* 1992; Ganjegunte *et al.* 2005). Furthermore, intensive grazing of lush vegetation and trampling of moist soils results in decreased vegetation cover and root mass, and increased bare ground (Scrimgeour and Kendall 2002).

The model predictions for SOMTC (Figure 6.2) decreased linearly with increase in grazing intensity. He *et al.* (2011) also showed linear decrease in soil carbon in both 0 – 10 cm and 10 - 30 cm soil layers with increasing stocking rates. In a mixed-grass ecosystem, Ingram *et al.* (2008) study reported 30% loss in soil carbon storage (0 – 60 cm) as a result of heavy grazing activity. Since removal of high amount of biomass significantly decreases the input of organic matter from aboveground biomass and roots, this partly explains the decrease in soil carbon with increase in grazing intensity (Johnson and Matchett 2001).

The grazing termination simulation (Scenario 2) of this study showed significantly higher aboveground biomass than simulation runs with continuous grazing by cattle for 8 years. The pattern predicted by the model is consistent with Milchunas and Lauenroth (1993) where grazing decreased net primary production in most systems with a long-term treatment history that lack a long evolutionary history of grazing. Significant differences were also observed in predicted belowground soil carbon and total plant system carbon in response to no grazing, variable grazing intensity and grazing termination. Grazing termination after 6-yr of grazing treatment showed significant increase in belowground soil carbon and total plant system carbon compared to modeling scenario (Scenario 1, Table 6.5) with long-term grazing at variable intensity. Results were similar to He *et al.* (2011) which also reported rapid increase in both carbon and nitrogen storage when grazing was terminated.

6.4.2 Sensitivity analysis

Modeling can help answer how much uncertainty is involved by looking at effect of specific parameters on plant processes essential to address management questions over large regions.

The CENTURY model calculates the plant available water based on the soil texture values provided in the model (Metherell *et al.* 1993; Parton *et al.* 2001), therefore the intent of soil texture sensitivity analyses was to see the effect on model predictions when soil parameters (% clay and % sand) within a same textural class (example, clay-loam soil) were changed in 5% increments. The model output showed statistical significance of variation of sand, silt and clay proportions within a soil textural class on plant ALB. Any change in sand and silt proportions within a soil texture will also affect its bulk density resulting in change in wilting point, field capacity and amount of water available to plants for growth (Rawls *et al.* 1982); this explains the variability in the ALB patterns for simulations based on changed soil parameters as predicted by the model. A statistical significant interaction between time and texture implies that plant productivity is a result of the interaction of environmental factors and management options, and that we should not expect a simple relationship between a single parameter and productivity (Vogt *et al.* 1996). For example, any change in soil texture properties had relatively little effect on averaged ALB, unless there was change in management conditions as well (Figure 6.7). This implies that model predictions were sensitive to the interaction between grazing and moisture status, thus emphasizing the need for site-specific knowledge of weather, soils and management history for better predictions (Riedo *et al.* 2000; Parton *et al.* 2001).

Changed soil parameters (% sand and % clay) within a soil texture class showed high amount of variability in the model output for SOMTC and TOTSYC (Appendix 3). Overall, both SOMTC and TOTSYC showed statistically significant decrease with change in sand and silt content within a soil textural class. Since soil carbon turnover rates are a function of soil's microclimate, physical, chemical (texture, pH, bulk density) and biological (microbial biomass, composition

and diversity) properties (Epstein *et al.* 2002), this explains the variability in SOMTC and TOTSYC during no grazing period. Additionally, a statistically significant interaction between time and texture on TOTSYC indicates importance of management options (No grazing VS grazing with different intensities) on carbon cycling in grasslands (Figure 6.8). Since grazing can alter the soil structure through trampling, altering soil porosity and organic matter of the soils (Goudie 2000), this explains the amount of variability in SOMTC and TOTSYC during grazing period. From a management perspective it is important to understand how variability in a soil texture as a result of land use options such as grazing may impact the ecosystem processes. This is because spatial distribution of soil moisture is highly dependent on the soil texture, which in turn influences the plant root growth and the above ground biomass organization (Ursino 2009).

Proper rangeland management is essential for sustaining the ecological integrity within a grazed system. This is because improper grazing pressure and stocking rate would severely degrade the grassland productivity, as well as affect processes such as carbon cycle within the ecosystem (Wang 2008). For example, grazing activities such as trampling can change the soil organic carbon in an ecosystem over time; however the effects vary with stocking rate (Schuman *et al.* 1999, 2001). Therefore it is important to estimate an appropriate grazing strategy in order to maintain a sustainable grassland ecosystem. The simulated results show that removal of live shoots per month (example, 30% to 80%) during a grazing event significantly affected the grassland net primary productivity (aboveground and belowground). When 80% live shoots were removed, the aboveground NPP decreased by 38% than when 40% live shoots were removed (17% decrease). This implies that proper grazing intensity would

stimulate the vegetation growth, while overgrazing would eventually degrade the plant productivity in the study area.

Furthermore, a change in ALB pattern was also observed in response to extreme grazing disturbance. For example, the model predictions showed a peak rate of vegetation growth around end of May, when ≥ 70% of live shoots during a month were removed compared to only 30% – 50% removal. Overgrazing can cause shifts in species composition (example, C_4 (warm season) to C_3 (cool season) dominated grasslands) (Seastedt *et al.* 1994). This will certainly be of concern to management of GNP where shift in plant species composition will also affect the amount of biodiversity within the park.

Change in *flgrem* value, a grazing parameter, also showed decrease in total soil and plant system carbon when more live shoots per month were consumed by grazers. For example, when 30% live shoots were removed during heavy grazing event per month, the model predicted about 2 - 5% soil organic carbon loss over a period of 14 years. In contrast, when a heavy grazing option with 80% removal of live shoots per month was selected, the model predicted about 10 - 15% loss of soil organic carbon over a 14 year period. Results were similar to Wang (2008) study which also showed significant decrease in soil organic carbon with increase in live shoots removed per month during a grazing event.

6.5 Applicability of results, limitations and research recommendations

The results from this study contribute towards understanding of effects of both soil texture and grazing disturbance over time on the grassland ecological processes. Additionally, the simulated results based on sensitivity analysis contribute to our understanding of importance of

knowing how much uncertainty exists in model predictions. For a modeller this information is crucial, as it will provide insight into the contribution of a specific parameter, as well as reveal which input parameter should be given a priority for precise measurements resulting in better model predictions. Additionally, it will also help in pointing out parameters that are unimportant and can be set to fixed value during model runs. From a manager perspective, the knowledge of uncertainty will not only result in more confidence in model predictions, but will also be vital in devising a site-specific sustainable grazing management plan both for short-term and long-term conservation by assessing the range of possible outcomes. Since GNP's plan is to increase heterogeneity of biomass to have a range of biodiversity, by knowing the range of effects on the carbon balance of grasslands as a result of change in stocking rates will certainly help in implementing grazing systems that are more sustainable in the long-term.

The simulated results also suggested that ≤ 40% removal of live shoots removed by grazing event per month would result in only slight changes to plant productivity, total soil and plant system carbon compared to >40% removal of live shoots, where ≥ 70% removal of live shoots will result in a significant loss of total soil and plant system carbon, as well as likely cause a shift in vegetation composition. Holechek *et al.* (2006) study also concluded that grazing in arid and semi-arid areas has a positive impact on grazing lands provided it is conservative and does not remove more than 40% of the plant growth. Therefore, it is important to restrict the grazer population within a specified intensity (light, moderate or heavy) not only to maintain the carrying capacity of grasslands for livestock but also to avoid any unintended vegetation changes causing homogeneity and leading to loss of biodiversity.

The most significant limitation of this study is the lack of site-specific soil carbon data which would have helped in knowing whether the model predictions were overestimated or underestimated. It is also acknowledged that a more rigorous soil texture data is required for the study area to reduce the uncertainty to some extent in the model predictions. Furthermore, since all the model simulations used stochastic weather based on past 30 years, it is acknowledged that the model predictions may not represent what will be experienced in coming years.

6.6 Conclusions

Based on modelling results using a long- term grazing scenario, moderate grazing (45 – 57%GI) is predicted to be the best option out of all three grazing intensities in terms of maintaining sufficient heterogeneity in ALB to support species diversity, as well as for carbon management in the mixed grassland ecosystem. Although the heavy grazing option leads to a predicted increase in ALB when higher fraction of live shoots value is used, there is substantial decrease in both soil and plant system carbon suggesting negative impact. Therefore, it does not seem to be a very sustainable option from a management or ecological point of view.

The model results conclude that when soil texture parameter is combined with management options, the model predictions show a discernible effect on the above-ground and below-ground productivity, as well as on the SOMTC and TOTSYC than one with soil texture alone. Overall, light and moderate grazing intensity showed higher ALB, SOMTC and TOTSYC compared to heavy grazing intensity. Finally, the sensitivity results based on change in fraction of live

shoots removed per month within the same grazing intensity option also concludes that grazing management decisions are important controls on the carbon balance in the GNP area.

7.0 Summary and Conclusions

The research presented in this thesis focused on examining the impact of variable grazing intensities on SM and ALB heterogeneity using semivariogram analysis at a range of scales using direct measurements and ground- and satellite-based remote sensing. In addition, the CENTURY model was used to assess the effect of variability in soil texture and grazing intensity on the grasslands' annual net primary productivity, total soil and plant system carbon over longer time periods. The modelling part of the research also looked at the sensitivity of model predictions on two input parameters, sand, silt and clay fractions within a given soil texture class and fraction of live shoots removed per month during a grazing event. Since GNP is considering terminating grazing in the East Block of GNP by 2013, the impact of grazing termination on ALB and carbon dynamics was tested.

Productivity in semi-arid regions responds to both climatic variability and grazing pressure. Analysis of field data quantified spatio-temporal patterns in SM and ALB. Significant impacts of intra-seasonal weather variability, slope position and grazing pressure were found for SM and ALB across a range of scales (plot and local (within pasture)). The statistical results conclude that some of the observed differences in plot scale SM and ALB are a result of introduction of grazing disturbance in addition to other factors. These results were based on only a few months of grazing and follow-up measurements would be needed to verify these results. However, these measurements and results provided the necessary baseline conditions for remote sensing-based analysis and ecosystem modelling that can be used to explore this landscape's ecological responses to changing climate and land use management.

Satellite based ALB estimates at a pasture scale were used to examine the impact of all five grazing intensities on the spatio-temporal pattern of ALB in mixed grasslands beyond the 2008 field season when grazing was initiated and field measurements were carried out. The results showed that vegetation spatial characteristics varied greatly by grazing intensity, time and slope position at a coarse scale (30 m). Overall, low to moderate grazing intensity was associated with increased ALB heterogeneity over time, whereas no change in ALB heterogeneity over time was observed under heavy grazing intensity. As expected, all grazing intensities showed decrease in semivariogram range (patch size) over time confirming that grazing is a patchy process (Vallentine 2001). The study demonstrates that cattle grazing with variable intensity both maintained and changed the spatial patterns of ALB in the studied mixed-grassland ecosystem. This is important from a biodiversity management perspective in GNP, as pastures with very light-to-moderate grazing intensity may provide suitable habitat for species preferring more vegetative cover, while heavy grazing may provide suitable habitat for species preferring less cover.

From the combination of field and satellite data analyses, it was found that factors such as grazing disturbance and local weather conditions cause heterogeneity in SM and ALB, and should be considered in the model for better site-specific predictions on productivity and total soil and plant system carbon. Past research has also shown that plant available water in a soil is affected by soil properties such as texture (Famiglietti *et al.* 1998; Wilson *et al.* 2005). For example, soils with high clay content have very large surface areas, resulting in the soil being able to hold more water and have greater nutrient retention than coarser textured soils. Despite the importance of soil texture, the difficulty of getting detailed regional soils data

especially with respect to the way it varies in space is an issue for understanding grassland ecosystem functioning. Since it is difficult to capture the dynamics of environmental processes using field experimentation alone, a modeling approach was also used in this thesis to acquire knowledge of ecological issues in GNP as a result of change in grazing intensity and spatial variations in soil texture.

The modelling part of the research demonstrates that plant productivity (both aboveground and belowground) and soil carbon and total plant system carbon over the longer time period are sensitive to soil texture and grazing intensity. Overall, based on simulation results, moderate grazing is the best option out of all three grazing intensities in terms of maintaining sufficient ALB essential to support diversity of species, as well as for carbon management in the mixed grassland ecosystem. Although the heavy grazing option predicts an increase in ALB, this is coupled with a substantial decrease in both soil and plant system carbon. Therefore, it does not seem to be a very sustainable option from a carbon management point of view. For example, ≥ 70% removal of live shoots per month will result in a 10 - 15% loss of total soil and plant system carbon, as well as cause a shift in vegetation composition. Any change in vegetation composition and structural characteristics will likely reduce biodiversity (Evans 1998) which does not complement Parks Canada's goal of increasing biodiversity. Holechek *et al.* (2006) concluded that grazing in arid and semi-arid areas has a positive impact on grazing lands provided it is conservative and does not remove more than 40% of the plant growth.

The impact of soil texture variations on ALB were marginal but when a change in soil texture parameters was combined with grazing disturbance, the effect on grassland productivity and

soil carbon was more discernible. This implies that model predictions were sensitive to the interaction between grazing and moisture status. The sensitivity of model predictions to a change in the fraction of live shoots removed per month within a grazing intensity category was substantial and indicates that grazing management decisions are important controls on the carbon balance and maintaining sufficient ALB in the GNP area. Overall, the modeling results indicate that for accurate predictions of plant productivity and soil and plant system carbon dynamics, knowledge of climate, soils and management history is essential.

Proper rangeland management is essential for sustaining the ecological integrity within a grazed system. This is because improper grazing pressure and stocking rate would severely degrade the grassland productivity, as well as affect processes such as carbon cycle within the ecosystem (Wang 2008). Satellite based analysis in this research showed how variation in stocking rates can result in ALB heterogeneity; however it was not possible to see the effect of variation in stocking rates on grassland processes such as carbon cycle. Since grasslands are important sources of carbon stocks, it is imperative to have an understanding of how different grazing intensities can affect the carbon cycle. This understanding is essential for development of an effective grazing system that increases the biodiversity without degrading the mixed grassland ecosystem within the park. The modelling component of this research complements the satellite based analysis in this context. Since GNP's plan is to increase heterogeneity of biomass to have a range of biodiversity, knowing the range of effects on the carbon balance of grasslands as a result of change in stocking rates will certainly help in implementing grazing systems that are more sustainable in the long-term. Overall, based on this research, low- to

high-moderate grazing (45 – 57%GI) is recommended in terms of maintaining sufficient heterogeneity in ALB to support species diversity and carbon management within GNP.

To conclude, this research contributes to the knowledge of how variation in stocking rates can cause heterogeneity in ALB and affect grassland productivity and carbon balance in a grassland ecosystem. This can be used as a tool in development of effective grazing system designs to maintain heterogeneity and restore biodiversity in grassland ecosystem, which is one of the main goals of Parks Canada. However, land management practices and processes must continually be monitored to ensure that sustainable productivity of grasslands is being maintained and enhanced. Organizations, and programs such as Parks Canada-Grasslands National Park of Canada (PC-GNP), Agriculture and Agri-Food Canada, Prairie Farm Rehabilitation Administration (AAFC-PFRA), and the Nature Conservancy in North America should strive to develop and use the best scientific information available to support ecological, economic and social sustainability. The knowledge base should be continuously improved through research and monitoring, to enhance the scientific understanding of ecosystems, including human use of land and to support decision-making and sustainable management of grassland areas.

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Appendices

Appendix 1: Field Analysis (Chapter 4)

A1.1 Averaged weather conditions in the East Block experimental area at the time of soil moisture data acquisition

Date of soil moisture Data acquisition	Pasture number	Air Temp. (°C)	Soil Temp. (at 4-6cm) (°C)	Relative Humidity (%)	PAR (µE)	Wind speed (m s^{-1})
May 16	P9	24.0	16.9	18.4	NA	2.45
May 17	P6	24.6	17.3	17.0	1,900	3.66
May 21	P8	15.2	12.0	33.0	500	5.80
May 22	P1, P6	14.5	9.5	34.0	1,000	10.72
May 25	P9	10.3	11.5	82.2	M	7.70
May 28	P8	12.8	10.7	52.8	M	5.03
May 30	P1	20.3	16.0	42.5	M	6.04
June 04	P8	14.2	15.5	64.3	M	5.30
June 06	P9	15.0	16.8	70.0	M	4.80
June 09	P6	12.5	12.9	67.3	1,174	1.80
June 10	P6	14.1	12.1	64.6	492	6.30
June 13	P1	20.1	16.5	41.2	1,767	6.10
June 15	P8	15.1	16.1	57.1	1,721	5.05
June 16	P6	21.3	17.3	36.3	1,828	4.07
June 23	P9	23.2	22.2	28.8	1,904	2.58

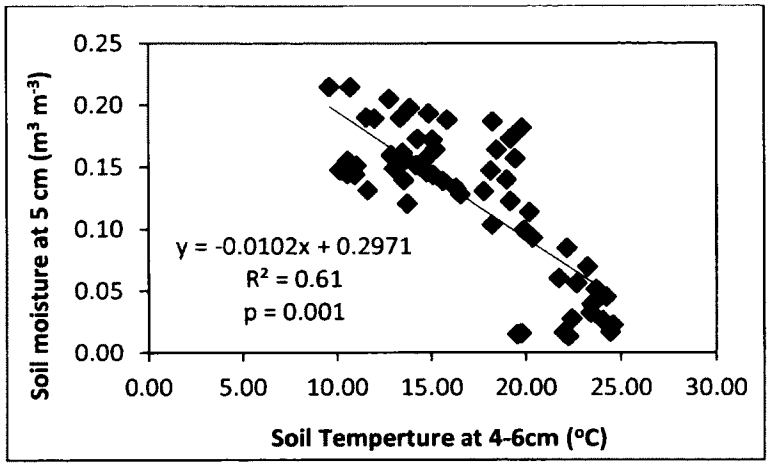
Note: SM = soil moisture; NA = data not available due to delay in instrument installation; M = data missing due to instrument malfunction.

A1.2 Averaged weather conditions in the East Block experimental area at the time of ALB data acquisition

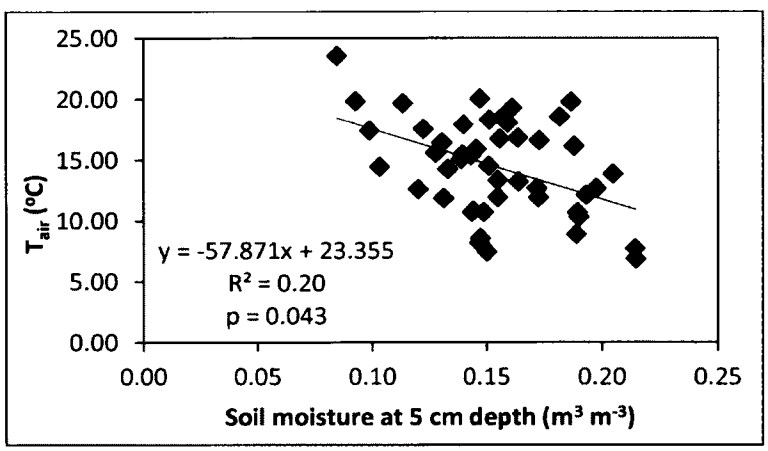
Date of ALB Data acquisition	Pasture number	Air Temp. (°C)	Soil Temp. (at 4_6cm) (°C)	Relative Humidity (%)	PAR (µE)	Wind speed (ms^{-1})	General Comments
May 17	P6	24.6	17.3	17.0	1,900	3.66	Sunny
May 19	P9	20.3	17.8	24.0	1,770	3.12	Sunny
May 27	P8	10.9	10.3	38.7	M	4.70	Sunny and Windy
May 29	P1, P8	17.5	15.3	53.6	M	2.10	Sunny with some cloudy periods, very warm with light breeze
May 31	P1	20.8	20.4	32.0	M	3.60	Sunny and windy at times
June 15	P8	15.1	16.1	57.1	1,721	5.05	Sunny
June 16	P6	21.3	17.3	36.3	1,828	4.07	Sunny with some cloud breaks
June 23	P9	23.2	22.2	28.8	1,904	2.58	Sunny
June 27	P1	17.8	20.2	33.4	1,700	11.30	Sunny and very windy
June 28	P9	22.3	24.2	36.8	1,741	3.87	Sunny

Note: ALB = above-ground live plant biomass; M = data missing due to instrument malfunction.

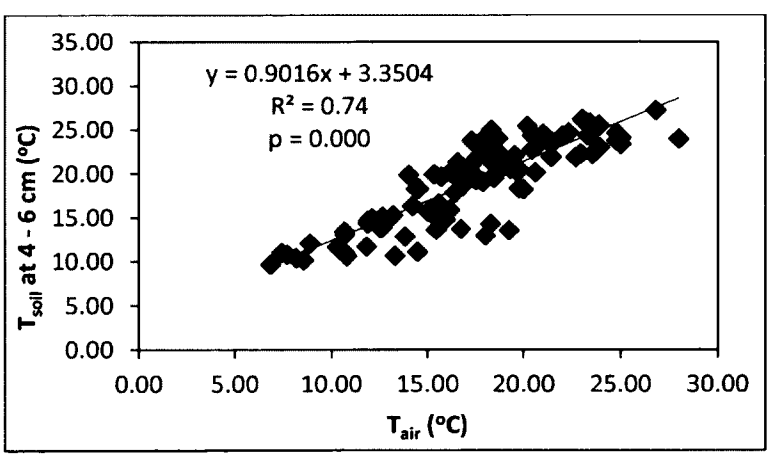
A1.4 Correlation between soil moisture at 5cm depth (m^3 m^{-3}) and soil temperature (T_{soil}) at 4-6 cm



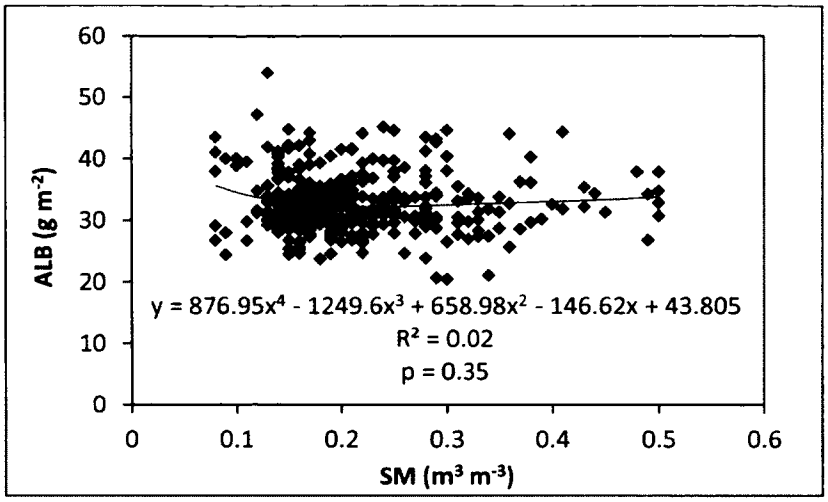
A1.5 Correlation between soil moisture at 5cm depth (m^3 m^{-3}) and air temperature (T_{air})



A1.6 Correlation between soil temperature (T_{soil}) at 4 -6 cm depth and air temperature (T_{air})



A1.7 Correlation between soil moisture and ALB



A1.8 Semivariogram parameters and other statistics for May SM in all the pasture plots

Pasture	Month/ Day	Nugget C_0	Sill $C+C_0$	Range A_0 (m)	MSH	CR	SDR	R^2	RSS
P9U	May-16	0.026	0.064	5.91	0.59	0.292	41.16 (M)	0.726	0.001
P9M	May-16	0.046	0.121	15.64	0.62	0.273	37.60 (M)	0.919	0.002
P9D	May-16	0.012	0.024	15.54	0.56	0.297	42.24 (M)	0.649	0.000
P6U	May-17	0.009	0.023	7.65	0.62	0.276	38.04 (M)	0.637	0.000
P6M	May-17	0.010	0.026	24.57	0.62	0.276	38.07 (M)	0.820	0.000
P6M-2	May-17	0.014	0.035	19.64	0.59	0.292	41.27 (M)	0.826	0.000
P8U	May-21	0.005	0.037	4.77	0.88	0.109	12.30 (S)	0.726	0.001
P8M	May-21	0.015	0.033	26.10	0.54	0.316	46.25 (M)	0.778	0.000
P8D	May-21	0.021	0.204	7.17	0.90	0.093	10.29 (S)	0.680	0.012
P6U	May-22	0.014	0.028	8.95	0.51	0.327	48.58 (M)	0.463	0.000
P6M	May-22	0.023	0.047	35.16	0.51	0.329	49.04 (M)	0.626	0.001
P6M-2	May-22	0.013	0.033	36.43	0.60	0.288	40.38 (M)	0.770	0.000
P1U	May-22	0.017	0.156	3.00	0.89	0.100	11.15 (S)	0.609	0.012
P1M	May-22	0.013	0.090	2.91	0.86	0.123	14.05 (S)	0.398	0.005
P1D	May-22	0.011	0.103	9.96	0.89	0.096	10.60 (S)	0.754	0.004
P9U	May-25	0.002	0.022	3.12	0.89	0.100	11.08 (S)	0.795	0.000
P9M	May-25	0.035	0.077	38.19	0.55	0.317	46.32 (M)	0.747	0.001
P9D	May-25	0.00	0.014	2.97	0.87	0.118	13.42 (S)	0.068	0.001
P8U	May-28	0.010	0.072	5.02	0.50	0.118	13.35 (S)	0.473	0.002
P8M	May-28	0.033	0.092	28.90	0.65	0.261	35.36 (M)	0.673	0.005
P8D	May-28	0.065	0.158	19.83	0.64	0.292	41.15 (M)	0.863	0.004
P1U	May-30	0.010	0.082	5.79	0.87	0.109	12.22 (S)	0.741	0.005
P1M	May-30	0.021	0.124	3.54	0.84	0.142	16.53 (S)	0.648	0.000

P1D	May-30	0.014	0.101	2.79	0.86	0.122	13.86 (S)	0.668	0.004

Note: U = upslope; M = midslope and D = downslope site. D/M = Date/Month; MSH = magnitude of spatial heterogeneity; CR = Correlation Ratio (values near to zero indicate continuity in spatial dependence); and SDR = Spatial Dependence Ratio (S=Strong; M=Moderate and W=Weak).

A1.9 Semivariogram parameters and other statistics for June SM in all pasture plots

Pasture	Month/ Day	Nugget C_0	Sill $C+C_0$	Range A_0 (m)	MSH	CR	SDR	R^2	RSS
P8U	Jun-04	0.006	0.049	1.800	0.88	0.11	12.17 (S)	0.75	0.001
P8M	Jun-04	0.016	0.037	21.30	0.58	0.30	42.24 (M)	0.72	0.000
P8D	Jun-04	0.007	0.072	5.340	0.901	0.09	9.90 (S)	0.62	0.001
P9U	Jun-06	0.041	0.041	29.44	0.00	0.50	100.00 (W)	0.10	0.002
P9M	Jun-06	0.025	0.076	15.12	0.67	0.25	33.22 (M)	0.76	0.001
P9D	Jun-06	0.004	0.035	7.56	0.88	0.11	11.78 (S)	0.64	0.001
P6U	Jun-09	0.007	0.053	6.18	0.87	0.12	13.07 (S)	0.62	0.000
P6M	Jun-09	0.003	0.049	1.35	0.94	0.06	5.97 (S)	0.20	0.001
P6M-2	Jun-10	0.019	0.053	15.41	0.64	0.27	36.23 (M)	0.64	0.002
P1U	Jun-13	0.000	0.028	2.56	0.99	0.20	22.39 (S)	0.45	0.005
P1M	Jun-13	0.010	0.031	11.280	0.68	0.24	31.67 (M)	0.17	0.005
P1D	Jun-13	0.006	0.021	17.370	0.71	0.22	28.67 (M)	0.79	0.001
P8U	Jun-15	0.002	0.019	2.850	0.86	0.08	8.66 (S)	0.45	0.000
P8M	Jun-15	0.002	0.018	2.580	0.90	0.10	10.56 (S)	0.62	0.000
P8D	Jun-15	0.001	0.033	2.880	0.96	0.04	4.19 (S)	0.47	0.005
P6U	Jun-16	0.003	0.020	3.06	0.87	0.12	13.22 (S)	0.16	0.000
P6M	Jun-16	0.002	0.014	1.20	0.88	0.11	12.36 (S)	0.19	0.000
P6M-2	Jun-16	0.003	0.035	2.01	0.91	0.08	8.91 (S)	0.23	0.001
P9U	Jun-23	0.002	0.014	1.49	0.88	0.10	11.18 (S)	0.85	0.000
P9M	Jun-23	0.013	0.082	7.65	0.85	0.13	15.41 (S)	0.66	0.003
P9D	Jun-23	0.002	0.018	1.78	0.88	0.11	12.31 (S)	0.69	0.000

Note: U = upslope; M = midslope and D = downslope site. D/M = Date/Month; MSH = magnitude of spatial heterogeneity; CR = Correlation Ratio (values near to zero indicate continuity in spatial dependence); and SDR = Spatial Dependence Ratio (S=Strong; M=Moderate and W=Weak).

A1.10 Semivariogram parameters and other statistics for ALB in all the pasture plots

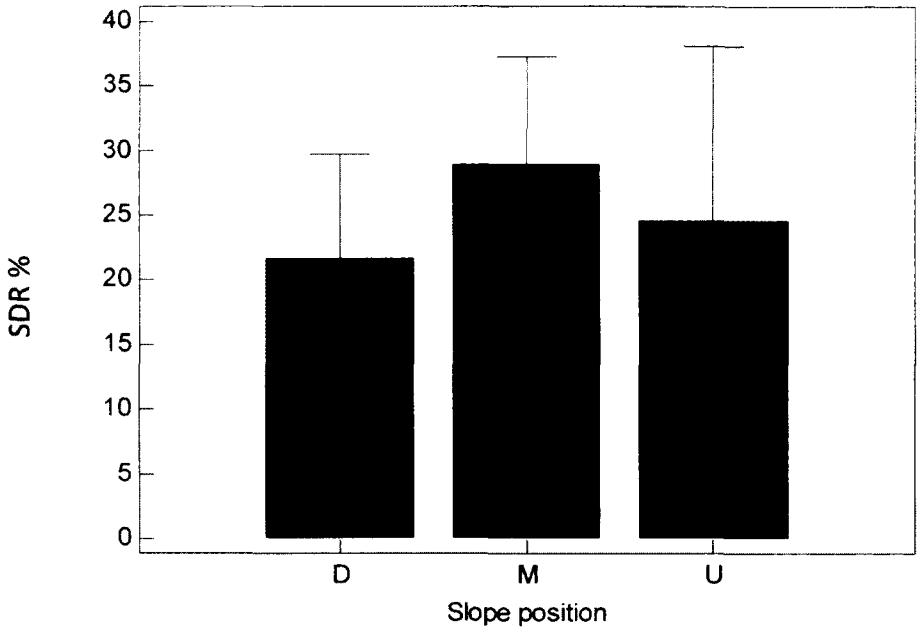
Pasture	Day/ Month	Nugget C_0	Sill $C+C_0$	Range A_0 (m)	MSH	CR	SDR	R^2	RSS
P6U	17-May	0.008	0.032	33.120	0.76	0.19	23.75 (S)	0.84	0.000
P6M	17-May	0.000	0.231	28.110	0.99	0.00	0.04 (S)	0.80	0.062
P6M-2	17-May	0.008	0.028	22.98	0.72	0.22	27.50 (M)	0.69	0.000
P9U	19-May	0.002	0.005	25.47	0.59	0.29	40.98 (M)	0.73	0.000

P9M	19-May	0.007	0.029	29.50	0.74	0.20	25.60 (M)	0.86	0.000
P9D	19-May	0.013	0.049	42.86	0.73	0.21	26.42 (M)	0.72	0.001
P8U	27-May	0.001	0.019	5.38	0.93	0.06	6.80 (S)	0.81	0.000
P8M	27-May	0.004	0.015	17.73	0.73	0.21	27.31 (M)	0.78	0.000
P8D	29-May	0.004	0.028	14.91	0.87	0.12	13.07 (S)	0.74	0.000
P1D	29-May	0.000	0.015	6.84	0.99	0.00	0.07 (S)	0.91	0.000
P1U	31-May	0.003	0.040	20.97	0.94	0.06	6.53 (S)	0.84	0.001
P1M	31-May	0.019	0.074	35.65	0.67	0.20	25.61 (M)	0.91	0.000
P8U	15-Jun	0.003	0.019	7.05	0.85	0.13	14.88 (S)	0.76	0.000
P8M	15-Jun	0.001	0.018	3.74	0.92	0.06	6.55 (S)	0.69	0.000
P8D	15-Jun	0.001	0.028	6.40	0.96	0.04	4.46 (S)	0.83	0.000
P6U	16-Jun	0.003	0.028	45.00	0.63	0.10	10.71 (S)	0.83	0.000
P6M	16-Jun	0.013	0.051	29.92	0.74	0.21	26.06 (M)	0.79	0.001
P6M-2	16-Jun	0.008	0.027	19.89	0.70	0.23	30.56 (M)	0.83	0.000
P9U	23-Jun	0.009	0.017	7.36	0.50	0.33	50.17 (M)	0.80	0.000
P9M	23-Jun	0.028	0.132	22.42	0.79	0.18	21.55 (S)	0.86	0.005
P9D	23-Jun	0.019	0.115	29.61	0.84	0.14	16.46 (S)	0.59	0.014
P1U	27-Jun	0.017	0.130	15.22	0.87	0.12	13.19 (S)	0.91	0.005
P1M	27-Jun	0.020	0.173	7.84	0.89	0.10	11.32 (S)	0.92	0.004
P1D	27-Jun	0.013	0.151	42.00	0.92	0.08	8.33 (S)	0.91	0.004
P9U	28-Jun	0.016	0.031	18.18	0.50	0.11	12.44 (S)	0.70	0.000
P9M	28-Jun	0.044	0.293	21.30	0.85	0.13	15.02 (S)	0.90	0.017
P9D	28-Jun	0.038	0.309	38.22	0.88	0.33	49.84 (M)	0.79	0.041

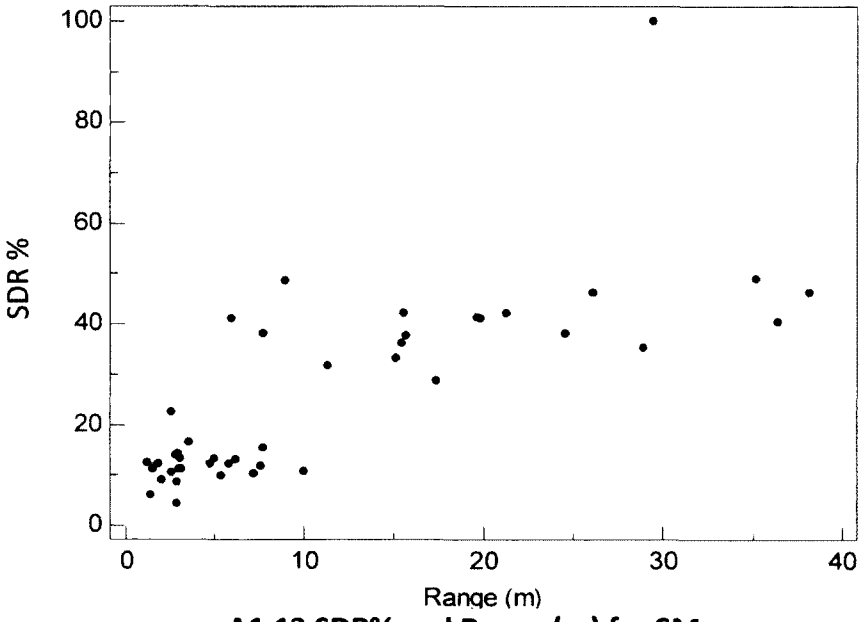
Note: U = upslope; M = midslope and D = downslope site. MSH = magnitude of spatial heterogeneity; CR = Correlation Ratio (values near to zero indicate continuity in spatial dependence); and SDR = Spatial Dependence Ratio (S=Strong; M=Moderate and W=Weak).

A1.11 Mixed effect model results for SM and ALB

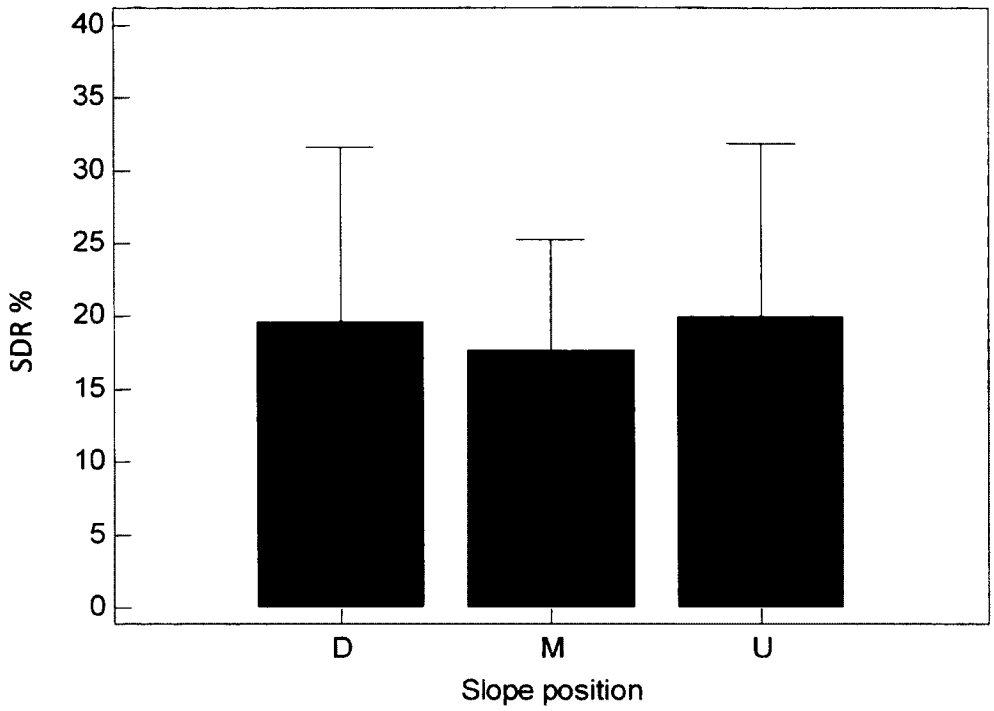
Variable	Effects	Degree of freedom (d.f.)	F-value	P-value	N
SM	Slope location	2, 5537	147.63	< 0.0001	5580
	Time (Date)	13, 5537	450.77	<0.0001	5580
	Slope location x time	26, 5537	21.13	<0.0001	5580
	Pasture	1, 5537	829.27	<0.0001	5580
	Slope location x pasture	6, 5568	110.62	<0.0001	5580
	Treatment	1, 222	125.74	<0.001	223
	Time	1, 222	727.06	0.02	223
	Treatment x time	1, 222	327.95	<0.05	223
ALB	Slope location	2, 2968	17.15	<0.001	2970
	Time (Month)	1, 2969	2677.25	<0.001	2970
	Slope location x time	2, 2969	30.11	<0.001	2970
	Pasture	3, 2964	88.85	<0.001	2970
	Slope location x pasture	6, 2964	9.59	<0.001	2970
	Treatment	1, 270	23.03	0.002	271
	Time	1, 270	2982.35	<0.0001	271
	Treatment x time	1, 270	37.83	<0.0001	271
	Treatment x slope location	2, 637	6.29	0.002	643



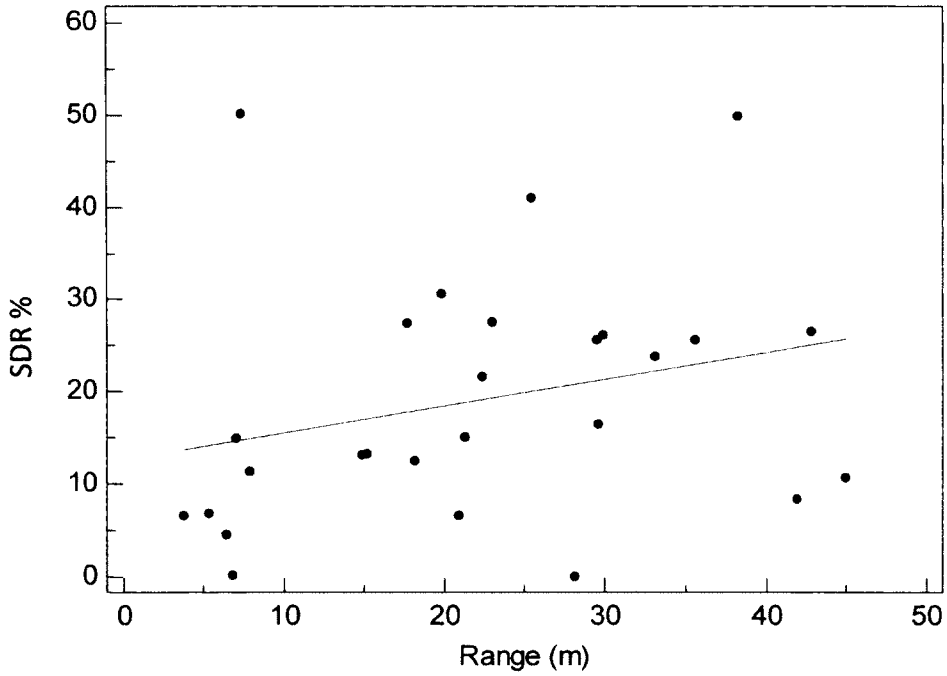
A1.12 SDR% and slope position for SM



A1.13 SDR% and Range (m) for SM

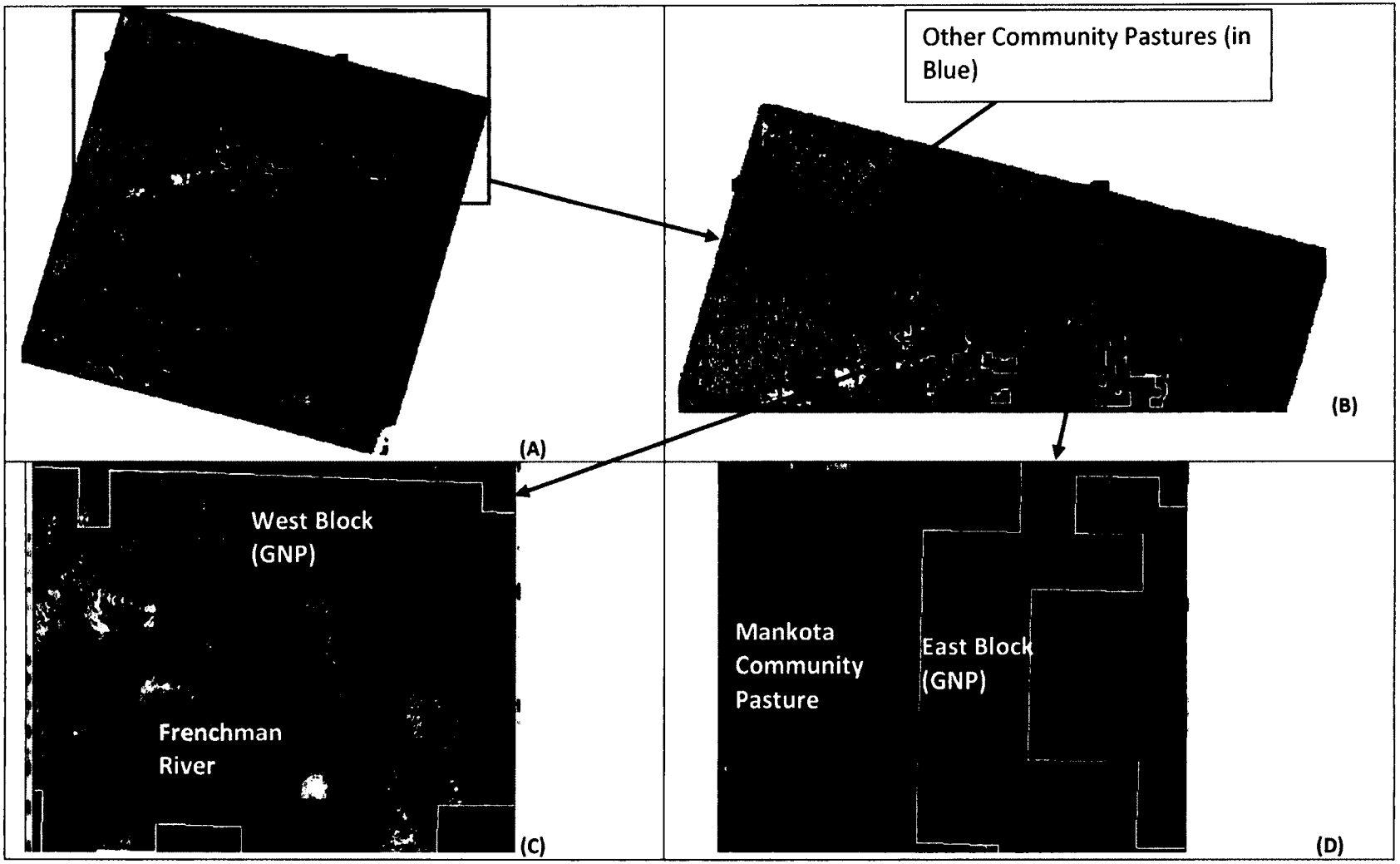


A1.14 SDR% and slope position for ALB



A1.15 SDR% and Range (m) for ALB

Appendix 2: Satellite based Analysis (Chapter 5)



A2.1 Landsat TM scene (July 2000) for the Study area in South-western Saskatchewan (A) Full Landsat scene in CIR showing parts of both south-western Saskatchewan and Northern Montana (B) Landsat scene showing only south-western Saskatchewan community pastures (in blue)and Grasslands National Park (in yellow) boundaries (C) CIR composite showing portion of West Block, GNP (ungrazed site) with Frenchman river and (D) CIR composite showing portions of East Block, GNP (ungrazed site) and Mankota community pasture (grazed site) in south-western Saskatchewan.

A2.2 Semivariogram statistics for ALB in pastures with variable grazing intensity (GI) (shown in subscript in column 'pasture with GI', where UG refers to ungrazed).

Here, SH % = Relative heterogeneity (SH % = MSH*100); CR = Correlation Ratio (values near to zero indicate continuity in spatial dependence); and SDR = Spatial Dependence Ratio (S = Strong; M = Moderate and W = Weak); RSS = Residual Sums of Squares (RSS provides an exact measure of how well the model fits the variogram data; the lower the reduced sums of squares, the better the model fits).

Note: During 2007, East block experimental pastures 1 to 9 were ungrazed so subscript BG for the grazed pastures (2, 3, 4, 6, 7, 8) = before grazing; CUG = controlled and ungrazed pastures 1, 5 and 9.

Pasture with GI	Year	Nugget C_0	Sill $C+C_0$	Range A_0 (m)	MSH	CR	SDR	R^2	RSS
$P1_{CUG}$	2007	0.028	0.128	729	0.782	0.18	21.79 (S)	0.97	0.00069
$P1_{CUG}$	2008	0.017	0.082	930	0.791	0.17	20.88 (S)	0.98	0.00021
$P1_{CUG}$	2009	0.012	0.047	1,059	0.751	0.20	24.89 (S)	0.98	0.00007
$P5_{CUG}$	2007	0.014	0.183	387	0.924	0.07	7.60 (S)	0.96	0.00174
$P5_{CUG}$	2008	0.004	0.083	351	0.948	0.05	5.21 (S)	0.95	0.00040
$P5_{CUG}$	2009	0.005	0.081	339	0.933	0.06	6.65 (S)	0.93	0.00095
$P5_{CUG}$	2010	0.005	0.072	228	0.926	0.07	7.44 (S)	0.96	0.00017
$P9_{CUG}$	2007	0.079	0.231	1,968	0.658	0.25	34.21(M)	0.82	0.00938
$P9_{CUG}$	2008	0.054	0.161	2,202	0.664	0.25	33.58(M)	0.84	0.00392
$P9_{CUG}$	2009	0.041	0.118	2,100	0.652	0.26	34.80(M)	0.83	0.00218
$P9_{CUG}$	2010	0.051	0.126	2,028	0.600	0.29	40.45(M)	0.82	0.00219
$P2_{BG}$	2007	0.003	0.148	249	0.903	0.02	1.69 (S)	0.97	0.00065
$P2_{20}$	2008	0.000	0.086	249	0.965	0.00	0.12 (S)	0.98	0.00014
$P2_{20}$	2009	0.001	0.067	252	0.988	0.01	1.19 (S)	0.94	0.00026
$P2_{20}$	2010	0.000	0.062	180	0.998	0.00	0.16 (S)	0.95	0.00012
$P6_{BG}$	2007	0.006	0.139	399	0.867	0.04	4.32 (S)	0.97	0.00068
$P6_{33}$	2008	0.005	0.064	360	0.922	0.07	7.85 (S)	0.97	0.00013
$P6_{33}$	2009	0.001	0.053	261	0.990	0.01	0.94 (S)	0.97	0.00008
$P6_{33}$	2010	0.000	0.065	234	0.998	0.00	0.15 (S)	0.98	0.00007
$P7_{BG}$	2007	0.016	0.187	180	0.915	0.08	8.49 (S)	0.86	0.0053
$P7_{45}$	2008	0.005	0.095	184	0.943	0.05	5.67 (S)	0.83	0.0017
$P7_{45}$	2009	0.001	0.071	168	0.999	0.00	0.14 (S)	0.83	0.0009
$P7_{45}$	2010	0.000	0.103	159	0.999	0.00	0.10 (S)	0.84	0.0015
$P3_{BG}$	2007	0.022	0.212	257	0.854	0.10	10.58 (S)	0.97	0.0016
$P3_{57}$	2008	0.017	0.107	324	0.838	0.14	16.23 (S)	0.99	0.0002
$P3_{57}$	2009	0.014	0.090	329	0.844	0.13	15.60 (S)	0.99	0.0002
$P3_{57}$	2010	0.010	0.133	212	0.925	0.07	7.45 (S)	0.99	0.0004
$P4_{BG}$	2007	0.025	0.249	222	0.898	0.09	10.18 (S)	0.97	0.0028
$P4_{70}$	2008	0.014	0.120	254	0.885	0.10	11.54 (S)	0.96	0.0009
$P4_{70}$	2009	0.011	0.092	203	0.891	0.11	12.36 (S)	0.97	0.0004

$P4_{70}$	2010	0.019	0.162	236	0.883	0.10	11.73 (S)	0.97	0.0012
$P8_{BG}$	2007	0.005	0.229	122	0.978	0.02	2.38 (S)	0.97	0.0023
$P8_{70}$	2008	0.000	0.136	121	0.999	0.00	0.07 (S)	0.96	0.0009
$P8_{70}$	2009	0.001	0.091	113	0.989	0.01	1.10 (S)	0.97	0.0004
$P8_{70}$	2010	0.000	0.143	124	0.999	0.00	0.07 (S)	0.98	0.0005

A2.3 Semivariogram statistics for ALB in Mankota community pastures 10, 12 and 13 with Low-moderate grazing intensity (45 – 50 %).

Pasture with GI	Year	Nugget C_0	Sill $C+C_0$	Range A_0 (m)	MSH	CR	SDR	R^2	RSS
$P10_{50}$	2000	0.030	0.071	1,209	0.586	0.29	41.37 (M)	0.91	0.0004
$P10_{50}$	2007	0.012	0.066	251	0.822	0.15	17.82 (S)	0.92	0.0003
$P10_{50}$	2008	0.002	0.039	249	0.944	0.05	5.55 (S)	0.97	0.0000
$P10_{50}$	2009	0.001	0.018	225	0.968	0.03	3.19 (S)	0.66	0.0001
$P10_{50}$	2010	0.004	0.065	198	0.967	0.06	6.34 (S)	0.96	0.0001
$P12_{50}$	2000	0.014	0.069	204	0.798	0.17	20.35 (S)	0.95	0.0002
$P12_{50}$	2007	0.032	0.154	900	0.759	0.19	24.09 (S)	0.97	0.0009
$P12_{50}$	2008	0.016	0.071	576	0.774	0.18	22.63 (S)	0.93	0.0004
$P12_{50}$	2009	0.020	0.066	816	0.697	0.23	30.34 (M)	0.94	0.0003
$P12_{50}$	2010	0.014	0.115	423	0.880	0.11	12.17 (S)	0.93	0.0010
$P13_{50}$	2000	0.016	0.097	152	0.830	0.14	16.46 (S)	0.96	0.0003
$P13_{50}$	2007	0.021	0.171	204	0.877	0.11	12.30 (S)	0.96	0.0012
$P13_{50}$	2008	0.011	0.096	162	0.885	0.10	11.46 (S)	0.97	0.0003
$P13_{50}$	2009	0.005	0.072	150	0.926	0.02	1.87 (S)	0.96	0.0001
$P13_{50}$	2010	0.002	0.096	165	0.981	0.02	1.87 (S)	0.98	0.0003

Appendix 3: Modeling (Chapter 6)

A3.1 Century model parameters

Most of the parameters in the mankota6.100 file were adjusted to account for the unique properties of the study area. However, some sets of parameters are more important than others. For example, climate and soil physical are very important but the initial organic matter and water parameters are not important if an equilibrium block in the schedule (.sch) file is included. We modified only the monthly precipitation, monthly minimum and maximum air temperature, sand/silt/clay fractions, bulk density, wilting point, and field capacity in the mankota6.100 file. Additionally, grazing parameters in grz.100 file were also modified to simulate effects of light, moderate and heavy grazing intensity of grassland productivity and total soil and plant system carbon.

Mankota6.100 file included the following parameters:

CENTURY model inputs

Climate Parameters:			
Parameter	Explanation	Value	References / Notes
PRECIP	Precipitation for January through December (cm)		Daily meteorological data from 1970 – 2007 observed from an Environment Canada station at Mankota were used to determine the precipitation and temperature data (Environment Canada 2010).
TMN2M	January through December minimum air temperature (°C)		
TMX2M	January through December maximum air temperature (°C)		
Site and Control parameters			
Parameter	Explanation	Value	
IVAUTO	controls how SOM pools are initialized --ivauto=0 the initial SOM values in your <site>.100 file are used --ivauto=1 an equation for native grass soil initializes SOM pools --ivauto=2 an equation for cropped/disturbed soils initializes SOM pools use Burke's equations to initialize soil C pools.	1	Mitchell and Csillag (2001)
NELEM	Controls number of elements (besides C) to be simulated = 1 simulate N = 2 simulate N and P = 3 simulate N, P, and S	1 (Only C and N were simulated)	Mitchell and Csillag (2001)
SITLAT	Latitude of site --Is used in the calculation of monthly	49.09917	

	potential evapotranspiration (cm)		
SITLONG	Longitude of site	107.02445	
SAND	Sand in soil layer (%)	Soil*	Based on field data (2008) and park's soil survey (Saskatchewan Institute of Pedology 1992)
SILT	Silt in soil layer (%)	Soil*	
CLAY	Clay in soil layer (%)	Soil*	
BULKD	Bulk density of soil (gcm^{-3}) used to compute soil loss by erosion, wilting point, and field capacity	Soil*	
pH	soil pH used to calculate the solubility of secondary P within the boundaries specified by phesp(1) and phesp(3) --phesp(1) = minimum pH for determining the effect of pH on the solubility of secondary P --phesp(3) = maximum pH for determining effect on solubility of secondary P	6.8	Saskatchewan Institute of Pedology 1992
NLAYER	Number of soil layers in water model (maximum of 9) --used only to calculate the amount of water available for survival of the plant	5.0	
NLAYPG	number of soil layers in the top level of the water model; determines avh2o(1), used for plant growth and root death --avh2o(1) = water available to grass/crop/tree for growth in soil profile	4.0	Field data 2008
DRAIN	the fraction of excess water lost by drainage; indicates whether a soil is sensitive for anaerobiosis (drain=0) or not (drain=1) --Excessively to moderately well drained, drain = 1.0 --Somewhat poorly drained , drain = 0.75 --Poorly drained , drain = 0.5 --Very poorly drained, drain = 0.25 --No drainage from solum, drain = 0.0	1.0	Mitchell and Csillag (2001)
BASEF	the fraction of the soil water content of layer NLAYER + 1 which is lost via base flow	0.3	Mitchell and Csillag (2001) *These parameters control monthly distribution of streamflow, but they have no effect on water balance, decomposition, or production*
STORMF	Is the fraction of excess water that runs off immediately in the current month; the remainder goes to the baseflow storage pool in asmos (nlayer+1).	0.6	
ROCK	Rock effect on field capacity and wilting point values.	0	A value of 0 means no rock effect
SWFLAG	Flag indicating the source of the values for	0	

	AWILT and AFIELD either from actual data from the site.100 file or from equations from Gupta and Larson (1979) or Rawls et al. (1982). swflag = 0 use actual data from the site.100 file swflag = 1 use G & L for both awilt (-15 bar) and afiel (-0.33 bar) swflag = 2 use G & L for both awilt (-15 bar) and afiel (-0.10 bar) swflag = 3 use Rawls for both awilt (-15 bar) and afiel (-0.33 bar) swflag = 4 use Rawls for both awilt (-15 bar) and afiel (-0.10 bar) swflag = 5 use Rawls for afiel (-0.33 bar) with actual data for awilt swflag = 6 use Rawls for afiel (-0.10 bar) with actual data for awilt		
AWILT	the wilting point (cm) of soil layer X, where X = 1-10 (fraction); used only if swflag = 0, 5 or 6	Soil*	
AFIELD	the field capacity (cm) of soil layer X, where X = 1-10 (fraction); used only if swflag = 0	Soil*	
Controls on Phosphorus Sorption			
SORPMX	Set the value for sorpmx to the maximum P sorption capacity for the soil (0-20 cm) expressed as g P sorbed / m^2 (extreme values are 1-3 for sands and 10-20 for high sorption capacity clays)	10	Mitchell and Csillag (2001)
External nutrient input parameters			
SIRRI	No irrigation	0	
EPNFA (2)	values for determining the effect of annual precipitation on atmospheric N fixation (wet and dry deposition) ($g/m^2/y$) (1) = intercept (2) = slope	0.007	Mitchell and Csillag (2001)
EPNFS (2)	values for determining the effect of annual precipitation on non-symbiotic soil N fixation; not used if nsnfix = 1 ($g/m^2/y$) (1) = intercept (2) = slope --nsnfix = equals 1 if non-symbiotic N fixation should be based on N:P ratio in mineral pool, otherwise non-symbiotic N fixation is based on annual precipitation	0.015	Mitchell and Csillag (2001)
SATMOS	values for atmospheric S inputs as a linear function of annual precipitation (g S /m^2/yr/cm precipitation) (1) = intercept (2) = slope	0	Mitchell and Csillag (2001)
SIRRI	S concentration in irrigation water (mgS/l)	0 (no	Mitchell and Csillag (2001)

		irrigation effect)	
Organic matter initial values			
---Initial litter and soil carbon storages (used only if IVAUTO = 0)			Mitchell and Csillag (2001)
Other Parameters			
No.	Parameter	Value	
1	Potential aboveground monthly production for study area (gm^{-2})	V*	Mitchell and Csillag (2001) Field data 2008
2	Optimum temperature for production (°C)	V*	Measured data from literature
3	Maximum temperature for production (°C)	V*	Measured data from literature
4	Initial SOM C/N, C/P, C/S ratios		Mitchell and Csillag (2001)
5	Grass/crop organic matter initial parameters such as aboveground (agliv), belowground (bgliv) and standing dead (stdede) for nitrogen, phosphorus and sulphur		Mitchell and Csillag (2001)
6	Effect of grazing on production (GRZEF)	1 and 2	Holland *et al.* 1992
7	Fraction of live shoots removed by a grazing event (*flgrem*) (range = 0.0 to 1.0)	0.2 to 0.8	Default as well as modified to see the effect
8	Fraction of standing dead removed by a grazing event (range = 0.0 to 1.0)	0.05	Default value

Note: V*, Soil* mean the parameters were related to vegetation type, soil type respectively.

A3.2 The graz.100 file will contain these parameters for each option:

flgrem -- fraction of live shoots removed by a grazing event
fdgrem-- fraction of standing dead removed by a grazing event
gfcret -- fraction of consumed C which is excreted in faeces and urine
gret(3) -- fraction of consumed E which is excreted in faeces and urine (should take into account E losses due to leaching or volatilization from the manure)
(1) = N (2) = P (3) = S
grzeff -- effect of grazing on production
= 0 no direct effect
= 1 moderate effect (linear decrease in production)
= 2 intensively grazed production effect (quadratic effect on production)
fecf(3) -- fraction of excreted E which goes into faeces (rest goes into urine)
(1) = N (2) = P (3) = S
feclig -- lignin content of feces

A3.3 Grazing parameters used to run simulations:

GL----Graze_low_intensity__no effect_on_production
0.10000 'FLGREM'
0.05000 'FDGREM'
0.30000 'GFCRET'

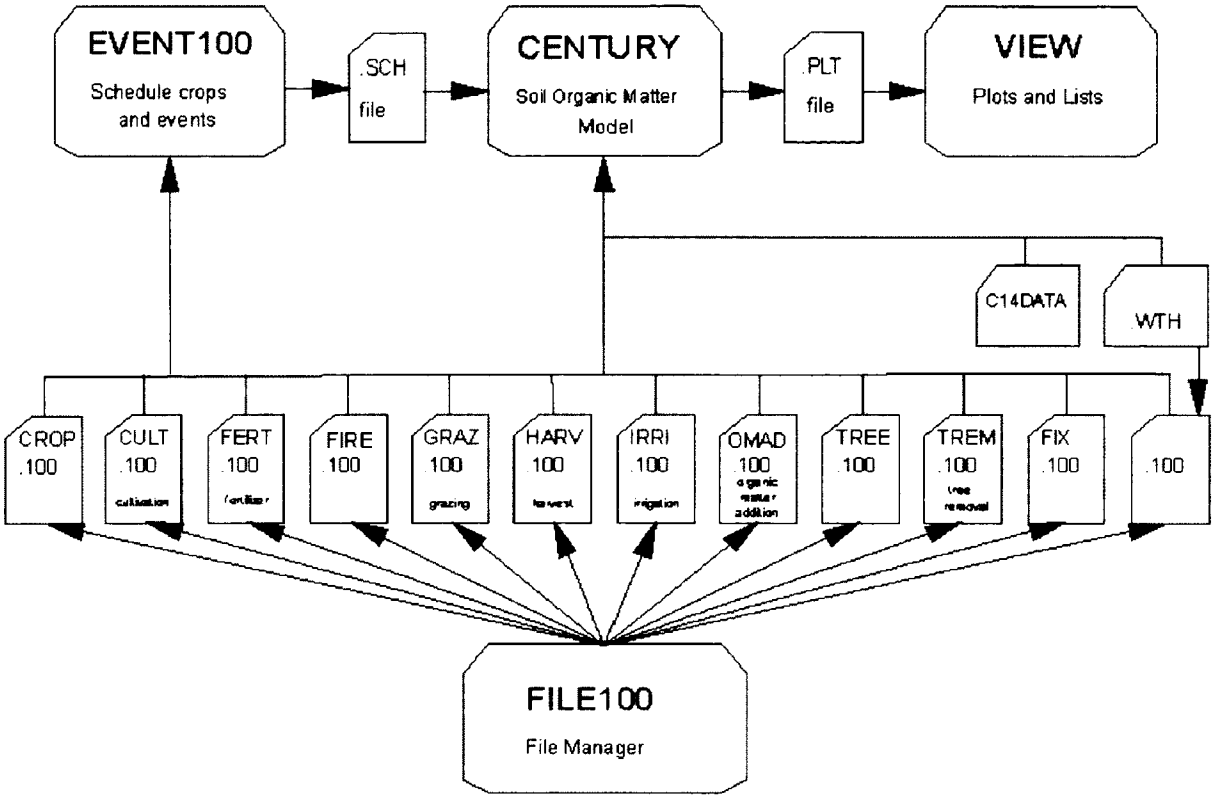
```
0.80000        'GRET(1)'
0.95000        'GRET(2)'
0.95000        'GRET(3)'
0.00000        'GRZEFF'
0.50000        'FECF(1)'
0.90000        'FECF(2)'
0.50000        'FECF(3)'
0.25000        'FECLIG'


GM---graze_moderate_intensity_linear_effect_on_root_shoot_ratio
0.1          'FLGREM'
0.01          'FDGREM'
0.30000        'GFCRET'
0.80000        'GRET(1)'
0.95000        'GRET(2)'
0.95000        'GRET(3)'
1.00000        'GRZEFF'
0.50000        'FECF(1)'
0.90000        'FECF(2)'
0.50000        'FECF(3)'
0.25000        'FECLIG'

GH---graze_high_intensity__quadratic_effect_on_production
0.30000        'FLGREM'
0.15000        'FDGREM'
0.30000        'GFCRET'
0.80000        'GRET(1)'
0.95000        'GRET(2)'
0.95000        'GRET(3)'
2.00000        'GRZEFF'
0.50000        'FECF(1)'
0.90000        'FECF(2)'
0.50000        'FECF(3)'
0.25000        'FECLIG'
```

A3.4 Schedule file for Simulations

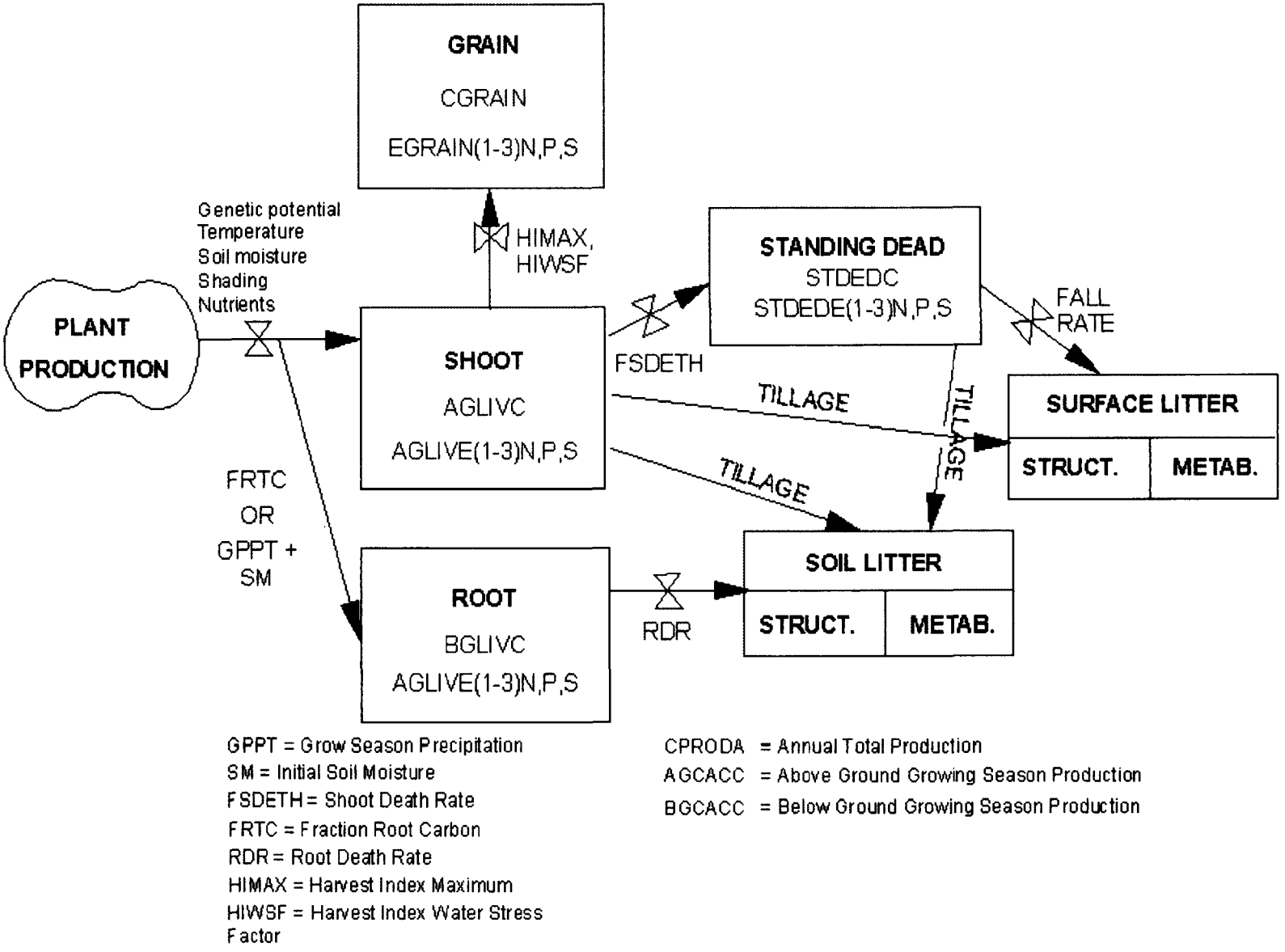
Block 1:	
Time:	0 - 1990
Management:	Continuous grass/grazing
Crop:	GNP2C3 (Mixed vegetation)
Life Cycle:	April (FRST, start of grass growth); Oct (LAST, end of grass growth); Nov (SENM, senescence)
Cultivation:	None
Fertilizer:	None
Grazing:	Winter Grazing (W) (Jan to April) ; Summer grazing (GM (Graze_low_intensity__moderate_(linear)_effect_on_production) (June - Oct.);Winter grazing (W) (Nov -Dec)
Harvest :	None
Weather:	M
Block 2:	
Time:	1991 - 2005
Management:	Continuous grass/No grazing
Crop:	GNP2C3 (Mixed vegetation)
Life Cycle:	April (FRST, start of grass growth); Oct (LAST, end of grass growth); Nov (SENM, senescence)
Cultivation:	None
Fertilizer:	None
Grazing:	None
Harvest :	None
Weather:	S, Stochastic
Block 3:	
Time:	2006 - 2012
Management:	Grazing (variable intensities; Low (GL), moderate (G), Heavy (GH, GH4))
Crop:	GNP2C3 (Mixed vegetation)
Life Cycle:	April (FRST, start of grass growth); Oct (LAST, end of grass growth); Nov (SENM, senescence)
Cultivation:	None
Fertilizer:	None
Grazing:	Summer grazing (GH, Graze_high_intensity__Quadratic_effect_on_production) (June - Oct.)
Harvest :	None
Weather:	S, Stochastic
Block 4:	
Time:	2013 - 2020
Management:	Continuous grass/No grazing
Crop:	GNP2C3 (Mixed vegetation)
Life Cycle:	April (FRST, start of grass growth); Oct (LAST, end of grass growth);Nov

	(SENM, senescence)
Cultivation:	None
Fertilizer:	None
Grazing:	None
Harvest :	None
Weather:	S, Stochastic

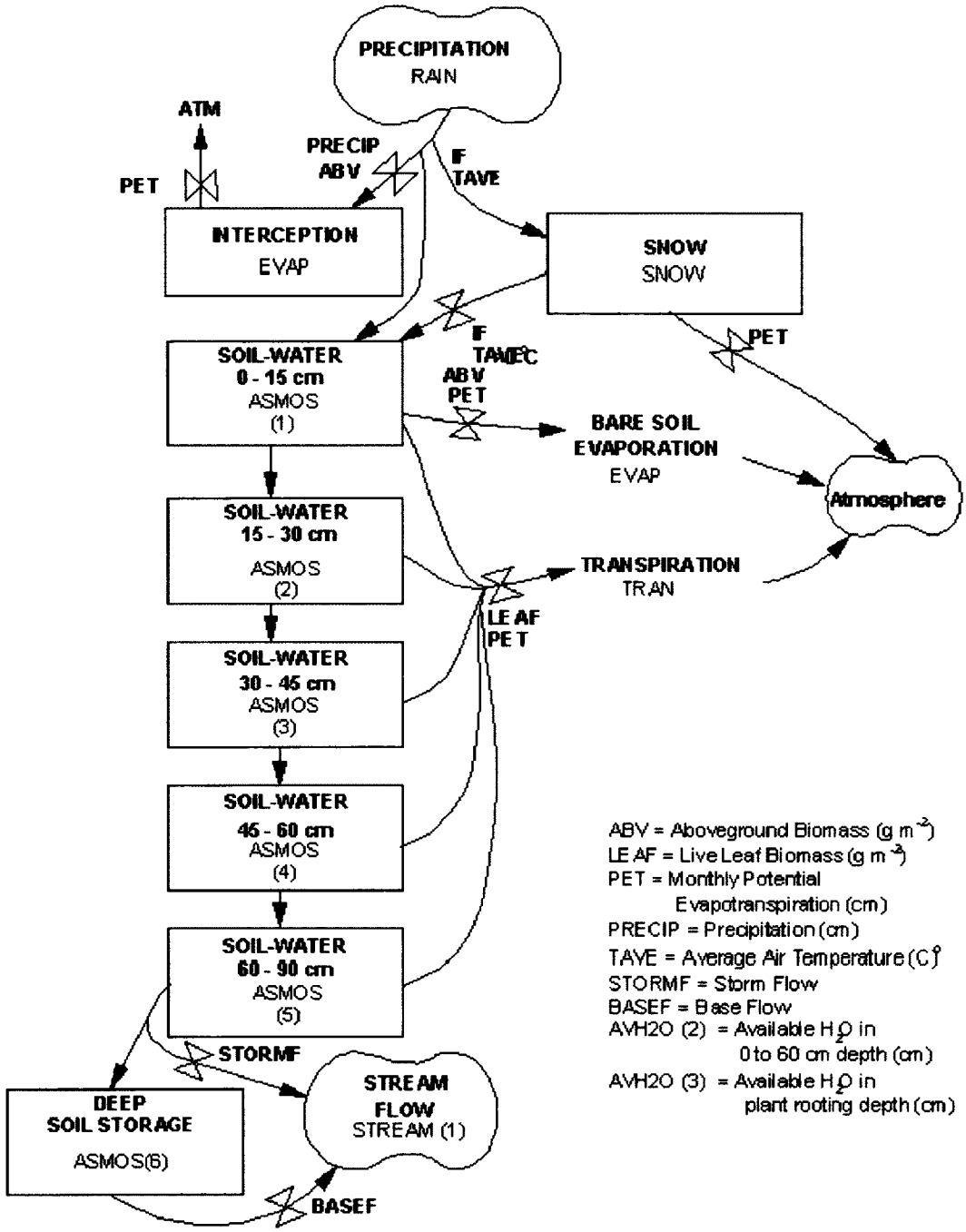
A3.5: Century model simulation structure and flow-charts for grassland/crop sub-model, water sub-model and flows of carbon in century model



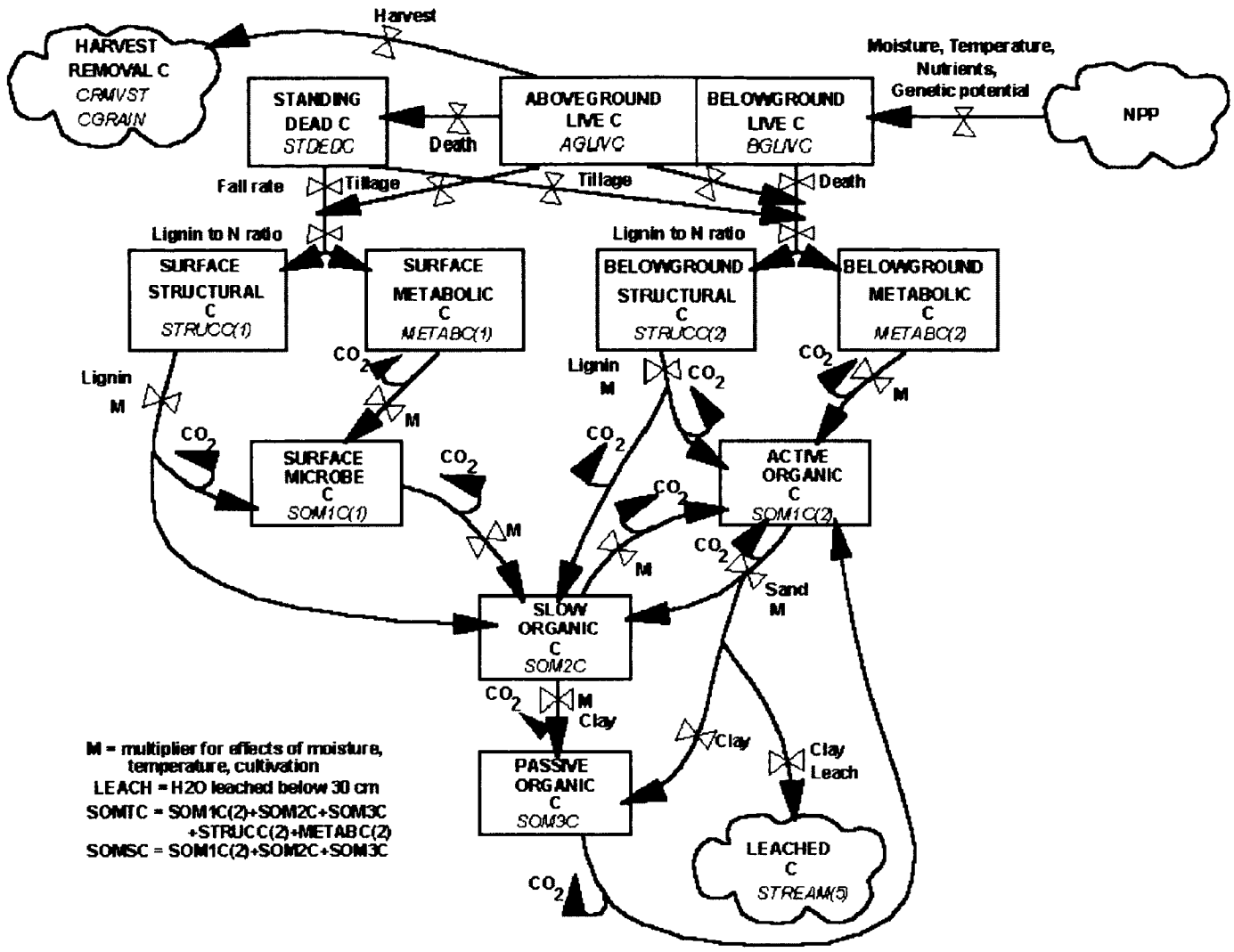
A3.5.1 The Century model environment showing the relationship between programs and the file structure (Metherell et al. 1993).



A3.5.2 Flow diagram for the grassland/crop sub-model (Metherell *et al.* 1993 and Parton *et al.* 2001).



A3.5.3 Flow diagram for the water sub-model. The structure represents a model set up to operate with NLAYER set to 5 (Metherell *et al.* 1993).



A3.5.4 The pools and flows of carbon in the CENTURY model. The diagram shows the major factors which control the flows (Metherell *et al*. 1993).

A3.6 Changes in Century 4.5 version from previous versions

Grazing change:

The GRET (1) parameter from the GRAZ.100 file is no longer being used. The value for GRET(1) now being used in the model equations is calculated based on soil texture so that the fraction of consumed N that is returned is now a function of clay content.

```
if (clay .lt. 0.0) then
  gret (iel) = 0.7
else if (clay .gt. 0.30) then
  gret (iel) = 0.85
else
  gret (iel) = line(clay, 0.0, 0.7, 0.30, 0.85)
endif
```

The line function returns the following value:

```
line = (y2 - y1) / (x2 - x1) * (x - x2) + y2
Where:
  x  = clay
  x1 = 0.0
  y1 = 0.7
  x2 = 0.30
  y2 = 0.85
```

Potential production calculation change:

Potential production is now taking into account the photo period effect on growth. In the fall, when the day length is decreasing, growth will slow down. The definitions for PRDX(1), CROP.100, and PRDX(2), TREE.100, have been changed. These parameters now represent the coefficient used when calculating the potential production as a function of solar radiation outside of the atmosphere. Potential grass/crop production is now being computed in the same manner as potential forest production using an estimate for total production rather than estimating potential aboveground production only. The allocation of aboveground to belowground production for the grass/crop is now based on the fraction of root carbon rather than the root to shoot ratio.

It is recommended to use a value of 0.5 for PRDX(1) and PRDX(2).

Fractional volume of rock used to modify field capacity and wilting point:

The ROCK parameter has been added to the <site>.100 file and will be used for modifying the AFIEL(*) and AWILT(*) values when SWFLAG is not equal to 0. This parameter value is set to 0.0 to run a simulation with no rock effect on field capacity and wilting point values.

A3.7 Mixed effect model results for Scenario 2: Grazing Termination

Descriptive Statistics for ALB, SOMTC and TOTSYC for light, moderate, heavy and grazing termination simulation

Note: LG = light grazing; MG = moderate grazing; HG = heavy grazing; BG = before grazing (1990 – 2005); AG = after grazing (2006 – 2012); GT = grazing termination (2013 – 2020).

	TREATMENT	Mean	Std. Deviation	Std. Error	95% Confidence Interval	
					Lower Bound	Upper Bound
ALB_LG	AG	28.38	25.48	2.77	22.85	33.91
	BG	46.99	40.61	2.53	42.03	51.96
	GT	42.73	39.29	3.58	35.69	49.77
SOMTC_LG	AG	8913.08	50.64	5.52	8902.09	8924.07
	BG	8905.07	50.68	3.90	8897.42	8912.73
	GT	8945.76	68.33	5.52	8934.91	8956.62
TOTSYC_LG	AG	9969.49	47.44	5.17	9959.19	9979.79
	BG	10051.86	68.08	5.04	10041.96	10061.77
	GT	10126.89	98.84	7.14	10112.86	10140.93
ALB_MG	AG	33.73	30.49	3.32	27.12	40.35
	BG	46.99	40.61	2.63	41.83	52.15
	GT	42.82	39.26	3.72	35.51	50.13
SOMTC_MG	AG	8923.55	52.22	5.69	8912.21	8934.89
	BG	8905.07	50.68	3.92	8897.37	8912.78
	GT	8977.56	67.73	5.56	8966.64	8988.48
TOTSYC_MG	AG	10005.67	40.63	4.43	9996.85	10014.49
	BG	10051.86	68.08	4.99	10042.07	10061.66
	GT	10162.11	90.94	7.07	10148.22	10176.00
ALB_HG	AG	30.68	27.55	3.01	24.69	36.65
	BG	46.99	40.61	2.57	41.95	52.04
	GT	42.84	39.34	3.64	35.69	50.00
SOMTC_HG	AG	8910.26	49.98	5.45	8899.41	8921.11
	BG	8905.07	50.68	3.89	8897.42	8912.72
	GT	8939.41	68.61	5.52	8928.57	8950.26
TOTSYC_HG	AG	9961.14	49.52	5.40	9950.39	9971.89
	BG	10051.86	68.08	5.09	10041.87	10061.86
	GT	10120.16	101.16	7.21	10105.98	10134.33

Multiple comparisons based on Tukey's HSD test:

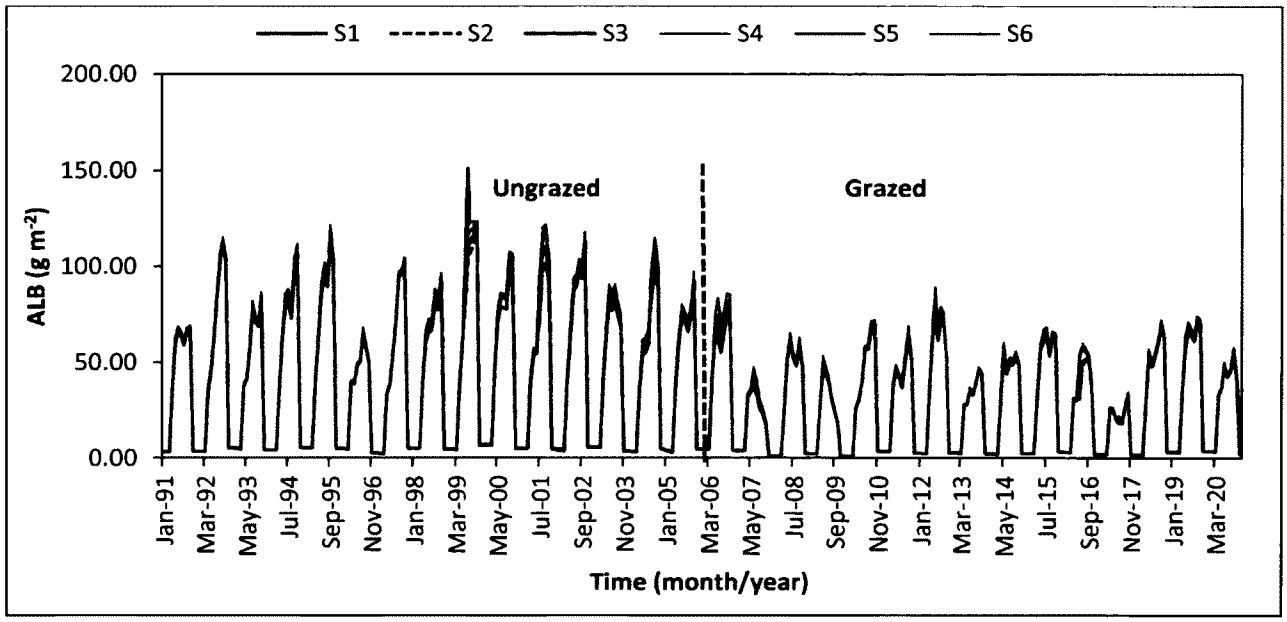
Note: LG = light grazing; MG = moderate grazing; HG = heavy grazing; BG = before grazing (1990 – 2005); AG = after grazing (2006 – 2012); GT = grazing termination (2013 – 2020).

Dependent Variable	(I) TREATMENT	(J) TREATMENT	Mean Difference (I-J)	Std. Error	Sig. (p-value)	95% Confidence Interval	
						Lower Bound	Upper Bound
ALB_LG	AG	BG	-19.40	3.63	0.000*	-27.94	-10.85
		GT	-15.14	4.43	0.002*	-25.56	-4.71
	BG	AG	19.40	3.63	0.000*	10.85	27.94
		GT	4.26	4.38	0.595	-6.05	14.57
	GT	AG	15.14	4.43	0.002*	4.71	25.56
		BG	-4.26	4.38	0.595	-14.57	6.05
SOMTC_LG	AG	BG	0.91	5.60	0.986	-12.26	14.08
		GT	-39.78	6.83	0.000*	-55.85	-23.71
	BG	AG	-0.91	5.60	0.986	-14.08	12.26
		GT	-40.69	6.76	0.000*	-56.58	-24.79
	GT	AG	39.78	6.83	0.000*	23.71	55.85
		BG	40.69	6.76	0.000*	24.79	56.58
TOTSYC_LG	AG	BG	-71.19	7.24	0.000*	-88.22	-54.16
		GT	-146.22	8.84	0.000*	-167.00	-125.44
	BG	AG	71.19	7.24	0.000*	54.16	88.22
		GT	-75.03	8.74	0.000*	-95.58	-54.48
	GT	AG	146.22	8.84	0.000*	125.44	167.00
		BG	75.03	8.74	0.000*	54.48	95.58
ALG_MG	AG	BG	-14.33	3.77	0.000*	-23.21	-5.46
		GT	-10.16	4.60	0.071	-20.99	0.67
	BG	AG	14.33	3.77	0.000*	5.46	23.21
		GT	4.17	4.55	0.630	-6.54	14.88
	GT	AG	10.16	4.60	0.071	-0.67	20.99
		BG	-4.17	4.55	0.630	-14.88	6.54
SOMTC_MG	AG	BG	26.30	5.63	0.000*	13.05	39.55
		GT	-46.18	6.88	0.000*	-62.35	-30.01
	BG	AG	-26.30	5.63	0.000*	-39.55	-13.05
		GT	-72.48	6.80	0.000*	-88.47	-56.49
	GT	AG	46.18	6.88	0.000*	30.01	62.35
		BG	72.48	6.80	0.000*	56.49	88.47
TOTSYC_MG	AG	BG	-23.83	7.17	0.003*	-40.68	-6.97
		GT	-134.08	8.75	0.000*	-154.64	-113.51
	BG	AG	23.83	7.17	0.003*	6.97	40.68
		GT	-110.25	8.65	0.000*	-130.59	-89.91
	GT	AG	134.08	8.75	0.000*	113.51	154.64
		BG	110.25	8.65	0.000*	89.91	130.59
ALB_HG	AG	BG	-17.11	3.69	0.000*	-25.79	-8.43
		GT	-12.96	4.50	0.012*	-23.55	-2.37
	BG	AG	17.11	3.69	0.000*	8.43	25.79
		GT	4.15	4.46	0.620	-6.32	14.63
	GT	AG	12.96	4.50	0.012*	2.37	23.55
		BG	-4.15	4.46	0.620	-14.63	6.32

SOMTC_HG	AG	BG	-5.37	5.60	0.603	-18.53	7.79
		GT	-39.71	6.83	0.000*	-55.76	-23.65
	BG	AG	5.37	5.60	0.603	-7.79	18.53
		GT	-34.34	6.75	0.000*	-50.22	-18.46
	GT	AG	39.71	6.83	0.000*	23.65	55.76
		BG	34.34	6.75	0.000*	18.46	50.22
TOTSYC_HG	AG	BG	-83.15	7.31	0.000*	-100.35	-65.95
		GT	-151.44	8.92	0.000*	-172.43	-130.46
	BG	AG	83.15	7.31	0.000*	65.95	100.35
		GT	-68.29	8.83	0.000*	-89.05	-47.54
	GT	AG	151.44	8.92	0.000*	130.46	172.43
		BG	68.29	8.83	0.000*	47.54	89.05

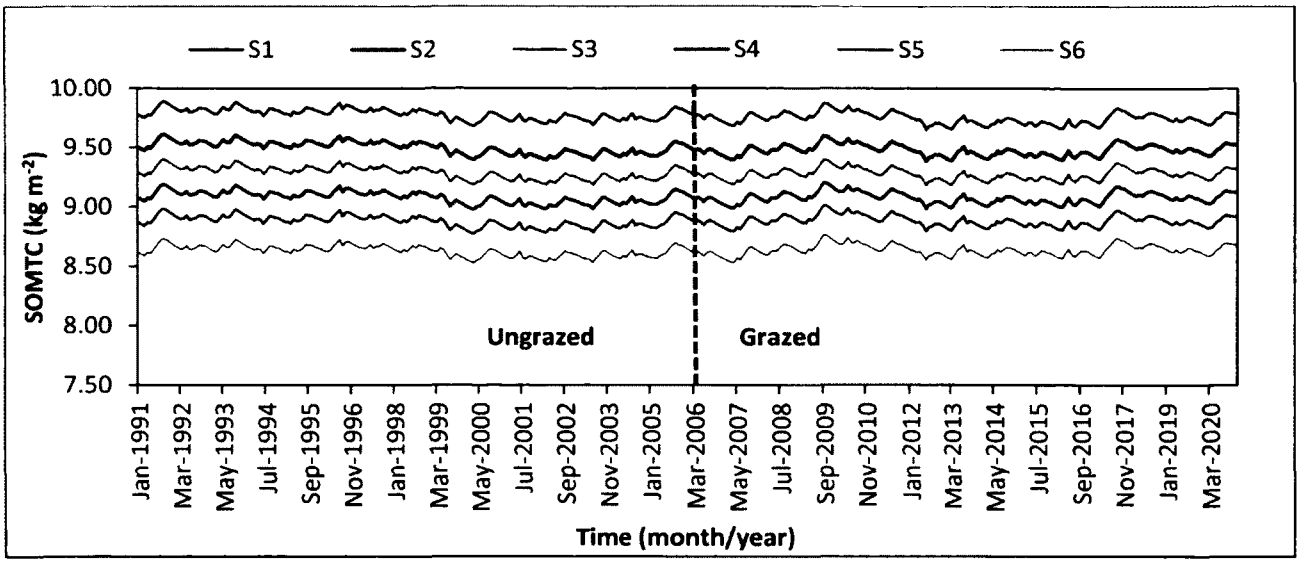
*The mean difference is significant at the 0.05 level.

A3.8 Sensitivity analysis results for variation within a soil texture class and fraction of live shoots removed during a grazing event



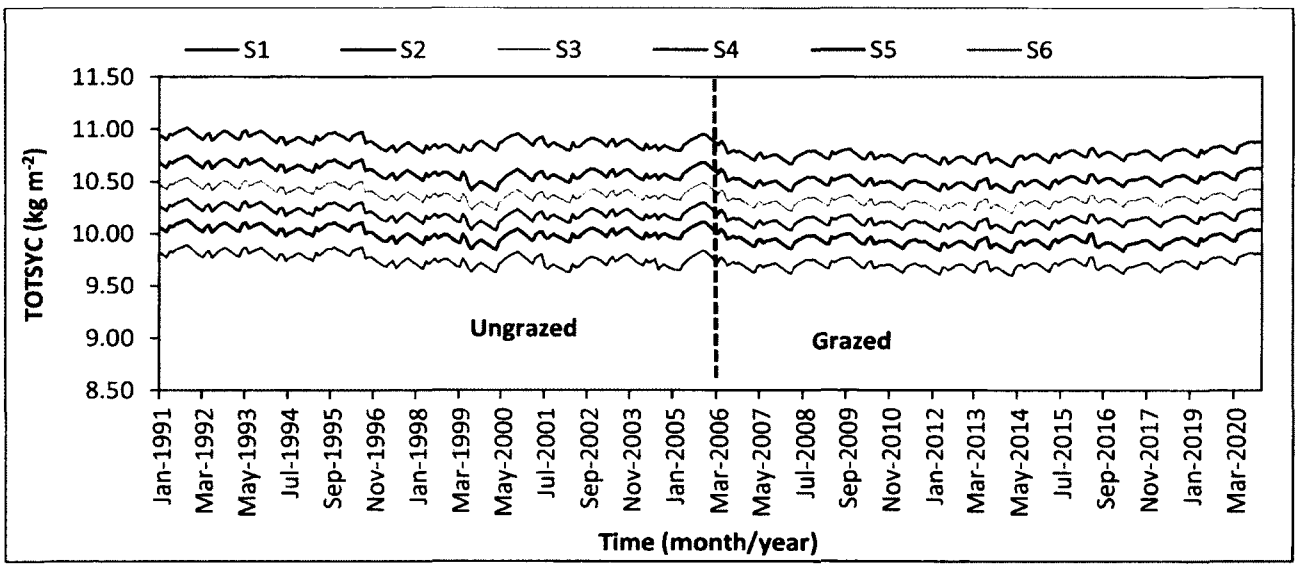
A3.8.1 Effect of change in % sand and % silt within a soil texture on ALB: An example of clay-loam soil texture.

Note: S1 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29; S2 = sand 27%, silt 37.06%, clay 35.94%, BD = 1.3; S3 = sand 31%, silt 33.06%, clay 35.94%, BD = 1.3; S4 = sand 35%, silt 29.06%, clay 35.94%, BD = 1.31; S5 = sand 39%, silt 25.06%, clay 35.94%, BD = 1.32; S6 = sand 44%, silt 20.06%, clay 35.94%, BD = 1.34. In this simulation % clay was held constant to see the effect of change in % sand and % silt within a soil texture on the model predictions.



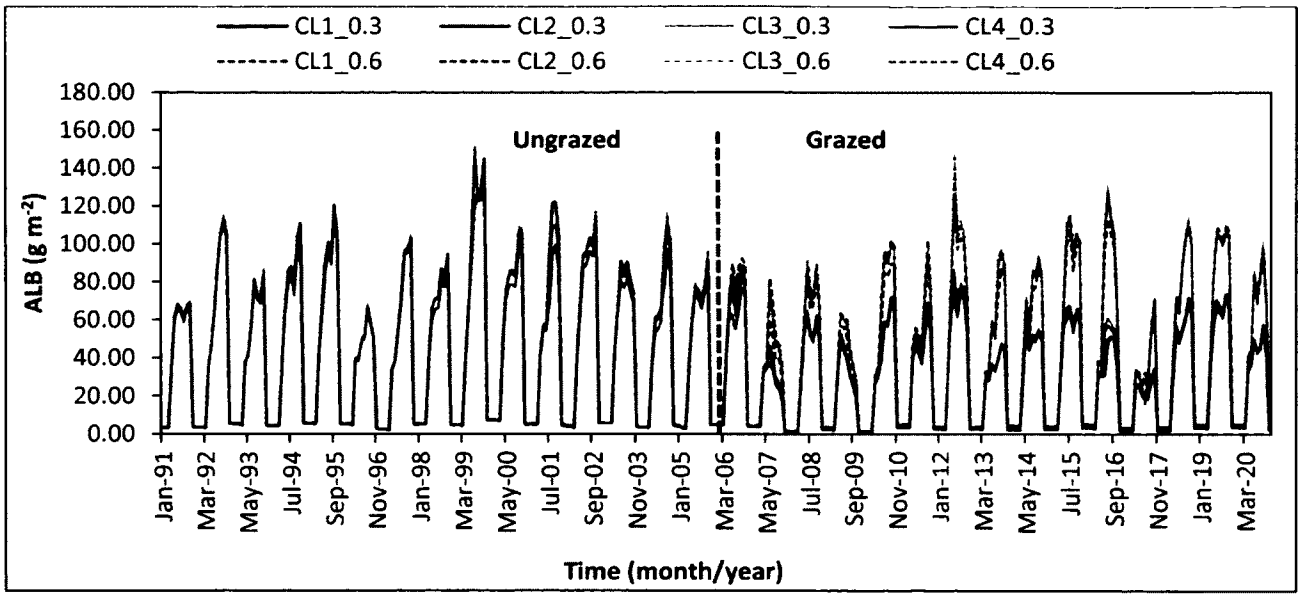
A3.8.2 Effect of change in % sand and % silt within a soil texture on SOMTC: An example of clay-loam soil texture.

Note: S1 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29; S2 = sand 27%, silt 37.06%, clay 35.94%, BD = 1.3; S3 = sand 31%, silt 33.06%, clay 35.94%, BD = 1.3; S4 = sand 35%, silt 29.06%, clay 35.94%, BD = 1.31; S5 = sand 39%, silt 25.06%, clay 35.94%, BD = 1.32; S6 = sand 44%, silt 20.06%, clay 35.94%, BD = 1.34. In this simulation % clay was held constant to see the effect of change in % sand and % silt within a soil texture on the model predictions.



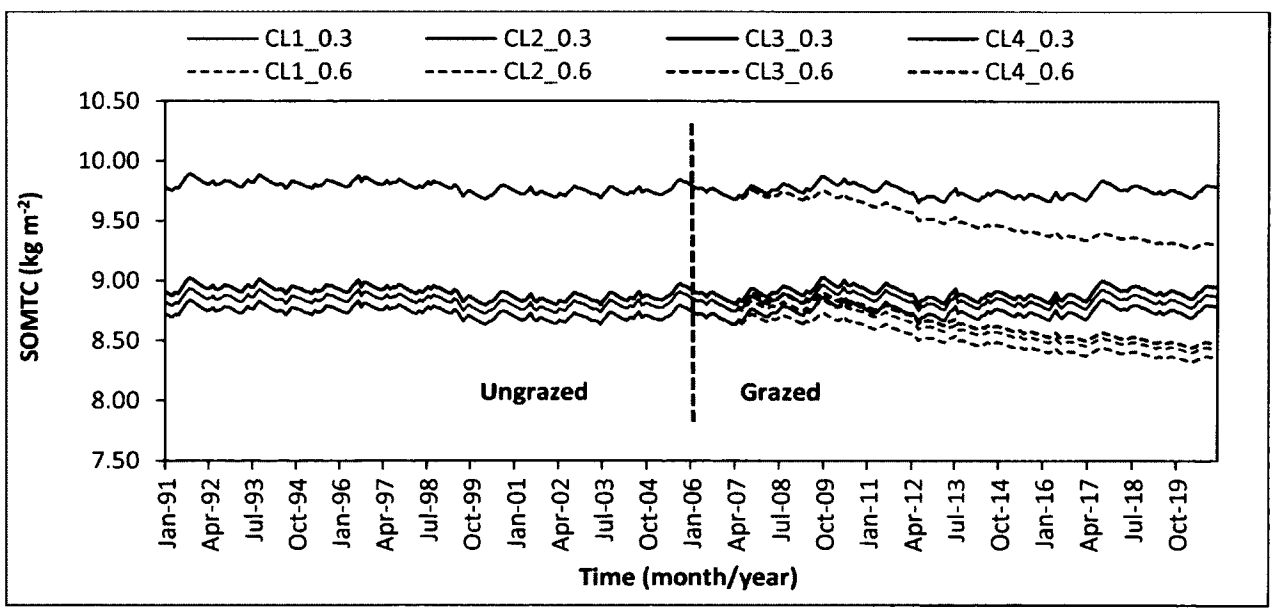
A3.8.3: Effect of change in % sand and % silt within a soil texture on TOTSYC: An example of clay-loam soil texture.

Note: S1 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29; S2 = sand 27%, silt 37.06%, clay 35.94%, BD = 1.3; S3 = sand 31%, silt 33.06%, clay 35.94%, BD = 1.3; S4 = sand 35%, silt 29.06%, clay 35.94%, BD = 1.31; S5 = sand 39%, silt 25.06%, clay 35.94%, BD = 1.32; S6 = sand 44%, silt 20.06%, clay 35.94%, BD = 1.34. In this simulation % clay was held constant to see the effect of change in % sand and % silt within a soil texture on the model predictions.



A3.8.4 Combined effect of variation within a soil texture and fraction of live shoots (flgrem) removed during a grazing event on ALB: An example of clay-loam soil texture.

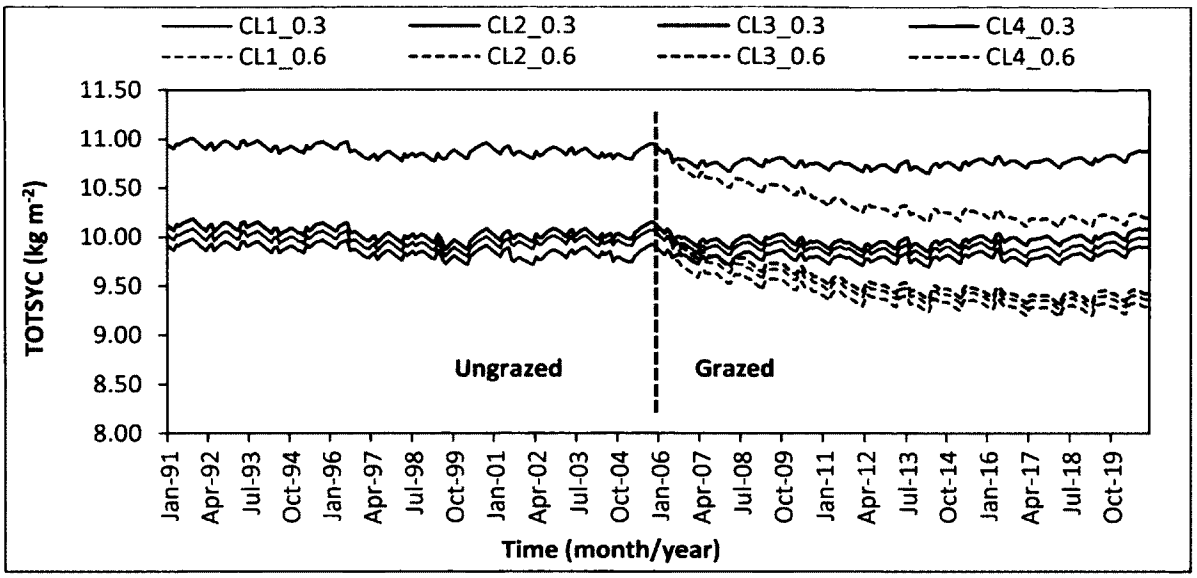
Note: ungrazed period = 1991 – 2005; Grazed period with heavy grazing = 2006 – 2020; CL = clay-Loam; CL1 = sand 40%, silt 24.06%, clay 35.94%, BD = 1.32; CL2 = sand 44%, silt 16.31%, clay 39.69%, BD = 1.31; CL3 = sand 35%, silt 35%, clay 30%, BD = 1.34; CL4 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29. 0.3 = model predictions based on flgrem value of 0.3; 0.6 = model predictions based on flgrem value of 0.6.



A3.8.5 Combined effect of variation within a soil texture and fraction of live shoots (flgrem) removed during a grazing event on SOMTC: An example of clay-loam soil texture.

Note: ungrazed period = 1991 – 2005; Grazed period with heavy grazing = 2006 – 2020; CL = clay-Loam; CL1 = sand 40%, silt 24.06%, clay 35.94%, BD = 1.32; CL2 = sand 44%, silt 16.31%, clay 39.69%, BD = 1.31; CL3 = sand 35%, silt

35%, clay 30%, BD = 1.34; CL4 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29. 0.3 = model predictions based on flgrem value of 0.3; 0.6 = model predictions based on flgrem value of 0.6.



A3.8.6 Combined effect of variation within a soil texture and fraction of live shoots (flgrem) removed during a grazing event on SOMTC: An example of clay-loam soil texture.

Note: ungrazed period = 1991 – 2005; Grazed period with heavy grazing = 2006 – 2020; CL = clay-Loam; CL1 = sand 40%, silt 24.06%, clay 35.94%, BD = 1.32; CL2 = sand 44%, silt 16.31%, clay 39.69%, BD = 1.31; CL3 = sand 35%, silt 35%, clay 30%, BD = 1.34; CL4 = sand 22%, silt 42.06%, clay 35.94%, BD = 1.29. 0.3 = model predictions based on flgrem value of 0.3; 0.6 = model predictions based on flgrem value of 0.6.