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The extent and value of carbon stored in mountain grasslands and shrublands globally, and the prospects for using climate finance to address natural resource management issues

Adrian Ward

B.Bus, PCert.EnergyEcon

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Abstract

The mitigation of climate change is a global priority. The United Nations Framework Convention on Climate Change *Paris Agreement* sets an ambitious goal of limiting global warming well below two degrees Celsius, and ideally at 1.5 degrees Celsius, by the end of the 21st Century. While recent publications have highlighted the important role that terrestrial forests, wetland forests, marine ecosystems (such as mangroves and seagrass meadows) and lowland grasslands play in global climate mitigation efforts, there has been few studies quantifying how mountain grasslands and shrublands might contribute in this regard. These fragile high-altitude (mostly above treeline) ecosystems cover around 6% (9.38 million km^2) of Earth's landmass, provide habitat for rare flora and fauna species while supplying water, food, fibre and economic opportunities to billions of people, many of whom are very poor. Like forests and marine ecosystems, mountain grasslands and shrublands are under threat from multiple anthropogenic stressors. Without a global assessment and understanding of the extent and value of carbon stocks in mountain grasslands and shrublands these ecosystems cannot be effectively integrated into international carbon budgets and climate policy. This thesis therefore helps address this issue by providing both an estimate of C stored in mountain grasslands and shrublands areas, and its relative economic value in climate regulation terms. This thesis also makes several recommendations for how this C pool might be factored into international climate policy frameworks and budgets, and how climate finance might be used to address the various drivers of degradation in mountains grassland and shrubland ecosystems around the world.

Using spatial analysis, this thesis commences by estimating there to be between 60.5 Pg C and 82.8 Pg of C contained within the biomass and soils of the world's mountain grasslands and shrublands in the year 2000. This is a significant amount when considering that global C pools in tropical Savannas and grasslands, temperate forests and tropical peatlands are estimated to be 326–330 Pg C, 159–292 Pg C and 88.6 Pg C respectively. This thesis found that mountain grasslands and shrublands C stocks are most likely not reliably accounted for in international carbon budgets. Building on this initial estimate, this thesis then models the exchange of CO_2 between mountain grasslands and shrublands and the atmosphere, clarifying both in biochemical and economic terms how land use and land use change impacts mountain grassland and shrubland C stocks. Analysis presented in this thesis estimates the value of CO_2 sequestered by mountain grasslands and shrublands to be between US$1.24 billion and US$11.8 billion per annum. It is also demonstrated that if land use was managed more sustainably mountain grasslands and shrublands could sequester up to an additional 8.4 Mt CO_2 per annum while contributing US$0.093 billion - US$0.89 billion annually in added value to society.

This thesis then investigates how climate finance might be used to support priority natural resource management actions in mountain grassland and shrubland ecosystems. The understandings and perspectives of experts on the risks and opportunities of using climate change in this respect, and what methodologies, institutional arrangements and enabling factors required, is presented. A top-down conceptual policy framework is then proposed to assist policy makers in developing key 'enabling factors' with the view of extending the eligibility of carbon markets and climate finance to natural resource management activities undertaken in mountain grasslands and shrublands in the same way that has been afforded to other ecosystems.

The results presented in this thesis have several implications for policy making. First, the results provide a sound first-step in developing global environmental accounts for C stored in mountain grassland and shrubland ecosystems. Moreover, these results provide a baseline estimate and methodology with which to monitor and manage these C stocks. This baseline has direct application for improving the precision of mountain grasslands and shrubland carbon accounting modalities issued by the Intergovernmental Panel on Climate Change, United Nations Framework Convention on Climate Change and carbon offset measurement methodologies e.g. the Verified Carbon Standard. Any improvement in this respect will also improve the reliability of the science, aiding progress towards the targets set by the *Paris Agreement*. Second, from an accounting perspective, this thesis could potentially provide input data into other global studies which have excluded estimates for C in alpine areas which until now has not been available. Third, when combined, the estimates for C stocks, CO_2 sequestration and economic value provided herein justify further investigation of how carbon markets and climate finance might be used specifically to address the factors influencing degradation in mountain grasslands and shrublands around the world.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

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Letters

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Incorporated in Chapter 4

Contributor	**Statement of contribution**
Adrian Ward (Candidate)	Designed experiments (90%) Executed experiments (95%) Literature review (100%) Wrote and edited paper (80%)
Paul Dargusch	Designed experiments (5%) Wrote and edited paper (5%)
Seth Thomas	Wrote and edited paper (5%)
Yan Liu	Executed experiments (5%) Wrote and edited paper (5%)
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Adrian Ward (Candidate)	Designed experiments (90%) Executed experiments (100%) Literature review (95%) Wrote and edited paper (90%)
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Contributions by others to the thesis

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Keywords

Mountain, grasslands, shrublands, global, land use, carbon, economics, policy, finance.

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List of acronyms and abbreviations

AFOLU	Agriculture, forestry and other land use
AGM	Adaptive grazing management
ALM	Agricultural land management
APD	Avoiding planned deforestation
TAR	Third Assessment Report (of the IPCC)
AR4	Fourth Assessment Report (of the IPCC)
AR5	Fifth Assessment Report (of the IPCC)
ARR	Afforestation, reforestation and revegetation
ASL	Above Sea Level
BAU	Business as usual
BioCF	BioCarbon Fund (World Bank)
C	Carbon
CAR	California Climate Action Reserve
CAS	Chinese Academy of Science
CBC	Convention for Biodiversity and Conservation
CCBA	Climate, Community & Biodiversity Alliance
CCER	Chinese Certified Emissions Reductions
CCX	Chicago Climate Exchange
CDM	Clean Development Mechanism
CERs	Certified emissions reductions (under CDM)
CF	Carbon Fund
CGLC	Cropland and grassland land-use conversions
CH_4	Methane
CHF	Crop Harvest Frequency
CIESIN	Center for International Earth Science Information Network
CO_2	Carbon dioxide
CONDESAN	Consortium for Sustainable Development of the Andean Ecoregion
COP	Conference of the Parties (under UNFCCC)
CRF	Common reporting format
DFID	Department for International Development (United Kingdom)
DNA	Designated national authority
DICE	Dynamic Integrated Climate and Economy
DM	Dry Matter

EF	Emissions factor
EP	Ecosystem preservation
ER	Ecosystem restoration
ERs	Emissions reductions
ERPA	Emissions reduction purchase agreement
ESC	Engineered soil conservation
ETS	Emissions trading scheme
EU	European Union
EU-ETS	European Union Emissions Trading System
FAO	Food and Agriculture Organization of the UN
FUND	Climate Framework for Uncertainty
GCF	Green Climate Fund
GCM	General Circulation Model
GDP	Gross Domestic Product
GEF	Global Environment Facility
GHG	Greenhouse Gas
GLC2000	Global Landscape Cover 2000
GIS	Geographical Information Systems
GIZ	Gesellschaft für Internationale Zusammenarbeit
GS	Gold Standard
GtC	Giga-tonnes of Carbon
HWSD	Harmonised World Soil Database
IAM	Integrated Assessment Model
IBM	Individual Based Model
ICIMOD	International Centre for Integrated Mountain Development
IPCC	Intergovernmental Panel on Climate Change
ISS	Institute of Soil Science
ISRIC	International Soil Reference and Information Centre
ISSCAS	Institute of Soil Science Chinese Academy of Science
IUSS	International Union of Soil Science
IUCN	International Union for the Conservation of Nature
JRC	Joint Resource Centre
LDCs	Least developed countries
LULUC	Land use, land use change
MBI	Market based instruments

MGS	Mountain grasslands and shrublands
Mha	Millions of hectares
MODIS	Moderate Resolution Imaging Spectroradiometer
MP	Mountain Partnership (of the UN FAO)
MRV	Measurement, reporting and verification
$MtCO_2$	Megatonne carbon dioxide
NAMA	Nationally appropriate mitigation action
NAP	National allocation plan
NAPAs	National adaptation programmes of action
N_2O	Nitrous oxide
NGO	Non-governmental organization
NIS	National Inventory Submission
NPP	Net Primary Productivity
NRM	Natural resource management
ODD	Overview-Design-Details
PA	Protected Areas
PAGE	Policy Analysis of the Greenhouse Effect
PDD	Project Design Document
PES	Payment for environmental services
Pg C	Petagrams of carbon
PIN	Project Idea Note
PoA	Programme of activities
POL	Policy
PS	Panda Standard
QC	Quality control
RCM	Regional Climate Model
REDD	Reducing emissions from deforestation and forest degradation
REDD+	"Plus" includes the role of conservation, sustainable management of forests, enhancement of forest carbon stocks and support to local communities dependent on forests
SBSTA	Subsidiary Body for Scientific and Technological Advice (UNFCCC)
SC	Sustainable cropping
SCC	Social cost of carbon
SGM	Sustainable grassland management
SMD	Sustainable mountain development

SMU	Soil mapping unit
SOC	Soil organic carbon
TEEB	The Economics of Ecosystems and Biodiversity
tCO_2e	Tonnes of CO_2 Equivalent
$tCha^{-1}$	Tonnes of carbon per hectare
TLU	Tropical livestock equivalent
UN	United Nations
UNCED	United Nations Conference on Environment and
UNDP	United Nations Development Program
UNEP	United Nations Environment Program
UNFCCC	United Nations Framework Convention on Climate Change
US	United States of America
USD	United States Dollars
USLE	Universal Soil Loss Equation
VCS	Verified Carbon Standard
VCUs	Verified Carbon Units
VS	Volatile Solids
WB	World Bank
WCI	Western Climate Initiative
WCMC	Wold Conservation Monitoring Centre
WDPA	World Database of Protected Areas
WWF	World Wildlife Fund

Chapter 1. *Introduction*

Mountains are one of the great geological landforms of our planet, and are of critical importance to humanity. In a global context, mountains cover approximately 24 percent of Earth's landmass (Kapos *et al,* 2000), are home to around 26 percent of the world's population (Meybeck *et al*, 2001) and are the economic backbone for an estimated 40 percent of people, many of whom are very poor (Ariza *et al*, 2013; Beniston, 2003). Moreover, mountains, which are often referred to as the 'water towers of the world', provide clean water to more than half of the world's population. Mountains also shelter nearly 50 percent of Earth's biodiversity 'hot spots' and afford important ecosystem-derived goods (e.g. food, fibre and medicines) and services (e.g. hazard reduction, recreation cultural and renewable energy) to people living both within and outside mountain areas (Ballabio *et al,* 2012; Beniston, 2003; Blyth *et al*, 2003; Costanza *et al,* 1997; Rashid *et al,* 2005; TEEB Synthesis, 2010; Singh, 2011). Importantly, six of the 20 plant species that supply more than three quarters of the world food supply originate in mountains (Panos Institute, 2002). Mountains also have a profound biophysical influence on regional weather patterns such as the South Asian summer monsoon which impacts more than a billion people (Turner and Annamalai, 2012). The significant contributions made by mountain regions is recognised by numerous internationally significant accords, and in particular, *Agenda 21* which was adopted at the first UN Conference on Environment and Development (UNCED) in 1992, and in *The Future We Want* publication adopted 20 years later at the UNCED RIO+20 conference.

> "Mountains are an important source of water, energy and biological diversity. Furthermore, they are a source of such key resources as minerals, forest products and agricultural products and of recreation. As a major ecosystem representing the complex and interrelated ecology of our planet, mountain environments are essential to the survival of the global ecosystem. Mountain ecosystems are, however, rapidly changing. They are susceptible to accelerated soil erosion, landslides and rapid loss of habitat and genetic diversity. On the human side, there is widespread poverty among mountain inhabitants and loss of indigenous knowledge. As a result, most global mountain areas are experiencing environmental degradation. Hence, the proper management of mountain resources and socio-economic development of the people deserves immediate action".
>
> *Agenda 21, Section 13.1 (UNCED, 1992)*

Despite their environmental and socioeconomic significance, many mountain ecosystems around the world are being degraded due to unsustainable land use and land use change (LULUCF) (Körner *et al*, 2005; Ward *et al*, 2015). According to Beniston (2003) and Jansky (2002) the underlying drivers of environmental degradation in mountains are globalisation, population and economic growth. Like for low-land economies, the rapid industrialisation of the 20th century has driven a rapid boom in mountain-based mining, tourism, hydro energy provision, forestry and agricultural intensification, all of which have had a direct and negative impact on fragile mountain ecosystems (Bradley *et al*, 2012; European Commission 2008; Grabherr, 1994; Rammig *et al* 2010; Schroter *et al*, 2005;). For many developing countries, this deterioration in the natural resource base has also increased poverty, unemployment, poor health, loss of culture, conflict and outmigration in mountain regions (Hurni, 1999; Jansky *et al*, 2002; Körner *et al*, 2005). Indirectly, climate change, which has been observed and forecast to drive greater temperature increases in mountains and at high latitudes, is serving to magnify this degradation by weakening the adaptive capacity and resilience of fragile mountain ecosystems (Woodwell, 2004).

Like for other ecosystems, the loss and damage caused to mountain ecosystems presents an economic cost to society which will be born mostly by future generations in the form of climate change impacts, a degraded natural resource base and other environmental and socioeconomic impacts (Hurni, 1999). From an ecological economics perspective, environmental degradation is considered an externality of intergenerational concern given it severely impacts ecosystem services which are of central importance to human health, livelihood, survival and well-being (Costanza *et al*, 1997; Millennium Ecosystem Assessment, 2005; TEEB Foundations, 2010; TEEB Synthesis, 2010).

As noted by *Agenda 21*, *Our Future We Want* and in much of the literature, strategic sustainable mountain development (SMD) policies and targeted natural resource management (NRM) action is needed urgently to address this loss and damage (Hurni, 1999; Price, 2007; UNCED, 1992). Of all mountain ecosystems this is particularly true for mountain grasslands and shrublands (MGS). MGS can be identified as biogeographically-derived ecoregions that exist at higher altitudes (typically but not always above treeline) and include Northern Andean Páramo, East African Montane Moorlands, Central Range Subalpine Grasslands, European Calcareous Grasslands, Sayan Alpine Meadows and Tundra, and Central Tibetan Plateau Alpine Steppe (Olson *et al*, 2001; WWF, 2000). To date, the bulk of mountain-focused academic literature and policy analysis has focused on the NRM of mountain forests rather than MGS. Perhaps this is due to difficulties in accessing these often remote, high-altitude and sparsely populated areas. Or, perhaps it is due to the lack of general knowledge, empirical data and/or an understanding of the economic value that MGS bring to society (Gurung *et al*, 2012; Körner *et al*, 2005). Though not the only aspect, understanding the relative and absolute value of ecosystem services is important if awareness for their relative importance to human wellbeing is to be acknowledged (Costanza *et al*, 2014).

This knowledge is also critical given that securing NRM funding remains a serious challenge for governments around the world, who are facing a multitude of NRM issues (WWF-MPO, 2003), and where mountains, which are often remotely located and thus 'out-of-sight' and 'out-of-mind', often seem to be at the end of the funding queue (Ward *et al*, 2015). In our public finance constrained world, many experts, development organisations and governments advocate the use of market-based instruments (MBIs) and unconventional funding mechanisms routed in ecological economic principles and the so-called 'green economy' in order to help solve mountain-based NRM issues (ICIMOD 2012; Richardson, 2013; Wentworth Group of Concerned Scientists, 2015; Worboys and Good, 2011). These mechanisms include carbon markets, green bonds, payments for ecosystem services (PES), impact investing and direct private equity investment (amongst other existing and emerging mechanisms). While such mechanisms offer part of the solution to addressing socio-ecological loss and damage, inherent scarcity means trade-offs about which ecosystems to support (and to what extent) will need to be made at some point in the decision making process (Leader-Williams *et al*, 2010). Making value judgements in this respect is also often unavoidable for government. Incorporating a transparent (though limited) valuation for ecosystems in this process is helpful if their benefits are to be considered in relative terms (Costanza *et al,* 2014). Knowing the value can also assist in the effective management of ecosystem services, particularly when the use of economic incentives are involved (Farley and Costanza, 2010).

A wealth of research into the economic value of ecosystem services has been undertaken since the early 1970s (Daily, 1997; de Groot, 1987; Ehrlich and Ehrlich, 1981; Ehrlich and Mooney, 1983; Odum, 1971; Westman, 1977). Recently, Costanza *et al* (2014) updated their original landmark study published in *Nature* (Costanza *et al,* 1997), putting the annual economic value of the world's natural capital and ecosystem services at US$125 trillion. This global study estimated the *non-market* value for many ecosystems, based on the regulating, provisioning, habitat and cultural services they provide (TEEB Synthesis, 2010). Thousands of other studies, too numerous to mention here, have built on the concepts proposed by Costanza's original 1997 paper, increasing both the number and resolution of economic estimates for many ecosystem services around the world (Braat and de Groot, 2012). Despite this surge of research there are very few studies enquiring as to the economic value for the ecosystems services derived from MGS, let alone a global estimate that would be particularly useful to policy makers at the national, international and regional levels (Ward *et al* 2015). Without this value, the contribution that MGS make to humanity cannot be considered fairly in the decision making process.

Establishing a reliable global estimate for the economic value of all ecosystem services provided by MGS is likely to be a challenging task given the lack of integrated systems research and the numerous data gaps that exist for mountains (Gurung *et al*, 2010; Jansky, 2002; TEEB Foundations, 2010; UN FAO 2015; Ward *et al* 2015). Moreover, these gaps are particularly apparent at the global scale (Debarbieux and Price, 2008). With this in mind, this thesis advocates a more pragmatic approach by focusing on valuing those ecosystem services which are connected to established economic incentives, with the view of using them to address the NRM challenges discussed above. Given the global scale of this study, focus should also be put on ecosystem services that have a global impact. Climate regulation ecosystem services, that is the biosequestration of carbon dioxide (CO_2) from the atmosphere into biomass and soils which serves to mitigate climate change, have been connected to carbon markets around the world for many years. Under the right conditions, climate finance incentives that encourage avoided deforestation, reforestation, and grassland and soil management, have supported global efforts to reduce GHG emissions and been proven to be co-effective in encouraging more sustainable NRM in both terrestrial and marine ecosystems (Thomas, 2013).

An initial assessment of climate policy discourse suggests that it has largely failed to consider the specific and critical role that MGS ecosystems play in international carbon accounts and global climate mitigation. This is evidenced by the results of an extensive literature review of key areas (discussed in relevant chapters in this thesis) which found relatively few empirical local studies in MGS carbon stocks, and no high-level global studies. This thesis seeks to fill a number of key literature gaps by answering the research questions below.

Research Question 1. What is the spatial distribution and significance of carbon stored in the world's MGS? How is it accounted for in global carbon budgets and international carbon accounting frameworks?

Research Question 2. To what extent is carbon globally exchanged between MGS and the atmosphere? How is this impacted by land use change? What is the economic value of these exchanges and as an ecological asset, when considering climate policy and broader sustainable perspectives?

Research Question 3. What are the stressors, NRM challenges and priorities related to carbon stocks in MGS? Why has climate finance not been utilized in this context? What is required to position these NRM activities eligible for carbon finance incentives, and in so doing, ensuring that MGS are more sustainably used and the aforementioned ecological economic value is maintained and/or improved?

This thesis begins (*Chapter 2*) by way of a general literature review, summarising the biophysical, ecological, social and economic aspects of mountains so one can better appreciate the dynamics and limitations to carbon storage in an MGS context. It also outlines the benefits and challenges in valuing the climate regulating services provided by MGS, the connection with ecosystem service valuation, and also how climate finance can be used to address NRM issues and SMD objectives.

Chapter 3 gives an overview of how the thesis methodology meets the overall research objective as-a-whole, and summarises how the methodologies presented in each of the key results driven chapters (*Chapters 4-6*) meet the specific research questions of the thesis.

Chapter 4 provides a spatially resolved estimate of C contained within the biomass and soils of the world's MGS, as published in *Global Environmental Change* by Ward *et al* (2014). It also puts this amount into perspective by comparing C stores in other terrestrial and marine ecosystems. *Chapter 4* then proceeds to review existing empirical studies and United Nations Framework Convention on Climate Change (UNFCCC) national greenhouse accounts, to ascertain if this C is reliably accounted for in international carbon budgets. This estimate is the first to provide a global point of reference, useful in developing future research and in climate policy discussions. *Chapter 4* concludes by briefly discussing how climate finance might be leveraged to support the sustainable management of these C stocks.

Chapter 5 makes use of an Individual Based Model (IBM) and the best available spatial input data (including the outputs of *Chapter 4*) to understand how land use and land use change (LULUC) influences Net Primary Productivity (NPP) and soil loss in MGS ecosystems, and subsequently estimate CO_2 fluxes between MGS and the atmosphere over a 15 year timeframe. It then translates this into an annual CO_2 sequestration estimate for MGS, before putting a range of economic values on this important ecosystem service by quantifying the avoided climate change induced damage to society. This chapter is currently under review with *Ecological Economics*.

Chapter 6 considers the magnitude of C stored by MGS ecosystems, and postulates how climate finance may be used to address specific LULUC and NRM stressors, what barriers exist to the implementation of climate finance, and proposes a framework to address these barriers and ultimately enable climate finance to be used to protect and enhance MGS ecosystems like it has been used for other biological C stores. Moreover, this chapter also considers how SMD focused experts understand the aforementioned points and if opportunities with respect to using climate finance for MGS NRM might have been missed. This chapter was published in *Climate Policy* (Ward *et al*, 2015).

Finally, *Chapter 7* summarises how the thesis has met the overall research objective and specific research questions, highlights the strengths and limitations of the study, and points to how the results of the thesis may contribute to better awareness and knowledge in the aforementioned critical areas of ecosystem services and climate finance, serve to provoke future research, and ultimately how it may aid in the sustainable management of MGS ecosystems.

Chapter 2. Context

2.1 Defining mountains

For many, the term 'mountain' conjures up images of towering snow-covered rocky monoliths, such as those peaks found in the Himalayas. For others, 'mountain' means something quite different, for example a relatively small and gentle rise in land covered by dense tropical rainforest. The academic definition of 'mountain' has been the subject of long-running debate amongst researchers. In 1990, Gerrard (p.7) noted that "Numerous definitions of what constitutes a mountain have been proposed, but mountains are extremely diverse landforms and it has proved difficult to achieve consistency in description and analysis. Several criteria have been used, such as elevation, volume, relief, and steepness, as well as spacing and continuity". The use of different criteria has caused wide-ranging estimates for global mountain coverage. For example, Fairbridge (1968) estimated that mountains, highlands and hill country cover 36 percent of the Earth's landmass, while Louis (1975) proposed that mountains cover only 20 percent. Messerli and Ives (1997) used altitude as the only criterion to estimate global coverage to be 48 percent. Kapos *et al* (2000) point out that, like previous studies, this last estimate is skewed as it takes into account large areas of mid-elevation plateau that are not really 'mountainous'. Though there is general consensus that altitude and steep slopes are key components of mountains (Ives *et al*, 1997), choosing an appropriate altitude threshold, such as one based on high-elevation and human physiological impact (i.e. oxygen availability), is often difficult because older and lower elevation mountain systems are consequently excluded (Kapos *et al*, 2000). The use of timberline and ecological variation is also problematic as both are impacted by altitude, latitude, the continent on which it's located and mountain mass (Holtmeier, 1994; Tranquillini, 1979; Troll, 1973).

More contemporary approaches have utilised GIS to achieve a more meaningful definition of what constitutes a 'mountain'. A recent study by Kapos *et al* (2000) used a Digital Elevation Model (DEM) to determine seven mountain classes (Figure 1). Using this method, elevation thresholds were developed based on the altitude above which humans are affected by oxygen depletion, slope angle for middle elevation mountains, and local elevation range as a way of including low-elevation and older mountain ranges. In this thesis, we adopt Kapos *et al's* (2000) definition of 'mountain'.

Figure 1. Mountain classes according to elevation range, and % of global landmass

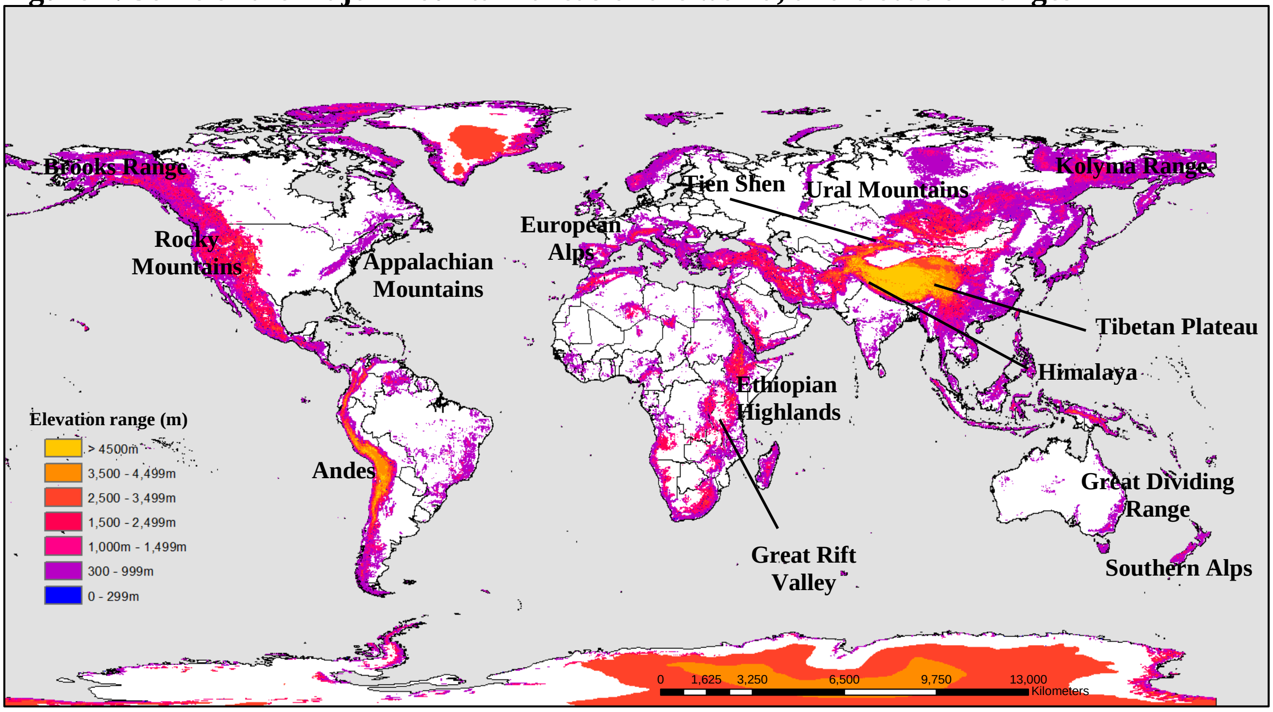
Class	Elevation range (m), slope (degrees)	% of global land mass
I	>4,500m	1.2
II	3,500 – 4,500m	1.8
III	2,500 – 3,500m	4.7
IV	1,500 – 2500m, Slope ≥ 2º	3.6
V	1,000 – 1,499m, Slope ≥ 5º or local elevation range (7km radius) > 300m	4.2
VI	300 – 1,000m and local elevation range (7 kilometre radius) 300m outside 23ºN-19ºS	8.8
VII	Isolated inner basins and plateaus less than 25 square kilometres in extent that are surrounded by mountains but do not themselves meet criteria 1–6	N/A

Adapted from: Kapos *et al*, 200l; Körner *et al*, 2005.

2.2 Mountains of the world

The Northern Hemisphere contains most of the world's mountains, particularly in the sub-tropic latitudes, boreal and subpolar zones (Figure 2). These mountain systems include the Altai, Brookes, Rockies and Tien Shen ranges. Notably, Eurasia is the most mountainous landmass on earth, home to all of the world's mountains above 7,000 metres, with the Himalayan range containing all peaks above 8,000m (Körner *et al*, 2005). The Eurasian landmass also features the extensive Tibet-Qingzian plateau (approx. 2,500,000 km^2), the most densely populated area above 2,500m (National Geographic, 2006). South America has the second most extensive area of mountainous area, and features the Andes range which forms an impressive high-elevation backbone of rock, ice and MGS ecosystems running approximately 7,000 km down the length of the continent. This is followed by Antarctica and Greenland, though this is difficult to observe as much of the mountainous terrain remains covered by thick icecaps. Other relatively small, though important, mountain systems include the New Zealand Alps, the Great Dividing Range in Australia and the Great Rift Valley of Africa.

Figure 2. Some of the major mountain areas of the world, and elevation ranges



Adapted from: Kapos *et al*, 200l; Körner *et al*, 2005.

Figure 3. Photos of some of the world's major mountain ranges

The Karakoram, Pakistan
Credit: Guilhem Vellut, 2016

Great Rift Valley, Africa
Credit: Msafiri, 2014

The Great Dividing Range, Australia
Credit: Buzzle, 2016

The Andes, Chile
Credit: Vagabondanna, 2016

2.3 Mountain grasslands and shrublands, the biophysical environment

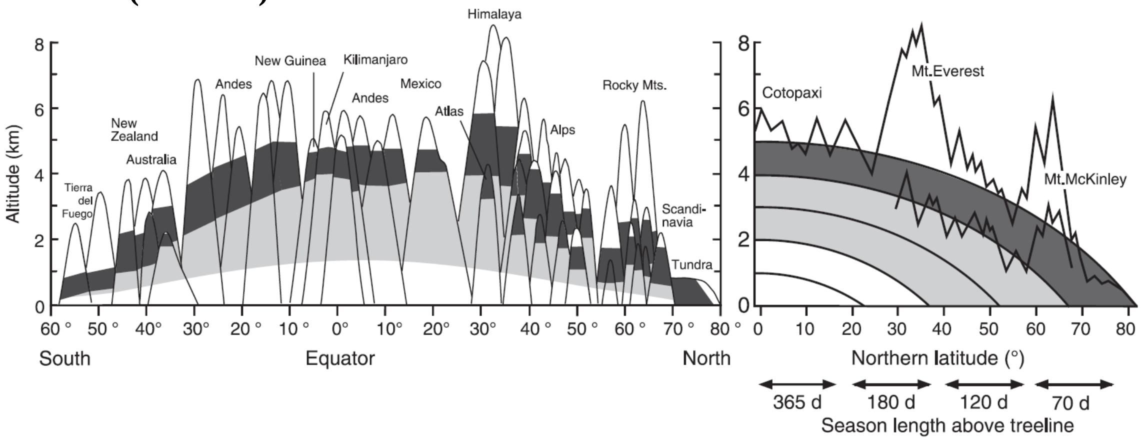
The biophysical environment encountered in mountainous areas is frequently described as ‘harsh’ and ‘unforgiving’, due to relative isolation, rapid and unpredictable changes in weather (wind, precipitation, temperature), relatively low temperatures (including snow and ice), low oxygen levels at higher elevations, extreme solar radiation, steep slopes and natural hazards (e.g. rock-fall and avalanches), dry conditions and shallow erosion-prone soils making agricultural production labour-intensive and often marginal (Jansky *et al*, 2002; Körner *et al*, 2005). As will be discussed in later chapters, these factors are critical to the objective of this study.

Due to their high elevation and vertical profile, mountains can also change air flow and influence local and regional weather. For example, the Himalayas are a key driver of India’s climate, sheltering it from cold northern winds and also influencing the annual South Asian monsoon (Turner and Annamalai, 2012).

Mountain environments encompass four distinctive altitudinal belts (Figure 4). At the highest altitudes exist the ‘nival belt’, a zone that is often referred to as the ‘cryosphere’ and which stores enormous amounts of water as snow and ice (Turner and Annamalai, 2012). These unique ‘water batteries’ release fresh water every year into many of the world’s great rivers, sustaining billions of people (Barnett *et al,* 2005). Lower down, the ‘alpine belt’ incorporates the zone that ranges from the tree-line to the seasonal snowline, where vegetation cover is below 20 percent and where trees fail to grow. The ‘subalpine belt’ is the zone between the upper timberline (where trees form a closed canopy and where height is 3m or more) and the start of the alpine zone. The ‘montane’ altitudinal belt extends from the upper timberline to the lower mountain limit and includes both trees and isolated pockets of grassland and shrubland ecosystems (Grabherr *et al,* 2003; Körner *et al,* 2003).

For the purposes of this thesis, ‘trees’ are defined as “as an upright woody plant with a dominant above-ground stem that reaches a height of at least 3m” (Körner 1998, p.445). MGS can be considered as plants that do not meet this criteria. While prevalent in the treeless alpine, MGS can also form tight assemblages around the stunted trees of the subalpine transitional zone and below within the forests and woodlands of the montane altitudinal belt due to a number of localised natural (e.g. ‘frost hollows’) and anthropogenic factors e.g. burning to clear land for agriculture (Körner *et al,* 2005).

Figure 4. Classic Humboldt Profile of the Latitudinal Position of Altitude Belts in Mountains across the Globe and Compression of Thermal Zones on Mountains, Altitudes and Latitudes. Grey is Montane; Black is Alpine/Subalpine and White is the Nival Belt (snowline).



Source: Körner, 2003.

Though the treeline is influenced by a number of factors (e.g. root zone temperature, radiation, moisture availability) it remains fairly consistent worldwide, though may not be visible on many mountains where forests have been converted to cropland or pasture, for example. For this reason, the montane belt may also include grassland and shrubland ecosystems which are thus subject to the relative biophysical factors described above (Körner *et al*, 2005). In this thesis, understanding the characteristics of the treeline is important given that most of the world's MGS are located above it.

While mountains feature a dynamic biophysical environment at the local and regional level, it is important to note the changes taking place at the global level. Mountains have already experienced three times more warming than the global average, with projections that temperatures are likely to increase dramatically by the end of the century, effecting hydrological and geomorphic processes, with negative flow-on impacts for society and the economy in both the highlands and lowlands (Benitson, 2002; Christiansen *et al*, 2007). These projections are however coarse and caution should be taken when interpreting the results as few General Circulation Models (GCMs) and Regional Climate Models (RCMs) have the fine resolution required to adequately simulate the topography found in mountain areas (Christiansen *et al*, 2007). Though many uncertainties exist, climate change is expected to cause an altitudinal lift in the global treeline, and eventually, the vegetation (and soils) found on mountain slopes (Beniston, 2001; UK Met Office, 2009).

2.4 Mountain grasslands and shrublands around the world

One of the key objectives of this thesis is to contribute to the general lack of data in the field of SMD in order to support better NRM for MGS. As featured in *Chapter 4*, Ward *et al* (2014) provides the first estimate for montane, subalpine, and alpine grasslands and shrublands coverage at approximately 9.38 million km^2 (around six percent) of the Earth's terrestrial landmass. Like for mountains in general, most MGS ecoregions[1] are located in the Northern Hemisphere (Figures 5 and 6), with significant areas in China (e.g. Tian Shan Montane Steppe and Meadows, South East Tibet Shrublands & Meadows, Tibetan Plateau Alpine Shrublands & Meadows, Central Tibetan Plateau Alpine Steppe, Quilian Mountains Subalpine Meadows), the Russian Federation (e.g. Cherskii-Kolyma Mountain Tundra, Trans-Baikal Bald Mountain Tundra), Greenland (e.g. Kalaallit Nunaat High Arctic Tundra), India (e.g. Eastern Himalayan Alpine Scrub and Meadows) and United States (e.g. British Range, Interior Yukon-Alaska Alpine Tundra, Alaska-St. Elias Range Tundra) (Olson *et al*, 2003; WWF, 2000).

South America contains the Southern Hemisphere's most extensive expanse of MGS ecoregions, and include Northern Andean Páramo, Cordillera de Merida Páramo and Central Andean Dry Puna. Following this is South Africa and its Drakensberg Montane Grasslands and Highveld Grasslands. Smaller and often isolated MGS ecoregions can be found around the world, such as throughout the west coast of North America (e.g. Pacific Coastal Mountain Tundra, Sierra-Nevada Alpine and Subalpine meadows, Rocky Mountain Alpine and Subalpine Meadows), Australia (Australian Alps Montane Grasslands), Europe (e.g. European Alps Alpine Meadows, Pyrenees Alpine Meadows and Upland Calcareous Grasslands), New Zealand (South Island Montane Grasslands), Papua New Guinea (Central Range Subalpine Grasslands), Africa (e.g. East African Montane Moorlands) and broader Asian region (e.g. Altai Alpine Meadows in Kazakhstan) (Olson *et al*, 2003; WWF, 2000).

[1] Ecoregions represent larger land size units than ecosystems, but both are constructed using the same biogeographical criteria. Ecoregions, also known as bioregions, can be thought of as a repetitive pattern of ecosystems that have common regional soil and landform characteristics (Brunckhorst, 2000)

Figure 5. The Tibet-Qingzian plateau in China, sometimes called the "third pole", contains the most extensive area of mountain grasslands and shrublands in the world.

Credit: C.Loyd, 2013.

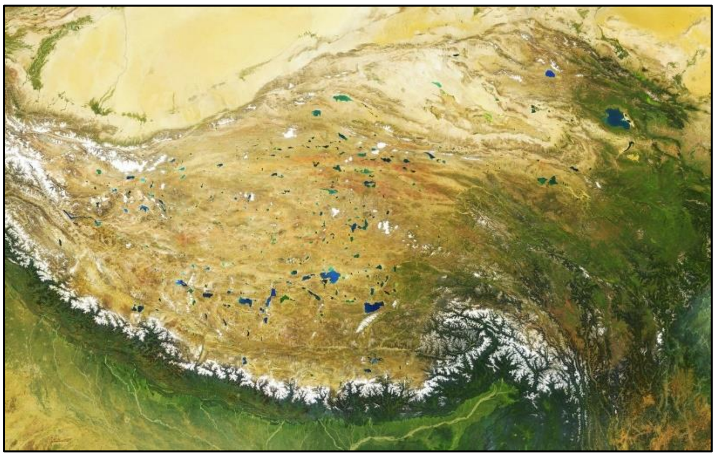
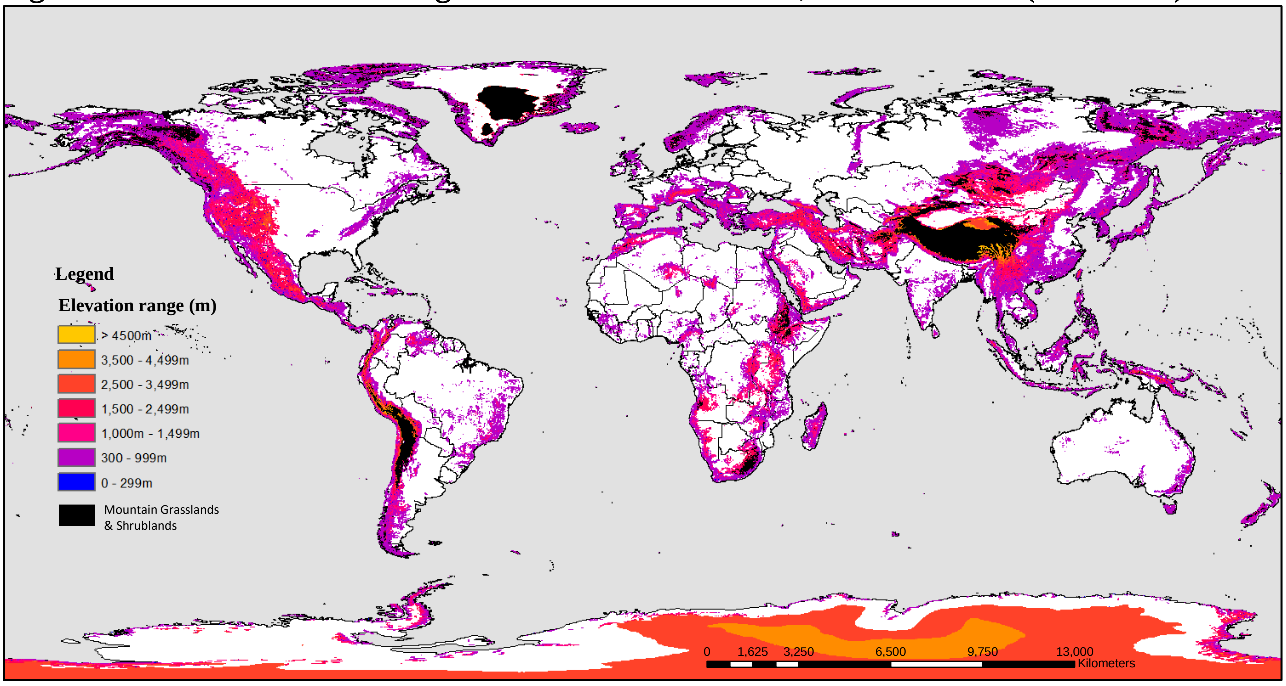


Figure 6. Location of mountain grasslands and shrublands, versus altitude (Year 2000)



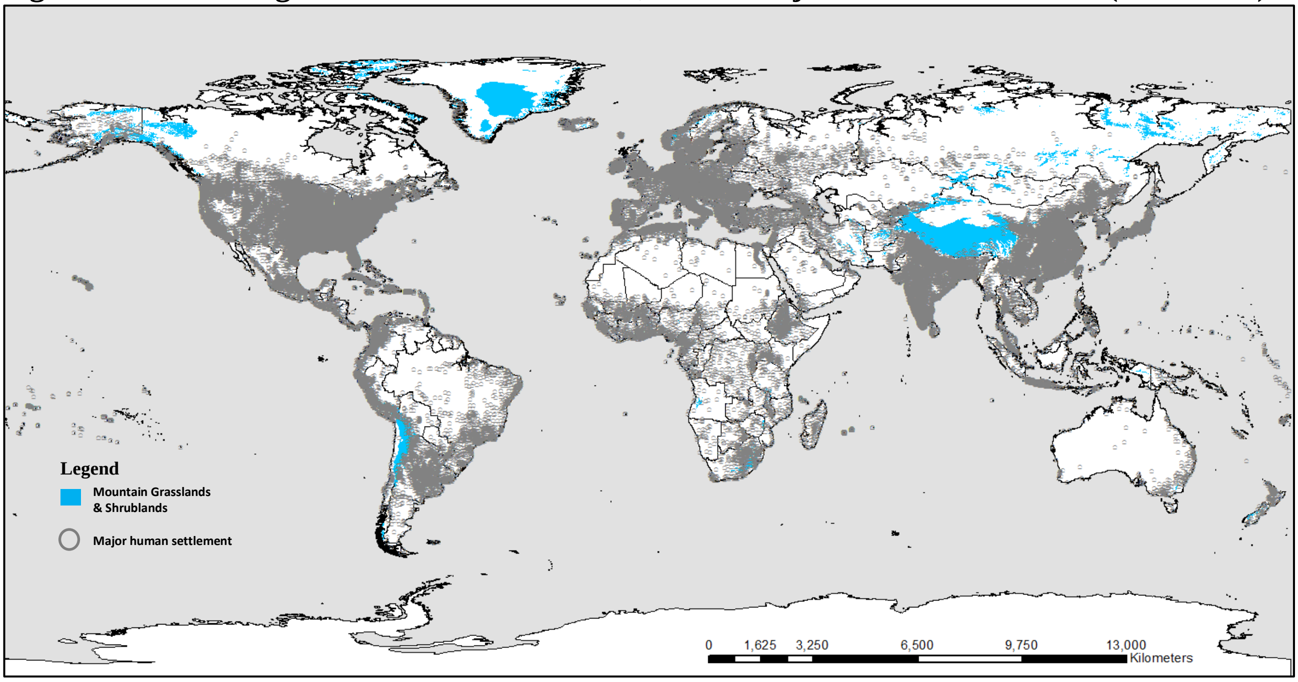
Data sources: WCMC, 2002; WWF, 2000.

2.5 Human populations and the role of mountain grasslands and shrublands in society

In 2000, the worldwide mountain population was estimated to be 1.2 billion, 20 percent of all people living on Earth (Körner *et al*, 2005). 70 percent of these people live below 1,500m above sea level (asl), with only eight percent (around 90 million) living above 2,500m asl. At this altitude the majority of land can be categorised as grazing, barren, sparsely vegetated or mix of all three. Generally, mountain population density decreases with altitude, with the lowest proportion of people living within the alpine and nival zones. Whilst there are no existing studies quantify the number of people living in MGS globally, through GIS mapping (Figures 7 and 8) it can be observed that many MGS (e.g. Tibet-Qingzian, Alaska and in Northern Russia) cover vast geographical areas that are isolated from major human settlements and located considerable distances from major roads (Huddleston *et al*, 2003).

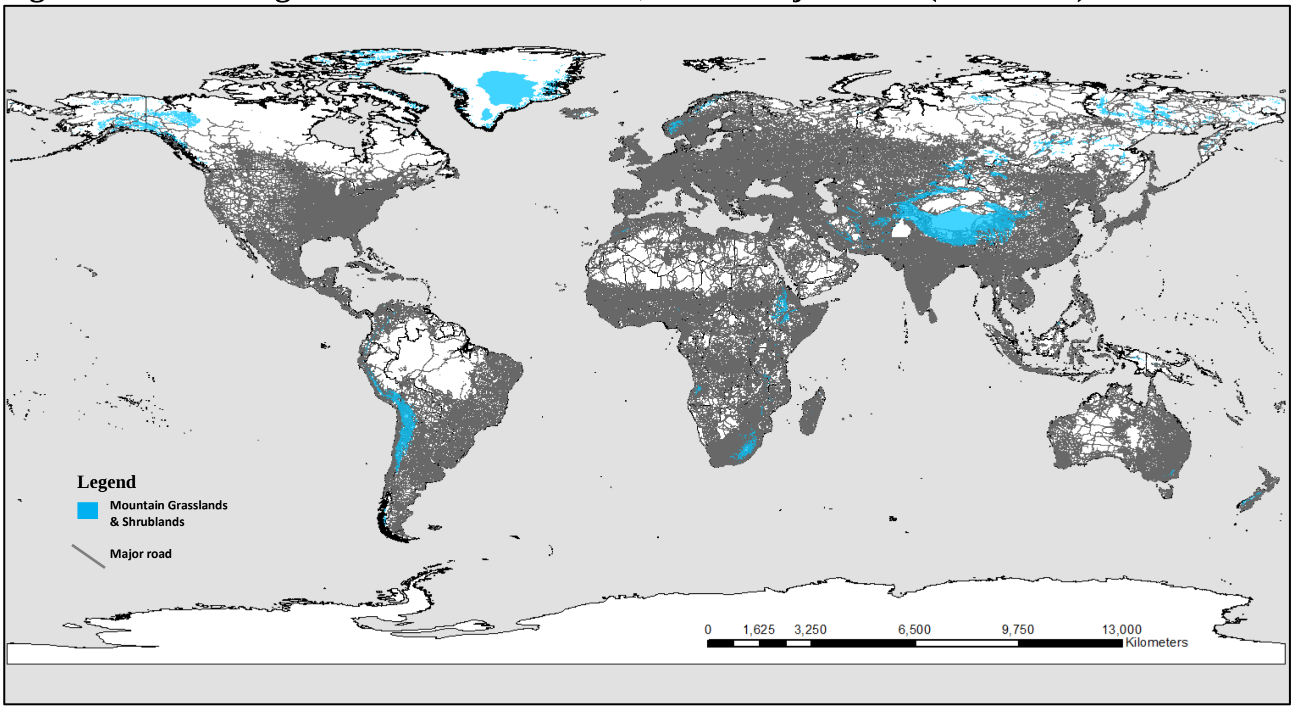
One of the main reasons for this lack of access is that traditionally MGS ecosystems have only generated relatively low value *in-situ* products (e.g. food and fibre) offer little incentive to build expensive links to markets (Körner *et al*, 2005). For developing countries, where food and fibre production is the predominant form of economic activity in most MGS areas, family-driven subsistence systems are common place (Körner *et al*, 2005). Access to market provides an opportunity to diversify into cash-crop systems, so without mountain roads, communities are at a disadvantage compared to their low-land counterparts (FAO, 2015). Moreover, if mountain communities *are* able to get their product (e.g. coffee) to market, as ‘price takers’ they are exposed to global commodity rates and higher transaction costs (Körner *et al* 2005; Wymann *et al*, 2013).

Figure 7. Mountain grasslands and shrublands, versus major human settlements (Year 2000)



Data sources: Balk *et al*, 2006; WCMC, 2002; WWF, 2000.

Figure 8. Mountain grasslands and shrublands, versus major roads (Year 2000)



Data sources: CIESIN, 2013; UNEP-WCMC, 2002; WWF, 2000.

These issues, combined with the low literacy rates, conflict, food insecurity, geographical fragmentation, different languages and cultures within the same regions, a general lack of accessible economic resources and the harsh biophysical factors already described above (especially low temperature) mean that MGS areas tend to have the highest poverty levels in the world (Starr, 2004; Wymann *et al*, 2013). Ives (1997) estimated that around 80 percent of the world's mountain population lives below the poverty line. The barriers that these issues present for utilising climate finance incentives to support SMD in MGS is discussed in detail in *Chapter 6*.

Notwithstanding these constraints, globalisation offers new opportunities for mountain communities to diversify into non-traditional goods and services. Non-traditional goods and services include, for example, growing markets for tourism and speciality mountain products (Wymann *et al*, 2013). Mountain tourism alone is estimated to worth US$70 to 90 billion per year (Panos Institute, 2002). However, while mountain tourism is an attractive and growing proposition, it can also have serious impacts on fragile MGS ecosystems (e.g. trampling of vegetation and exposure of soils) and in limiting sustainable development by competing with traditional subsistence needs, such as the harvesting of biomass for household energy requirements (Grabherr, 1982).

In recent years, the growth of high-altitude mining has also had a significant impact on local mountain communities, mainly due to advances in technology that have enabled machinery to work more reliably and efficiently in low-oxygen environments (Kraul, 2014). Increased competition for traditional agricultural lands, the contravention of the human rights of indigenous peoples, conflict between local and migrant workers, child labour, additional workplace hazards associated with the extreme environment (e.g. Acute Mountain Sickness) and the widespread degradation to the fragile MGS are becoming more common (Kraul, 2014; Mackay, 2006; Verrier and Greenberg, 2011). Public concern about the social and environmental impacts of high-altitude mining have primarily been centred on the Páramo ecosystems in Colombia and elsewhere in South America, and the extensive grasslands of the Tibetan plateau which are relied upon by local herdsman.

2.6 Biodiversity in mountain grasslands and shrublands, and protected areas

Mountains and MGS exhibit some of the highest species richness in the world, contributing around 25 percent of Earth's biodiversity and 50 percent of its biodiversity hotspots (Moser *et al*, 2005; Spehn and Körner, 2005; Singh, 2011; Väre *et al*, 2003). For example, the Indian Himalaya has more than 8,000 species, including snow leopards and Ibex (IIRS, 2003). Alpine ecosystems alone are estimated to contain about four percent of the world's flowering plant species (Körner, 2004). These statistics are surprising if one is to consider the aforementioned harshness of the biophysical environment. Conversely, it is these factors that sustain conditions supportive of highly adapted and diverse lifeforms.

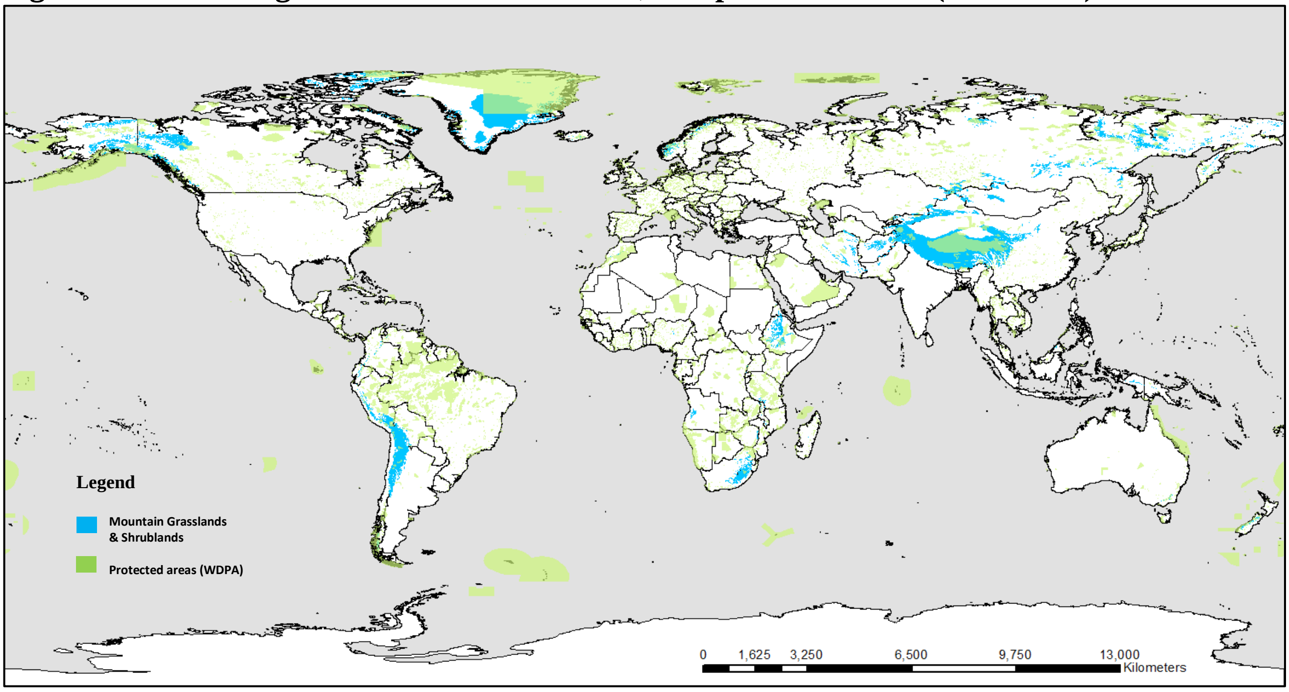
The climate on any particular mountain slope can vary greatly over a relatively short horizontal distance due to the sharp vertical profile that mountains commonly feature (Beniston, 2003). The thermal gradient is about 600 to 1,000 times greater in mountains compared to the conforming latitudinal temperature gradient. Such a rapid spatial change in climate also corresponds to a rapid spatial change in vegetation and hydrology (Whiteman, 2000). Slope angle and gravity-induced erosion has a significant impact on shallow mountain soils and therefore vegetation (Körner, 2004). For these reasons, MGS exhibit high biodiversity with clearly observable ecotones governing the transition between vegetation types (Beniston, 2003).

Due to their relative geographic isolation and a long cultural history of mountain communities protecting fauna and flora, genetic diversity is also notably high in MGS ecosystems. This isolation has motivated many experts to coin 'mountains' as "islands" in a sea of human influenced landscapes, serving as the last refuges for the endemic species they contain (Körner *et al*, 2005). The vertical profile of mountains cause native MGS plant species to be found in some of the world's most unexpected places, such as in the alpine zone of the Malaysia's Mount Kinabalu which has more 4,000 species of alpine plants (Kohler and Maselli, 2009).

MGS are also important from an agrobiodiversity perspective, with six of the 20 crops that supply 80 percent of the world's food supply (e.g. barley, maize and potatoes) originating from mountains (Singh, 2007). This includes around 4,000 varieties of native potatoes and 13,000 medicinal plant species (Wymann *et al*, 2013). Equally, MGS ecosystems have supported the agrobiodiversity amongst animals e.g. Yaks, alpacas, llamas, sheep and goats (Singh, 2007). For this reason, while maintaining healthy, 'natural' MGS ecosystems above the treeline is desirable to ensure the conservation of endemic biodiversity, it is equally important to consider the sustainable management of 'semi-natural' MGS ecosystems of anthropogenic origin, particularly those rangelands below treeline which have economic high importance to indigenous herdsman (Gao *et al*, 2014; Körner *et al*, 2005). The latter are dominant in many parts of the European Alps and Tibetan plateau, and have been shaped and sustainably managed over the course of thousands of years. Many semi-natural/managed MGS in the upper montane, subalpine and alpine zones reflect traditional and sustainable farming regimes that host rich, diverse and highly adapted native flora and fauna communities (Körner *et al*, 2005). Moreover, many are the target of protected area establishment because they exhibit incredibly high levels of biodiversity (Körner, 2004).

However, as highlighted in *Chapter 6*, despite their relative isolation, anthropogenic threats to biodiversity from unsustainable tourism, high-altitude mining and agricultural intensification are of real concern given MGS ecosystems are highly vulnerable to global change (Macchi, 2010; Price and Butt, 2000; Ward *et al* 2015). According to Beresford *et al* (2010), Chape *et al* (2005; 2008) and Jenkins and Joppa (2009) protected areas are the most cost-effective and common means to protect biodiversity. Under the Convention on Biological Diversity (CBD), member governments agreed that 17 percent of terrestrial landmass and inland rivers should be protected by 2020. This included a theme focused on mountains. While the 17 percent protected area (PA) target has been met for mountains on a global scale (noting that 32 percent of all protected areas can be considered 'mountainous'), NRM and connectivity at the local and regional scales remains uneven and insufficient to the long-term preservation of many MGS species (Rodríguez-Rodríguez *et al*, 2011) (Figure 9). The merits, limitations and challenges of using PAs to conserve MGS is discussed later in this thesis, as is the urgent need for novel approaches to NRM strategy and funding.

Figure 9. Mountain grasslands and shrublands, and protected areas (Year 2000)



Data source: IUCN and UNEP-WCMC, 2000; UNEP-WCMC, 2002.

2.7 Mountain grassland and shrubland plant ecology

As already highlighted, mountains exhibit extreme environmental conditions which ultimately governs the flora species found on any given slope, at any given altitude and on any given aspect. As the climate becomes more extreme (e.g. as altitude, exposure and marginal soil conditions increase) plants respond accordingly with the diversity and structure of vegetation communities becoming more woody, stunted, and eventually only close-growing to the barren and rocky ground e.g. lichens (Körner, 1999; Woodwell, 2004).

Alpine plant life can be likened to artic plant life in that both have a relationship with the treeline, whether this is a latitudinal band of boreal forests that encircle the Earth or a distinct altitudinal transition line that may exist high up on the side of mountain (Billings and Mooney, 1968). Though the treeline remains the most reliable indicator to distinguish between artic and alpine vegetation, there are limitations, as both types may occur below treeline due to natural and anthropogenic reasons. This is especially true for the Southern Hemisphere and the Andes in central Chile in particular where the boundary between alpine and subalpine zones is difficult to ascertain. Expert judgement therefore plays a key role in supporting the use of empirical, GIS and other data in determining exactly what constitutes *alpine* vegetation, and therefore, what was considered a MGS for the purpose of this study (Billings and Mooney, 1968).

Regardless of geographical location, terrestrial arctic and alpine plants can usually be categorised as either Angiosperms (flowering plants, mostly perennial), lichens, bryophytes or ferns. More simply put, one can say that MGS vegetation generally consists of low perennial herbs and dwarf shrubs, noting that composition, structure, floristic complexity and functional traits vary widely depending on whether a mountain resides in a tropical, sub-tropical or temperate climates (Billings and Mooney, 1968; Cornelissen *et al*, 2003; de Bello *et al*, 2010; Grime, 2001; Pickering and Venn, 2013).

A common and unique characteristic of alpine plants is their ability to not only survive but also metabolise, grow and reproduce at very low temperatures. Many herbaceous perennials are adapted to make the most of the short growing season, shooting quickly during the spring thaw (Hodgson, 1966; Körner *et al*, 2005; Ssrensen, 1941). These abilities are critical as alpine plants suffer frigid and powerful winds, heavy precipitation and runoff due to slope, temperature fluctuations, extreme solar radiation and low partial pressures (oxygen and CO_2) on an almost daily basis, all while clinging to steep slopes where they are rooted in shallow mountain soils.

The height of alpine plants is determined by a number of limiting factors. First, intense winter winds and the spatial distribution and depth snow, which serves to force trees to grow into woody, strong but small *Krummholz* shrubbery. Second, and at the other extreme, exposure to harsh and often dry summer conditions which limits cambial growth and seed production through moisture deficiency and exposure to punishing levels of solar radiation (Daubenmire, 1954; Billings and Mooney, 1968; Hustich, 1948). It can be assumed that the more severe the mountain environment, the greater likelihood shrubs will be prostrate (lie just above the ground) or be confined to snowbeds that serve as insulation. Moreover, the most extreme alpine microclimates lie at high elevations within late-lying snow banks and on windswept ridges. Here, growth is limited to mostly cryptogams (lichens and mosses) which are more adapted to strong wind and low night time temperatures than vascular plants – with cushion plants and small herbs making the occasional appearance in some snowbeds where they can gain some protection from the elements (Billings and Mooney, 1968).

Though MGS species can be considered hardy and resilient, there is a clear correlation between the decline in taxonomic diversity and altitude (Körner, 2004). As discussed above, climate is one of these factors. Other factors include: adaptation limits (scarcity of adaptive or pre-adaptive taxa); limited physical area (mountains often exhibit more rock, ice and unstable/steep surfaces at altitude); limited functional space (geological time and growing period); seasonal time constraints (e.g. reduction in growing season due to draught); and, geological time constraints (e.g. episodic evolutionary periods interrupted by glaciation) (Hawkins, 2004; Körner, 2000; Körner, 2003). Due to these characteristics and limitations, MGS are considered to be exceptionally fragile and very slow to recover from disturbances such as trampling and wildfire (Beniston, 2003; Körner *et al*, 2005). In many cases, once damage occurs, it may be irreversible. Hamilton (2002) concluded that the majority of these ecosystems currently exist in their pristine state and provide a number of important functions and ecosystem services to humanity. However, Körner (2004) also noted that while pressure on MGS was subsiding in industrialised countries it was increasing (and sometimes rapidly) in developing countries. This thesis investigates the latest trends in MGS ecological health in *Chapter 6*.

2.8 Mountain grassland and shrubland soils

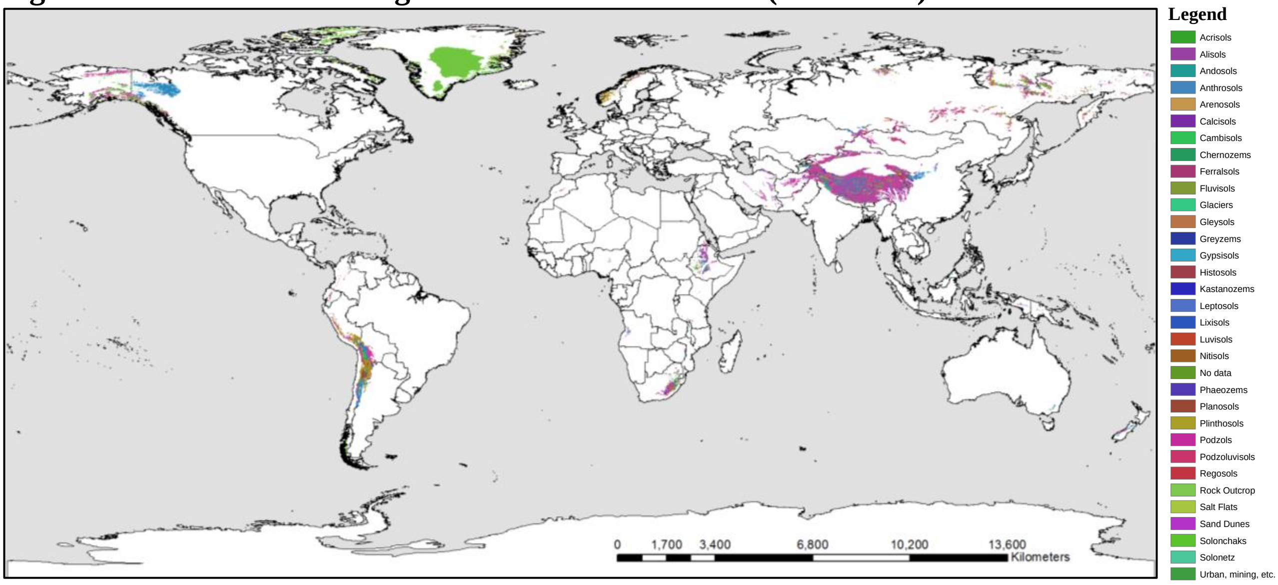
Commonalities also exist between soils formed in artic and alpine ecosystems. In both cases soils are slow to form, are influenced by a short growing seasons and are sensitive to climatic change (Hagedorn *et al*, 2010a). However important differences exist. First, soils in MGS ecosystems have in many locations been intensively used and modified over the course of history, especially in Central Europe. Second, soils in high-elevation temperate alpine climates are better insulated from frost, and commonly receive higher rainfall and warmer winter temperatures compared to their artic counterparts. Third, soils on mountain slopes are usually thin but drain well, therefore moist soils and peatlands generally only occur in wet coastal and/or geologically old mountain ranges such as the Snowy Mountains of Australia. Also, due to gravity and the extreme climatic conditions, mountain soils (particularly in the alpine) are more susceptible to disturbance caused by erosion, avalanches, landslides, wind, runoff and unsustainable land use practices (Hagedorn *et al*, 2010a; FAO 2015). Lastly, as for vegetation, mountains exhibit the highest number of soil types per area (‘Vertical Soil Zonality’) due to climate, slope, aspect and physical area available for soil deposition. The formation and distribution of mountain soils is also influenced by the available parent materials, prevailing winds, geological timeframe, chemical, and retreating glaciers. For many years it has been understood that mountain soils thus present a high degree of spatial variability, which in-turn relates to the soil’s properties (e.g. texture and structure), available organisms, composition (i.e. percentage sand versus percentage silt versus percentage clay), vegetation cover and ecosystem function (Bonefacio, 2013; Dokuchaev, 1899; FAO 2015).

The FAO (2003) estimates only 22 percent of mountain areas to be suitable for agriculture. Given the number of people living in mountain areas, substantial stress is placed on mountain soils in the form of overgrazing and agricultural intensification that, combined with extreme climatic conditions and geomorphic processes (that rarely exist in lowland areas) commonly causes irreversible soil loss, ecosystem degradation and adverse socio-economic impacts e.g. loss of agricultural land and slope instability, which causes a risk to humans (Hurni, 1999). So damaging is this soil loss that it has been deemed by some SMD practitioners as *the* most threatening process to MGS agriculture and the communities which rely on it (FAO, 2015; Hurni, 1999). The loss of dense native vegetation cover is a key factor driving soil erosion and thus CO_2 emissions from MGS ecosystems, the impact of LULUC which is modelled and discussed in *Chapter 5* of this thesis.

According to the FAO (2015) mountain soils "ensure food security and nutrition to 900 mountain people around the world and benefit billions more living downstream" (p.5). The importance of soils has been recognised in four of the recently adopted Sustainable Development Goals (SDGs): food security; drinking water; climate mitigation and adaptation; and, for their significance to terrestrial ecosystems and in halting biodiversity loss and environmental degradation (United Nations, 2016). The ecosystem services provided by MGS soils is further discussed below.

A map showing the spatial distribution of soil reference groups can be found in Figure 10. The World Reference Base for Soil Resources is the recognised international system for classifying soil into 32 taxonomic groups (e.g. Histosols, Cryosols and Fluvisols) according to 10 different identification classes e.g. 'soils with thick organic layers' (Histosols) and 'soils influenced by water' (Fluvisols) (IUSS Working Group, 2006). This grouping is used by the Harmonised World Soil Database (HWSD) which underpins the results produced in *Chapter 4* and in Ward *et al* (2014). Understanding soils is a critical component in this study, as Ward *et al* (2014) estimated that on average 97 percent of ecosystem carbon in MGS is SOC.

Figure 10. Soils in mountain grasslands and shrublands (Year 2000)



Data source: HWSD, 2011; UNEP-WCMC, 2002.

2.9 Ecosystem functions and services of mountain grasslands and shrublands

Extending from discipline of conservation biology, the concept of ecosystem function focuses on the geochemical, biological, physical processes that occur within an ecosystem (Sekercioglu, 2010). The specific functions of MGS ecosystems include, for example: securing soil and the stabilisation of headwaters of major river systems (such as the Mekong River Basin which flows from the Tibetan Plateau), resulting in good water quality flowing downstream into the rivers and streams; fixing carbon dioxide and nitrogen from the atmosphere into biomass and soils; protecting mountain soils (including permafrost) from erosion; providing habitat; producing oxygen; and, harvesting water from the atmosphere (Körner *et al*, 2005).

Ecological function is of the upmost importance to the provision of ecosystem services, which are defined by Daily (1997, p.2) as “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life”. Kremen (2005, page 3) described ecosystem services as simply “the set of ecosystem functions that are useful to humans”. An ecosystem services can be categorised as one of three types: i) *Provisioning Services*, for example the extraction of resources to benefit local and downstream communities e.g. water for irrigation and drinking, metals and minerals, medicinal plants and timber; ii) *Regulating Services and Supporting Services* e.g. habitat for biodiversity, climate change mitigation, soil fertility and erosion control (agricultural productivity) and hazard reduction; and, iii) *Spiritual and Cultural Services*, such as recreation (hiking, fishing, skiing etc.), worship, 'having a sense of place' and indigenous knowledge, biodiversity viewing and aesthetic appreciation (Körner *et al*, 2005; ICIMOD, 2012; Sekercioglu, 2010).

Ecosystem services provided by MGS vary depending on the altitudinal belt (Figure 11 and Figure 12) and include: climate regulation (storing carbon in soils and biomass); mitigating risks from natural hazards through slope stabilisation (e.g. landslides and avalanches); storing water and releasing it slowly, thus reducing peak stream flow and ensuring a year-round source of water for irrigation, drinking and energy generation; providing biomass for fuel and construction; facilitating food and fibre production; and, a place of spiritual meaning, culture and recreation. Maintaining function and diversity of MGS is paramount to the provision of ecosystem services and in guarding against system failure (Körner, 2004).

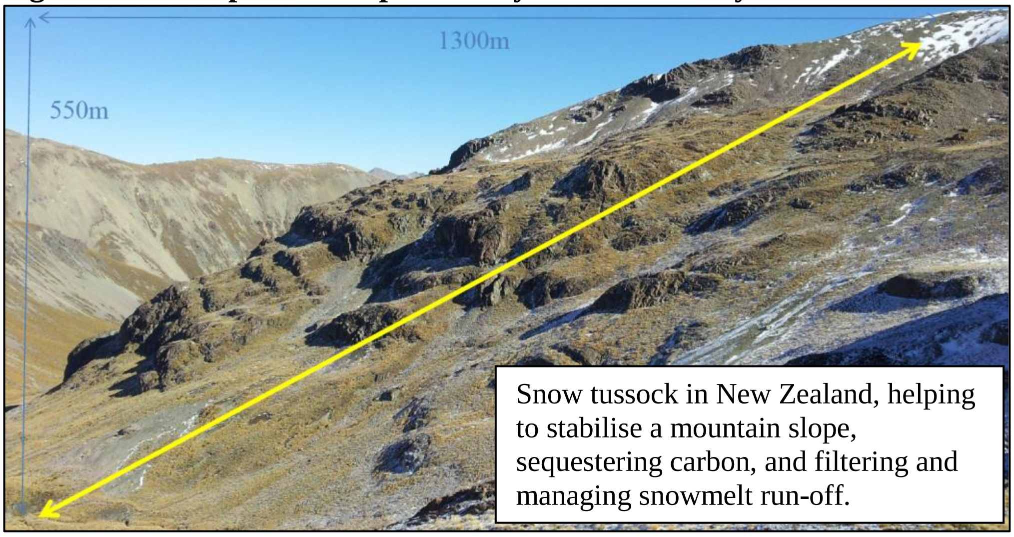
Figure 11: The importance of ecosystem services provided by mountains

	Ecosystem Service										
Altitudinal Belt	Downslope Safety		Water		Food		Fibre		Medicinal		Cultural
	Safety	Dams	Fresh water	Energy	Grazing	Crops	Fuel	Timber	Wild	Cultivar	Recreation
Alpine	●	●	●	●	◕	•	•	•	●	•	◕
Sub-alpine /Montane	●	▲	●	●	●	●	◕	◕	●	●	◕

Key: ● Very Important ◕ Important ▲ Relevant • Not Relevant

Adapted from: Rashid *et al*, 2005.

Figure 12. MGS provide slope stability and other ecosystem functions



Credit: Ward, 2012.

Many of the benefits provided by ecosystem services can be considered intangible, in that there is often no market for them. As such, putting a price on them is problematic (ICIMOD, 2012). Moreover, these ecosystem services are generally dynamic, co-dependent and multifunctional, varying over temporal and geographic space. For example, MGS mitigate climate change but at the same time stabilise slope, which has flow on benefits for human safety, crop cultivation and improved catchment water quality (Rasul *et al*, 2011; Ring *et al*, 2010). The value of MGS ecosystem services also varies depending on scale. For example, while at the macro level they are important for global, regional and nation economic growth, they are often critical in sustaining the livelihoods of local communities (ICIMOD, 2012).

2.10 Climate regulation and carbon storage in mountain grasslands and shrublands

Climate regulation, defined here as the "influence of ecosystems on near-surface climatic conditions, such as air temperature and moisture" (West *et al*, 2010, page 126), is recognised as an important ecosystem service by many studies. Ecosystems regulate the global climate system via both biochemical (CO_2 sequestration) and biophysical (balancing heat and moisture) processes (Foley *et al*, 2003; Körner *et al*, 2005; ICIMOD, 2012; Piekle *et al*, 2002). Though an ecosystem's biophysical contribution to climate regulation is important, this study focuses only on climate change mitigation as a result of biochemical carbon sequestration processes, which have been effectively converted into a commodity and traded on carbon markets around the world (Hungate and Hampton, 2012) - this is a core research question answered by this study.

Plant functional traits (discussed below) both drive and limit CO_2 sequestration in MGS ecosystems (De Deyn 2008). Local climate is especially influential. Like for lowland and marine ecosystems, alpine, subalpine and montane ecosystems remove CO_2 from the atmosphere and sequester it in alpine meadows and steppe, herbs, sages and other vegetation during a short but intense growing season, before slowly storing it in the soil (European Commission, 2008; ICIMOD, 2012; Kohler *et al*, 2012). Like for other ecosystems, MGS also release CO_2 back into the atmosphere via natural respiration, fire or other disturbance (Hungate and Hampton, 2012).

Numerous studies show that the *net* C stored and sequestered in above and below ground biomass of MGS ecosystems to be much less than for most forest types (Gullison *et al*, 2007; Rose and Sohngen, 2011; Ruesch *et al*, 2008; Ward *et al*, 2014). This is also reflected in internationally accepted methods and modalities for calculating C stocks and fluxes in various ecosystems (IPCC, 2003). Conversely though, a number of studies have led researchers to infer that SOC may be greater in treeless arctic and alpine ecosystems (Leifeld *et al*, 2009; Perruchoud *et al*, 2000; Sjogersten *et al*, 2003; Weiss *et al*, 2000). This may be due to a number of reasons, including that these soils are isolated and relatively inaccessible compared to lower elevation montane and (especially) lowland soils, where agricultural development is considered economically more attractive and where such soils have evolved and laid undisturbed for relatively long geological time periods (FAO, 2015). It may also be because alpine and arctic tundra ecosystems tend to allocate a higher proportion of plant C to the soil (Jackson *et al*, 1996). Moreover, the results of some studies also imply that in some artic (and therefore alpine) ecosystems, SOC may be higher than in forests ecosystems (Ping *et al*, 2008). Furthermore, De Deyn *et al* (2008) suggests that SOC pools are greatest in wet biomes, such as tropical tundra ecosystems or coastal temperate MGS ecosystems (Britton *et al*, 2011). As investigated in Ward *et al* (2014) and summarised in Table 2 (*Chapter 4*) there are few studies estimating carbon stocks at a local level for MGS ecosystem, let alone at the global scale which may be more useful for strategic decision making.

Considering this and the aforementioned fragility of MGS ecosystems, C stores in mountain soils tend to be relatively substantial but also highly vulnerable to global change (Hagedorn *et al*, 2010a; Ward *et al* 2015). Like for vegetation in MGS, once soil is disturbed it is unlikely that it can be restored to its original condition (Körner *et al*, 2005). Noting that many MGS ecosystems remain isolated and protected from anthropogenic land use and land use change (LULUC), climate change may be the biggest threat to mountain soils. Hagedorn *et al* (2010) suggests that global temperature rise may increase SOC decomposition, turning alpine soils into a source of CO_2 rather than a sink, particularly for permafrost soils.

2.11 MGS, climate regulation, and the ecological economic and market perspectives

Ecological economics seeks to address the relationships between ecosystems and economic systems, and focuses on six distinct themes that serve to distinguish it from conventional neoclassical economic approaches: sustainability; multiple values; intergenerational equity; uncertainty; methodological pluralism; and land ethics or 'utilitarianism' (Costanza, 1991). Given the complex and competing social, environmental and economic priorities associated with climate change, ecological economics can provide a useful interdisciplinary approach to considering not only cost-effective mitigation and adaptation strategies but also in ensuring that any climate action is equitable and compatible with other development goals (Scrieciu *et al*, 2011).

A number of recent studies recommend that traditional neoclassical economic approaches to climate change, such as single-discipline focused cost-benefit analysis, need to shift to a "new inter-disciplinary and multi-disciplinary risk analysis" (Barker 2008, p.174) that also considers the social values associated with the future climate system and not just purely economic ones (Jaegar *et al*, 2008). Specifically, in his 2008 critique of the Stern Review, Barker (2008) sets out four issues of critical importance to climate change that in his view traditional economic modelling fails to contemplate, namely that: i) the global economy and its relationships with energy and GHG emissions is complex and dynamic, with future climate effects and technological pathways exhibiting a high-degree of uncertainty; ii) intergenerational equity associated with climate change is a key ethical issue that should be informed by moral philosophy; iii) the study of economics and engineering history is central to identifying energy-related low-carbon possibilities; and, iv) the politics of climate change often results in government trade-offs which are currently not adequately factored into mainstream social welfare functions.

From an ecological economics perspective, the 'green economy' fundamentally differs from the 'grey economy' (and underlying conventional economics) in two ways. First, the green economy does not treat the environment as a subset of the economy. Rather, it puts it front-and-place, with the economy as a subset of the environment (Daly and Farley, 2010). Second, the green economy recognises that the externalities (e.g. land degradation) caused by economic growth need to be internalised in order to address market failure and promote equity and the other key principles of ecological economics (Costanza, 1991; Richardson, 2013). If one is to take an ecological economics view of MGS then these ecosystems should be considered alongside other ecosystems (marine, forests etc) as an ecological asset that is also valuable in the green economy, based not only on the relative benefits provided to society but also their intrinsic worth next to other lifeforms that exist within Earth's biosphere. Furthermore, the green economy offers the opportunity to recognise the benefits of ecosystems in a monetary form (e.g. carbon markets) that can provide financial incentives to improve the health of natural capital.

However, just how valuable are MGS ecosystems? And, just how warranted is the need to put a value on something that maybe considered 'priceless'? In answering the first question, economic valuation presents one method that can assist in answering this question. And in answering the second question, TEEB (2010) concluded that "natural resources are economic assets, whether or not they enter the marketplace" and that "conventional measures of national economic performance and wealth…. fail to reflect natural capital stocks of flows of ecosystem services contributing to the economic visibility of nature" (p.26). The report went on to recommend that "an urgent priority is to draw up consistent physical accounts for forest stocks and ecosystem services, both of which are required, e.g. for the development of new forest carbon mechanisms and incentives" (p.26). From this perspective, valuation makes sense if one is to build better environmental accounting systems that can be used to improve decision making i.e. 'you can't manage what you have not measured'.

In the case of valuing and assessing climate change policy options for carbon stored in MGS we can generally assert economics as a relevant discipline as it explains why reward-driven economic choices made by humans can lead to lower, or indeed higher, climate change impacts (Robbins, 1932). However, as noted above, taking a neoclassical versus ecological economic approach also provides contrasting determinations. For example, Nordhaus' (2007, p.177) neoclassical cost-benefit analysis approach to the climate change problem concluded that "the Gore and Stern proposals… are more costly than [doing] nothing". In his survey of 80 peer-reviewed estimates, Tol (2005, p.2073) made a similar finding in that "One can therefore safely say that, for all practical purposes, climate change impacts may be very uncertain but [it] is unlikely that the marginal damage costs of carbon dioxide emissions exceed [US$14/t$CO_2$] and [they] are likely to be substantially smaller than that". By contrast Stern's (2007) multi-dimension risk analysis concluded that the benefits of early and strong GHG mitigation action would outweigh the substantial costs imposed by climate change on the economy, environment and human life while considering intergenerational equity as a key factor. Stern (2007) subsequently estimated this ‘Social Cost of Carbon’ (SCC) to be US$126/t$CO_2$, therefore much higher than Tol’s estimate. As a more recent and popularised term, SCC can be defined as “estimate of the monetized damages associated with an incremental increase in carbon emissions in a given year” whose purpose is to “allow agencies to incorporate the social benefits of reducing carbon dioxide (CO_2) emissions into cost-benefit analyses of regulatory actions that impact cumulative global emissions” (US Government, 2015 p.2).

Estimations of SCC fall somewhere in between Stern and Tol’s estimate given above. The US government, for example, put its central estimate for SCC at US$36/t$CO_2$ in 2015 (2007 dollars), with estimates given for higher (US$11) and lower discount rates[2] (US$105) (US Government, 2015). Some ecological economists advocate that where outcomes cannot be adequately expressed in monetary terms that a zero or close-to-zero discount rate should be applied with the premise that intergenerational equity discards the time preference and that a resource or benefit is worth the same in the future as it is now (Cline, 1994; Howarth, 2003; Stern, 2007). This approach to the climate change problem popularised by Stern encapsulates the key elements of ecological economics through the use of multi-criteria analysis (for example) which attempts to evaluate non-monetised (or *non-tradable*) values such as human suffering damage, damages to nature and the risks and uncertainties of social decision-making. This thesis takes a similar perspective, and accounts for these issues when estimating the non-tradable economic value of carbon stored in MGS ecosystems, with the intention of providing policy makers with an acceptable proxy through applying SCC.

To date a number of ecological economics based studies have tackled similar problems as posed by this thesis. For example, Costanza e*t al* (1997) estimated the value of world’s ecosystem services and natural capital to be approximately US$46 trillion per annum in 2007 dollars (updated to US$125 trillion in Costanza *et al* 2014). Withstanding the morale and intrinsic issues arising from putting a price on what many consider priceless values (McCauley, 2006), Costanza aimed to quantify the benefits to humanity from a multitude of natural resources and processes that are provided by the world’s ecosystems which had not at that time been quantified in economic terms nor considered in contemporary financial decision making frameworks (Ackerman and Heinzerling, 2004). While the

[2] According to EDF, IPI and NRDC (2016) “the discount rate is how economists measure the value of money over time the tradeoff between what a dollar is worth today and what a dollar would be worth in the future” and “social cost of [2] According to EDF, IPI and NRDC (2016) “the discount rate is how economists measure the value of money over time the tradeoff between what a dollar is worth today and what a dollar would be worth in the future” and “social cost of carbon pollution estimate decreases as the discount rate increases because a higher discount rate implies that people care less about future generations than they do about the present”.

article generated criticism for the aforementioned valuation problems it did serve to heighten the understanding of important ecosystem services and the associated economic benefits to human welfare, commencing a discussion amongst scientists, economists and policy makers on how best to prioritise, protect and enhance natural capital. Another study by Pendleton *et al* (2012) sought to quantify the SCC for global Blue Carbon fluxes with the explicit intent of: providing the first evaluation of economic impacts associated with the clearing of mangroves, seagrass and marshes; and; highlighting how Blue Carbon could be incorporated into international carbon market protocols.

There are however only a handful of ecological economic orientated studies for mountain forests, and even fewer focused on MGS ecosystems. At the local scale, ecosystem valuation studies have been completed for a number of locations in the European Alps (Baumgart, 2005; Grêt-Regamey *et al*, 2007a; Getzner, 2000; Gliick and Kuen, 1977; Glos *et al*, 2006; Golo *et al*, 2005; Grêt-Regamey *et al*, 2007b; Hackl and Pruckner, 1997; Jaggin, 1999; Lowenstein, 1995; Tangerini and Soguel 2004). The majority of these studies have used contingent valuation methods to value just one ecosystem service (e.g. scenic beauty, avalanche protection, recreation), with only two attempting to value multiple ecosystem service benefits (including carbon sequestration). All of these studies focused on just one discrete geographical location e.g. Davos Switzerland. Grêt-Regamey *et al* (2008) point out that there is scope and potential benefits to policy makers in broadening valuation frameworks (beyond this narrow focus) to support planning processes, particularly when considering the most appropriate location for a new development. Grêt-Regamey and Kytzia (2007) go further and advocate the benefits that ecological economic valuation can contribute to regional planning and development. At the global level, no such studies exist for MGS ecosystems.

A literature gap thus exists. For C stored in MGS, a similar outcome is sought whereby policy-makers, researchers and potential carbon market participants are made aware of the value of C stored in MGS ecosystems. By estimating the relative value (*Chapter 5* of this thesis), one of the objectives of this thesis is to make "the values of nature visible and accountable for in economic decision making" (Akerman and Peltola, 2012 p.1). Another objective of this thesis is to contribute to carbon market protocols, so to enable climate finance to be applied to MGS NRM issues.

2.12 Natural resource management challenges for mountain grasslands and shrublands

Climate change, wild fires, agricultural intensification, tourism and urban development are substantial threats for alpine, subalpine and montane vegetation and the climate regulation services they provide (Beniston, 2001; Godde *et al*, 2000; Körner *et al*, 2005). A global report by a number of notable government and research organisations was presented to stakeholders at the recent United Nation's Rio+20 Summit in Brazil and emphasized the importance of dealing with these threats, stating that "Twenty years after Rio, the challenge of sustaining the provision of these goods and services has never been greater. The global community must act – a new agenda and strengthened institutional framework for mountain development is urgently required" (Kohler *et al,* 2012 p.5).

The establishment of protected areas is the most commonly favoured conservation strategy in mountain landscapes, with 17 percent of the world's mountains outside Antarctica being protected under such regimes (Rodríguez *et al,* 2011). Despite this, when compared against international targets for biodiversity conservation (e.g. the Global 200 Ecoregions, Olson and Dinerstein, 2002) existing mountain protected areas remain poorly connected and insufficient in size, with endemic biodiversity in many of these zones remains poorly represented (Rodríguez *et al,* 2011). The establishment of protected areas adversely impacts the financially poor local communities that inhabit high-mountain regions through restrictions placed on economic development opportunities (Adams *et al*, 2005; Huber *et al*, 2005; Daily *et al*, 2009; Htun *et al*, 2011). In addition, the costs of managing these protected areas for pests, fire and illegal activities are usually high and underfunded by government (James *et al*, 1999; Green *et al*, 2012; McCarthy *et al*, 2012).

Another important consideration is the trans-boundary nature of mountains. A mountain range may be protected by local authorities in one country, whilst in another the same mountain range may be open to unchecked development. The consequence is usually a decline in the ecoregion's health as a whole. This highlights the need for NRM policies that can apply across geo-political boundaries (Kohler *et al*, 2012). Despite the apparent logic and need for cross-boundary management of mountains, governments have to date struggled to achieve such outcomes, and have largely failed to adequately balance conservation and development needs in mountains worldwide. Intergovernmental and other SMD focused organisations and experts are increasingly advocating for novel and broad-based approaches to the NRM of mountains and MGS ecosystems (FAO, 2009; ICIMOD, 2012; Price, 2007). To this end, the introduction of climate policy-based economic incentives could provide a much needed NRM funding source through encouraging developers and investors to consider the real value of ecosystems services provided by mountains (Tobey, 1996) and enable local communities to be genuinely engaged in SMD opportunities (Draper, 2000).

2.13 Climate finance and carbon markets

The commoditisation of CO_2 (and effectively climate regulating activities) through national and international quantity-based emissions trading and carbon pricing schemes over the last decade has presented an economic incentive to address both biodiversity loss and climate change (Orlando *et al*, 2002; Thomas, 2011). In 2012, the global market for tradable carbon credits was valued at US$176 billion (World Bank, 2012), which included carbon compliance units and voluntary offsets created under schemes and standards such as the Clean Development Mechanism (CDM), European Emissions Trading Scheme (EU-ETS) and Verified Carbon Standard (VCS). The latest estimate, which also includes non-market direct climate financing (e.g. traditional loans and low-interest loans) puts this figure much higher. Buchner *et al* (2015) calculated total climate finance pools in 2014 to be US$391 billion, US$243 billion of which is estimated to come from the private sector and US$7 billion of which was directed to land use projects.

Significantly, governments around the world have also committed US$100 billion per annum by 2020 by way of the Green Climate Fund which will provide upfront grants to support climate change mitigation and adaptation projects in developing countries (Abbott and Gartner, 2011). Significantly, the signing of the UNFCCC Paris Agreement in late 2015 has buoyed hope for more investment in land-based carbon mitigation projects and has set the stage for long-term financing which supports the use of novel and rapidly growing economic incentives, such as green bonds, to meet the goal of US$16.5 trillion which is needed to keep global warming below 2 degrees Celsius (Climate Policy Initiative, 2016).

The utilisation of climate finance can yield valuable and sustainable co-benefits e.g. provision of infrastructure to the poor, lower particulate emissions and better health when using cook stoves (Freeman and Zerriffi, 2014; Lin *et al*, 2013). Carbon offsets are not only regarded as an important carbon management option for firms that are required to reduce their net reportable emissions under domestic regulatory and voluntary schemes, but when properly considered, a mechanism by which important sustainability benefits can be also realised (Dargusch and Thomas, 2012; Lin *et al*, 2013). While a number of institutional, governance and technical barriers remain, there is a growing opportunity to leverage carbon markets and finance to support conservation, improve ecosystem services and alleviate poverty through investment in clean technology and ecological restoration projects (Crossman *et al,* 2011; Lipper and Cavatassi, 2004; Peterson *et al,* 2012). Under the right conditions, carbon markets may also be 'stacked' with other Payment for Ecosystem Service (PES) credits, such as those for hydrological services, with the aim of arresting further biodiversity loss (Hein *et al*, 2013; Gold Standard Foundation, 2015).

From a natural ecosystem perspective, climate policy has focused on the use of economic incentives (e.g. grants, subsidies, market-based instruments) to encourage developers and government to reduce GHG emissions through a number of means. These include sequestering carbon in ecosystems (predominantly forests), avoiding the clearance of native vegetation and the disturbance of soils, and minimising the impact of wildfires and pest species on indigenous plants through focused management regimes (Sedjo and Sohngen, 2012; Venter *et al,* 2009). These incentives have arguably been successful in mobilising funds to support new reforestation, avoided deforestation, fire management, grassland rehabilitation and other biological carbon offset projects worldwide. This heightened interest in carbon offset mechanisms and carbon finance pools has led to a greater need for robust data to support carbon accounting, monitoring, reporting and verification principles to ensure that emissions are genuinely being reduced (Cowie *et al,* 2007; Gupta *et al,* 2012).

A growing body of evidence on lowland terrestrial, marine and coastal ecosystems is pointing to the prevalence of substantial carbon sinks which are under threat from anthropogenic and natural forces. This has important implications for international carbon budgets and the role these ecosystems should play in climate policy (Beer *et al*, 2010; Dixon *et al,* 1994; Fourquerean, 2012; McCleoud *et al,* 2011; Zhao and Running, 2000). There is increasing recognition of the importance of natural landscapes to climate change mitigation and adaptation. The concept of 'Blue Carbon', for example, has highlighted the opportunity to store C in marine and coastal ecosystem vegetation and sediment e.g. mangroves, seagrass and tidal marshes (De Blas 2009; Djukic *et al,* 2010; Mackey *et al*, 2008; Pendleton *et al,* 2012; Siikamäki *et al*, 2012; Smith, 2010).

Without a global assessment and understanding of carbon stocks in MGS, these ecosystems cannot be effectively integrated into international carbon budgets and climate policy. This shortcoming is important to recognise because the aforementioned carbon incentive schemes cannot be extended to MGS ecosystems without the support of robust and standardised carbon accounting data that reflects the biochemical and biophysical characteristics of MGS (Bumpus, 2011; Wilcock, 2013). It also represents a barrier to market proponents and researchers who without recent underlying scientific information cannot accurately measure and assess the opportunity of developing carbon finance related projects against other economic opportunities (Mcleoud *et al*, 2011). Critically, no estimates for economic (non-tradable) value for carbon in MGS had been published as yet. It is important to evaluate the economic value of these stocks globally, but it is acknowledge that any value derived at a global scale can vary dramatically for different stakeholders and pose morale challenges on valuing what can intrinsically be regarded as priceless (Sagoff, 2008).

2.14 Climate finance implementation issues for mountain grasslands and shrublands

The use of climate finance to support NRM and sustainable development in MGS ecosystems is an attractive proposition, however many challenges, barriers and risks exist. One of the key issues is whether (or not) climate finance can provide an economically feasible alternative to BAU development activities with various studies highlighting the priority that governments put on achieving financial objectives over environmental orientated goals (Strassburg *et al*, 2009; Osborne and Kiker, 2012). For example, a study on whether the sale of carbon offsets could provide a sufficiently economic alternative to large-scale logging in Guyana, Osbourne and Kiker (2012) concluded that a relatively low carbon price could provide competitive returns compared with logging, provided benefits gained from forest protection. However, other studies reveal that although carbon finance and offset markets remain a promising option to support conservation goals and enhance ecosystem services, a number of hurdles remain including the high transaction costs of getting carbon offset units to mark, how long-term permanence requirements may inhibit uptake, the risk-of-reversal associated with landholders clearing land in the future, lack of institutional and skills capacity, and, the risk of emissions happening in another location e.g. carbon leakage (Cacho *et al*, 2013; Dargusch *et al*, 2010; Galik and Jackson, 2009).

Thomas (2011) investigated carbon mitigation project constraints further and categorised these into four categories: (i) capacity; (ii) finance; (iii) governance; and, (iv) regulation. Firstly, with regards to capacity, individuals and organisations can be constrained by knowledge deficiency due to factors such as a lack of knowledge of climate policy, technologies and market opportunities (Dargusch *et al*, 2010), a shortage of data and/or information (Pelletier *et al*, 2010), and, inadequate technical and operation skills (Thomas and Dargusch, 2010). Capacity could also be adversely affected by limited access to networks (Suneetha *et al,* 2011) and the local availability of physical and technological infrastructure e.g. electrical grid connectivity for installing renewables and heavy machinery that may be required to prepare carbon offset project sites (Rochecouste and Dargusch, 2011; Sirohi, 2007).

Secondly, financial constraints such as limited access to capital, high transaction costs and delayed returns were all identified as major hurdles to carbon mitigation project development, particularly with regard to bio sequestration projects. Capital investment requirements are equal to the costs of new technology, materials, machinery and training, while transaction costs are usually up-front and relate to project origination (e.g. prefeasibility studies, technical expertise and data verification); depending on how stringent the rules of a particular carbon offset scheme are, both capital requirements and transaction costs can be very high (such as in the case of the CDM) and can thus inhibit the development of offset projects (Thomas, 2011). Delayed returns are a common constraint to the development of biological carbon offset projects as any carbon credits are generally only issued when carbon is sequestered, with associated revenue from the sale of these credits to follow once registration, verification and monitoring procedures have be completed. For example, an afforestation carbon offset project developer may need to wait between three and five years before any carbon credits are issued. Depending on how project finance is structured (debt versus equity, for example), the holding costs associated with interest rates and bank fees can dilute the required rate of return. For this reason, traditional grants (upfront payments) may be more attractive for developers than seeking carbon offset credits.

Thirdly, the legitimacy and effectiveness of governance structures also present key barriers to using climate finance. Ensuring that stakeholders from local and downstream communities and varying levels of governments have a positive perception of a carbon mitigation project is vital to its legitimacy as an accepted development opportunity i.e. without legitimacy projects can face wide ranging community opposition. At a higher level, legitimacy also governs the broader acceptance of climate policy which in-turn affects the collaboration amongst stakeholders and ultimately the successful implementation of the project (Rosendal, 2011). Legitimacy can be adversely impacted by institutional fragmentation and policy overlap therefore policy mechanisms must be continuously evolved on a national basis in-line with international rules to ensure climate mitigation strategy is effective on a global scale (Medrilzam *et al*, 2011). The effectiveness of carbon offset projects can be gauged on the extent of emissions reductions and ancillary benefits such as technology transfer, capacity building and more environmentally and socially sustainable industries; however the challenge of balancing self-serving commercial interests with broader sustainable development goals remains.

Lastly, regulatory barriers such land tenure and the application of additionality and permanence provisions under the rules of various offset schemes often pose a significant hurdle to the development of carbon offset projects (Thomas, 2011). Land tenure relates to the statutory or customary right of an individual or group to own land or exploit its natural resources according to a set of conditions and time limit. Much of the global MGS carbon stocks exist in developing countries where they are exploited by local communities for fuel, food, water and shelter under unclear tenure (Ward *et al*, 2014). While many governments may have the option of relocating these communities and protecting the natural resources to meet emissions avoidance goals, the failure to effectively engage with local communities and resolve tenure issues can ultimately lead to negative consequences for carbon offset project development e.g. illegal logging (Lasco, *et al* 2011; Medrilzam *et al*, 2011).

Another regulatory barrier is that concerning the concept of 'additionality'. The intent of the additionality clause administered under the CDM (for example) is to ensure that any emissions reductions from an offset project would not have otherwise occurred for regulatory reasons or in the absence of revenue derived from the sale of offset credits (UNFCCC, 2010). There are a number of reasons why 'additionality' can be prohibitive to carbon offset project development: (1) It may be a disincentive for countries to implement improved environmental regulation (Bode and Michaelowa, 2003); (2) The generic approach to determining additionality does not take into account the varying financial margins associated with the needs and characteristic of different project types (Rochecouste and Dargusch, 2011); and; (3) The individual assessment of additionality is highly subjective, leading to project applications that lack robustness and credibility and that get rejected by independent verifiers (Schneider, 2009). Of critical note for biological carbon offset projects is the notion of 'permanence' where fire, logging and other hazards are regarded by the CDM rules (and other schemes) as risks that could lead to the reversal of stored carbon. The CDM regards biological carbon offsets as either 'temporary' or 'long-term', but not 'permanent' (as is the case with renewable energy projects) meaning that these credits will eventually expire and be unable to be traded on the market. The exclusive application of permanence to ecosystem carbon further encumbers investment into biological carbon offset projects (IETA 2009; Thomas *et al*, 2010), particularly when one considers the uncertainty associated with the impact of climate change on MGS C stocks. This aspect of carbon offset regulation needs to be reviewed if the co-benefits associated with biological projects are to be realised (Boyd *et al* 2009; Dutschke and Anglesen, 2008).

Regardless of the aforementioned constraints and barriers, biological carbon offset offer considerable climate change mitigation and sustainable development opportunities, and under the right conditions present attractive financial returns and enhanced socio-economic and environmental outcomes (Trumper *et al*, 2009). As is the case for Green and Blue Carbon, overcoming the barriers to these benefits will require due regard to conceptual research, capacity building and pilot project development and implementation (Thomas, 2011).

In the case of conceptual research, the application of additionality and permanence as part of a future policy framework needs to be considered as to the specific ecological and economic characteristics of MGS C stocks. Likewise, building capacity will necessitate the strengthening of governance mechanisms to enhance credibility, and, establishing public-private partnerships and networks to improve technology transfer and access to markets and skills training programs. The development and implementation of a MGS carbon mitigation project could also support conceptual research and capacity building by providing a practical opportunity for global institutions and national governments to experimentally manage the various concerns identified above. This project will take all these aspects of climate policy into account when assessing MGS C as a viable carbon mitigation project that can potentially attract climate finance.

Chapter 3. *Research questions & overarching methodology*

3.1 Research objectives

This thesis seeks to achieve the following broad research objectives: The extent, significance and economic value of carbon stored in mountain grasslands and shrublands, and the importance of this carbon pool for climate policy; the stressors on this carbon pool and the potential opportunities for, barriers to, and pathway forward to using climate finance to address these stressors.

As for studies and reports looking at other ecosystems, understanding these aspects is critical if C in MGS ecosystems is going to be recognised for their importance to global climate change mitigation goals, and if climate finance is to be used to support NRM activities and more sustainable land use.

3.2 Research questions

This thesis will achieved these objectives by addressing the following three research questions:

Research Question 1. What is the spatial distribution and significance of carbon stored in the world's MGS? How is it accounted for in global carbon budgets and international carbon accounting frameworks?

Research Question 2. To what extent is carbon globally exchanged between MGS and the atmosphere? How is this impacted by land use change? What is the economic value of these exchanges and the *in-situ* stock as a whole when considering climate policy and broader sustainable perspectives?

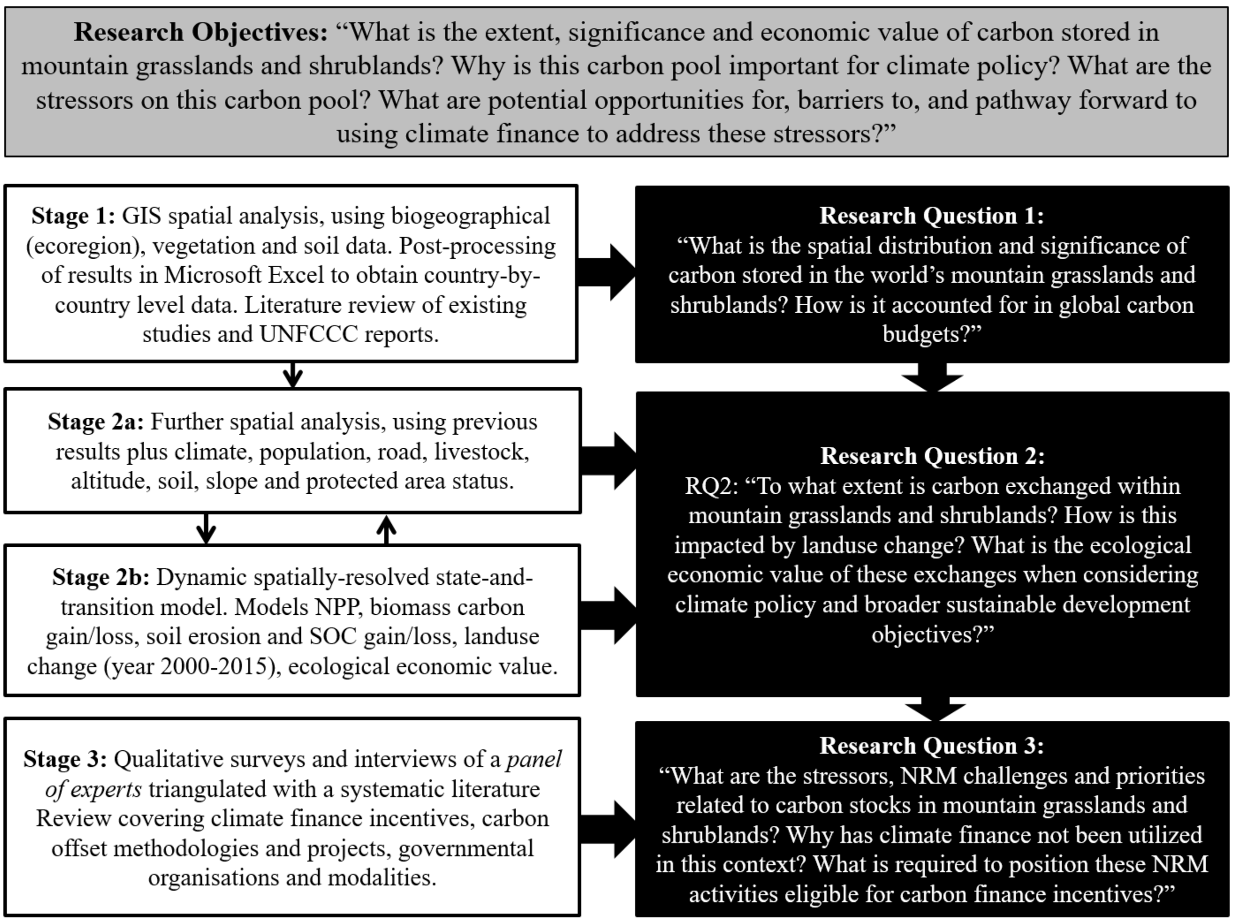
Research Question 3. What are the stressors, NRM challenges and priorities related to carbon stocks in MGS? Why has climate finance not been utilized in this context? What is required to position these NRM activities eligible for carbon finance incentives, and in so doing, ensuring that MGS are more sustainably used and the aforementioned economic value is maintained and/or improved?

3.3 Methodology overview

Addressing these research objectives involved three broad methodological stages, as highlighted in Figure 13 and as discussed below.

Note: A detailed methodology for each stage is presented in each of the relevant chapters.

Figure 13. Overview of Methodology & Research Questions



3.3.1 Stage 1

Stage 1 (*Chapter 4*, Ward *et al*, 2014) initially involved a systematic quantitative literature review (Pickering and Byrne, 2014) to understand the current state of knowledge and the availability of data for MGS, with respect to geographical extent and C pools contained within biomass and soils. The literature review included the following sources: empirical studies; government policy and technical reports; spatial datasets; and, non-government organization reports, and, spatial datasets. Following this, GIS and spatial analysis was used to identify the geographical extent of MGS worldwide, and a global estimate of *in-situ* carbon stocks. This approach was biogeographical in nature (Olson *et al*, 2001), and resulted in the identification of 875 different MGS ecoregions worldwide e.g. Central Range Subalpine Grasslands and Sayan Alpine Meadows. This initial map was overlaid with 16,000 individual Soil Mapping Units (representing 16 different soil types) derived from the Harmonised World Soil Database (HWSD) (FAO, IIASA, ISRIC, ISSCAS and JRC, 2012). For each individual SMU, SOC was calculated for topsoil (0–30 cm depth) and subsoil (30–100 cm depth) using the empirically-based spatial data provided by HWSD at that location. Biomass C stocks were estimated using the GLC2000 spatial dataset (EC JRC, 2003). The last step in this stage, was to determine how robustly MGS C stocks is incorporated into international carbon accounting standards. To achieve this, 19 UNFCCC National Inventory Submissions (NIS) were evaluated in accordance with seven criteria with the aim of making a judgement on: (i) whether mountain MGS C had been estimated; (ii) and if so, were estimates wholly delineated into a separate category or consolidated within a broader category (e.g. 'Grassland' or 'Other Land'); and (iii) whether a default IPCC or nationally derived emissions factor was used (Ward *et al*, 2014).

In summary, this stage answered RQ1 through producing a map and accompanying dataset of the geographical extent of the world's MGS, and the biological C (biomass and SOC) contained within.

3.3.2 Stage 2

Stage 2 (*Chapter 5*) built on Stage 1 and involved three sub-stages. The results of the literature review and spatial data outputs from Stage 1 provided the basis for GIS analysis, where additional datasets were overlayed and a number of calculations were run to determine a set of spatially-resolved input parameters. These inputs were used in a performance Individual Based Model (IBM) to estimate the monthly impact of LULUC on Net Primary Productivity (NPP) and soil loss, and the resulting influence MGS on net accumulations of C stocks (biomass and soil) and CO_2 exchange dynamics between the years 2000 and 2015. Two different simulation scenarios were run. The first simulation scenario represented current land use regimes (BAU). The second simulation scenario provided an indication of the additional CO_2 that could be sequestered if MGS were managed more sustainably. Finally, an economic assessment was undertaken by applying a range of economic values (at different discount rates) to the CO_2 sequestration results of two different simulation scenarios, providing a conservative estimation of the economic value of MGS climate regulation based on avoided damage to society (using Social Cost of Carbon as a proxy). This estimate of economic value was provided for the annual contribution that MGS make to climate regulation (net CO_2 sequestration) and the value of total *in-situ* C stock in MGS ecosystems as at 31 December 2015 (the end of the model run).

In summary, this stage answered RQ2 through modelling the influence of LULUC on MGS C stocks globally, and by estimating the economic contribution that MGS made in a climate regulation context.

3.3.3 Stage 3

Stage 3 (*Chapter 6*, Ward *et al*, 2015) was undertaken concurrently with Stage 2, as it did not require Stage 2's outputs to be completed. Stage 3 initially involved a survey of experts to obtain an insight into the priority NRM stressors for mountain grasslands and shrublands, and to ascertain how they judged their own understanding of carbon markets and climate finance. This qualitative survey method was indicative but not prescriptive. A survey was then used to identify the issues, priorities, and challenges facing natural resource managers and policy makers who are responsible for the NRM of mountain grasslands and shrublands, and to provide an understanding of the extent to which these experts have considered using carbon markets and climate finance to support NRM. The process for selecting an 'expert panel' of survey respondents was non-probabilistic and purposive, an approach deemed suitable for research where the objective is to understanding complex social phenomena (Marshall, 1996; Small, 2009) and where the sampling size is small but targeted (Gideon, 2012). Potential experts were initially identified through the Food and Agriculture Organisation (FAO) Mountain Partnership's network of practitioners, as successful sampling requires 'assembling a sample of persons with demonstrable expertise in given area' (Gideon, 2012, p. 400). Experts were selected for the survey if publically available information confirmed they met five criteria. The final panel consisted of 20 experts from a range of organizations. Follow-up interviews were used to gather more detailed data.

Finally, a literature review was used to verify the survey results and interview discussion. The literature review was systematic and quantitative (Pickering & Byrne, 2014) and included examining secondary data sources (publically available databases and registries, government and non-government reports, and peer-reviewed journal articles) covering (1) sustainable mountain development strategies; (2) climate policy documents, existing and pending carbon offset methodologies and projects; and (3) international climate change mitigation funds that were undersubscribed as of January 2015, and that could be used in supporting NRM activities in mountain grasslands and shrublands. The literature review was conducted using online databases including Summon, Science Direct, the Web of Science, EBSCO, and ProQuest. The primary keyword search terms consisted of a combination of ‘mountain’, ‘montane’, ‘alpine’, ‘tundra’, ‘subalpine’, ‘grasslands’, ‘shrublands’, ‘carbon’, ‘markets’, ‘funding’, ‘natural resource management’, ‘climate’, and ‘policy’. Secondary search terms included ‘climate change’, ‘biomass’, ‘soil’, ‘ecosystem services’, ‘environmental services’, and ‘ecological services’. All in all, this stage reviewed around 2000 peer-reviewed journal articles, policy, and technical documents.

In summary, this stage answered RQ3 through shedding light on how experts in sustainable mountain development understand the barriers, risks and opportunities associated with carbon markets and climate finance, and how they might be used to support specific NRM actions in MGS ecosystems. It also provides a systematic top-down conceptual policy framework for responding to these barriers and risks, with the aim of enabling climate finance to be used in an MGS context.

Chapter 4. *A global estimate of carbon stored in mountain grassland and shrublands, and the implications for climate policy*

Ward, A. Dargusch, P. Thomas, S. Lui, Y. Fulton, E. 2014. A global estimate of carbon stored in mountain grasslands and shrublands, and the implications for climate policy. *Global Environmental Change*, 28:14 – 24.

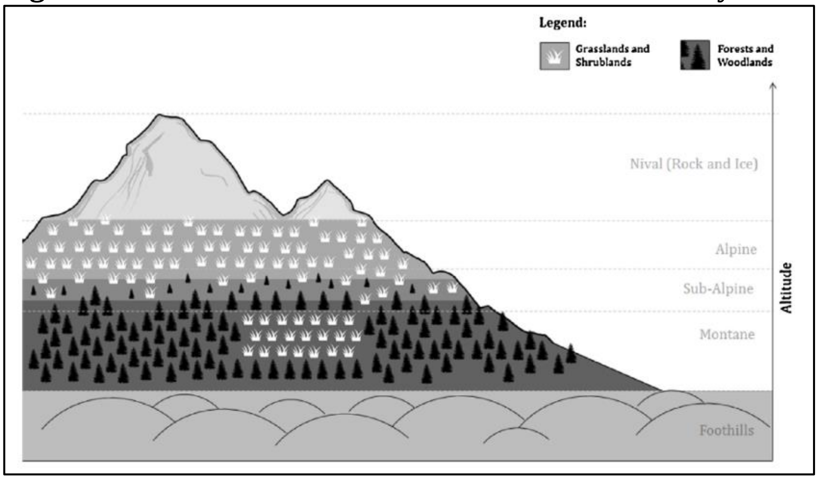
4.1 Chapter Summary

In this chapter, spatial analysis is used to estimate that there is between 60.5 Pg C and 82.8 Pg of C contained within the biomass and soils of the world's MGS. To put this in perspective, globally tropical Savannas and grasslands, temperate forests and tropical peatlands are estimated to contain 326–330 Pg C, 159–292 Pg C and 88.6 Pg C respectively. A subsequent review of existing empirical studies and of United Nations Framework Convention on Climate Change (UNFCCC) national greenhouse accounts is presented, and suggests that this C is not reliably accounted for in international carbon budgets. This estimate is the first to provide a global point of reference, useful in developing future research and in climate policy discussions. This chapter concludes by briefly discussing how climate finance might be leveraged to support the sustainable management of these C stocks, and in so doing uphold the other important socioeconomic benefits provided to humanity.

4.2 Introduction

Nearly one quarter of the Earth's landmass is covered by mountains which provide clean water to over 50% of the world's population, shelter almost half of the world's biodiversity 'hot spots', and afford important ecologically derived goods and services (Dixon *et al*, 1994; Kapos *et al*, 2000). Like mountain forests, healthy and well-functioning MGS (Figures 14 and 15) provide important benefits to humanity that are often unique due to the presence of steep slopes, extreme weather, and soil types.

Figure 14. Differentiation of different mountain ecosystems



Source: Ward *et al*, 2014.

Figure 15. Examples of mountain grasslands and shrublands

Alpine tussock and soils, Southern Alps, New Zealand (Credit: Adrian Ward)

Grasslands and meadows, Andes Mountains, Chile (Credit: Solange Godoy)

Alpine and subalpine grasslands and herbfields, Australian Alps, Australia (Credit: Adrian Ward)

Alpine plants, Nærøyfjord Mountains, Norway (Credit: Adrian Seligman)

Source: Ward *et al*, 2014.

These services include downslope safety and stable arable terrain, high-quality water for drinking and energy generation, pasture for grazing, recreational opportunities, medicinal plants and a buffer against the spread of bushfires (Hassan *et al*, 2005; Worboys and Good, 2011). Critically, these ecological goods and services are relied upon by some of the most impoverished people in the world who are often marginalized due to cultural factors and remoteness (Gerlitz *et al*, 2012). They are also incredibly important in regions that rely on the 'Alpine Economy' (Ariza *et al*, 2013). One discretely important ecological service performed by these ecosystems is climate regulation. Carbon dioxide (CO_2) is removed from the atmosphere and sequestered in the biomass of dwarf shrubs, heaths, alpine meadows, sages and other vegetation during a short but highly productive growing season, before that C is slowly stored in soil (Djukic *et al*, 2010). Though this process is relative slow due to tough growing conditions, over time significant amounts of C has accumulated in both the shallow (e.g. Leptosols) and deep soils (e.g. Histosols) of the world's mountain ranges. While similar to soils located in lowland and boreal regions, mountain soils are subject to a number of additional factors including: historical intensive use by humans; especially in Central Europe; higher rainfall, thicker snow cover (insulation) and thus warmer winter ground temperatures; the prevalence of steep well-drained slopes inhibiting widespread peatlands formation, and natural disturbances such as soil erosion, rock fall, spring snow thaw and avalanches (Hagedorn *et al*, 2010). The perception that mountains are remote, inaccessible and thus not under threat has contributed to government neglect and a lack of national and international conservation action (Ariza *et al*, 2013). In particular, the potential for more severe erosion make mountain soils more vulnerable to anthropogenic stressors (e.g. grazing) than their lowland and boreal equivalents. Moreover, studies have shown these soils, and the C contained within, are most effectively protected from erosion when there is overlying natural shrubland and grassland vegetation (Luz *et al*, 2002).

Natural (and semi-natural) mountain grassland and shrubland vegetation is under increasing threat, mainly in developing countries where rapid population growth is contributing to the intensification of agriculture (e.g. potatoes and maize), the expansion of grazing, growth in tourism and expanded high-altitude mining (Körner *et al*, 2005). While many of these stressors are declining in wealthier nations, others, for instance the infiltration of exotic plant and animal species and the expansion of tourism infrastructure, are having a substantial impact on natural vegetation within protected areas (Booth and Cullen, 2001; Körner *et al*, 2005; Worboys and Good, 2011). These invasions are not disconnected from the greatest of pressures, climate change, which is projected to have a disproportionate influence at higher latitudes and altitudes (Schroter *et al*, 2005; Nogues-Bravo *et al*, 2007). Many of the invading species are range extending, pushing up into alpine areas as lower levels become warmer and less hospitable due to global warming (Benitson, 1994). While one might immediately ponder the potential C gains from a change in vegetation type (i.e. trees), counterintuitively, there is a growing body of evidence to suggest that the expansion of trees into mountain grassland and shrublands may actually result in a net loss of C due to a lower capacity of forests to support soil organic carbon (SOC) compared to tundra (Hartley *et al*, 2012; Qian *et al*, 2010). The complex interaction of other factors, such as the suppression of indigenous burning regimes and introduction of grazing, can also lessen the extent of mountain grassland and shrubland vegetation, resulting in higher biomass loads, more frequent and intense fires, and the subsequent loss of additional SOC due to the erosion of exposed soils (Gross and Coppoletta, 2013).

Mountain vegetation plays a critical role in the control of soil erosion. When this vegetation is degraded SOC becomes particularly vulnerable to erosive forces (such as rain, runoff, wind and gravity), risking eventual decomposition and therefore a strong negative impact on the global carbon cycle (Lal, 2003). Without stabilization this process is likely to accelerate and lead to a positive feedback relationship whereby heightened erosion causes loss of more vegetation, which in turn causes more erosion, and so on (Juying *et al*, 2009). The environmental conditions prevalent at altitude discussed above mean that erosion rates in the mountains can be at least three times that of the lowlands (Ariza *et al*, 2013). This powerful force degrades biodiversity values, ecological goods and services (e.g. water quality, slope stability and provision of medicinal plants) and local upland communities. Furthermore, erosion has been linked with severe socio-economic and political disturbance in downstream low-land communities (Egziabher, 1991; Lal, 2003).

Damage to mountain vegetation and soils is often irreversible (Jansky *et al*, 2002) and there needs to be polices in place to encourage the conservation and restoration of the existing natural resource stock by ensuring mountain ecosystems remain healthy and more resilient to climate change and other stressors (Ariza *et al*, 2013). The establishment of protected areas is a widespread conservation strategy, but when implemented as a standalone policy it has been shown to be largely ineffective in managing these threats through creating economic disadvantage amongst local communities (Gaston *et al*, 2008). For mountains, trans- boundary and biophysical complexities provide further challenges for this approach. Moreover, the increasing shortage of available financial resources is repeatedly cited as one of the major constraints to arresting biodiversity loss (Waldron *et al*, 2013) and engaging in sustainable natural resource management (Worboys and Good, 2011). Therefore, the question here is ‘are there other non-traditional sources of funding that could support the sustainable management of mountain grassland and shrublands’?

It is well known that other ecosystems such as terrestrial forests, peatlands, lowland grasslands and shrublands, mangroves, and seagrass meadows store large amounts of C within biomass and soils (Beer *et al*, 2010; Dixon *et al*, 1994; Donato *et al*, 2011; Fourqurean *et al*, 2012; Scurlock and Hall, 1998; Siikamaki *et al*, 2012). Carbon markets and climate finance schemes (such as the Clean Development Mechanism, REDD+, and the emerging US$100 billion Green Climate Fund) are being investigated for their role in protecting and replenishing C stocks while achieving enhanced sustainability outcomes (Alongi, 2011; Gibbs *et al*, 2007; Pendleton *et al*, 2012; Smith, 2010; Ullman *et al*, 2013). We propose that the same action be taken for MGS. Efforts to investigate how international carbon markets and climate finance may support marine ecosystems, existing forests and degraded lands are underpinned by studies estimating C stocks at a global level (Donato *et al*, 2011; Fourqurean *et al*, 2012; Siikamaki *et al*, 2012). By contrast the distribution, extent, volume and density of mountain grassland and shrubland C pools remain largely unknown on a global scale. This presents a number of issues: one, it raises the question as to whether these C pools are adequately accounted for in the IPCC's global carbon budget; two, without an accurate baseline of these C pools it is difficult to track and assess the potential 'loss and damage' arising from CO2 emissions due to anthropogenic stressors; and three, without a global perspective we cannot evaluate where and how carbon markets and climate finance might be used most effectively to address the aforementioned stressors, improve ecosystem resilience and arrest emissions from MGS. In meeting the UNFCCC's ultimate objective of avoiding dangerous climate change, international climate policy will need to ''cover all relevant sources, sinks and reservoirs for greenhouse gases'' (UNFCCC, 1992). Britton et al. (2011, p. 287) recognize this ''urgent need'' to ''quantify the stocks of C'' held in alpine ecosystems.

Here we assemble the first global estimate of C stored in MGS. We present our review of existing literature, conduct a comprehensive Geographical Information System (GIS) analysis, and show that C stocks in these ecosystems (ecoregions) are neither accurately nor adequately accounted for in Annex I country UNFCCC national accounts. Finally, we briefly discuss ways in which carbon markets and climate finance might be leveraged to support conservation while considering sustainable development outcomes for some of the world's poorest people. Ultimately our aim is to bridge the nexus between climate policy and MGS by providing a sound reference point that will serve to underpin future scientific and economic studies, and encourage climate policy discourse – a precursor to leveraging carbon markets and climate finance.

4.3 Materials and methods

Mountain grasslands and shrublands

While prevalent in the treeless alpine altitudinal belt, MGS can also form communities of similar endemic plant species around the stunted trees of the subalpine transitional zone, and below within the forests and woodlands of the montane altitudinal belt, due to a number of localized natural phenomenon (e.g. frost hollows) and anthropogenic factors e.g. the influence of exotic herbivores (Körner *et al*, 2005; Benitson, 1994; Worboys and Good, 2011; WSDNR, 2011). This is illustrated in Figure 14 with photographic examples of vegetation and locations provided in Figure 15.

We undertook a systematic quantitative literature review (Pickering and Byrne, 2014) using online databases including Summon, Science Direct, the Web of Science, EBSCO, and ProQuest. The purpose of the review was to understand the current state of knowledge and the availability of data for MGS, primarily in relation to their geographical extent and C pools. In order of priority, the review sought to identify the following sources: (a) empirical studies; (b) government policy and technical reports, and, spatial datasets; and (c) non-government organization reports, and, spatial datasets. These databases covered the major literature sources across the biological, geographical, and social sciences, along with governance, policy and GHG reporting. The primary keyword search terms consisted of a combination of 'carbon', 'mountain', 'montane', 'alpine', 'tundra', 'subalpine', 'grasslands', 'shrublands'. Secondary search terms included 'climate change', 'biomass', 'soil', 'ecosystem services', 'environmental services' and 'ecological services'.

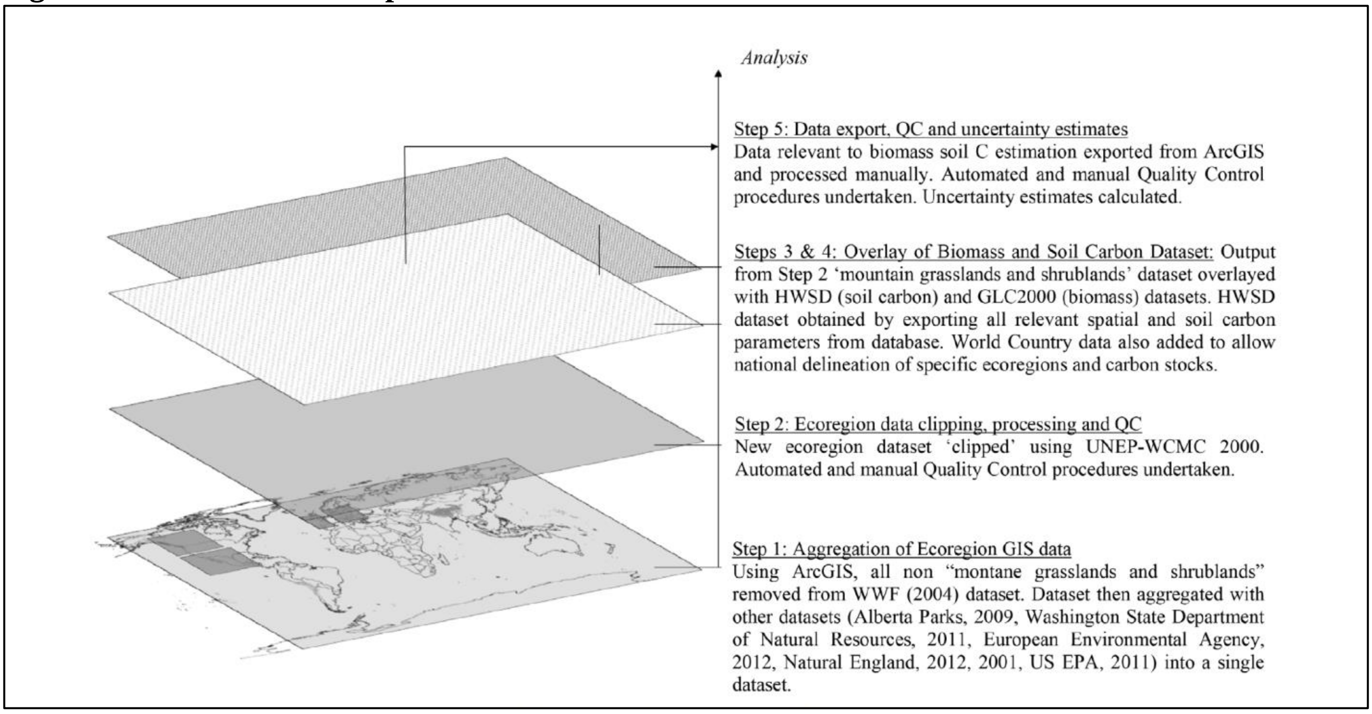
Estimating the spatial distribution of mountain grasslands and shrublands

We used GIS spatial analysis to identify natural and semi-natural MGS areas worldwide (Figure 16). Our analysis primarily used two readily available datasets (UNEP-WMCM, 2000; WWF, 2004) to determine the global extent of montane grassland and shrubland ecoregions. Ecoregions represent areas of land (150,000 km2 on average) that contain ''a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change'' (Olson *et al*, 2001, p. 933). Traditional biophysical models, such as Holdridge Life Zones (which is widely used in climate impact studies today), have been shown to poorly represent how natural vegetation responds to climate change in comparison to biogeographical models (Yates *et al*, 2000). This may be because such models only take into account average precipitation and biotemperature, and ignore seasonal variations.

Testing of the Holdridge Life Zones model has also shown it to misrepresent or even omit transitional mountain vegetation zones such as shrub-steppe (Yates *et al*, 2000). Ecoregions, by contrast, are ‘‘built on the foundations of classical biogeography. . .’’ and thus ‘‘. . .are likely to reflect the distribution of species and communities more accurately than do units based on global and regional models derived from gross biophysical features’’ (Olson *et al*, 2001, p. 933). Where spatial data was available, we sought to improve geographical coverage by adding smaller and more fragmented montane, subalpine and alpine grasslands and shrubland areas which are common in continental Europe, the United Kingdom and North America (Alberta Parks, 2009; Benitson, 1994; European Environmental Agency, 2012; IPCC, 2003; Natural England, 2012; US EPA, 2011; WSDNR, 2011). Ecoregions represent larger land size units than ecosystems, but both are constructed using the same biogeographical criteria. Ecoregions, also known as bioregions, can be thought of as a repetitive pattern of ecosystems that have common regional soil and landform characteristics (Brunckhorst, 2000).

We identified 875 different ecoregions including Northern Andean Páramo, East African Montane Moorlands, Central Range Subalpine Grasslands, European Calcareous Grasslands, Sayan Alpine Meadows and Tundra, and Central Tibetan Plateau Alpine Steppe, using GIS (Table 1). This dataset was ‘clipped’ using UNEP-WMCM’s (2000) dataset which defines the extent of Earth’s mountain zones based on slope and local elevation range (Kapos *et al*, 2000). We then applied a spatial filter to this dataset in order to remove ecoregions located in the subalpine and montane altitudinal belts predominantly populated by forests and woodlands, those denoted as ‘rock and ice’, and montane forest-grassland mosaics where the grassland/ shrubland to forest ratio was uncertain. We manually removed any spatial data relating to ‘‘arctic tundra’’ ecoregions in the arctic mountain areas in order to exclude C stocks that have already been estimated in other studies (Ping *et al*, 2008; Walker *et al*, 2005). Mountain ranges located in Turkey, Iran, Liechtenstein, the Antarctic and a number of small remote islands (e.g. Heard Island) were excluded due to the lack of reliable spatial data.

Figure 16. Overview of GIS procedures undertaken



Source: Ward *et al*, 2014.

Table 1. Summary of empirical studies on C storage in mountain grasslands and shrublands

Mountain Area / Country	C stocks density *range* per hectare (t C ha^{-1})	Reference
Global (no specific mountain range)	40-207	Schlesinger 1977
Sierra Nevada, USA	156-191	Norton *et al* 2011
Rocky Mountains, USA	60-70	Seastedt, 2011
Scottish Highlands, UK	115-498	Britton *et al* 2011
Alps, Austria	133-385	Djukic *et al* 2010
Alps, Switzerland	53-115	Leifeld *et al* 2009
Alps, Austria	14 - 140	Körner *C et al* 1993
Alps, Switzerland and Austria	40-220	Körner & Thompson 2003
Pyranees, Spain	59-299	Garcia-Pausus *et al* 2007
Hallingskarvet, Norway	207 (No Range)	Martinsen *et al* 2011
All, China	140-207	Ni 2002
Tibetan Plateau, China	31-290	Wang *et al* 1998
Tibetan Plateau, China	12-565	Wang *et al* 2002
Tibetan Plateau, China	10-137	Ohtsuka *et al 2008*
Southern Alps, New Zealand	16-56	Landcare Research 2012
Andes, Columbia	135-521	Pena *et al* 2011
Andes, Columbia	259-486	Hofstede *et al* 2003
This study	*15-530*	
Range	**10-565**	

Source: Ward *et al*, 2014.

Determination of soil and biomass C stocks (including uncertainty estimates)

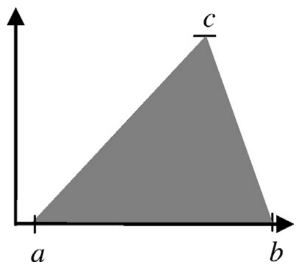
We overlaid our mountain grassland and shrubland ecoregion spatial dataset with the soil mapping data extracted from the Harmonized World Soil Database (HWSD). This resulted in over 16,000 individual Soil Mapping Units (SMUs). The SMUs measure bulk density and SOC to one metre depth at 1 _ 1 km (30 arcsec) resolution for 26 soil types as defined by the Food and Agriculture Organisation (FAO) (FAO, IIASA, ISRIC, ISSCAS and JRC, 2012). The HWSD was chosen over other datasets (such as outputs from CMIP5 based earth systems models) because: (a) it has been shown to produce more conservative estimates of global SOC stocks (Todd-Brown et al., 2013); and (b) it has been deemed suitable for making broad-scale SOC assessments as per IPCC Tier-I guidelines (Batjes, 2010). For each individual SMU we computed SOC separately for topsoil (0–30 cm depth) and subsoil (30–100 cm depth) layers using the following equation (Hiederer and Kochyl, 2011):

$$\mathrm{SOC_S} = \mathrm{SOC_C} \times \mathrm{BD}\left(1 - \frac{\mathrm{VS}}{100}\right) \times \mathrm{LD} \times 10^{-2} \ \ (\mathrm{t\ \ ha^{-1}})$$

where SOCS is the total amount of soil organic carbon to given depth (t ha^{-1}), SOCC is the soil organic carbon content for given depth (%), BD is the dry bulk density (g cm^{3}), VS is the volume of stones (%), and LD is the depth of soil layer (m). Note: as SOCC was provided in %, it was divided by 100 to give t C ha^{-1}. This data was aggregated at a national level. Normal probability distribution based uncertainty analysis has not been performed as the required data points (i.e. sample size, minimum and maximum estimates) were not readily available and would be a substantial challenge to obtain (Todd-Brown *et al*, 2013). Instead, we adopted a triangular fuzzy number framework to make uncertainty estimates for the 26 FAO soil types we included in this study. Fuzzy number frameworks have been used in conservation biology applications where the degree of measurement error is unknown (AkCakya *et al*, 2000). Triangular probability distributions were constructed by determining the 'minimum' (a), 'maximum' (b) and 'best estimate' (c) value for each FAO soil type. The 'minimum' and 'maximum' values were allocated based on the soil type C density values from the HWSD database. The 'best estimate' value was allocated based on the mean C density spatial output value from the GIS procedure. Where the FAO database contained no data for a particular location, we applied the mean C density emissions factor from the relevant empirical study (Table 1).

The triangular probability density function is given below:

$$f(x) = \begin{cases} 0, & x < a \\ \dfrac{2(x-a)}{(b-a)(c-a)}, & a \le x \le c \\ \dfrac{2(b-x)}{(b-a)(b-c)}, & c \le x \le b \\ 0, & x > b \end{cases}$$



We used these triangular distributions to develop uncertainty estimates which we applied nationally (by FAO soil type) to give an overall uncertainty for density (t C ha^{-1}) and a range for absolute SOC stocks.

To estimate biomass stocks we utilized the Global Landscape Cover (GLC) 2000 database (JRC, 2003) which contains IPCC-Tier I compatible data on aboveground and belowground biomass (t C ha^{-1}) for 124 different aboveground living biomass carbon zones (Ruesch and Gibbs, 2008). The GLC2000 is the product of a joint collaboration between the Joint Research Centre of the European Commission and more than 30 institutions from around the world. It is used as the core dataset for the Millennium Ecosystem Assessment and provides a bottom-up approach using data from Earth Observing Satellites and the Land Cover Classification System developed by the FAO. A recent accuracy assessment of GLC2000 found the product to have an overall accuracy of 68.6% (Mayaux *et al*, 2006). We adopt this figure here as the basis for our uncertainty estimates for biomass C and combined these values with those for the FAO database discussed above to give an overall uncertainty estimate for each country (ESRI, 2011) and also globally. A full list of GIS data used can be found in Table 5.

Quality Control

Quality control (QC) consisted of automated and manual procedures performed throughout the GIS analysis according to a QC checklist and broader Quality Assurance plan. Initially ArcGIS geoprocessing tools (e.g. 'Check Geometry', 'Repair Geometry' and 'Import and Clean Lines') were used to check and (where necessary) repair datasets. Record-by-record manual checks of all 'Attribute Tables' were then undertaken to ensure all non-relevant ecoregions (as discussed above) were removed from both the source datasets and the final dataset. Other manual QC checks included comparing feature counts between the source datasets and the final dataset, assessing outliers, confirming that duplicate records were removed, checking for poorly formed geometry, evaluating sliver polygons (very small polygons resulting from overlay analysis, as used by this study), and, ensuring carbon calculation formulas were correctly applied. All-in-all more than 221,000 records were checked manually.

Review of UNFCCC reports and estimates

It is not possible to directly compare our estimates to those in other studies, as none are available at a global scale. However, we can take a qualified approach to review national GHG accounts to understand the extent to which this C is captured. The UNFCCC reporting process requires Annex I countries to compile national greenhouse inventories on an annual basis (SBSTA, 2006). These reporting requirements include a section on Land Use Land Use Change and Forestry (LULUCF) and a subsection specific to Grasslands (IPCC, 2003). Annex I countries report using a standardized Common Reporting Format (CRF) and may utilize default emissions factors in the absence of country specific data. We methodically evaluated 19 National Inventory Submissions (NIS) for 2013 for relevant Annex I countries to determine the degree to which each had referenced data sources and estimates specific to national MGS. Our focus was on three areas: (i) whether mountain MGS had been estimated; (ii) and if so, were estimates wholly delineated into a separate category or consolidated within a broader category (e.g. 'Grassland' or 'Other Land'); and (iii) whether a default IPCC or nationally derived emissions factor was used. In order to judge the level of reporting we evaluated each NIS against the following seven criteria. These criteria reflect what we would deem to be ''reliable reporting'' for C stocks in MGS at a national level when considering the UNFCCC's ultimate objective (discussed in Section 4.2). In this context, we considered ''reliability'' to be of crucial importance (Golafshani, 2003).

I. Clear delineation from other land categories: whether MGS were explicitly delineated from other LULUCF land categories in the NIS report i.e. ''Grasslands'' and ''Other Lands'';
II. Estimate of geographical extent: whether a discrete estimate of the spatial extent (e.g. hectares) of MGS was provided in the NIS report;
III. Estimate of C stocks: whether a discrete estimate of C stocks for MGS was provided in the NIS report;
IV. Relevance of underlying data: whether the empirical studies or data used (e.g. GIS) was specific to MGS;
V. Underlying data: whether the empirical studies or data used was relatively recent and therefore likely to reflect the current situation (e.g. taking into factors such as land use change);

VI. Geographical coverage: whether spatial estimates were likely to be a complete geographical representation of MGS for that country; and

VII. Appropriateness of emissions factors: whether the emissions factors or data underlying the estimate of C stocks suitably accounted for the location and type of mountain grassland and shrubland ecosystem reported on. Estimates based on spatial data, such as provided by the HWSD, constituted the highest level of appropriateness in the absence of a geographically specific empirical dataset.

Using our newly developed global dataset we identified the following Annex I countries containing MGS, forming the basis of our review: Australia, Austria, Bulgaria, Canada, Finland, France, Italy, Kazakhstan, New Zealand, Norway, Poland, Romania, Slovakia, Slovenia, Spain, Sweden, Switzerland, United Kingdom of Great Britain and Northern Island, and, United States of America. We did not evaluate Non-Annex I countries as there was very little detail available from the UNFCCC (i.e. submitting NIS reports is not currently a requirement under the convention). Given the mandated use of the CRF, our assumption was that Annex I NIS reports would provide a strong indication of the overall level of reporting for MGS at a global level.

4.4 Results

Our literature review identified 18 studies on C storage in MGS (Table 2). Generally these studies were species-specific, small in scale, or focused on a narrow set of locations. However, what these studies show is that C stocks are substantial in many locations and therefore potentially of global significance compared to other ecosystems. For example, mountains adjacent to coastal areas exhibit very large amounts of C per hectare (up to 498 t C ha^{-1}) due to the moist oceanic-alpine environment (Britton *et al*, 2011). In the Andes Mountains in South America C stocks can be between 239 and 479 t C ha^{-1}, owing to the presence of high-density Páramo vegetation and carbon rich soils (Hofstede *et al*, 2003). In other high-mountain regions C stocks show a much higher variability, for example the Qinghai-Tibetan Plateau exhibits between 15 and 548 t C ha^{-1} (Wang *et al*, 2002) and the European Alps between 14 (Körner, 2003) and 385 t C ha^{-1} (Djukic *et al*, 2010) due to the localized effects of slope, temperature, rainfall, and wind (Körner, 2004).

Our global analysis shows that MGS are estimated to cover an area of 9.38 million km^2 of land. We estimate that these ecoregions contain between 60.5– 82.8 Pg C, 98.1% of which is contained within the top 1 m of soil, the remainder in the aboveground and belowground biomass. Table 2 provides a snapshot of absolute C stocks (and uncertainty estimates) for 64 of the world's mountain countries where MGS represent substantial ecoregions. These estimates are both more specific and of greater resolution compared to other widely used models e.g. Holdridge Life Zones. As discussed above, ecoregions are founded in biogeographical science and take into account a much wider set of parameters that can influence their spatial distribution. We also estimated the mean C density rate (Table 2) to be between 15 and 685 t C ha^{-1} (mean 86.5-14.8 t C ha^{-1}) which is presented as a global map in Figure 17. Taking into account outliers, the nominal range is between 15 and 538 t C ha^{-1} and falls within the range provided by the studies listed in Table 1 (10-548 t C ha^{-1}). This range is comparable to average C intensity rates in temperate and tropical forests on non-peatlands of between 150 and 321 t C ha^{-1} (Rose and Sohngen, 2011) and sea grass meadows at between 9 and 628 t C ha^{-1} (Fourqurean *et al*, 2012), but less than for mangroves at between 101 (Pendleton *et al*, 2012) and 1074 t C ha^{-1} (Donato *et al*, 2011). When compared to existing studies (Table 1) our estimates are likely to be at the lower end of the range, but until more empirical research is available we judged it appropriate to take a conservative approach.

Table 2. National estimates of carbon stored in mountain shrubland and grassland ecoregions

Country	Land area (ha)[a]	C stock range (Pg C)	Mean C density (t C ha ^{1}, CI95)	
Afghanistan	7,068,260	0.33–0.44	54.5 ± 7.1	54.5 ± 7.1
Angola	1,877,989	0.09–0.11	52.5 ± 6.8	LDC[c]
Argentina	36,974,014	0.7–0.97	22.6 ± 3.6	[c]
Australia	837,776	0.07–0.1	99.6 ± 16.8	[c]
Austria	970,779	0.11–0.39	261 ± 142.6	Djukic et al. (2010)[b]
Bhutan	682,494	0.03–0.05	58.7 ± 7.9	LDC[c]
Bolivia	24,213,163	0.7–0.97	34.4 ± 5.7	
Bulgaria	2,259,398	0.3–0.46	168 ± 37	Leifeld et al. (2009), Körner (2003) and Garcia-Pausus et al. (2007)[b]
Canada	119,363,612	5.48–7.59	54.6 ± 9.2	
Chile	13,240,959	0.57–0.83	52.9 ± 9.8	[c]
China	299,140,816	17.51–23.61	68.7 ± 10.2	Ni (2002), Wang et al. (2002), Wang et al. (2008)[b,c]
Columbia	1,532,922	0.31–0.43	239.7 ± 38.4	
Costa Rica	8000	0.003–0.004	392.5 ± 62.9	Hofstede et al. (2003)[b]
DR Congo	73,461	0.005–0.007	77.6 ± 14.3	[c]
Ecuador	1,578,426	0.22–0.33	171.6 ± 35.2	Hofstede et al. (2003)[b]
Eritrea	674,633	0.02–0.02	28.6 ± 3.7	LDC[c]
Ethiopia	17,859,327	0.65–0.9	43.4 ± 6.8	LDC
Finland	1,876,918	0.1–0.14	63.9 ± 9	[c]
France	1,719,485	0.23–0.35	168 ± 37	Leifeld et al. (2009), Körner (2003), Garcia-Pausus et al. (2007)[b,c]
Germany	258,186	0.03–0.05	168 ± 37	Leifeld et al. (2009), Körner (2003) and Garcia-Pausus et al. (2007)[b]
Greenland	17,423,385	0.94–1.43	68.1 ± 14	
India	10,244,045	0.48–0.64	55 ± 7.7	
Indonesia	977,019	0.07–0.1	86.4 ± 13.2	[c]
Italy	2,205,114	0.29–0.45	168 ± 37	Leifeld et al. (2009), Körner (2003) and Garcia-Pausus et al. (2007)[b]
Kazakhstan	7,906,666	0.44–0.58	64.6 ± 8.9	
Kenya	81,927	0.01–0.02	209 ± 53.2	
Kyrgyzstan	13,594,983	0.58–0.75	48.9 ± 6.6	[c]
Lesotho	3,817,199	0.19–0.24	56.4 ± 7.6	
Madagascar	96,331	0.01–0.01	69.8 ± 10.2	LDC
Malawi	689,800	0.04–0.05	66.2 ± 7.8	LDC
Malaysia	16,285	0.001–0.001	65.9 ± 9.9	[c]
Mongolia	13,242,180	0.93–1.22	81.2 ± 11.1	
Morocco	873,519	0.05–0.07	69 ± 9.2	[c]
Mozambique	118,805	0.004–0.005	37 ± 4.7	
Myanmar	666,758	0.03–0.04	54.8 ± 7.1	LDC
Nepal	2,946,719	0.14–0.19	55.3 ± 7.7	LDC
New Zealand	4,228,334	0.21–0.28	57.8 ± 8.8	Landcare Research (2012)[b]
Nigeria	278,170	0.01–0.02	58.4 ± 7.5	
Norway	21,803,199	0.87–1.3	49.8 ± 9.9	[c]
Pakistan	10,945,241	0.52–0.68	54.8 ± 7.3	
Papua New Guinea	468,389	0.03–0.05	88.3 ± 13.7	
Peru	18,681,557	1.07–1.5	68.9 ± 11.4	Hofstede et al. (2003)[b]
Poland	680,618	0.09–0.14	168 ± 37	Leifeld et al. (2009), Körner (2003) and Garcia-Pausus et al. (2007)[b]
Romania	3,302,098	0.43–0.68	168 ± 37	
Russian Federation	150,526,780	12.88–17.75	101.7 ± 16.2	
Rwanda	57,775	0.02–0.02	348.7 ± 71.5	LDC[c]
Slovakia	1,715,340	0.17–0.27	127 ± 27.9	[c]
Slovenia	738,274	0.1–0.15	168 ± 37	Leifeld et al. (2009), Körner (2003), Garcia-Pausus et al. (2007)[b,c]
South Africa	4,408,441	0.13–0.17	34.7 ± 4.1	
Spain	827,701	0.11–0.14	152.8 ± 17.3	Körner (2003)[b]
Sudan	54,327	0.001–0.001	15.3 ± 2	[c]
Swaziland	408,489	0.03–0.04	79.7 ± 11.3	
Sweden	5,288,620	1.15–1.55	255.3 ± 37.5	
Switzerland	940,000	0.07–0.1	89 ± 12.6	[b,c]
Tajikistan	9,075,393	0.27–0.35	34.1 ± 4.5	Hofstede et al. (2003)[b,c]
Tanzania	558,832	0.03–0.04	68.2 ± 11.5	LDC[c]
Turkmenistan	88,741	0.002–0.002	21.8 ± 3.1	[c]
Uganda	151,947	0.02–0.02	135.6 ± 27.1	LDC
United Kingdom	12,409	0.001–0.005	243.9 ± 134.2	Britton et al. (2011)[b]
United States	94,037,599	10.46–13.77	70.6 ± 9.6	Norton et al. (2011), Schlesinger (1977) and Seastedt (2001)[b]
Uzbekistan	129,718	0.01–0.01	54.7 ± 7.1	[c]
Venezuela	314,421	0.13–0.2	530.2 ± 112.9	Hofstede et al. (2003)[b,c]
Zambia	76,381	0.004–0.005	60.8 ± 6.9	
Zimbabwe	675,565	0.04–0.05	61.3 ± 7.8	
Total/range	937,585,714	60.52–82.84		

This table provides a range of national estimates for total C stored in mountain grassland and shrublands and for C density (t C ha ^{1}). The range and uncertainty estimates are based on HWSD and GLC2000 databases. References used to estimate C (where no GIS data was available) are listed here. Nations that have been assessed to have a biodiversity conservation funding shortfall are also identified, as are those who are Least Developed Countries; carbon markets and climate finance could be particularly attractive in supporting natural resource management and sustainable development in order to meet funding shortfalls/social objectives (for example).

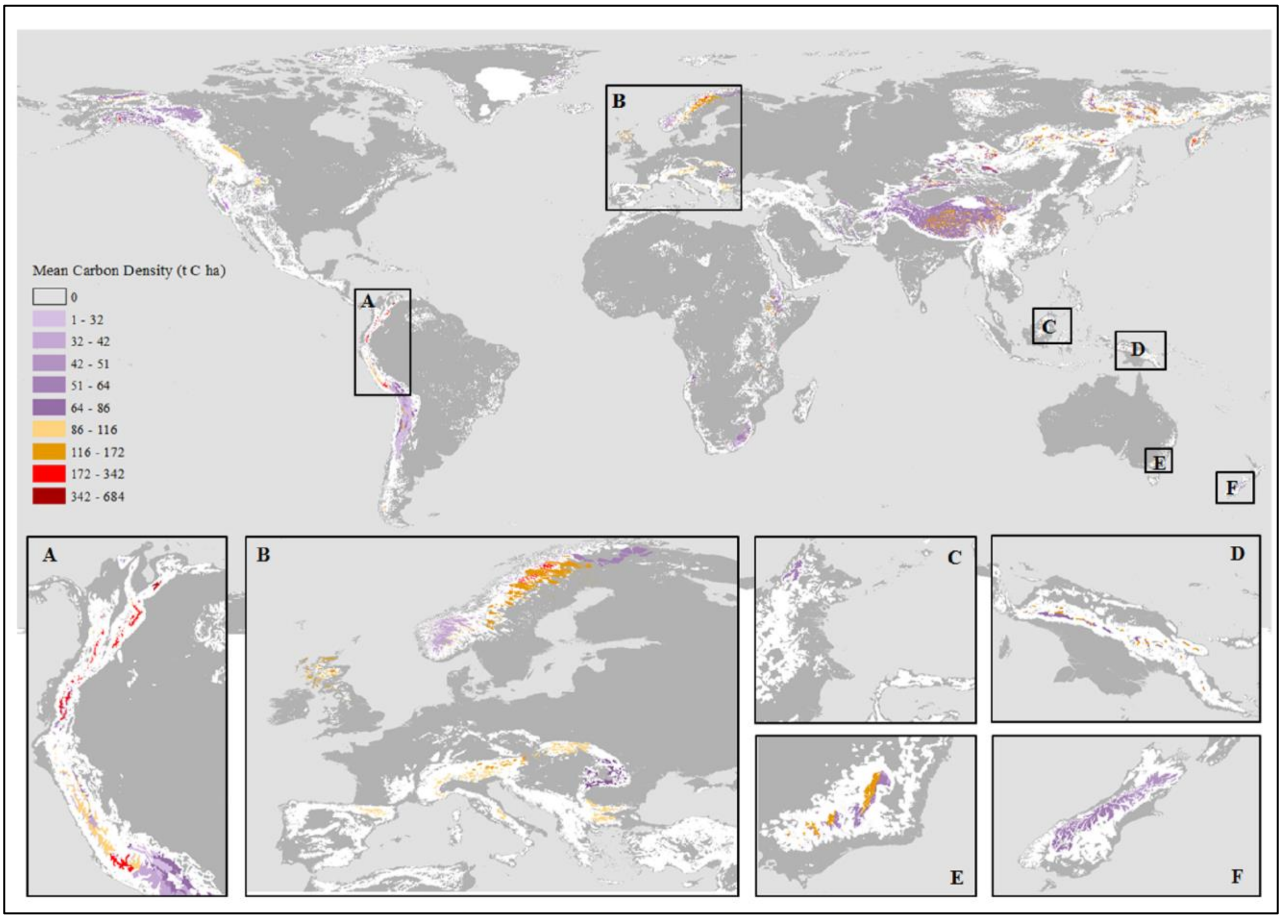
[a] Land area estimated using GIS datasets to 1 × 1 km resolution.

[b] (Ref): C estimated using empirical studies (Table 2), not FAO or GLC2000 databases.

[c] Underfunded for biodiversity conservation, LDC: Least Developed Country.

Source: Ward *et al*, 2014.

Figure 17. Global distribution and C storage of mountain grasslands and shrublands.



Source: Ward *et al*, 2014.

Our review of reporting for C in MGS by UNFCCC Annex I countries revealed a number of accounting issues. First and foremost, most Annex I countries did not report these C stocks in a robust and reliable manner. Five of the 19 (26 percent) 2013 NIS reports reviewed provided no C estimate at all. This exclusion was either explicitly stated in the NIS, or by virtue by the fact that there was simply no estimate provided for ‘Grassland’ and/ or ‘Other Land’ (IPCC categories in which these C stocks would fall). This was particularly the case with ‘Other Land’ where the category description extended to mountain ecosystems, and, which was generally not evaluated for C. We also discovered that seven countries (36 percent) did not discretely include MGS, but may have consolidated these C stocks within the ‘Other Land’ categories. However, those MGS falling within ‘Other Land’ were generally deemed ‘unmanaged’ and thus not reported (Cowie *et al*, 2007). In any case, if these C stocks had been included a default IPCC factor was used in the calculation. New Zealand and the United Kingdom provided more reliable estimates however in both instances either there was no delineation from the broader categories such as ‘Grassland’, or, that the emissions factors used were not nationally/ecologically specific. Austria and Switzerland provided the most reliable estimates as the area of MGS was well defined and emissions factors were based on locally focused empirical studies. A key limitation however was that SOC was only estimated for 0–50 cm and 0–30 cm respectively. Overall, we found that C stocks were either poorly calculated, or not calculated at all, thus generally unreliable considering the UNFCCC's objective discussed below. The ability that Annex I countries have to select which LULUCF sources to include or exclude in their NIS reports is likely to be a major reason as to why this data is incomplete. In general, the reporting of greenhouse gases emissions and sinks amongst Non-Annex I countries is typically less reliable compared to Annex I countries (Gibbs *et al*, 2007; Umemiya *et al*, 2010), thus we can assume that it is highly unlikely that these mountain grassland and shrublands are reliably accounted for in international carbon inventories due principally to the lack of GHG reporting capacity (Cowie *et al*, 2007).

4.5 Discussion

Mountain grasslands and shrublands and the carbon accounting gap

Current studies on C stocks in MGS are geographically limited so we developed a global estimate using GIS with the intent of filling this knowledge gap. In the process we discovered that this C store is not adequately accounted for in international carbon accounts. MGS C was only discretely accounted for by two (out of 19) Annex I countries and in both cases C estimates were made by applying a representative emissions factor (t C ha^{-1}) to an area size (ha) for a narrow range of mountain grassland and shrubland ecosystem types. The application of generic emissions factors is problematic, as the results are accompanied by a high level of uncertainty (Gibbs *et al*, 2007). This is because, unlike our GIS-based spatially explicit estimates, the use of generic factors does not take into account important variations within each category (e.g. soil type and land-use history). Consequently, our estimates are the first to provide a spatially based global inventory for C stored in MGS. Moreover they offer much higher resolution and a greater accuracy than currently available. Due to their foundations in biographical science our estimates offer a higher degree of accuracy than those used by the IPCC models e.g. Holdridge (Olson *et al*, 2001; Yates *et al*, 2000).

The implications for international climate policy

Understanding the degree to which these estimates are already captured in international carbon accounting is of high importance when considering issues such as double-counting and tracking progress towards international mitigation targets. The UNFCCC explicitly states that its objective ''is to stabilize greenhouse gas (GHG) concentrations in the atmosphere at a level that would prevent and reduce dangerous human-induced interference with the climate system'' and that ''the ability of the international community to achieve this objective is dependent on an accurate knowledge of GHG emissions trends, and on our collective ability to alter these trends'' (UNFCCC, 2013). Despite the inherent uncertainties involved with carbon in ecosystems, to build useful knowledge of emissions trends it is first necessary to establish reliable baselines against which they can be measured (Alongi, 2011). By their nature, MGS are located within extremely dynamic and heterogeneous environments influenced by steep slopes, high altitude, aspect effects, rainfall, soil type and land use history. Thus, in order to provide an accurate measure of the C within these ecosystems we must recognize the differing characteristics from their low-land equivalents. In addition, building political will and determining what types of policy interventions might be suitable to these mountain ecosystems will require high resolution for the specific situation and challenges at hand (Gibbs *et al*, 2007). Lin *et al* (2014) highlighted that (biophysically specific) GIS information is also particularly useful in climate policy formation, whereby tools such as 'multi-criteria decision making analysis' and 'spatial targeting' can aid in determining where climate finance might be most efficiently and effectively targeted e.g. areas of the highest C density and greatest co-benefits.

Our estimate of 60.5–82.8 Pg C stored in MGS is not insignificant. To put this in perspective, globally tropical Savannas and grasslands, temperate forests and tropical peatlands are estimated to contain 326–330 Pg C, 159– 292 Pg C and 88.6 Pg C respectively (IPCC, 2001; Page *et al*, 2011). We also note that this is more than three times the C estimated to be contained within Blue Carbon ecosystems (Donato *et al*, 2011; Fourqurean *et al*, 2012; Siikamaki *et al*, 2012). In recent decades marine and coastal ecosystems have also been impacted by anthropogenic stressors, with scientists and policy-makers creating institutions, expanding research and developing specific carbon baseline and abatement calculation methodologies in order to leverage financial opportunities created by climate finance and carbon markets with the intention of funding ecosystem restoration and protection (Ullman *et al*, 2013).

International climate finance pools are substantial, with a recent study valuing annual investment at US$343–385 billion (Buchner *et al*, 2012). Moreover, the UNFCCC is establishing its Green Climate Fund, which aims to provide US$100 billion per annum by 2020 to support climate change mitigation and adaptation projects in developing countries. It is also worth noting that there is around US$27 billion in carbon offset units traded annually on international carbon markets (World Bank, 2012). 'Blue Carbon' advocates are seeking to tap into these funds by developing carbon offset projects that sequester or avoid emissions in the LULUCF sector by either planting new mangroves, or, through a REDD+ type model that conserves existing C stocks e.g. not clearing mangroves for fish farms (Ullman *et al*, 2013). Similarly in the rangelands realm, Booker *et al* (2013) suggests that climate policy and rangeland management activities focus on the long-term retention of C stocks (and the broader socioecological benefits) in grassland ecosystems via conservation.

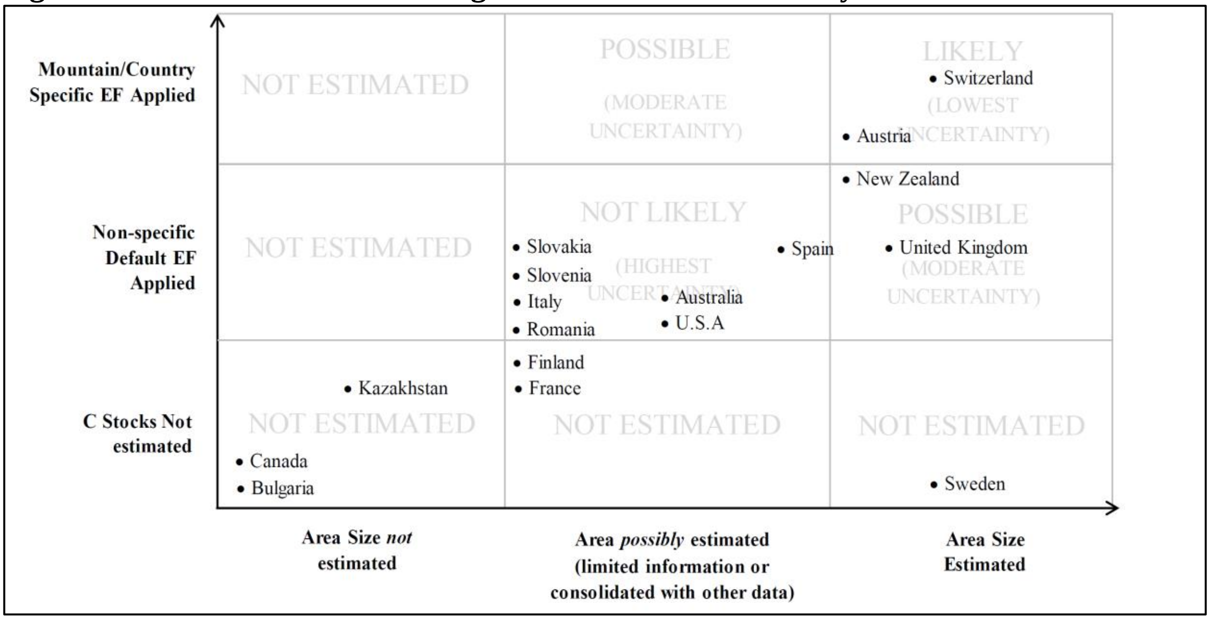
In this view, we propose that similar climate policy mechanisms be investigated for MGS. While we have initiated this process with our global estimates, operationalizing such mechanisms would be a lengthy process, requiring the expansion of empirical studies, capacity building, and the establishment of new institutions, governance structures and technical methodologies (amongst other requirements). There are also other considerations surrounding carbon and biodiversity which need to be addressed in order to meet enhanced socioecological outcomes (Thomas *et al*, 2013). The reasons to do so though are compelling. Firstly, climate finance pools are expected to increase substantially in the coming years. Natural resource managers are in need of additional financial resources to effectively manage mountain grassland and shrubland ecosystems against the aforementioned stressors; not taking advantage of these funding streams would be a missed opportunity. Second, of the 64 nations for which we estimated C stocks for, 25 have recently been assessed as underfunded for biodiversity conservation (Table 2, note ’c’) (Waldron *et al*, 2013). Moreover, 12 of the 64 nations listed are categorized as Least Developed Countries (LDCs) (Table 2). The role that climate finance and carbon markets might play in climate mitigation and adaptation, environmental stewardship and sustainable development has become increasingly important for LDCs in recent years. When carefully considered and implemented, carbon mitigation can act as an effective proxy wherein the conservation and restoration of these C stocks maintains other ecological goods and services e.g. biodiversity (Lawrence, 2012; Thomas *et al*, 2013).

4.6 Conclusion

MGS have distinct biophysical features (e.g. steep slopes) and associated ecological services compared to their lowland equivalents (e.g. slope stability); are relied upon by some of the world’s most vulnerable people; and have a strong influence on the quality (and cost) of water available to downstream communities for irrigation, drinking, energy generation and other industrial processes. There are many stressors on these ecosystems. The most immediate of these include agricultural intensification, the infiltration of exotic pest species, and, the rapid growth of tourism and associated infrastructure. In the medium to long-term, the greatest stressor is likely to come from anthropogenic climate change. Understanding, conserving and restoring these ecosystems is of critical importance. Healthy ecosystems are also more resilient to these stressors and more capable of arresting negative impacts e.g. erosion. By our estimates, there is a substantial amount of C stored in MGS that is not reliably accounted for in the international carbon budget. Our recommendation is that this study be used to strengthen UNFCCC national carbon inventory estimates for mountain shrublands and grasslands, and in so doing, improve the tracking of effective action on climate change at an international level. Lastly, we recommend that the results of this study also be used to assess the role of climate mitigation and adaptation funding in the sustainable management of these important resources.

4.7 Supplementary information

Figure 18. International accounting for C stored in MGS ecosystems



This figure shows the likelihood that C estimates for mountain grassland and shrublands are *included* in the relevant UNFCCC NIS report. It also highlights the *accuracy* of national C estimates. For example, countries shown in the top right most square are likely to include the most accurate estimates of C stocks based on the application of mountain/country specific emissions factor (EF) to delineated area (ha) estimates for mountain grassland and shrubland ecosystems. By contrast, countries in the middle square are not likely to present an accurate estimate because these ecosystems have been included within broader categories (e.g. ‘Grasslands’) and thus have been estimated using a default emissions factor that is non-specific to mountain ecosystems. **Available at:** http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/7383.php

Table 3. MGS ecoregions considered by this study

Country	Ecoregion
Afghanistan	Sulaiman Range Alpine Meadows, Ghorat-Hazarajat Alpine Meadows, Northwestern Himalayan Alpine Scrub (et
Angola	Angolan Montane Grasslands
Argentina	Southern Andean Steppe, Central Andean Puna, High Monte, Southern Andean Steppe
Australia	Australian Alps Montane Grasslands
Austria	European Alps Alpine Meadows
Bhutan	Eastern Himalayan Alpine Shrub & Meadows, Yarlung Tsangpo & Steppe
Bolivia	Central Andean Puna, Central Andean Wet & Dry Puna,
Bulgaria	Rila & Pirin Mountain Alpine Meadows
Canada	Pacific Coastal Mountain Ice fields & Tundra, Davis Highlands Tundra, Ogilvie-Mackenzie Alpine Tundra Alaska-St Elias Range Tundra, Rocky Mountain alpine meadows, Torngat Mountain Tundra
Chile	Southern Andean Steppe, Central Andean Dry Puna
China	Tian Shan Montane Steppe and Meadows, South East Tibet Shrublands & Meadows, Tibetan Plateau Alpine Shrublands & Meadows, Central Tibetan Plateau Alpine Steppe, Quilian Mountains Subalpine Meadows, Altai Alpine Meadow & Tundra, Eastern Himalayan Alpine Shrub & Meadows, Ordos Plateau Steppe, Yarlung Tsangpo Arid Steppe, Western Himalayan Alpine Shrub & Meadows, North Tibetan Plateau-Kunlun Mountain Alpine Desert, Pamir Alpine Desert & Tundra
Columbia	Northern Andean Paramos - Cordillera Central páramo, Santa Marta páramo, Cordillera de Merida páramo, Northern Andean Páramo.
DR Congo	Rwenzori-Virunga Montane Moorlands
Ecuador	Northern Andean Páramo, Cordillera Central Páramo
Eritrea	Ethiopian Montane Grasslands
Ethiopia	Ethiopian Montane Moorlands, Ethiopian Montane Grasslands & Shrublands
Finland	Low Alpine Grasslands
France	European Alps Alpine Meadows
Germany	Eastern Alps Subalpine & Alpine Meadows
Greenland	Kalaallit Nunaat High Arctic Tundra
India	Eastern Himalayan Alpine Shrub & Meadows, Northwestern Himalayan Alpine Shrub & Meadows, Karakoram-West Tibetan Plateau Alpine Steppe, Central Tibetan Plateau Alpine Steppe
Indonesia	Central Range Subalpine Grasslands
Italy	European Alps & Dolomites Alpine Meadows
Kazakhstan	Tian Shan Montane Steppe & Meadows, Altai Alpine Meadow & Tundra
Kenya	East African Montane Moorlands
Kyrgyzstan	Tian Shan Montane Steppe &Meadows, Pamir Alpine Desert & Tundra
Lesotho	Drakensberg Montane Grasslands, Highveld Grasslands
Madagascar	Madagascar ericoid thickets
Malawi	Southern Rift Montane Grasslands
Malaysia	Kinabalu Montane Alpine Meadows
Mongolia	Altai Alpine Meadows & Tundra, Khangai Mountain Alpine Meadow
Morocco	Mediterranean High Atlas Juniper Steppe
Mozambique	Eastern Zimbabwe Montane Grasslands, Southern Rift Montane Grasslands
Myanmar (Burma)	Eastern Himalayan Alpine Shrub & Meadows
Nepal	Eastern Himalayan Alpine Shrub & Meadows, Western Himalayan Alpine Shrub & Meadows
New Zealand	South Island Montane Grasslands
Nigeria	Jos Plateau Grasslands
Norway	Scandinavian Montane Grasslands
Pakistan	Sulaiman Range Alpine Meadows, Northwestern Himalayan Alpine Shrub & Meadows, Pamir Alpine Desert & Tundra, Karakoram-West Tibetan Plateau Alpine Steppe
Papua New Guinea	Central Range Subalpine Grasslands
Peru	Central Andean Puna (Wet & Dry), Cordillera Central Páramo
Poland	Tatra Mountain Alpine Grasslands
Romania	Făgăraş Montane Grasslands
Russian Federation	Cherskii-Kolyma Mountain Tundra, Trans-Baikal Bald Mountain Tundra, Kamchatka Mountain Tundra
Rwanda	Rwenzori-Virunga Montane Moorlands
Slovakia	Tatra Mountain Alpine and Subalpine Grasslands
Slovenia	Slovenia Alps Alpine Meadows
South Africa	Drakensberg Montane Grasslands, Highveld Grasslands
Spain	Pyrenees Alpine Meadows
Sudan	Ethiopian Montane Grasslands
Swaziland	Drakensberg Montane Grasslands
Sweden	Scandinavian Montane Grasslands
Switzerland	Montane, Subalpine and Alpine Grasslands
Tajikistan	Pamir Alpine Desert & Tundra, Hindu Kush Alpine Meadow, Karakoram-West Tibetan Plateau Alpine Steppe
Tanzania	East African Montane Moorlands, Southern Rift Montane Grasslands
Uganda	Rwenzori-Virunga Montane Moorlands, East African Montane Moorlands
United Kingdom	Upland Calcareous Grasslands
United States	Brookes British Range, Interior Yukon-Alaska Alpine Tundra, Alaska-St. Elias Range Tundra, Pacific Coastal Mountain Tundra, Sierra-Nevada Alpine and Subalpine meadows, Rocky Mountain Alpine and Subalpine meadows, Wasatch Montane Grasslands, Ogilvie-MacKenzie Alpine Tundra, Davis Highlands Tundra
Uzbekistan	Tian Shan Montane Steppe & Meadows, Pamir Alpine Desert & Tundra
Venezuela	Northern Andean Páramo, Cordillera de Merida Páramo
Zambia	Southern Rift Montane Grassland
Zimbabwe	Eastern Zimbabwe Montane Grassland

Table 4. Ecoregion source data

Name	Description	References	Date Published	Spatial Resolution/Scale	Geographical Coverage
	Datasets used to identify and delineate major ecoregions				
WWF_Ecoregions	Depicts the 825 terrestrial ecoregions of the globe, based on the work of Olsen *et al* (2001). Dataset distributed by WWF.	Olson *et al*, 2001, WWF, 2004	2004	1km / 30 arc seconds	Global
Mountain_and_Forests_in_Mountains_2000	Digital Elevation Model (DEM) developed by UNEP-WCMC that identifies global mountain area based on local elevation range and slope. Analysis by Kapos *et al* (2000).	Kapos *et al*, 2000, UNEP-WCMC, 2000	2000	1km / 30 arc seconds	Global
	Datasets used to identify smaller ecoregions (e.g. Level IV) and other conservation planning units				
ESAs2009	Environmentally Significant Areas (ESAs), Provincial Update 2009.	Alberta Parks, 2009	2009	1:1000,000	Canada
Ca_eco_l4	Level IV ecoregions for California, USA.	US EPA, 2011	2009	1:250,000	USA
Co_eco_l4	Level IV ecoregions for Colorado, USA.	US EPA, 2011	2009	1:250,000	USA
Mt_eco_l4	Level IV ecoregions for Montana, USA.	US EPA, 2011	2009	1:250,000	USA
Nv_eco_l4	Level IV ecoregions for Nevada, USA.	US EPA, 20111	2009	1:250,000	USA
Nm_eco_l4	Level IV ecoregions for New Mexico, USA.	US EPA, 2011	2009	1:250,000	USA
Or_eco_l4	Level IV ecoregions for Oregon, USA.	US EPA, 2011	2009	1:250,000	USA
Ut_eco_l4	Level IV ecoregions for Utah, USA.	US EPA, 2011	2009	1:250,000	USA
Wa_eco_l4	Level IV ecoregions for Washington State, USA.	US EPA, 2011	2009	1:250,000	USA
Wy_eco_l4	Level IV ecoregions for Wyoming, USA.	US EPA, 2011	2009	1:250,000	USA
Upland_Calcareous_Grassland_v2_1	Describes the geographic extent and location of upland calcareous grassland priority habitat in England.	Natural England, 2012	2012	10-100m	United Kingdom
Natura2000_End2011	Natura 2000 is an ecological network of designated conservation sites in Europe.	European Environment Agency, 2012.	2012	1:100,000	Continental Europe
NVC_SCOTLAND	British National Vegetation Classification (NVC) for Scotland.	Scottish Natural Heritage, 2010.	2010	5m	United Kingdom
	Datasets used to determine biomass and soil carbon stocks				
HWSD_Soil_Carbon	Harmonized World Soil Database (HWSD) Version 1.21	FAO, IIASA, ISRIC, ISSCAS & JRC, 2012	2012	1km / 30 arc seconds	Global
Carbon_zones	Global Land Cover 2000 (GLC2000) Version 1.1	JRC, 2003.	2000	1km / 30 arc seconds	Global
	Other datasets used in the analysis				
World_Countries	Represents the political boundaries of the world as at December 2012.	ESRI, 2011.	2012	N/A	Global

Photo: Adrian Ward
Location: Andes, Chile

Photos: Adrian Ward
Location: Southern Alps, NZ (summer and winter)

Chapter 5. *The global economic impacts of land use change on carbon stored in mountain grasslands and shrublands*

Ward, A. Yin, K. Dargusch, P. Fulton, EA. Abdul Aziz, A. 2016. The global economic impacts of landuse change on carbon stored in mountain grasslands and shrublands (Under review at *Ecological Economics*).

5.1 Chapter summary

As proposed in *Chapter 4,* C stores in MGS is not well understood and therefore not commonly factored into climate policy and natural resource management (NRM) decision making. One reason for this is the lack of general knowledge and data, including around the impact of LULUC, and, the exchange of carbon dioxide (CO_2) between MGS and the atmosphere. Another reason is that MGS are often remotely located and exhibit only minimal vegetation above the treeline, leading policy makers to mistakenly conclude that they store little C and thus have, from a climate regulation perspective, a relatively low economic impact for society. This chapter attempts to clarify, both in biochemical and economic terms, the role that MGS play in global climate regulation and therefore international climate policy. Using an Individual Based Model (IBM), the best available spatial input data, and while considering a broad range of assumptions, it is estimated that under current land use regimes MGS sequester on average 112 million tonnes of CO_2 ($MtCO_2$) per annum for the years 2000 to 2015. We estimate the equivalent economic value (based on avoided social cost of carbon) to be between US$1.24 billion and US$11.8 billion per annum, depending on the discount rate applied. If land use is managed more sustainably, MGS ecosystems could sequester up to an additional 8.4 Mt CO_2 per annum while contributing US$0.093 billion - US$0.89 billion annually in added economic value to society. These results vary substantially from ecoregion-to-ecoregion and from country-to-country. When considered for the total *in-situ* stock, it is estimated that MGS ecosystems contain at least 252 ± 39 gigatonnes CO_2 (68 ± 11 petagrams C) as at 31 December 2015. The equivalent ecological asset value of this C stock is assessed to be between US$2.5 (± 0.43) trillion and US$26.5 (± 4.1) trillion (2007 dollars). Building on the results in the previous chapter, these figures represent the most comprehensive and up-to-date global biochemical and economic estimate provided for MGS C stores and associated CO_2 fluxes.

5.2 Introduction

Mountain grasslands and shrublands (MGS) provide numerous ecosystem services including plant and animal biodiversity, the provision of clean water (for drinking, sanitation, irrigation and energy), food, culture and recreation, and as is the focus here, climate regulation through the continued storage of existing carbon (C) stocks and sequestration of carbon dioxide (CO_2) in high-altitude vegetation and soils (Körner, 2005; Ward *et al*, 2014). Ward *et al* (2014) estimated that between 60.5 Pg C and 82.8 Pg of C was stored across 64 mountain countries in the year 2000 (excluding Antarctica). This C pool plays an important role in international-level carbon budgets and climate regulation, and has a substantial economic value (Ward *et al*, 2015). Likewise, any net sequestration of CO_2 by MGS over a period of time will also have an economic benefit to society through mitigating climate change.

Ecological economics-based valuations have been made for forests, lowland grasslands and marine ecosystems (e.g. mangrove forests and seagrass meadows) with the aim of building more robust environmental accounts and drawing attention to the climate regulating importance of these areas. Proponents also advocate that such estimates enable more effective natural resource management (NRM) decision-making through the use of spatial targeting to determine where to best focus limited financial and technical resources, enabling climate finance to be used to fund more sustainable land use (Costanza *et al*, 2014; Braat and de Groot, 2012; Lin *et al*, 2013; Pendleton, *et al* 2012; TEEB, 2010). However, unlike for other C pools, an economic value for both C *in-situ* stocks (the C that is there now) and net CO_2 sequestration (the additional C that is stored over time) has not been estimated for MGS at any national level, let alone at the global scale.

In this regard, a lack of understanding of trends in land use and land use change (LULUC), CO_2 flux, and the associated economic values, has potential for adverse decision-making implications for the management of MGS (Costanza *et al*, 2014; TEEB, 2010). Advocates of ecosystem valuation argue that such studies help make sense of complex socioecological interactions, allowing for the incorporation of the value of natural capital into public decision making processes (TEEB, 2010). Perhaps more critically, unsustainable LULUC practices more often than not exert a negative impact on C pools, and in addition, the ongoing capacity of vegetation and soils to sequester CO_2. This is particularly the case for MGS, which are fragile and slow to recover from degradation (Benitson, 2003). Such degradation has the potential to undermine international climate change mitigation targets, such as the recent *Paris Agreement,* whereby the degradation of C stocks and CO_2 biosequestration capacity (be it forests, marine or MGS ecosystems) offsets greenhouse gas (GHG) reduction gains in other areas e.g. energy generation using low emissions technologies. To this end, avoiding emissions from unsustainable LULUC practices can be considered both logical and desirable. Avoiding emissions may also offer previously unrecognised carbon mitigation potential through mechanisms such as the Reduced Emissions from Deforestation and Degradation (REDD) and the Verified Carbon Standard which provide a financial incentive to manage MGS more sustainably (Ward *et al*, 2015).

Critical knowledge and data gaps impede the resolution of many mountain-related issues, including for LULUC in MGS and associated CO_2 dynamics (Jansky *et al*, 2002; Ward *et al*, 2015). There are only a handful of ecological economic orientated studies for mountain forests, and even fewer focused on MGS ecosystems. At the local scale, ecosystem valuation studies have been completed for a number of locations in the European Alps (Grêt-Regamey *et al*, 2007; Getzner, 2000; Gliick and Kuen, 1977; Glos *et al*, 2006; Goio *et al*, 2005; Hackl and Pruckner, 1997; Jaggin, 1999; Tangerini and Soguel 2004). The majority of these studies have used contingent valuation methods to value a single ecosystem service (e.g. scenic beauty, avalanche protection, recreation). Only two of the studies (Goio et al, 2005; Grêt-Regamey *et al*, 2007) attempted to value multiple ecosystem services, including carbon sequestration. All of these studies focused on just one discrete geographical location e.g. Davos Switzerland. Grêt-Regamey *et al* (2012) point out that there is scope and potential benefits to policy makers in broadening valuation frameworks (beyond this narrow focus) to support planning processes, particularly when considering the most appropriate location for a new development. Grêt-Regamey and Kytzia (2007) go further and advocate the benefits that economic valuation can contribute to regional planning and development. At the global level, no such studies exist for how LULUC might impact C stores in MGS ecosystems.

What is known, however, is that MGS are amongst the world's most vulnerable ecosystems, with climate change, overgrazing, tourism, wildfires and intensive cropping posing a growing threat to the C pools contained within these ecosystems (Körner *et al*, 2005; Ward *et al*, 2015). Putting aside the direct impacts of climate change, the expansion and intensification of cropping and grazing are considered by many experts to be the most significant anthropogenic stressors facing MGS ecosystems today (Körner *et al*, 2005; Ward *et al*, 2015). These land use types dominate the economic makeup of many mountain countries around the world, yet the extent to which these activities influence MGS ecosystems, the C stored within and the rates of CO_2 sequestered, has not been quantified or analysed at a global scale.

Here we present the results from the first global model to estimate the impact of LULUC on C stored and CO_2 exchanged by MGS ecosystems. This model considers MGS ecoregions in 48 mountain countries (98% of the MGS land area identified by Ward *et al*, 2014) for the years 2000 to 2015. We then go one step further and offer the first estimate of the economic value of this C, using Social Cost of Carbon (SCC) as a proxy for the avoided damage to society (as used in similar studies e.g. Pendleton *et al*, 2010). Though there are considerable uncertainties, due mainly to limited data availability, our objective is to provide the first comprehensive global estimate of CO_2 emissions from MGS and associated economic value. It is our hope that policy-makers will be able to use this estimate to make more informed and equitable decisions when considering the absolute and relative benefits of these important ecosystems from a climate policy perspective, and that it spurs further research in building more robust environmental-economic accounts.

5.3 Materials & methods

The method for this study consists of three stages. First, we used the spatial data outputs from Ward *et al* (2014) as the basis for Geographical Information Systems (GIS) analysis in *Esri ArcGIS*, where we overlayed additional datasets and ran a number of calculations to determine a set of spatially-resolved input parameters critical to stage 2 (Table 10). Second, using these input parameters, a performance model was developed in *AnyLogic* to estimate the monthly impact of LULUC on Net Primary Productivity (NPP) and soil erosion, and its influence on MGS C stocks and CO_2 exchange dynamics between 1 January 2000 and 31 December 2015. Finally, using *Microsoft Excel*, we undertook an economic assessment by applying a range of SCC values to absolute C stocks for 31 December 2015 and to annual net CO_2 exchanges over the simulation timeframe as per the outputs from our *AnyLogic* model. This assessment takes into account changes in biomass and soil C over 15 years under anthropogenic (current land use regime) versus non-anthropogenic scenarios (sustainable management), providing a conservative estimation of the value of MGS climate regulation based on potential for avoiding economic damage to society. Below we present the details of the GIS procedure, and the model, the latter being based broadly on Overview-Design-Details (ODD) protocols for Individual Based Models (IBMs) as established by Grimm *et al* (2006). Though our model is not exclusively an IBM in the sense of a cellular automata (as there is no direct interaction between individual ecoregions) it does share many of the same characteristics that such IBMs exhibit, where for example, the characteristics of each ecoregion are tracked through time (Reynolds, 1997). The model also shares the same purpose of many IBMs which is to provide an insight into how local actions translate into global consequences. Therefore, in the absence of a better framework, the ODD protocols were judged to be fit for this purpose. We then describe the economic assessment process.

5.4 Model purpose

The purpose of the model is to use the best available input data to gain a high-level global insight into how MGS C stocks might change over time under current LULUC practices and trends ('Anthropogenic Scenario') compared to a situation where natural MGS ecosystems might experience minimum disturbance ('Non-anthropogenic Scenario'). Understanding the difference in absolute C stocks between these two scenarios as at 31 December 2015 will infer important information about annual CO_2 exchanges in MGS globally, a measure that has not yet been estimated. The output data is then used to make an economic assessment of the value of this C using a range of SCC scenarios as metric for avoided economic damage to society.

5.5 Input parameters, variables and scales

The model consists of three hierarchical levels: ecoregion (individual), country (group) and global (simulation). Ecoregions are considered individuals and are initially characterised by a number of input parameters, the data of which was derived from the biogeographically-derived outputs of the study by Ward *et al* (2014).

Considering the limitations of available computer processing power, MGS surface area was divided into 20,798 land area vectors (of varying area) based primarily on ecoregion boundary. Important input parameters for each of these ecoregions include: proportion of land use for each ecoregion; mean soil bulk density; mean organic content; mean climatic factors important to determining NPP (rainfall, temperature and snow coverage); crop harvest frequency (CHF); and, factors critical to determining soil loss using the *Universal Soil Loss Equation* (USLE) e.g. mean rainfall erosivity, mean soil structure and mean land cover protection factor. The model input parameters are detailed in (Table 10), and utilise the data outputs from a recently published biogeographically-focused study (Ward *et al*, 2014) to define the extent of MGS ecosystems and associated carbon stocks (above and below-ground biomass and soil carbon to one metre depth) for the year 2000. The input parameters drive the variables of the model, through a series of monthly time steps as described below. These dynamic variables influence and change each ecoregion's input parameters, incorporating any feed-back that may be present in the system (Table 11). The model is of global scale, and utilises spatial input data from a variety of sources (Table 5). While the activities in one ecoregion may in theory influence the state of LULUC practices elsewhere (e.g. through competition, regulatory measures, sequential adoption of technology etc), it was felt there was too little information available to suitably parameterise such dynamic components. Consequently, interactions between ecoregions were omitted from this first estimate of MGS C stocks.

Table 5. Sources of spatial data

Input Data Type (Name)	Temporal and Spatial Resolution (data type)	Literature/Data Sources
World Countries	Year 2012 (Feature)	ESRI, 2011.
MGS Ecoregions Extent (WWF Ecoregions, Mountains of the world)	Year 2000, 1km (shapefile)	Olson et al, 2001; WWF, 2004; Kapos *et al*, 2000; UNEP-WCMC, 2000; Ward *et al*, 2014.
Soil Organic Carbon (Harmonised World Soil Database)	Year 2000, 1km (shapefile)	FAO, IIASA, ISRIC, ISSCAS & JRC, 2012.
Biomass Carbon (GLC2000)	Year 2000, 1km (shapefile)	JRC, 2003.
Major roads around the wold (Global Roads)	1980-2010 (Feature)	CIESIN, 2013.
Extent of MGS pasture (Global Agricultural Lands: Pastures)	Year 2000, 10km (Raster)	Ramankutty *et al*, 2008.
Extent of MGS cropland (Global Agricultural Lands: Croplands)	Year 2000, 10km (Raster)	Ramankutty *et al*, 2008.
Monthly temperature and rainfall (WorldClim)	Year 2000-2015 (monthly), 1km (Raster)	Hijmans *et al*, 2005.
MGS Protected Areas (World Database on Protected Areas)	Year 2000, 1km (shapefile)	IUCN and UNEP-WCMC, 2015.
Population Density (Global Rural-Urban Mapping Project)	Year 2000, 1km (Feature)	Balk et al, 2006.
Settlement Points (Global Rural-Urban Mapping Project)	Year 2000, 1km (Feature)	Balk et al, 2006.
Global Cattle, Buffalo, Sheep Density (Gridded Livestock of the World)	Year 2000, 1km (Feature)	FAO, 2005.
Snow cover (MODIS snow cover fraction)	Year 2000 (monthly), 500m Raster	NASA, 2006.

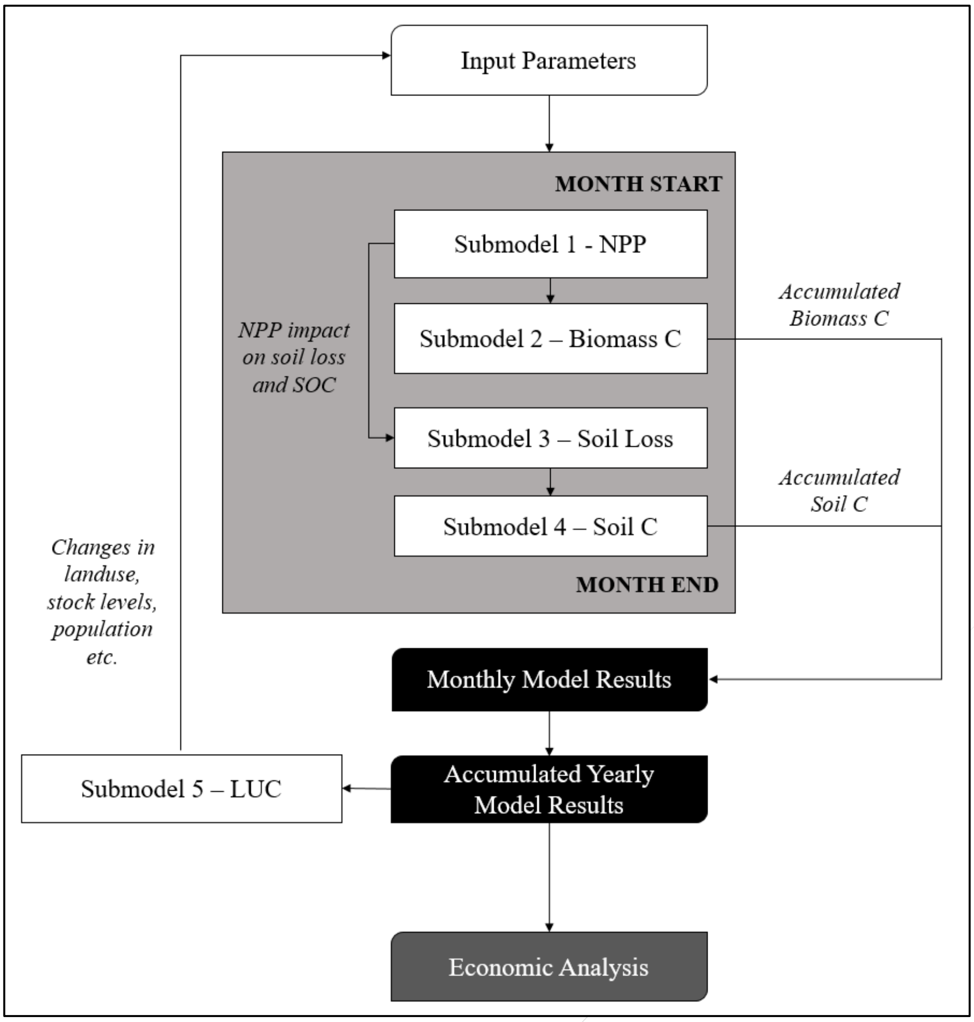
Most datasets in the model start in the year 2000, which corresponds to the simulation start time. The simulation timeframe was set from 1 January 2000 to 31 December 2015. This timeframe was chosen on the basis: i) that the year 2000 provides the opportunity to utilise the most reliable and consistent baseline data (e.g. HWSD, GLC2000 and WWF Ecoregions); and ii), that empirical data published until the year 2015, would be used to validate the model. Politically, 2015 also represents an important year from a climate policy perspective given the establishment of the *Paris Agreement* at the most recent United Nations Framework Convention on Climate Change (UNFCCC) Conference of Parties.

The model's spatial resolution varies by dataset, from 10km (5 Arc Mins) to 1km (30 Arc Secs). Ecoregion, climate, biomass and soil C data is of the finer 1km resolution, which is important when considering that rainfall, temperature and snowfall are highly variable in mountains areas, influencing Net Primary Productivity (NPP) and soil loss, and thus CO_2 sequestration from site-to-site (Beniston, 2003). Roads, livestock density and initial land use data resolution was more coarse, however when considering that the model is of global scale, the resolution was judged to be adequate for the stated purpose. Overall, we argue that the model largely achieves spatial and temporal heterogeneity by utilising the best global scale data available for MGS for the year 2000.

5.6 Process overview and scheduling

The model proceeds in monthly time steps (Figure 19). Within each month, five key submodels are processed in the following sequence: NPP (Submodel 1); Biomass C accumulation (Submodel 2); soil loss (Submodel 3); Soil C accumulation (Submodel 4); and, LUC (Submodel 5). This sequence is completed for each of the 27,000+ ecoregions individually, for the period 1 January 2000 to 31 December 2015, with monthly outputs generally accumulating with each time step for each ecoregion. Key outputs (e.g. 'Change in total ecoregion C') are aggregated on a country (national) and then at a global (simulation) level on an annual basis. The LUC submodel feedbacks changes into the input parameters and associated variables on an annual basis. Each of these submodels is described below. Note that because we chose not to include dynamic interaction between ecoregions the model is insensitive to the order of execution of the ecoregions.

Figure 19. Model design overview (including submodels)



5.7 Design concepts

The model is deterministic (non-stochastic) and focused on simulating five main relationships (submodels) for each individual ecoregion, where the following general assumptions have been made:

i. low mean monthly temperatures (especially winter), low rainfall and proportionally high snow coverage limit net NPP and therefore net biomass C accumulations;
ii. NPP is adversely impacted by high livestock stocking rates (TLU/ha), high rural population density (people/ha) and more frequent crop harvesting;
iii. lower net NPP has a relatively minor direct impact on soil C, through indirectly contributing to more sparse vegetation and lower soil protection, and thus higher rates of soil and C loss over time;
iv. rates of soil erosion and C loss are higher for MGS ecoregions located in areas with steeper and shorter slopes and where carrying capacity is exceeded; and

v. average annual trends in national cropland, pasture, protected area, population and livestock expansion, as limited by biophysical and social factors (e.g. biomass C and SOC limits, erosion protection, protected area status and proximity to major roads), change the land use makeup of each ecoregion, and therefore net NPP, soil erosion, and ultimately ecoregion C accumulated over time.

The model takes into account many of the most important drivers of land use change, as identified by the Economics of Land Degradation (ELD) Initiative (2015). These drivers include those categorised as *Proximate* (topographic, land cover, climate, and soil erodibility), *Underlying* (population density, market access), *Natural* (topographic, land cover, climate, soil erodibility) and *Anthropogenic* (land cover, unsustainable land management).

5.8 Initialization

The model was run twice for each of the 48 mountain countries and associated individual ecoregions concerned, with the simulation timeframe set from 1 January 2000 to 31 December 2015. Under the first simulation, all input parameters for the Year 2000 (Table 10) sourced from Ward *et al* (2014) and using GIS (Table 5) remained unchanged. Under the second simulation, key anthropogenic-orientated input parameters influencing reductions in NPP and increased soil loss were manually set to '0'. These parameters were: land cover protection factor; livestock and rural population density; CHF; and, annual trends in population, livestock, cropland, pasture and protected area expansion. Setting these parameters to '0' effectively negates anthropogenic influence under this scenario, for example, if livestock density is set to zero then there are no cattle consuming biomass and therefore no loss of NPP and higher gains in biomass and soil C.

5.9 Input

The model is largely driven by monthly changes in temperature, rainfall and snowfall - important environmental conditions for the accumulation of C in MGS (Benitson, 2003). These conditions are spatially explicit for each ecoregion, and influence the dynamics of the model's dynamic variables e.g. NPP. Model inputs are discussed in detail in *Section 6.0 Supplementary Materials*. .

5.10 Submodels

The submodels are run in the following sequence: NPP; net biomass C accumulation; soil and SOC loss; net SOC accumulation; and, land use change. The specific approach, mathematical formulas, assumptions and references for each submodel are detailed below, with formulas and further explanation provided in *Chapter 5.16, Supplementary materials*.

5.10.1 Net Primary Productivity (NPP)

According to Running *et al* (2000), terrestrial biological productivity (or primary productivity) is the single most fundamental measure of global change of practical interest for humankind. NPP is the measure of C intake by plants during photosynthesis, and this measure is an important indicator for studying the health for plant communities. To determine biomass NPP for each ecoregion, the first stage in our model, we used a modified version of Leith's well-known *Miami Model* which was first published in 1975 (Leith, 1975). This empirical model links NPP with either long-term mean rainfall or temperature, with the assumption that the smaller value of the two be applied to achieve the most conservative estimate for NPP on the basis that any increase in NPP is assumed to be limited by these two factors (Grieser *et al*, 2006). Though relatively simple to implement, the *Miami Model*'s ability to generate reasonable global NPP estimates (as done so in other studies) made it a suitable starting point for this study (Zaks *et al*, 2007).

While the *Miami Model* is a relatively reliable method of estimating NPP in general terms, it does require further modification for MGS. A regression analysis by Lin (2014) found that the Miami Model overestimated high values of observed values of NPP for MGS ecosystems of the Qinghai-Tibetan Plateau. A probable reason for this is that the Miami Model does not take into account the extent to which NPP is constrained by winter snow coverage through limiting the light response to CO_2 uptake (Körner, 1982). Another potential issue is that the *Miami Model* does not take into account changes in atmospheric CO_2 concentrations and other environmental factors, such as the CO_2 response coefficient and scale translation, plant life span, and the steady-state partitioning coefficient concerning the proportionate distribution of new biomass to the leaf, branch, stem and roots of MGS vegetation (Cure and Acock, 1986; Grace *et al*, 2006; King *et al*, 1997; Walker *et al*, 2013). To take these factors into account, we adopted Grace *et al's* (2006) and King *et al's* (1997) suggestions for modifying the *Miami Model,* then adapted it further for our purposes by including the NPP limiting impact of monthly snow coverage, and by subtracting NPP consumed by the local rural population and livestock.

5.10.2 Biomass C submodel

In this model, we assume that C may only continue to accumulate in vegetation if the assumed maximum theoretical storage value for biomass C has not been reached on a per hectare basis. Each ecoregion has been assigned a biomass C limit ($tCha^{-1}$) based on the highest maximum C density per hectare ($tCha^{-1}$) given by the GLC2000 (Table 5) spatial dataset for the relative ecoregion type (e.g. Tian Shan Montane Steppe and Meadows) and respective country (e.g. China). The limit reflects C storage in the most intact, healthy and undisturbed (and often remote) ecoregions.

5.10.3 Soil loss (USLE) submodel

According to Ward *et al* (2014), around 98 percent of MGS ecosystem C is contained within the soil globally. It is therefore critical to consider the influence of LULUC on erosion rates, and the extent to which SOC is destabilised and/or exposed to oxygen and thus released as a GHG into the atmosphere. The Universal Soil Loss Equation (USLE) is a processed-based mathematical model, developed from erosion plot empirical studies and rainfall simulations, commonly used to estimate long-term annual soil erosion within a given area or slope (Hudson, 1993; Lane *et al*, 1988; Wischmeier and Smith, 1978). Like the *Miami Model*, USLE is a relatively simple approach to modelling erosion and soil loss, which has been used at the local to global scale (King *et al*, 1997).

5.10.4 Soil C submodel

In this model, we assume that C may only continue to accumulate in soil if the assumed maximum theoretical value for SOC (down to 1m depth) has not been reached on a per hectare basis. Each ecoregion has been assigned a C limit based on the highest maximum C density per hectare ($tCha^{-1}$) given by the HWSD (Table 5) spatial dataset for the relative ecoregion type and respective country.

5.10.5 Land use change submodel

We assumed national-level LUC growth indicators (cropland, pasture, TLU, rural population), coupled with constraints associated with market access (i.e. major roads) and protected area status, to be sufficient for the purposes of incorporating LUC trends into this model. National-level LUC data for each of the 48 mountain countries concerned was obtained from FAOSTAT (FAO, 2015a) for the following variables for the years 2000 to 2015: annual percent change in agriculture (crop) area; annual percent change in pasture area; annual percent change in protected area; annual percent change in cattle and buffalo herd size; and, annual percent change in rural population.

At the end of each simulation year, the model updates the key input parameters accordingly by way of a *state-and-transition* approach i.e. change in percent of ecoregion land use for cropland versus pasture versus natural (protected), percent increase or decrease in rural population and TLU per hectare. This has flow on effects for NPP and soil loss. For example, an upward trend in cattle density (TLU/ha) increases NPP consumed (decreasing biomass C) while also potentially exceeding the ecoregion's carrying capacity, and therefore adversely affecting the USLE land cover protection factor (i.e. more bare ground) and ultimately leading to greater erosion, soil and SOC loss.

Each ecoregion's proximity to road and its protected areas status also has a bearing on whether or not LUC trends were applied. Explicitly, if the borders of the concerned ecoregion were within 1km proximity of a major road (Table 5), then we applied the relevant trend for cropland, pasture, cattle and population expansion to the ecoregion as described above. If not, then we assumed these parameters for the ecoregion did not change. Likewise, we did the same for protected area status, where the ecoregion was not protected by a recognised convention (Table 5) then the input parameters would change (and vice versa). In summary, LUC trends were only applied if the ecoregion: was within 1km of a major road and not a protected area.

5.11 Estimation of economic value

Using the simulation results, we adopted a quasi-option value *avoided cost* approach, as presented by TEEB (2010) and TEEB (2013), to estimate the cumulative and annual global economic value of CO_2 sequestered by MGS. According to TEEB (2010) avoided cost is the most common approach to quantifying the value of regulating ecosystem services (Lescuyer, 2000; Pendleton *et al*, 2012). The approach "relies on the assumption that damage estimates are a measure of value" (TEEB, 2010, p.32). We use SCC as the key metric to value avoided damage. According to the US Government (2015, p. 2) SCC can be defined as "estimate of the monetized damages associated with an incremental increase in carbon emissions in a given year" whose purpose is to "allow agencies to incorporate the social benefits of reducing carbon dioxide (CO_2) emissions into cost-benefit analyses of regulatory actions that impact cumulative global emissions".

To determine the avoided damage currently being provided by MGS, and the potential for avoiding additional damage should MGS be more sustainably managed, the US Government's (2015) estimates for SCC were applied in 2015 (Table 6) to the outputs of the model (annual and accumulated *net CO_2 exchange* between 2000 and 2015, and *total in-situ CO_2* storage in MGS at the end of 2015). Providing an economic estimate of net CO_2 sequestration and total in-*situ* CO_2 storage provides an important metric on the value of MGS in terms of its ecosystem services and as an environmental asset. SCC pricing scenarios are given below (US Government, 2015), with the value of US$36/$CO_2$ (at three percent discount rate) considered our central estimate.

Table 6. SCC Pricing Scenarios 2015 (2007 US$ / CO_2)

Discount Rate	**5.0%**	**3.0%**	**2.5%**	**3.0%**
Model Value	Mean value	Mean value	Mean value	*95th Percentile*
SCC Estimate	US$11/$CO_2$	$36/$CO_2$	$56/$CO_2$	$105/$CO_2$

The first three values (at discount rates of five, three and 2.5 percent) represent the mean SCC estimate for 2015 (in 2007 US$ per tCO_2) as generated by three of the most commonly used global integrated assessment models (IAMS) which seek to quantify the damage to society caused by climate change e.g. impacts of temperature rise agriculture and sea-level rise on infrastructure. These IAMs are the Dynamic Integrated Climate and Economy (DICE), Policy Analysis of the Greenhouse Effect (PAGE) and Climate Framework for Uncertainty (FUND). The 95th percentile SCC estimate given for a three percent discount rate represents higher-than-expected damage across the three IAMs (US Government, 2015).

5.12 Testing & validation

We tested the model by observing the monthly results of each submodel as it was processed, how the impact of these results were accounted for the next submodel processed by the system (and-so-on), and by observing the outputs of the model as a whole on a monthly, yearly and year-to-year basis. We manually calculated the results outside the model to crosscheck these results. Testing consisted of manual procedures performed throughout the model development, testing and implementation according to a checklist, for example: Identifying poorly formed geometry and evaluating sliver polygons (very small polygons resulting from overlay analysis, as used by this study) before extracting spatial data from *ArcGIS*, cell-by-cell checks of all data input spreadsheets for null, outlying and duplicate values, and, ensuring calculation formulas were correctly applied within *ArcGIS*, *AnyLogic* and within *Excel*.

The simulation results fall largely within the range of recently published empirical values for C density (t C ha^{-1}) for the relative MGS country (Table 7), and in most cases are conservative. Most of these results are directly related to the initial input parameters, so while the results may support the validity of the model, they mostly confirm that the input parameters were set appropriately. Other results from the model emerge from more complex inter-relationships. For example, annual soil loss is influenced by rainfall, existing vegetation type and coverage, annual net biomass (which is influenced by temperature, rainfall, snow coverage, crop harvest frequency, biomass consumption by livestock and human populations) and overgrazing. This in turn provides positive feedback concerning biomass and soil C for each ecoregion as a whole, and therefore nationally and globally when aggregated.

Table 7. Comparison of studies on ecosystem C storage dynamics (C density) in mountain grasslands and shrublands with results provided by this model

Country	Study results (biomass and soil)	Study Reference	Avg.model result at end of 2015
Global average	40 - 207 t C ha^{-1}	Schlesinger 1977	60 t C ha^{-1}
China	31 - 290 t C ha^{-1}	Wang *et al* 1998	67 t C ha^{-1}
	12 - 565 t C ha^{-1}	Wang *et al* 2002	
	10 - 137 t C ha^{-1}	Ohtsuka *et al 2008*	
New Zealand	16 - 56 t C ha^{-1}	Landcare Research 2012	62 t C ha^{-1}
Columbia	135 - 521 t C ha^{-1}	Pena *et al* 2011	232 t C ha^{-1}
Venezuela	259 - 486 t C ha^{-1}	Hofstede *et al* 2003	430 t C ha^{-1}

5.13 Limitations and uncertainties

For each ecoregion, we estimated the total CO_2 that could be sequestered (or released) annually. Due to the lack of data quality this study did not utilise more sophisticated methods for estimating decomposition processes over time e.g. exponential delay modelling (VCS, 2014). The temporal aspects of shallow SOC decomposition (0-1m depth) is an area of significant research with many complexities (Pendleton *et al*, 2012). We took a conservative approach when estimating SOC sequestration and losses, only focusing on SOC in the top metre of soil (Pendleton *et al,* 2012). Our assumption is that these pools are most prone to LUC disturbances, and thus most important to include in the model. The USLE model has a number of limitations: the model applies only to sheet erosion caused by rainfall (not mass erosion events); has not been verified using empirical data from steep slopes > 25º; the mathematical relationship between kinetic energy and rainfall intensity is likely to be underestimated (meaning soil loss could be much higher in MGS areas with high orographic rainfall influence); and, USLE uses decadal averages and does not take into account individual rainstorms (FAO, 2016). In estimating NPP, the *Miami Model* does not take into account leaf mass index, leaf area index (LAI), solar radiation, daytime length, the presence of persistent snow banks into spring and other factors important to the photosynthesis of alpine plants (Körner, 2003).

We did not account for other greenhouse gas emissions, such as methane ($CH_{4,}$) from livestock or nitrous oxide (N_2O) from fertiliser. Nor did we account for possible exchanges of CO_2 from one ecosystem to another. Mountain ranges located in Turkey, Iran, Liechtenstein, the Antarctic and its remote islands (e.g. Heard Island), and other relatively small and isolated MGS ecoregions in the Rockies Mountains and European Alps were excluded due to the lack of reliable spatial data. However, taking into account these exclusions, 98 percent of the MGS land area identified by Ward *et al* 2014 was included. We also did not take into account the biophysical influence that MGS have on climate change as suggested by Hungate and Hampton (2012). In this case, we did not apply a discount rate to the economic analysis, however if the simulation was run over a greater timeframe then this would be necessary. It should also be noted that SCC should be considered an economic estimate for avoided climate damage, and not necessarily what the market is willing to pay for emissions reductions (US Government, 2010). Considerable uncertainty also stems from the rates of land use conversion in MGS, and more generally for mountain areas as identified by a number of studies (Gurung *et al*, 2010; Jansky, 2002; Ward *et al*, 2015).

All-in-all, given that our model is of global scale, with the objective of providing an initial and conservative estimate of climate regulation value to stimulate future research and policy discourse, we argue that our model approach and assumptions are fit-for-purpose while acknowledging that improvements could be made in the future.

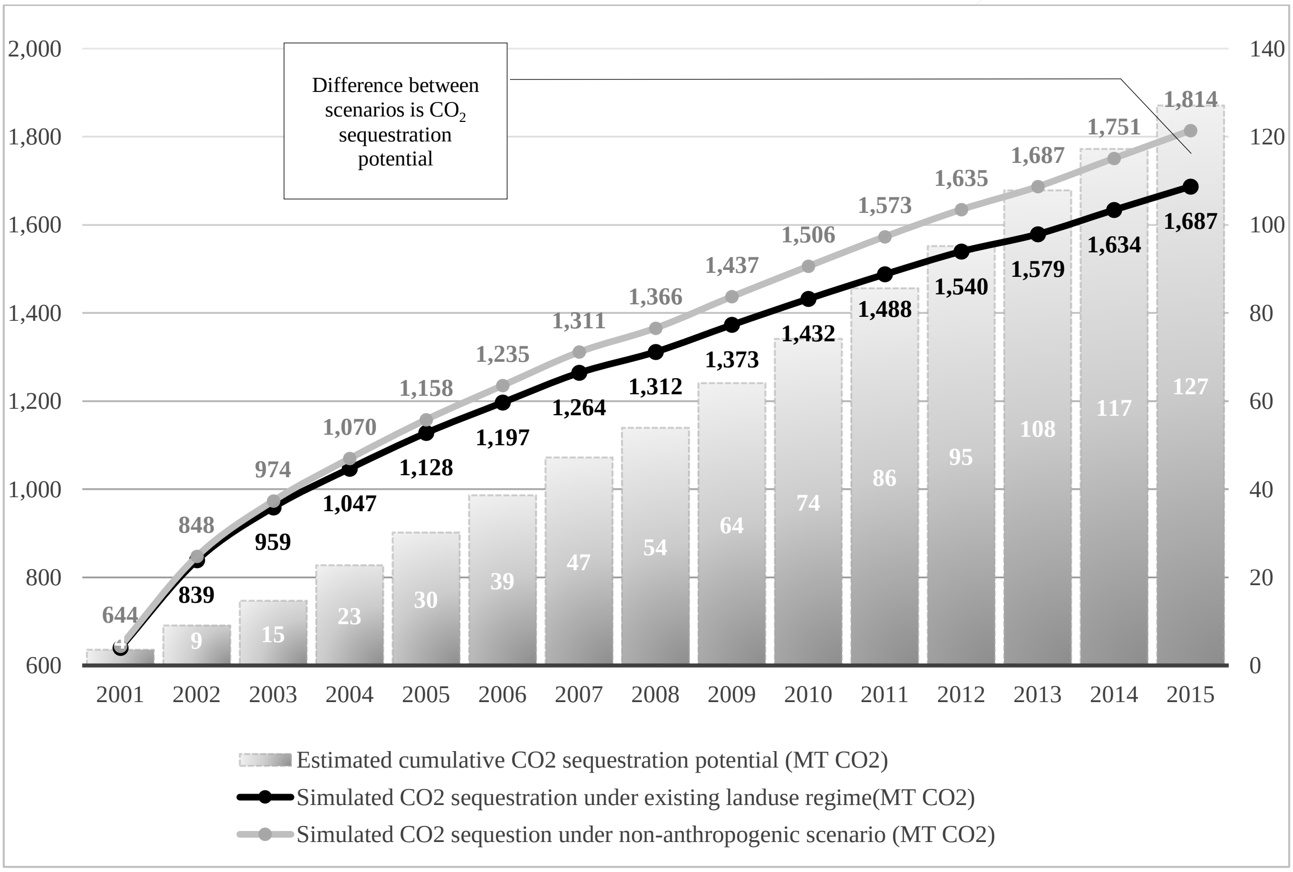
5.14 Results and Discussion

We ran two scenarios using the model: in the first scenario all input parameters remained unchanged. Under the second scenario key anthropogenic-orientated input parameters influencing reductions in NPP and increased soil loss were manually set to ‘0’. As no global baseline of MGS CO_2 exchange currently exists, the first objective was to determine the difference in net CO_2 sequestration between the two scenarios, providing an initial estimate for *potential CO_2 by MGS,* and subsequently, potential avoided economic damage using SCC.

The simulation results (Figure 20) reveal that between 1 January 2000 and 31 December 2015, under current land use regimes we estimate MGS sequestered 1,687 million tonnes of CO_2 ($MtCO_2$), or an average of 112 $MtCO_2$ per annum. If land use is managed more sustainably, MGS ecosystems could sequester *up to* 8.4 Mt in additional CO_2 per annum (total of 121 $MtCO_2$ per annum, or 1,814 $MtCO_2$ over the 15 year simulation timeframe). The results vary substantially from MGS ecoregion to ecoregion and from country to country (Table 8). For example, MGS ecoregions located in China account for around 54 percent of this potential, while MGS in the Democratic Republic of the Congo are relatively small and thus account for approximately two percent of global MGS CO_2 sequestration potential.

These estimates support the postulation made in Ward *et al* (2014) that when MGS ecoregions are maintained in a sustainable ecological state, where LUC influences were minimal compared to the present situation where many MGS are being managed under various levels of land use and land use intensity, MGS ecoregions can sequester additional CO_2 emissions. MGS represent a relatively minor (though still comparable) *annual* CO_2 sink when compared to mangroves (for example) where LULUC is estimated to result in reduced CO_2 sequestration of between 0.8 and 21 MT C per annum (Bridgham *et al*, 2006; Pigeon, 2009).

Figure 20. Global cumulative CO_2 sequestration potential of mountain grasslands and shrublands vs CO_2 sequestration under current land use regime (between 2001-2015)



Note: This figure shows the potential cumulative CO_2 sequestration by MGS between 2000 and 2015. This is the difference in CO_2 sequestered between simulated baseline sequestration under existing land use regime scenario and a scenario where anthropogenic influence is minimised (i.e. largely intact and healthy MGS ecosystems reach maximum CO_2 uptake).

Translating these outputs into an economic value, between 1 January 2000 and 31 December 2015, under *current* land use regimes the total CO_2 sequestered worldwide (avoided SCC) was estimated to be between \$18.5 billion and \$177.14 billion (2007 dollars, based on 1,687 $MtCO_2$ being sequestered over the 15 years). On an annual basis, we estimate the average avoided SCC to be between US\$1.24 billion and US\$11.8 billion (based on 112 $MtCO_2$ per annum and depending on the discount rate chosen). If MGS were more sustainably managed, the *potential additional value* (Table 8) of MGS climate regulation (avoided SCC) is estimated to be between US\$1.3 billion and US\$13.34 billion over the simulation timeframe (1 January 2000 and 31 December 2015), and on average US\$0.093 billion - US\$0.89 billion *per annum*, with a central estimate (at three percent discount rate) of US\$0.30 billion per annum as shown in Table 8.

Table 8. Range of economic values (2007 US$ million) for potential additional CO_2 sequestration by MGS ecosystems 1 Jan 2000 - 30 Dec 2015 (total and annual)

	SCC Scenarios 2015 (US$ million 2007)						
	Est. *total* tCO_2 sequest. 2000-2015	*Potential add. sequest value for 2000-2015 (15 years)* @US$ 11 2.5% Avg.	@US$ 36 3% Avg.	@US$ 56 5% Avg.	@US$ 105 3% Avg.	*Ann. mean sequest. value* Est. mean *annual* tCO_2 sequest.	@US$ 36 3% Avg.
China	69,385,174	$763.24	$2,497.87	$3,885.57	$7,285.44	4,625,678	$166.52
Ethiopia	24,071,830	$264.79	$866.59	$1,348.02	$2,527.54	1,604,789	$57.77
South Africa	5,428,528	$59.71	$195.43	$304.00	$570.00	361,902	$13.03
Lesotho	4,814,405	$52.96	$173.32	$269.61	$505.51	320,960	$11.55
Argentina	3,665,588	$40.32	$131.96	$205.27	$384.89	244,373	$8.80
Peru	3,387,267	$37.26	$121.94	$189.69	$355.66	225,818	$8.13
Angola	2,882,927	$31.71	$103.79	$161.44	$302.71	192,195	$6.92
Kyrgyzstan	2,070,181	$22.77	$74.53	$115.93	$217.37	138,012	$4.97
Tanzania	1,396,714	$15.36	$50.28	$78.22	$146.65	93,114	$3.35
Ecuador	1,237,171	$13.61	$44.54	$69.28	$129.90	82,478	$2.97
Chile	1,216,556	$13.38	$43.80	$68.13	$127.74	81,104	$2.92
Bolivia	1,107,641	$12.18	$39.88	$62.03	$116.30	73,843	$2.66
Nepal	1,095,111	$12.05	$39.42	$61.33	$114.99	73,007	$2.63
Colombia	1,074,193	$11.82	$38.67	$60.15	$112.79	71,613	$2.58
Kazakhstan	913,809	$10.05	$32.90	$51.17	$95.95	60,921	$2.19
Nigeria	830,376	$9.13	$29.89	$46.50	$87.19	55,358	$1.99
Malawi	515,237	$5.67	$18.55	$28.85	$54.10	34,349	$1.24
New Zealand	453,263	$4.99	$16.32	$25.38	$47.59	30,218	$1.09
Eritrea	392,105	$4.31	$14.12	$21.96	$41.17	26,140	$0.94
Indonesia	345,306	$3.80	$12.43	$19.34	$36.26	23,020	$0.83
Tajikistan	331,867	$3.65	$11.95	$18.58	$34.85	22,124	$0.80
Bhutan	280,561	$3.09	$10.10	$15.71	$29.46	18,704	$0.67
PNG	213,088	$2.34	$7.67	$11.93	$22.37	14,206	$0.51
Morocco	187,197	$2.06	$6.74	$10.48	$19.66	12,480	$0.45
Zimbabwe	158,461	$1.74	$5.70	$8.87	$16.64	10,564	$0.38
Zambia	149,255	$1.64	$5.37	$8.36	$15.67	9,950	$0.36
India	123,164	$1.35	$4.43	$6.90	$12.93	8,211	$0.30
Kenya	118,613	$1.30	$4.27	$6.64	$12.45	7,908	$0.28
Mongolia	96,216	$1.06	$3.46	$5.39	$10.10	6,414	$0.23
Venezuela	90,145	$0.99	$3.25	$5.05	$9.47	6,010	$0.22
Madagascar	57,480	$0.63	$2.07	$3.22	$6.04	3,832	$0.14
Afghanistan	53,722	$0.59	$1.93	$3.01	$5.64	3,581	$0.13
Norway	40,239	$0.44	$1.45	$2.25	$4.23	2,683	$0.10
United States	35,542	$0.39	$1.28	$1.99	$3.73	2,369	$0.09
Russ. Fed.	25,877	$0.28	$0.93	$1.45	$2.72	1,725	$0.06
Turkmenistan	14,823	$0.16	$0.53	$0.83	$1.56	988	$0.04
DR Congo	3,705	$0.04	$0.13	$0.21	$0.39	247	$0.01
Finland	4,304	$0.05	$0.15	$0.24	$0.45	287	$0.01
Canada	- 12,456	-$0.14	-$0.45	-$0.70	-$1.31	- 830	-$0.03
Pakistan	- 1,155,505	-$12.71	-$41.60	-$64.71	-$121.33	- 77,034	-$2.77
Totals	**127,099,701.21**	**$1,398.10**	**$4,575.59**	**$7,117.58**	**$13,345.47**	**8,473,313**	**$305.04**

Australia, Greenland, Malaysia, Mozambique, Myanmar, Rwanda, Swaziland, Sweden and Uganda considered to have relatively stable C stores (thus not included above). Negative values indicate CO_2 sequestration is likely greater under the anthropogenic scenario.

Historically, the C within MGS has been stored over a long geological timeframe (FAO, 2015b). Changes are much more rapid now. The negative impact of anthropogenic stressors be it agricultural intensification or climate change are projected to increase over time for MGS due to growing populations, new technologies and greater affluence (Ward *et al*, 2015). Much of this C could be lost if we are not conscious of its significance, both in terms of its biochemical relevance to climate change policy and economic value to society. Therefore, when considering the economic value of avoided damage through climate regulation by natural ecosystems, it is important to recognise MGS as a long established C pool, and not just for the C stored annually (TEEB, 2010; Pendleton *et al*, 2012).

Taking into account the scenario where CO_2 is sequestered over the simulation timeframe, and using the original estimates provided by Ward *et al* (2014) we estimate there to be at least 68 ± 11 Petagrams C (252 ± 39 Gigatonnes CO_2) stored in MGS vegetation and soils worldwide at 31 December 2015.

The equivalent *in-situ* value (Table 9) of this C stock is estimated to be between US$2.5 ± 0.43 trillion and US$26.5 ± 4.1 trillion (2007 dollars). This range would be even wider if we considered the full array of estimates for SCC ($7–126 t$CO_2$) as published in the literature (Foley *et al*, 2013; Stern, 2007; US Government, 2010). Even at the most conservative estimate of US$2.5 trillion, MGS are arguably significant environmental assets when only avoided SCC is considered. Obviously, if one was to include a value for water provision, slope stability and the many other ecosystem services provided by MGS ecosystems, then this estimate would be far higher (Gret-Regamey and Kytzia, 2007).

Our premise for providing this first initial estimate of MGS climate regulation value is to highlight the potential damage to society should MGS be degraded. In other words, for every tonne of C that is not released from MGS there are benefits to humanity. From this point of view, and when considering the goals of *the Paris Agreement*, more sustainable LULUC should be considered alongside other climate policy options just as it has been considered for other ecosystem.

Table 9. Range of economic values (2007 US$ million) for absolute in-situ carbon storage by MGS ecosystems, as estimated at 31 December 2015, by country

	Mt CO2 storage equivalent (adapted from Ward *et al*, 2014)			**SCC Valuation Scenarios 2015 (2007 US$ million)** **@ US$ 11** **2.5% Avg.**			**@ US$ 36** **3% Avg. (central est.)**			**@ US$ 56** **5% Avg.**			**@ US$ 105** **3% (95th percentile)**		
Afghanistan	1,413	+/-	184	$15,545	+/-	$2,027	**$50,874**	**+/-**	**$6,633**	$79,137	+/-	$10,318	$148,381	+/-	$19,346
Angola	343	+/-	47	$3,774	+/-	$518	**$12,353**	**+/-**	**$1,694**	$19,215	+/-	$2,636	$36,029	+/-	$4,942
Argentina	3,036	+/-	495	$33,394	+/-	$5,442	**$109,291**	**+/-**	**$17,809**	$170,008	+/-	$27,703	$318,765	+/-	$51,944
Australia	306	+/-	52	$3,370	+/-	$568	**$11,028**	**+/-**	**$1,859**	$17,154	+/-	$2,892	$32,164	+/-	$5,422
Bhutan	145	+/-	20	$1,596	+/-	$217	**$5,224**	**+/-**	**$710**	$8,126	+/-	$1,104	$15,237	+/-	$2,070
Bolivia	3,054	+/-	505	$33,589	+/-	$5,558	**$109,928**	**+/-**	**$18,190**	$170,999	+/-	$28,295	$320,623	+/-	$53,053
Canada	23,978	+/-	3,884	$263,759	+/-	$42,719	**$863,210**	**+/-**	**$139,809**	$1,342,772	+/-	$217,480	$2,517,697	+/-	$407,776
Chile	2,563	+/-	477	$28,192	+/-	$5,245	**$92,265**	**+/-**	**$17,165**	$143,523	+/-	$26,701	$269,105	+/-	$50,064
China	74,968	+/-	11,198	$824,645	+/-	$123,181	**$2,698,840**	**+/-**	**$403,136**	$4,198,195	+/-	$627,101	$7,871,616	+/-	$1,175,814
Columbia	1,341	+/-	216	$14,751	+/-	$2,375	**$48,276**	**+/-**	**$7,774**	$75,096	+/-	$12,093	$140,805	+/-	$22,674
DR Congo	21	+/-	4	$230	+/-	$43	**$753**	**+/-**	**$139**	$1,171	+/-	$216	$2,195	+/-	$406
Ecuador	986	+/-	204	$10,841	+/-	$2,245	**$35,481**	**+/-**	**$7,348**	$55,192	+/-	$11,431	$103,486	+/-	$21,432
Eritrea	68	+/-	9	$752	+/-	$102	**$2,462**	**+/-**	**$333**	$3,830	+/-	$518	$7,182	+/-	$971
Ethiopia	2,666	+/-	445	$29,329	+/-	$4,896	**$95,986**	**+/-**	**$16,023**	$149,312	+/-	$24,925	$279,960	+/-	$46,734
Finland	440	+/-	62	$4,842	+/-	$681	**$15,846**	**+/-**	**$2,228**	$24,649	+/-	$3,466	$46,216	+/-	$6,499
Greenland	4,353	+/-	898	$47,882	+/-	$9,875	**$156,705**	**+/-**	**$32,319**	$243,763	+/-	$50,273	$457,056	+/-	$94,262
India	2,067	+/-	290	$22,732	+/-	$3,195	**$74,396**	**+/-**	**$10,455**	$115,726	+/-	$16,263	$216,987	+/-	$30,494
Indonesia	307	+/-	47	$3,381	+/-	$521	**$11,064**	**+/-**	**$1,705**	$17,210	+/-	$2,653	$32,269	+/-	$4,974
Kazakhstan	1,868	+/-	257	$20,551	+/-	$2,827	**$67,259**	**+/-**	**$9,251**	$104,625	+/-	$14,390	$196,172	+/-	$26,981
Kenya	62	+/-	16	$683	+/-	$176	**$2,235**	**+/-**	**$575**	$3,476	+/-	$895	$6,517	+/-	$1,678
Kyrgyzstan	2,423	+/-	329	$26,650	+/-	$3,614	**$87,218**	**+/-**	**$11,826**	$135,673	+/-	$18,396	$254,387	+/-	$34,493
Lesotho	757	+/-	107	$8,325	+/-	$1,179	**$27,246**	**+/-**	**$3,857**	$42,383	+/-	$6,000	$79,468	+/-	$11,250
Madagascar	24	+/-	4	$267	+/-	$40	**$874**	**+/-**	**$130**	$1,360	+/-	$202	$2,550	+/-	$379
Malawi	164	+/-	20	$1,806	+/-	$218	**$5,912**	**+/-**	**$714**	$9,196	+/-	$1,111	$17,243	+/-	$2,084
Malaysia	4	+/-	1	$43	+/-	$6	**$142**	**+/-**	**$21**	$220	+/-	$33	$413	+/-	$62
Mongolia	3,943	+/-	538	$43,369	+/-	$5,918	**$141,936**	**+/-**	**$19,368**	$220,789	+/-	$30,128	$413,979	+/-	$56,490
Morocco	220	+/-	30	$2,418	+/-	$326	**$7,915**	**+/-**	**$1,067**	$12,312	+/-	$1,660	$23,085	+/-	$3,112

Mozambique	16	+/-	2	$177	+/-	$23	**$581**	**+/-**	**$74**	$903	+/-	$115	$1,694	+/-	$215
Myanmar	134	+/-	17	$1,475	+/-	$191	**$4,826**	**+/-**	**$626**	$7,507	+/-	$973	$14,075	+/-	$1,825
Nepal	592	+/-	84	$6,510	+/-	$921	**$21,305**	**+/-**	**$3,015**	$33,142	+/-	$4,690	$62,140	+/-	$8,794
New Zealand	893	+/-	137	$9,819	+/-	$1,507	**$32,135**	**+/-**	**$4,933**	$49,988	+/-	$7,673	$93,728	+/-	$14,388
Nigeria	54	+/-	8	$590	+/-	$84	**$1,932**	**+/-**	**$275**	$3,006	+/-	$427	$5,636	+/-	$801
Norway	3,981	+/-	790	$43,786	+/-	$8,685	**$143,301**	**+/-**	**$28,424**	$222,913	+/-	$44,215	$417,962	+/-	$82,904
Pakistan	2,223	+/-	293	$24,457	+/-	$3,227	**$80,041**	**+/-**	**$10,561**	$124,508	+/-	$16,428	$233,453	+/-	$30,803
PNG	150	+/-	24	$1,654	+/-	$259	**$5,412**	**+/-**	**$848**	$8,418	+/-	$1,320	$15,784	+/-	$2,474
Peru	4,705	+/-	783	$51,759	+/-	$8,611	**$169,394**	**+/-**	**$28,182**	$263,502	+/-	$43,839	$494,065	+/-	$82,198
Russ. Fed.	56,203	+/-	8,939	$618,237	+/-	$98,333	**$2,023,323**	**+/-**	**$321,816**	$3,147,391	+/-	$500,602	$5,901,358	+/-	$938,629
Rwanda	74	+/-	15	$813	+/-	$167	**$2,662**	**+/-**	**$546**	$4,140	+/-	$849	$7,763	+/-	$1,593
South Africa	519	+/-	66	$5,709	+/-	$731	**$18,685**	**+/-**	**$2,392**	$29,065	+/-	$3,721	$54,497	+/-	$6,977
Swaziland	119	+/-	17	$1,314	+/-	$186	**$4,302**	**+/-**	**$608**	$6,691	+/-	$945	$12,546	+/-	$1,773
Sweden	4,955	+/-	728	$54,509	+/-	$8,010	**$178,392**	**+/-**	**$26,215**	$277,499	+/-	$40,778	$520,311	+/-	$76,459
Tajikistan	1,132	+/-	150	$12,449	+/-	$1,653	**$40,743**	**+/-**	**$5,409**	$63,379	+/-	$8,414	$118,835	+/-	$15,777
Tanzania	130	+/-	24	$1,425	+/-	$260	**$4,663**	**+/-**	**$850**	$7,254	+/-	$1,323	$13,602	+/-	$2,480
Turkmenistan	7	+/-	1	$77	+/-	$11	**$251**	**+/-**	**$36**	$390	+/-	$56	$732	+/-	$105
Uganda	76	+/-	15	$832	+/-	$167	**$2,723**	**+/-**	**$545**	$4,236	+/-	$848	$7,942	+/-	$1,589
United States	44,478	+/-	6,072	$489,258	+/-	$66,790	**$1,601,209**	**+/-**	**$218,586**	$2,490,769	+/-	$340,022	$4,670,192	+/-	$637,541
Venezuela	611	+/-	130	$6,724	+/-	$1,434	**$22,005**	**+/-**	**$4,692**	$34,231	+/-	$7,298	$64,183	+/-	$13,684
Zambia	16	+/-	2	$177	+/-	$21	**$579**	**+/-**	**$70**	$900	+/-	$109	$1,688	+/-	$204
Zimbabwe	151	+/-	19	$1,660	+/-	$213	**$5,434**	**+/-**	**$698**	$8,453	+/-	$1,085	$15,850	+/-	$2,035
Totals	**252,739**	**+/-**	**38,654**	**$2,780,132**	**+/-**	**$425,192**	**$9,098,614**	**+/-**	**$1,391,538**	**$14,153,399**	**+/-**	**$2,164,615**	**$26,537,623**	**+/-**	**$4,058,653**

The following countries have been omitted from the original dataset provided by Ward *et al* (2014) due to the absence of suitable spatial data: Austria, Bulgaria, Costa Rica, France, Germany, Italy, Poland, Romania, Spain, United Kingdom, Slovakia, Slovenia, Switzerland and Uzbekistan.

Perhaps more pragmatically, by placing a value on something, it can be argued that more informed and equitable public policy decisions can be made due to a better understanding of full costs and benefits (Costanza *et al*, 1997; Costanza *et al*, 2014; Daily, 1997; TEEB, 2013). Policy makers and industry need to make decisions all the time, and the economic premise is that every decision is underpinned by weighing the values among different alternatives (Bingham *et al*, 1995). In valuing natural capital and ecosystem services such as MGS, policy makers can be supported in: discovering areas of market failure; setting up markets to address these failures (e.g. for carbon), untangling uncertainty surrounding future natural resource use, designing innovative ecosystem conservation programs (e.g. payment for ecosystem services) and establishing robust natural capital accounts (TEEB, 2010). Moreover, placing a value on a natural capital stock and its ecosystem services can also serve to guide policy makers in making what are often tough trade-offs in our resource constrained world (Costanza *et al*, 2014).

TEEB (2010) concluded that "natural resources are economic assets, whether or not they enter the marketplace" and that "conventional measures of national economic performance and wealth…. fail to reflect natural capital stocks of flows of ecosystem services contributing to the economic visibility of nature" (p.26). In this sense, valuation studies are critical to the long-term sustainable management of MGS and other ecosystems. Likewise, valuation studies have been instrumental in raising awareness of the world's natural capital compared to the world's gross economic output, such as in Costanza *et al*'s landmark study in 1997, and the many thousands of studies that have been spurred by it. Conversely, we acknowledge that there are important ethical and altruistic reasons to conserve MGS carbon ecosystems, and both moral and intrinsic issues arising from putting a price on what many consider priceless values (McCauley, 2006).

5.15 Conclusion

Here we have presented the first global economic valuation of C stored in MGS, filling critical gaps in knowledge about MGS ecosystems (Gurung *et al*, 2010; Jansky, 2002). Our principle aim was to determine the economic significance of these C stocks, and in so doing, contribute to enhanced public policy formation and setting future research directions. For example, the estimate made here provides a more robust global measure of MGS C stocks and CO_2 fixes compared to what is currently available, which is critical in tracking progress towards the emissions reduction targets set out by the *Paris Agreement*. Importantly, although the majority of MGS are located in remote and inhospitable areas, if managed more sustainably, these ecoregions could help avoid climate-induced damage to our society by between US$0.09billion and US$49 billion per annum. By the same token this is the estimated social cost of carbon that is being borne globally year-on-year. By estimating the relative value of carbon stored by MGS, we are attempting to make “the values of nature visible and accountable for in economic decision making” (Akerman and Peltola, 2012 p.1). Our results provide pointed information for national government as to which MGS ecoregions are under the greatest stress from LULUC. Lastly, our results also support the business case for developing the required methods, modalities, legal frameworks and institutions if climate finance is to be used to incentivise the sustainable management of MGS in the future (Ward *et al*, 2015).

5.16 Supplementary materials

5.16.1 Submodels

Net Primary Productivity (NPP)

The standard three equations of the Miami Model is provided below.

$$\boldsymbol{NPP_T = \frac{3000}{1+e^{1.315-0.119T}}} \qquad \textbf{(1)}$$

$$\boldsymbol{NPP_P = 3000\,(1 - e^{-0.000664P})} \qquad \textbf{(2)}$$

$$\boldsymbol{NPP = \min(NPP_{T,} NPP_P)} \qquad \textbf{(3)}$$

Where

NPP_T and NPP_P represent *NPP* as functions of mean annual temperature T (ºC) and average annual precipitation P (mm) (*Equations 1 and 2*),

Equation 3 selects the minimum of NPP_T and NPP_P which provides a measure of NPP in *gDM/m²/year/ºC* and *g(DM)/m²/year/(mm/year)* respectively.

Our enhanced method for predicting NPP in MGS ecoregions is given by the following equation:

$$\boxed{\boldsymbol{NPP_{MGS} = SF \times \min\left(NPP_{T,} NPP_P\right) \times PL\left(ST \times CO2_{Resp} \times \frac{p-p0}{p0}\right) - nppC_{pop,liv,hvs} \times 0.01 \times areaHa}}$$

Where

NPP_{MGS} is the net NPP for the given month,

SF is the 'snowless fraction' (%) of the ecoregion,

PL is the 'plant lifespan' (years) of MGS vegetation,

ST is the 'scale translation' factor,

$CO2_{Resp}$ is the 'CO_2 response coefficient',

p and *p0* is the atmospheric concentration of CO_2 (ppm) at the start (year 2000) and end of the simulation (year 2015) respectively,

$nppC_{pop,liv,hvs}$ is the biomass consumed by the local population (for energy), livestock and crop harvest respectively,

areaHa is the total area of the ecoregion (hectares).

The snowless fraction for each ecoregion is spatially derived from NASA (2006). Plant life is assumed to be 1 year (Grace *et al*, 2006). The inclusion of changes in atmospheric CO_2 is important because NPP is a function of both climate and CO_2 concentration (King *et al*, 1997). The CO_2 response coefficient also known as the biotic growth factor (Bacastow and Keeling, 1973), refers to the leaf photosynthesis response, and is a key factor in determining NPP (Polglase and Wang, 1999). King *et al*, 1997 suggest that a scaling translation factor of 0.6 be applied to the CO_2 response coefficient in order to apply it at a global simulation level, rather than at an ecosystem level, for which it has not been verified (Grace *et al*, 2006). The formula for calculating the CO2 response coefficient is provided in *Equation 4*.

$$\boldsymbol{CO2_{Resp} = \frac{3\beta CO2_{Comp}}{(\beta - CO2_{Comp})(\beta - 2CO2_{Comp})}} \quad \textbf{(4)}$$

Where

B is the intercellular CO_2 concentration, and $CO2_{Comp}$ is the CO_2 compensation point.

The intercellular CO_2 concentration factor is assumed to be 0.7 and represents the proportion of CO_2 captured by the leaves of plants (Leegood, 2001; King *et al*, 1997). The CO_2 compensation point' (Equation 5) refers to the point at which the uptake of CO_2 via photosynthesis is matched to the respiration of CO_2 (Long and Bernacchi, 2003).

$$\boldsymbol{CO2_{Comp}} = \frac{\boldsymbol{0.5 * CO2_{ppress}}}{\boldsymbol{RubiscoS}} \qquad \textbf{(5)}$$

Where

$CO2_{ppress}$ is CO_2 partial pressure and *RubiscoS* to 'Rubisco' (mmol mol^{-1}).

CO_2 partial pressure is assumed to have a value of 0.36 for MGS which often exist at high altitude (Billings and Mooney, 1968; Körner *et al*, 2005). Rubisco (ribulose-1,5-bisphosphate carboxylase-oxygenase) is an enzyme which influences the first step of carbon fixation and here is considered to have a mid-range value of 100 mmol mol^{-1} for *Carex Curvula*, a typical alpine grass (Körner, 2013). Biomass consumed by livestock and for energy (e.g. cooking) by the local rural population is given by equations (6) and (7) below.

$$\boldsymbol{nppC_{pop}} = (\boldsymbol{popHa} \times \boldsymbol{areaHa}) \times (\mathbf{1} \times \mathbf{365}) \div \mathbf{1000} \times \mathbf{0.083} \qquad \textbf{(6)}$$

$$\boldsymbol{nppC_{liv}} = \frac{(\boldsymbol{TLUha} \times \boldsymbol{areaHa} \times \mathbf{4.6})}{\mathbf{12}} \qquad \textbf{(7)}$$

Where

nppC(pop) is the biomass consumed by the local rural population within the ecoregion,

nppC(liv) is the biomass consumed by livestock for any given month, *areaHa* is the size of the concerned ecoregion,

TLUha corresponds to Tropical Livestock Units (TLU) equivalent per hectare.

Energy consumed (i.e. fuelwood extraction) is assumed to equal 1kg of fuelwood per person per day (.03 tonnes/person per month) (Lambin, 1988; Stephenne and Lambin, 2001). TLU is a commonly used conventional stock unit, with one TLU equal to either one horse, one cattle, five donkeys, 10 sheep or 10 goats (Boudet, 1975; Pieri, 1989). Based on average dietary requirements, we assumed that one TLU consumes 4.6 tonnes of dry biomass per year (0.38 tonnes/TLU per month) (Behnke and Scoones, 1993; de Leeuw and Tothill, 1993; Le Houérou and Hoste, 1977).

NPP lost to harvesting is governed by the Crop Harvest Frequency (CHF). CHF was based on national averages derived from Ray and Foley (2013). In the model we assume that when a crop harvest event takes place, all accumulated NPP (and carbon) for the proportion of the ecoregion under cropland land use is lost. Ray and Foley (2013) suggest that CHF varies substantially around the world, and in many countries can occur well below one cycle per year (CHF<1.0), and sometimes just once in two years (CHF<0,5). CHF is highly sensitive to temperature variations (common in mountains), thus we adopted the assumption that if monthly temperature fell below 10 degrees Celsius, CHF would be set to the minimum value of either the national average or '1'. The given formula for NPP lost to CHF is given below.

$$\boldsymbol{nppC_{hvs}} = \boldsymbol{NPP_{ACC}} \times \boldsymbol{CHF} \times \boldsymbol{Crop_{frac}} \quad \textbf{(8)}$$

$$\boldsymbol{T_{mth}} < \mathbf{10}℃,\ \boldsymbol{CHF} = \mathbf{min}(\mathbf{1}, \boldsymbol{CHF_{nat}}) \quad \textbf{(9)}$$

$$\boldsymbol{T_{mth}} \geq \mathbf{10}℃, \boldsymbol{CHF} = \boldsymbol{CHF_{nat}}) \quad \textbf{(10)}$$

Where

NPP_{ACC} is accumulated NPP since model start,

$Crop_{frac}$ is the percentage of cropland for the ecoregion,

T_{mth} is the temperature for a given month in the model, *CHF* is Crop Harvest Frequency (e.g. '1.0' is once per year, '2.0' is twice per year, '0.5' is every two years),

CHF_{nat} is the national CHF average.

Biomass C submodel

Biomass C is given by the following formula.

$$\varphi\ BiomassC_{MGS} < BiomassC_{lim}\ \vartheta\ BiomassC_{ha} = BiomassC_{AG} + Biomass_{BG}$$

$$BiomassC_{AG} = (NPP_{MGS} \times 0.6 \times 0.5 \times areaHa) \quad \textbf{(11)}$$

$$BiomassC_{BG} = (NPP_{MGS} \times 0.4 \times 0.5 \times areaHa) \quad \textbf{(12)}$$

Where

$BiomassC_{AG}$ is net accumulation (or loss) of aboveground biomass C, *$BiomassC_{BG}$* is net accumulation of belowground biomass C (tC) for the given month and ecoregion.

NPP is multiplied by 0.5 to convert it to dry biomass C, which is then apportioned to aboveground (leaf) and belowground (root) based on a ratio of 3:2 for grasslands as described by Grace *et al* (2006).

Soil loss (USLE)

USLE is given by the following equation.

$$A = R \times K \times LS \times CP \times P$$

Where

A is annual soil loss (t ha^{-1} y^{-1}),

R is rainfall erositivity (MJ mm h^{-1} ha^{-1} y^{-1}),

K is soil erodibility (t ha h MJ^{-1} mm^{-1}),

LS is the topographic factor,

CP is the land cover protection factor,

P is the erosion control factor.

Each of the calculation components of RUSLE is detailed below. Rainfall erositivity (R) is first calculated as follows:

$$\boldsymbol{R} = (\mathbf{4.17} \times \boldsymbol{MFI} - \mathbf{152}) \times \mathbf{17.02} \qquad \textbf{(13)}$$

$$\boldsymbol{MFI} = \frac{\sum \boldsymbol{\rho}^{2}}{\boldsymbol{P}} \qquad \textbf{(14)}$$

Where

R is rainfall erositivity (t ha^{-1} y^{-1}),

MFI is the Modified Fournier Index,

ρ^2 is monthly rainfall,

P is annual rainfall for the respective ecoregion.

Rainfall erositivity takes accounts for rainfall intensity and the kinetic energy of rain drops. The greater intensity and duration of a rain storm, the higher the erosion potential (Arnoldus, 1977; OMAFRA, 2015). The function is calculated using monthly WorldClim (Hijmans *et al*, 2005) spatial data for each respective region (Table 1).

Soil erodibility (K) is then calculated as follows:

$$\boldsymbol{K_{mth}} = \frac{\mathbf{2.1} \times \mathbf{10^{-4}} \times (\mathbf{12} - \boldsymbol{OM}) \times \boldsymbol{M}^{\mathbf{1.14}} + \mathbf{3.25} \times (\boldsymbol{S} - \mathbf{2}) + \mathbf{2.5}(\boldsymbol{P} - \mathbf{3})}{\mathbf{7.59} \times \mathbf{100}} \times \mathbf{0.083} \times \boldsymbol{areaHa} \qquad \textbf{(15)}$$

Where

K_{mth} is soil erodibility (t ha h MJ^{-1} mm^{-1}) for the given month for the ecoregion concerned,

OM is organic matter (%),

M is given by (% silt + % very fine sand) x (100 - % clay),

S is soil structure,

P is soil permeability.

Soil erodibility provides an indicative measure of the potential for soil particles to become detached and transported by rainfall and runoff (OMAFRA, 2015). Soil erodibility is highly impacted by organic matter (OM), soil structure and permeability (Bonifacio, 2013; FAO, 1978). OM values were derived for each ecoregion from the HWSD (Table 5). Soil structure and permeability were also assigned a value (between 1 and 4, 1 and 6 respectively) based on the dominant FAO soil type for each ecoregion (Table 12), which is also spatially derived from the HWSD.

The length-slope (LS) factor is then calculated. L is first calculated as follows:

$$\boldsymbol{L = \left(\frac{\lambda}{22.1}\right)^{m}} \qquad \textbf{(16)}$$

Where

L is the slope Length factor,

λ is the horizontal plot length,

m is a variable exponent calculated from the ratio of rill-to-interrill erosion which varies between 0.5 (slopes ≥ 5%) and 0.2 (slopes $<$ 1%).

S is then calculated as follows:

$$\boldsymbol{Slope_{grad} \leq 0.09,\ S = 10.8\sin\theta + 0.03} \qquad \textbf{(17)}$$

$$\boldsymbol{Slope_{grad} > 0.09,\ S = 16.8\sin\theta - 0.50} \qquad \textbf{(18)}$$

Where

$Slope_{grad}$ is the average slope gradient for the ecoregion,

S is the Slope factor,

θ is the average slope angle.

Length (L) and Slope (S) factors represent the influence of slope length and slope steepness on erosion rates. Together, *LS* are commonly referred to as the topographic factor (FAO; 1978; OMAFRA, 2015). Equations 14 and 15 were undertaken for each ecoregion using *ArcGIS* in accordance with the procedure and functions described by Pelton *et al* (2014). Broadly, this included: creating a depressionless Global Digital Elevation Model (GDEM); clipping the GDEM using the 'flow direction' tool against the geospatially referenced boundaries of all ecoregions; calculating flow accumulation using the *ArcGIS* 'flow accumulation' tool; and finally, calculating average slope for the ecoregion using the 'slope' tool.

Once R,K and LS values are calculated, USLE requires two final values to be factored in. First, the land cover protection (CP) factor, which provides a simple mathematical relationship of the impact of rainfall erosion under a particular cropping system versus the impact of erosion on bare ground (FAO, 1978). The factor represents the effectiveness of different land cover types in protecting soil from erosion (OMAFRA, 2015). Values range from 1 to 0, with 1 signifying the highest impact and lower numbers (for example 0.01) being observed for dense canopied forests. Here we adopt a CP factor value of 0.024 for natural MGS ecosystems, as described by Herbert *et al* (2008). Given that each ecoregion may also contain cropland and pasture, we also adopt CP factor values of 0.20 and 0.003 respectively, applying each as a weighting against the relative proportion of land use which makes up each ecoregion e.g. 0.024 multiplied by 0.4 if the ecoregion contains 40% natural ecosystems, and 0.20 multiplied by 0.6 if the remainder of the ecosystem (60%) is cropland. Second, the erosion control factor (P), which is sometimes also called the 'support practice factor', represents the effect that land management practices (e.g. straight-row farming, cross-slope cultivation, contour farming) have on minimising erosion. Here, due to the lack of spatially referenced land management data, we adopted a P factor value of 0.12 for all MGS ecoregions as described by Hebert *et al* (2008) for alpine vegetation, cultivation, grassland and shrubland.

Exceeding an ecoregion's carrying capacity, that is a situation where stock numbers cannot be supported *without* causing rangeland degradation, serves to heighten erosion and soil loss rates (Boudet, 1975; Stéphenne and Lambin, 2001). Though estimating rangeland degradation in this respect is complicated, our model attempts to account for whether or not each MGS ecoregion is being overgrazed, as per equation 19. This is important, as pastoralism is one of the key economic activities found in MGS worldwide (Körner *et al*, 2005). If the ecoregion is deemed to be overgrazed (OverG value of ≥ 0) then the USLE land cover protection factor is set to bare ground (0.0045), reflecting the absence of MGS vegetation to protect the soil from erosion. Carrying capacity (*CC*) is set to 1.25ha/TLU (Stéphenne and Lambin, 2001).

$$\boldsymbol{OverG} = \frac{\boldsymbol{TLU \times areaHa\left({}^{areaHa}/_{CC}\right)}}{\boldsymbol{{}^{areaHa}/_{CC}}} \tag{19}$$

Where

OverG is dimensionless and indicator of whether (≥ 0) or not the given ecoregion is being overgrazed (< 0), CC is the carrying capacity for the given ecoregion.

Soil C submodel

Net accumulations (or loss) of SOC is thus given by the following formula.

$$\boldsymbol{\varphi\ SOC_{MGS}\ < SOC_{lim}\ \vartheta\ SOC_{MGS} =\ SOC_{MGS}}$$

$$\boldsymbol{SOC_{MGS}} = \left(\left(\boldsymbol{A} \times \frac{\boldsymbol{BD}}{\boldsymbol{100}} \times \frac{\boldsymbol{OC}}{\boldsymbol{100}}\right) + (\boldsymbol{Biomass_{MGS}} \times \boldsymbol{0.007})\right) \times \boldsymbol{areaHa} \qquad \textbf{(20)}$$

Where

SOC_{MGS} is net accumulation (or loss) of SOC for the given month and ecoregion (tC),

A is the annual soil loss (t ha^{-1} y^{-1}),

BD is soil bulk density (%),

OC is soil organic content (%).

Estimates for bulk density and organic content were spatially derived from the HWSD. We assumed that for each tonne of C accumulated in biomass, 0.007 would also accumulate in the soil (Ernst-Detlef *et al*, 2001). To determine gains or losses, we also assumed that 100% of the SOC would be atomised and emitted to the atmosphere as CO_2 (using a conversion factor of 3.67). Though 100% may be considered to be an extreme assumption (Pendleton *et al*, 2012), it should be acknowledged that given the inherently steep slopes found in many MGS areas, the probability that disturbed SOC material is reburied is considered to be relatively low.

5.16.2 Summary of model input parameters and dynamic variables

Table 10. Model input parameters

Ecoregion Input Parameter Description	Notation	Dimension
MGS Ecoregion Name e.g. "Southern Andean Steppe"	EcoName	dimensionless
Country in which Ecoregion is located e.g. "Australia"	CountryName	dimensionless
Land area	areaHa	ha
Initial dominant land use, as at 1 Jan 2000 e.g. "Natural"	initialLUState	dimensionless
Percent of land area in natural state	percNatural	%
Percent of land area under cropping	percCrop	%
Percent of land area under grazing (pasture)	percPast	%
Dominant protected area status e.g. "Protected"	protectedAreaStatus	dimensionless
Avg. tonnes soil C per hectare	soiltCha	$tCha^{-1}$
Max. tonnes SOC per hectare (steady-state limit)	soilChaLimit	$tCha^{-1}$
Avg. tonnes biomass C per hectare	biomasstCha	$tCha^{-1}$
Max. tonnes biomass C per hectare ('steady-state limit')	biomassChaLimit	$tCha^{-1}$
Avg. soil bulk density	soilBulkDen	%
Dominant soil type e.g. "Histosols"	soilType	dimensionless
Avg. soil organic content	soilOC	%
Avg. rainfall erosivity (USLE 'R' factor)	R	MJ mm h^{-1} ha^{-1} y^{-1}
Avg. soil erodibility (USLE 'K' factor)	erosion	t ha h MJ^{-1} mm^{-1}
Avg. soil permeability (USLE 'K' factor)	erosion	dimensionless
Avg. soil structure (USLE K factor)	erosionS	dimensionless
Avg. protection (landcover) factor (USLE 'P' factor)	erosionProFac	dimensionless
Crop harvest frequency	CHF	dimensionless
Avg. Tropical Lifestock Units (TLU) per hectare	tluHa	TLU
Avg. number of people per hectare (rural population)	popHa	people
Road within 1km of Ecoregion boundary,e.g."Yes"	RoadWithin1km	text
Avg. monthly temperature	tempJan, tempFeb, …	degrees C
Avg. monthly rainfall	rainJan, rainFeb, …	millimetres
Avg. snowfall coverage	snowJan, snowFeb…	%
Avg. annual rainfall	rainAnn	millimetres
Modified Fournier Index	MFI	millimetres
Topographic factor (USLE 'LS' factor)	LS	dimensionless
Avg. annual change in crop land for country	CropTrend	dimensionless
Avg. annual change in grazing land for country	PastTrend	dimensionless
Avg. annual change in protected area for country	ProTrend	dimensionless
Avg. annual change in TLU for country	TLUtrend	dimensionless
Avg. annual change in rural population for country	POPtrend	dimensionless

Table 11. Model dynamic variables

Variable Description	Notation	Dimension
Erosion cover protection factor (USLE 'C' factor)	erosionCovFac	dimensionless
Crop Harvest Frequency for relevant country	CHFminTemp	degrees C
Mean min monthly temperature for year	minTemp	degrees C
Mean monthly temperature at current model time	currentTemp	degrees C
Mean monthly rainfall at current model time	currentRain	millimetres
Mean monthly snow coverage at current model time	curSnowFrac	%
Mean monthly snowless coverage at current model time	snowlessFrac	%
Mean monthly rainfall at current model time	NPPrain	mm
Mean monthly temperature at current model time	NPPtemp	degrees C
Mean monthly NPP lost to grazing	NPPlossgrazing	gDM
Mean monthly NPP lost to energy use by local populations	NPPlossEnergy	gDM
Mean monthly NPP lost to harvesting	NPPlossHarvest	gDM
Net Primary Productivity	NPP	gDM
CO_2 response coefficient	CO2respCoeff	dimensionless
CO_2 partial pressure	CO2partialPress	mmHg
CO_2 compensation point	CO2compPoint	dimensionless
RubiscoS (Ribulose-1,5-bisphosphate carboxylase-oxygenase	RubiscoS	mmol mol^{-1}
Mean monthly Soil Organic Carbon loss	SoilCloss	tC
Mean monthly Soil Organic Carbon gain	SoilCgain	tC
Annual Carbon to NPP ratio	annCtoNPPratio	dimensionless
Overgrazing indicator	overG	dimensionless
Accumulated biomass C at current model time	AccBiomassC	tC
Accumulated soil C at current model time	AccSoilC	tC
Maximum biomass C that can accumulate for ecoregion	maxCbioLim	$tCha^{-1}$
Maximum soil C that can accumulate for ecoregion	maxCsoilLim	$tCha^{-1}$
Total ecosystem carbon (biomass C + SOC)	EcoTotalC	tC
Total ecosystem carbon (biomass C + SOC) per hectare	EcoTotalCha	$tCha^{-1}$
Total CO_2 sequestered by ecoregion	EcoTotalCO2e	CO_2
Change in CO_2 sequestered compared to model start time	changeEcoTotalCO2e	%

5.16.3 Soil permeability and structure

Table 12. USLE Soil texture and permeability factors

FAO Soil Type	Permeability	Structure
Leptosols	1	3
Regosols	2	2
Solonetz	5	4
Anthrosols	2	3
Gypsisols	2	3
Fluvisols	5	4
Calcisols	2	2
Chernozems	2	3
Kastanozems	5	3
Gleysols	5	2
Cambisols	3	3
Phaeozems	3	2
Solonchaks	4	1
Luvisols	2	3
Podzols	5	4
Podzoluvisols	5	4
Histosols	6	4
Arenosols	1	2
Planosols	4	3
Alisols	5	4
Nitisols	2	3
Vertisols	4	4
Andosols	2	3
Acrisols	3	3
Ferralsols	4	4
Lixisols	2	2

Values adopted using the World Reference Base for soil resources (IUSS Working Group, 2006).

Chapter 6. *Using carbon finance to support climate policy objectives in high mountain ecosystems*

Ward, A. Dargusch. P. Grussu, G. and Romeo, R. 2015. Using carbon finance to support climate policy objectives in high mountain ecosystems. ***Climate Policy,*** **6:1-20.**

6.1 Chapter Summary

This chapter investigates the stressors, challenges, and priorities related to the NRM of carbon stocks in mountain grasslands and shrublands; why carbon markets and climate finance have not yet been utilized in this context; and, what is required to position mountain-based NRM activities as eligible for carbon finance incentives. Using surveys and interviews triangulated with a systematic literature review, this chapter concludes that carbon finance incentives are not well understood, both amongst mountain-focused experts and in the literature. This chapter also highlights that the required technical methodologies, policy frameworks, and data to be largely undeveloped. This chapter concludes by article proposing a top-down conceptual policy framework that can be used to develop key ‘enabling factors’ with the view of extending the eligibility of carbon markets and climate finance to NRM activities undertaken in mountain grasslands and shrublands in the same way that has been afforded to other ecosystems.

6.2 Introduction

Mountain grassland and shrublands provide unique ecological functions and economic benefits to society. For instance, the high-altitude rangelands of the Hindu Kush Himalayan Region support major livestock production in Asia and in turn provide livelihood opportunities for impoverished local herdsman (Gao, Li, Xu, Wan, and Jiangcun, 2014), and the Páramo ecosystems of the Andes Mountains supply and regulate the hydrological flows critical for irrigation, industrial processes, power generation, and drinking water in the expanding regional and urban centres of Peru, Venezuela, Columbia, and Ecuador (Buytaert *et al*, 2006). Ward, Dargusch, Thomas, Lui, and Fulton (2014) estimate that montane, subalpine, and alpine grasslands and shrublands globally cover 9.38 million km^2, or about 6% of the Earth’s terrestrial landmass. Without these ecosystems, the livelihoods and survival of millions of people (many of whom are very poor) would be put at great risk (Körner *et al*, 2005; Mountain Partnership, 2014). MGS are also important from a climate policy perspective, storing between 60.5 Pg and 82.8 Pg carbon worldwide (Ward *et al*, 2014), which is approximately the same amount as is stored in tropical peatlands (Page *et al*, 2011).

However, these ecosystems are being degraded due to the influence of multiple anthropogenic stressors, ranging from the intensification of agriculture and mining to the growth in tourism, urbanization, exotic species, and – most notably – climate change (Ariza, Maselli and Kohler, 2013; Buytaert *et al*, 2006; Körner, 2004; Marquis, Baldassarri, Hofer, Romeo and Wolter, 2012). The damage leads to serious socio-economic consequences, including desertification, increased poverty, and more frequent conflict (Ariza *et al*, 2013; Körner *et al*, 2005). This degradation is also having a significant impact on carbon stocks. As is already the case for forests, lowland grasslands, and marine ecosystems (e.g. mangroves), reducing vegetation and carbon loss in mountain grassland and shrubland ecosystems must also feature in international climate policy discourse.

The United Nation's General Assembly recently reaffirmed (United Nations, 2013) its view that the 'management of mountain resources and socio-economic development of the people' will best be achieved through better natural resource management (NRM) (UNCED, 1992, Ch. 13). However, a number of constraints exist. In particular, global socio-economic changes, coupled with disproportionate levels of poverty, have limited the capacity of developing-country governments to address unsustainable land management practices in mountain regions (Körner *et al*, 2005; Wehrli, 2014).

Industrialized countries also face challenges, such as the reintroduction of cattle into the Alpine National Park in Australia, where there is the potential to exacerbate existing and underfunded issues such as the spread of weeds, erosion, and damage to vegetation caused by feral horses (Worboys and Good, 2011). Although long-term financing is not the only requirement for a successful NRM project, it remains a critical component. Since the 1990s, NRM finance has declined in relative terms, meaning that NRM practitioners have 'had to be more innovative and systematic in their search for financing options' (WWF-MPO, 2003, p.7).

This study focuses on five categories of MGS focused NRM activities that could potentially contribute to broader climate policy outcomes: (1) adaptive grazing management (e.g. rotation grazing and destocking); (2) sustainable cropping (e.g. fertilizer management); (3) ecosystem preservation (e.g. fencing-off of sensitive areas, environmental buffer zones, fire management, avoided clearance); (4) ecosystem restoration (e.g. revegetation, exotic plant and animal control); and (5) engineered soil conservation measures (e.g. terracing). These five were chosen due to their common ability to influence the amount of carbon stored in such ecosystems and to provide other environmental, economic, and social benefits.

In recent years there has been a marked increase in finance flowing to projects that reduce GHG emissions. In 2013, global climate finance (non-market incentives such as low-interest loans and private equity) was estimated to be worth US$331 billion (Buchner *et al*, 2014). In the same year, an estimated $31 billion in carbon offset units were traded on international carbon markets (Peters-Stanley and Gonzalez, 2014; World Bank, 2014b). Although the majority of climate finance and carbon market transactions have centred on renewable energy, energy efficiency, and industrial efficiency projects, there has been a noticeable push by conservationists and natural resource managers to tap into these funds to support biological and land-based carbon offset projects (Booker, Huntsinger, Bartolome, Sayre, and Stewart, 2013; Buchner *et al*, 2014; Ullman, Bilbao-Bastida and Grimsditch, 2013). Notably, Nationally Appropriate Mitigation Actions (NAMAs) – strategic plans focused on large-scale emissions reductions in developing countries – are also growing in importance as a precursor to accessing climate finance, including from the UN's Green Climate Fund (Wurtenberger, 2013).

The objectives of this study are: (1) to revisit and consider the stressors, challenges, and priorities related to the NRM of MGS, as perceived by experts working in the area; (2) to develop an understanding as to why carbon markets and climate finance have not yet been utilized in this context; and (3) to provide a preliminary assessment of the barriers, opportunities, and 'enabling factors' that will probably be required to position mountain-based NRM activities as eligible for climate finance and carbon markets. This is the first study to explicitly highlight the important role that MGS might play in international climate policy. It is also the first to propose how international climate policy mechanisms might support much needed NRM activities in these areas, with the aim of setting the scene for future research and discussion.

6.3 Materials and methods

To meet these objectives a survey of experts was undertaken to obtain an insight into the priority NRM stressors for MGS, and to see how they judged their own understanding of carbon markets and climate finance. Follow-up interviews were used to gather more detailed data. This qualitative survey method is indicative and not prescriptive, although it has credibility due to the expert input and triangulation methods used, as described below. The survey (*Chapter 6.7*, Table 15) was designed to identify the issues, priorities, and challenges facing natural resource managers and policy makers who are responsible for the NRM of MGS, and to provide an understanding of the extent to which these experts have considered using carbon markets and climate finance to support NRM. To achieve the first aim, the survey included the following types of qualitative question: what was the perceived trend in the health of these ecosystems; how experts rated different anthropogenic stressors; and the types of priority actions that are currently being used to manage these stressors, e.g. slope stabilization through revegetation. In achieving the second aim, the types of questions asked included: whether experts had considered the role of carbon markets and climate finance in supporting NRM activities; if experts understood the differences between carbon market (market) and climate finance (nonmarket) incentives, both before and after completing the survey; and what the barriers to developing carbon offset projects might be, e.g. technical capacity.

The process for selecting an 'expert panel' of survey respondents was non-probabilistic and purposive, an approach deemed suitable for research where the objective is to understanding complex social phenomena (Marshall, 1996; Small, 2009) and where the sampling size is small but targeted (Gideon, 2012). Potential experts were initially identified through the Food and Agriculture Organisation (FAO) Mountain Partnership's network of practitioners, as successful sampling requires 'assembling a sample of persons with demonstrable expertise in given area' (Gideon, 2012, p. 400). Experts were selected for the survey if publically available information confirmed they met the following five criteria: (1) currently working in a senior position (within a relevant government, non-government, or private-sector organization), or had done so up until five years ago; (2) had at least five years' experience working exclusively on mountain resource management issues; (3) had published a relevant peer-reviewed journal article and/or policy paper for a reputable organization; (4) had obtained a qualification that was relevant to policy and/or NRM implementation in MGS; and, (5) when considered as part of the expert panel, represented a diverse geographical and/or policy perspective. The final panel consisted of 20 experts from a range of organizations, including the International Centre for Integrated Mountain Development (ICIMOD), the Consortium for Sustainable Development of the Andean Ecoregion (CONDESAN), the Peruvian Government, the Government of Bhutan, the Government of Afghanistan, the New Zealand Government, and the United Nations Development Programme. Their position titles included 'Professor Emeritus', 'Senior Policy Officer', 'Regional Coordinator', 'Chief Scientist', and 'Professor of Environmental Law'. More detail is presented in Figure 21. Experts came from either a policy formation (POL) or on-the-ground NRM background, some working at the local scale and some at a larger regional scale (as represented by the approximate 'hectares' shown for each expert). The authors deemed it necessary to cover a broad range of perspectives, given that successful utilization of carbon finance in the NRM context will require effective interaction between policy making and on-the-ground NRM implementation (Campbell and Sayer, 2003). The survey was voluntary, anonymous, and confidential, and was conducted using an online questionnaire tool.

A number of survey questions (as indicated in Table 15) asked experts to either rank or rate variables in the system. Rating averages (e.g. Question 9) were obtained on a weighted basis and according to the following equation:

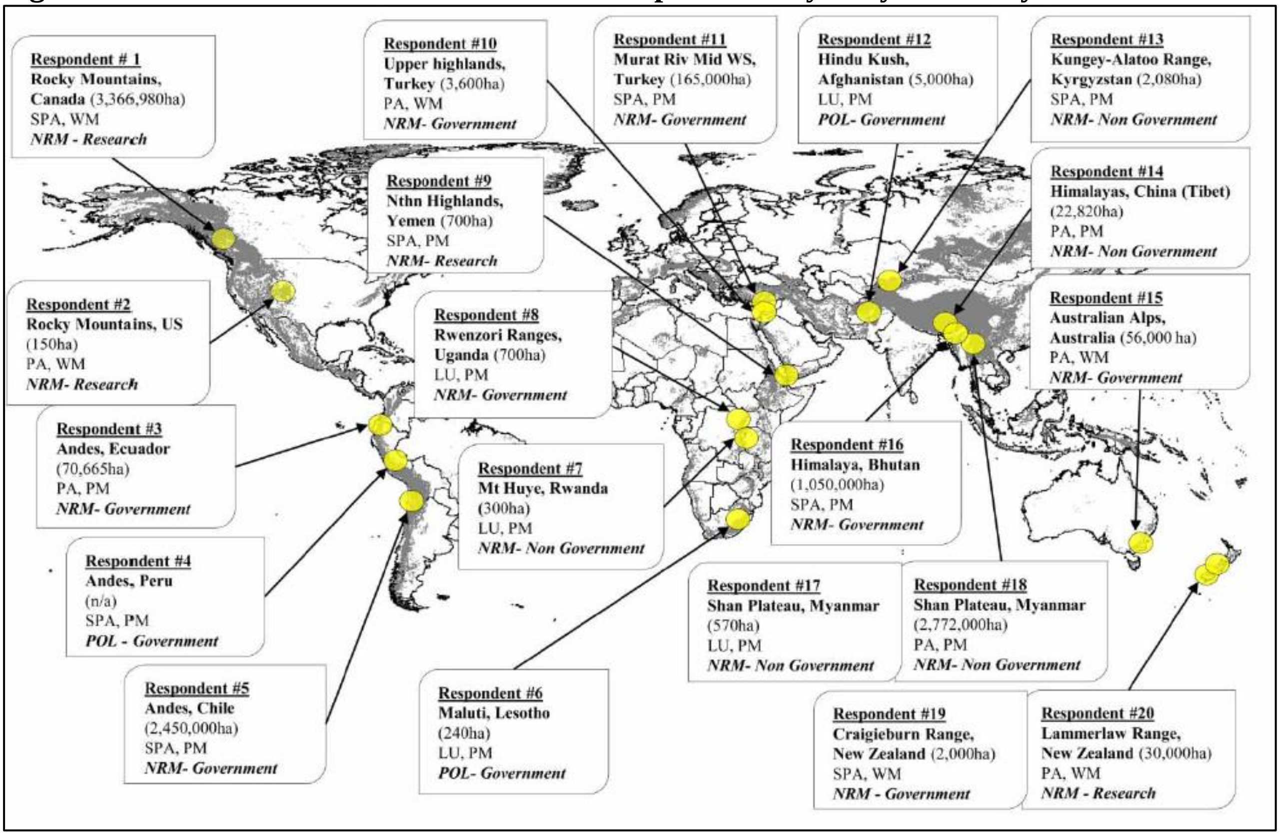
$$\frac{X_1W_1 + X_2W_2 + X_3W_3 \ldots X_nW_n}{\text{Total}}$$

where X is the weight of the answer choice and W is the response count for the answer choice. Ranking averages (e.g. Question 7) were obtained according to:

$$\frac{X_1W_1 + X_2W_2 + X_3W_3 \ldots X_nW_n}{\text{Total}}$$

where X is the weight of the ranked position and W is the response count for the answer choice. The results were methodologically triangulated through the use of interviews and a literature review to increase reliability (Gideon, 2012). Using theoretical sampling (Gideon, 2012), eight of the most experienced experts surveyed were selected and interviewed at length. These experts were asked to elaborate on their answers to the survey (the interviews were semi-structured and face-to-face) and also invited to provide additional comments. The interview process was used to discuss, verify, and better explain the survey results, such as the extent to which experts understood the differences between carbon markets and climate finance at the beginning and at the end of the survey.

Figure 21. Distribution and characteristics of experts surveyed by this study



Notes: This figure shows: (1) the geographical area (mountain range, country) where each expert works; (2) the approximate size (in hectares) of the mountain grassland and shrubland area in which they work (as an approximate and relative measure of working scale among the experts surveyed); (3) whether they work predominantly at a natural resource management (NRM) or policy (POL) level; and (4) whether they are from a government, non-government, or research organization. Grey areas show the distribution of the world's mountain ranges. **Source:** Ward *et al*, 2015.

Finally, the objectives of the literature review were (1) to verify the survey results and interview discussion, (2) to understand the extent to which carbon markets and climate funds are currently being applied in the context of MGS; and (3) to highlight any knowledge gaps. The literature review was systematic and quantitative (Pickering and Byrne, 2014) and included examining secondary data sources (publically available databases and registries, government and non-government reports, and peer-reviewed journal articles) covering (1) sustainable mountain development strategies; (2) climate policy documents, existing and pending carbon offset methodologies and projects developed under the Clean Development Mechanism (CDM), Verified Carbon Standard (VCS), Gold Standard (GS), Climate Action Reserve (CAR), Plan Vivo and the Panda Standard; and (3) international climate change mitigation funds that were undersubscribed as of January 2015, and that could be used in supporting NRM activities in MGS. Funds focused only on climate change adaptation were not reviewed. Particular attention was paid to information that addressed the key issues identified in the survey results and interviews. The literature review was conducted using online databases including Summon, Science Direct, the Web of Science, EBSCO, and ProQuest. The primary keyword search terms consisted of a combination of ‘mountain’, ‘montane’, ‘alpine’, ‘tundra’, ‘subalpine’, ‘grasslands’, ‘shrublands’, ‘carbon’, ‘markets’, ‘funding’, ‘natural resource management’, ‘climate’, and ‘policy’. Secondary search terms included ‘climate change’, ‘biomass’, ‘soil’, ‘ecosystem services’, ‘environmental services’, and ‘ecological services’. Although the information generated by this study was sourced from the most recent literature and reputable, qualified and experienced professionals, the findings must be considered indicative. All in all, this study reviewed around 2000 peer-reviewed journal articles, policy, and technical documents.

6.4 Results

6.4.1 Panel of expert survey

Ecosystem characterization, perceived level of degradation, and protection status

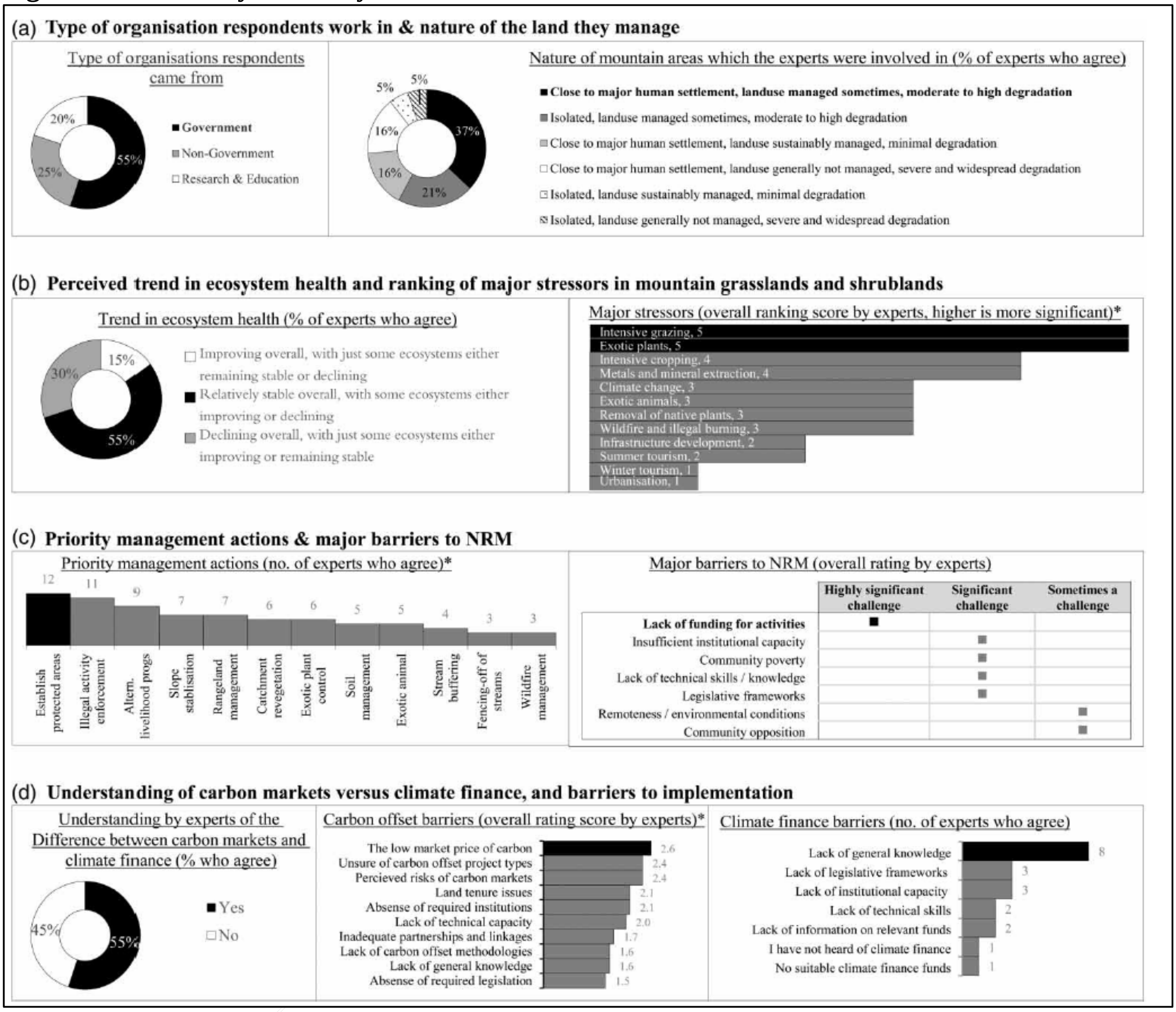
The majority of experts judged the mountain grassland and shrubland areas they managed as ‘close to major settlement, land use managed sometimes, and moderate to high degradation’ (Figure 22a). Only the experts from Australia and Yemen classified land as suffering from ‘minimal degradation’, with the other seven classifying land as suffering from either ‘moderate to high’ or ‘severe and widespread’ degradation.

The majority of areas worked on by the experts were judged to be ‘poorly managed’ and ‘largely unprotected’, with ‘some’ protected areas (Figure 21). Experts from industrialized nations were most likely to report their area of interest as either ‘a protected area that is well managed’ or ‘an area with some protected areas that is well managed’. The most common level of protection reported by experts from developing nations was ‘areas with some protected areas that are poorly managed’.

Ecosystem stressors

Of the experts, 55% classified the trend in ecosystem health within their areas of interest as ‘relatively stable overall’, with 30% reporting it as ‘declining overall’ and the remaining 15% as ‘improving overall’ (Figure 22b). When taking into account the level of significance, the conversion of land to intensive grazing and the increasing prevalence of exotic plants were rated as the most significant stressors for these ecosystems. It is also worth noting here the differences in responses from industrialized and developing nations. Experts from industrialized nations tended to rate climate change and the increasing prevalence of exotic plant and animals as the most significant stressors, while those from developing nations generally rated the conversion of land to intensive agriculture (cropping and grazing) as the most significant factor affecting ecosystem health, followed by the increasing incidence of minerals and metals extraction.

Figure 22. Summary of survey results



Notes: *Based on their experience and judgement, experts were asked to choose one or more options, or rank or rate all options based on the level of significance or level of priority ('high' through to 'not applicable'). ^ The numbers shown here (and in the text) represent the ranking average or rating average based on the selection made by all experts. The higher the number the better. **Source:** Ward *et al*, 2015.

Priority NRM actions and barriers

Experts rated the establishment of protected areas, policing and preventing illegal activities, establishment of alternative livelihood programmes, slope stabilization, rangeland management, and catchment revegetation as priority direct and indirect NRM actions (Figure 22c). These priorities are compatible with the five NRM carbon mitigation activities identified above, particularly those focused on ecosystem preservation. The priorities differ between developing and industrialized countries. In Australia and New Zealand, NRM priorities are the control of exotic plants and animals. In the US and Canada it is wildfire management. In most developing nations the NRM priorities are establishing protected areas, slope stabilization, followed by rangeland management and enforcement of illegal activities.

In understanding the barriers to implementing NRM actions, the experts noted overwhelmingly that a lack of funding was identified as a 'highly significant challenge' (Figure 22c). Insufficient institutional capacity, community poverty, lack of technical skills and knowledge, and a lack of legislative frameworks were considered by experts as 'significant challenges'. Experts indicated that the most significant challenges for industrialized nations were lack of funding and community opposition. For developing nations, the most significant challenges were access to funding, a lack of technical skills, institutional capacity and networks, and community poverty. These findings were backed up by the interviews and literature review.

Understanding of carbon markets and climate finance

Half of the experts surveyed indicated that they did not understand the difference between carbon markets and climate finance (Figure 22d). The significance of this is discussed later. Overall, the experts ranked the low carbon market price as the greatest barrier to participating in carbon markets. Other highly ranked barriers included the perceived risks associated with carbon markets, issues associated with ascertaining land tenure, a lack of technical skills, inadequate partnerships, and the absence of suitable carbon offset methodologies. Experts made the following comments during the interview stage: 'the opportunities have not received enough attention', 'it is often difficult to obtain data', and 'there is a need for a major organization to lead and coordinate the whole project where small partner organizations can take part in'. For climate finance, the perceived barriers were very different. Experts indicated the most significant barrier to be a general lack of knowledge around climate finance. Experts commented in the interviews that 'government leadership was missing in this area', and that 'politicians needed to be better informed of the opportunities associated with climate finance'.

Finally, experts were asked whether they understood how NAMAs might support mountain NRM actions. According to GIZ (2012, p. 7) NAMAs are 'an instrument within the global climate change architecture for implementing the necessary mitigation activities to keep the mean global temperature rise due to anthropogenic emissions of greenhouse gases below 2 degrees C’. NAMAs range from project-based mitigation actions (e.g. soil management) to nationally strategic objectives (e.g. greater energy efficiency across a specific economic sector). NAMAs are developed according to country-specific sustainable development priorities (e.g. peatland management in Indonesia) and are likely to form the basis for attracting climate mitigation finance in the future, especially from large multilateral financing mechanisms such as the Green Climate Fund. In general, there seemed to be a moderate understanding of NAMAs, with four out of 20 experts signalling that they were currently investigating the role they could play, and nine indicating that they understood what NAMAs were but had not yet investigated their role in mountain NRM. Three experts indicated that NAMAs would not be applicable, while the last four signalled that they did not understand what NAMAs were and/or how they might apply to NRM in MGS.

6.4.2 Carbon offset methodologies and projects in mountain grasslands and shrublands

The literature review of various carbon offset project registries (CDM, VCS and CAR) revealed that no carbon offset projects (e.g. revegetation, soil management) have been developed in mountain grassland or shrubland areas (Table 13). This is underscored, at least until recently, by a lack of the appropriate methodologies required to develop NRM-focused carbon offset projects under the major carbon offset standards. Only three suitable methodologies have been approved under the VCS, and a number of other potentially suitable methodologies being developed under the GS, Panda Standard, CAR, and the emerging Chinese Certified Emissions Reduction (CCER) scheme.

Table 13. Mountain grasslands and shrublands: stocktake of relevant carbon offsets methodologies and projects (approved and under development)

	Suitable methodologies/protocols		Projects in MG&S	
Standard	Approved (NRM type, methodology no.)	In development (NRM type)	Approved (no. of projects)	In development (no. of projects)
Verified Carbon Standard (VCS)	AGM, SC, ER, EP, ESC (VM0017, VM0021, VM0026)	SC, EP		–
China Certified Emissions Reduction (CCER) scheme		SC, AGM	–	–
Clean Development Mechanism (CDM)[c]	–	–	–	–
Panda Standard	–	AGM, SC, EP, ER (SC, AGM)[a]	–	–
Gold Standard	–	AGM, SC[a]	–	–
Climate, Community & Biodiversity Alliance (CCBA)[b]	N/A	N/A	–	–
Plan Vivo[b]	–	–	–	–
Climate Action Reserve (CAR)[c]	–	EP, ER, AGM	–	–

NRM activities: Adapted Grazing Management, AGM; Sustainable Cropping, SC; Ecosystem Preservation, EP; Ecosystem Restoration, ER; Engineered Soil Conservation, ESC

Notes: This table shows the status of the following as of 5 February 2015: (1) carbon offset methodologies (approved or in development) under widely accepted standards that could most likely be applied to mountain grasslands and shrubland (MG&S) NRM activities; (2) the number of carbon offset projects (approved or in development) under widely accepted standards.

[a]Limited information available, indicative only.

[b]Methodology from other standard used/can be used.

Source: Ward *et al*, 2015.

These methodologies could potentially support adapted grazing management, sustainable cropping, ecosystem preservation, and ecosystem restoration. However, the stage of development for many remains uncertain, particularly for those methodologies being developed under the Panda Standard where the last official update was provided in 2011 (Panda Standard Association, 2011). A grassland management methodology is set to be released the CAR in mid-2015, although there are still many details to be finalized (Climate Action Reserve, 2015). There is also uncertainty as to the suitability of these methodologies for mountain grassland and shrubland landscapes rather than lowland landscapes, where there is a risk that important geomorphic differences that influence CO_2 emissions, such as erosion rates which Hurni (1999) states are five to ten times greater in mountainous areas), are not recognized. Although there are several methodologies that could currently be used to generate credible carbon offsets, no carbon offset projects have been developed in MGS ecosystems. Critically, the VCS and CAR may be applied in both industrialized and developing countries, whereas the other schemes may only be applied in developing countries. The opportunity to develop carbon offset projects in industrialized countries is thus limited to methodologies approved under the VCS and

CAR. Also, because many mountain areas are already protected, any carbon abatement would not be considered ‘additional’ under the eligibility rules for these schemes.

6.4.3 Climate finance for mountain grasslands and shrublands

There are a number of climate mitigation funding mechanisms that could be used to support NRM activities in mountain grassland and shrubland areas (Table 14). Incentive types vary from co-financing payments to grants, concessional loans, and non-financial technical assistance measures. For example, the Global Environment Facility (GEF) disbursed around US$300 million during 2010 (Buchner *et al*, 2014) in grants, while the World Bank’s innovative Green Bond Programme has raised over US$6.4 billion through the issuance of AAA rated ‘Green Bonds’ that support eligible climate mitigation and adaptation projects while providing a commercial return to investors (World Bank, 2014a). It should be noted that climate finance mechanisms generally apply to developing countries only, another limiting factor for industrialized countries.

Table 14. Mountain grasslands and shrublands: examples of relevant climate mitigation funds

Fund name	Organization	Incentive available (US$)	Incentive type/ status	Target high-mountain countries/ regions
NAMA Facility	Government of Germany	$6.8–21 million	Grants	Developing countries with NAMAs
GEF Medium & Large Project Financing	Global Environment Facility	Over $2 million (large) Up to $2 million (medium)	Co-financing, grant	All developing countries
GEF Small Grants Programme	Global Environment Facility	Up to $50,000	Grant	All developing countries
Nordic Climate Facility	Nordic Development Fund	$205,000–680,000	Co-financing, grant	Bolivia, Kenya, Kyrgyz Republic, Malawi, Mongolia, Mozambique, Nepal, Pakistan, Rwanda, Tanzania, Uganda, Zambia, Zimbabwe
BioCarbon Fund	World Bank	Varies/not specified	Technical assistance, grant, results-based	All developing countries
Carbon Partnership Facility	World Bank	Varies/not specified	Results-based	All developing countries
Green Climate Fund[a]	United Nations Framework Convention on Climate Change	Varies/not specified (total fund $100 billion annually by 2020)	Co-financing, grant, concessional loans	All developing countries
International Climate Initiative	Government of Germany	Varies/not specified (total fund $900 million, $165 million per annum)	Grant, concessional loans	All developing countries, countries in transition, newly industrialized
Enhancing Capacity for Low Emissions Development Strategies	Varies, e.g. US, Japan	Varies/not specified	Technical assistance	All developing countries
World Bank Green Bonds	World Bank	Varies/not specified	Quarterly payments	All developing countries

Notes: This table highlights examples of open climate change mitigation funds (as at 5 February 2015) that could be used to financially and/or technically support the implementation of SNRMs in high-mountain areas.
[a]Not yet operational.
Sources: UNDP, World Bank (2015); UNEP (2015); BMUB (2014).

Source: Ward *et al*, 2015.

While this list provides a snapshot of potential funds that could support the aforementioned NRM activities, studies suggest that the actual transacted value of climate finance is difficult to quantify and is likely to be much larger (Stadelmann, Axel and Timmons Roberts, 2013). For example, this list excludes financing opportunities that are privately negotiated and that are not currently open for application. As such, this list represents just a fraction of the US$331 billion per annum estimated by Buchner *et al* (2014). Nevertheless, it conservatively highlights a substantial opportunity to leverage climate finance to support NRM activities in MGS, particularly for those areas in developing countries.

6.5 Discussion

6.5.1 The NRM context

The results emphasize many of the same stressors, NRM priorities, and barriers identified by other studies (ICIMOD, 2013; Odermatt, 2004; Price, 2007; Price and Kim, 1999; Worboys and Good, 2011). The literature review revealed that many of the stressors identified by the expert panel have the potential to cause both significant and negative impacts on carbon stocks (Gao *et al*, 2014; Landcare Research, 2012; Li, Li, Singh, Rengel and Zhan, 2007; Wangchuk, Gyaltshen, Yonten, Nirola and Tshering, 2013; Ward *et al*, 2014).

The failure to address these stressors could have a significant effect on biomass and soil organic carbon stored in MGS. Depending on geographical location, slope, rainfall, vegetation, soil type and depth, mountain grassland and shrubland ecosystems are estimated to contain between 15 and 685 tonnes of carbon per hectare (t C ha^{-1}),most of which is typically contained within the soil (Ward *et al*, 2014). This carbon is largely unaccounted for in international carbon budgets (Ward *et al*, 2014), and its loss into the atmosphere would undermine the ultimate climate policy objective of the United Nations Framework Convention on Climate Change (UNFCCC), which is 'to stabilize greenhouse gas concentrations in the atmosphere at a level that would prevent and reduce dangerous human-induced interference with the climate system' (UNFCCC, 2013). This study suggests that adaptive grazing management, sustainable cropping, ecosystem preservation and restoration, and engineering soil conservation could play an important NRM role in addressing these stressors. Moreover, because these activities can achieve carbon mitigation outcomes they are likely to be eligible for climate finance and carbon markets. However, the question remains as to why this has not happened.

6.5.2 Risks, barriers, and challenges in using carbon markets and climate finance

Almost half of the experts surveyed here were not aware of the difference between climate finance and carbon markets. It is an important distinction to make. Carbon markets generally relate to the development carbon offset projects (e.g. planting trees), which create tradable credits. These credits can be sold to organizations who either have a regulatory obligation to reduce emissions (e.g. under the EU-Emissions Trading Scheme) or that want to voluntarily offset their emissions in response to stakeholder and sustainability concerns (Dargusch and Thomas, 2012). On the other hand, climate finance refers to unilateral, bilateral, multilateral, and private-sector funding schemes, which provide upfront grants, subsidies, and low-interest loans (for example) to eligible countries to develop projects or sectoral activities that reduce emissions e.g. the Green Climate Fund (Stadelmann *et al*, 2013); i.e. there is no trading.

Among the experts there was a better understanding of carbon markets than climate finance, although there was also a perception that carbon offset projects were a technically difficult and risky undertaking. As of January 2015, land-based NRM carbon offset projects represented just over 1% of all CDM projects (UNEP, 2015), none of which were undertaken in MGS. The poor representation of NRM carbon offset projects is due to numerous factors, including future carbon price uncertainty; access to capital; delayed returns on investment; high transaction costs; burden to prove financial and regulatory additionality; long-term permanence requirements of carbon stocks; risk-of-reversal associated with landholders clearing land in the future; lack of institutional and skills capacity; absence of suitable methodologies; the risk of emissions occurring in another location, i.e. carbon leakage; and unintended negative impacts such as those on local communities and biodiversity (Cacho, Lipper and Moss, 2013; Dargusch, Harrison and Thomas, 2010; Galik and Jackson, 2009; Thomas *et al*, 2013).

The confusion that carbon markets and climate finance are 'one-and-the-same' could lead to the assumption that the risks of pursuing direct climate finance are of the same magnitude as for carbon markets, and thus equally unappealing development propositions. While both funding pathways seek to support the development of projects that reduce emissions and improve sustainable development outcomes, there are distinct differences. Unlike for carbon offset projects, the use of climate finance has no requirement to sell tradable credits, and therefore no market risk. That is not to say climate finance is risk-free, as risk-of-reversal and carbon leakage remain key issues. This confusion, combined with an absence of general knowledge (as per the questionnaire responses), provide one explanation as to why climate finance was not widely considered by experts. This study reinforces two recent investigations that found climate finance to be difficult to quantify, report, and track precisely (Stadelmann *et al*, 2013; Würtenberger, 2013). This suggests that similar misperceptions may also exist for experts working in other areas.

Another barrier is the measurement and monitoring of soil organic carbon. Almost all carbon (on average 98.1 percent) in mountain grassland and shrubland ecosystems is contained within the soil (Ward *et al*, 2014). The successful application of carbon markets and climate finance depends heavily on accurate and cost-effective monitoring of changes in soil carbon stocks. Although precise soil measurement methodologies are well-established, high spatial variability presents a financial barrier to potential carbon mitigation project developers (i.e. detailed studies are cost prohibitive; Conant *et* *al*, 2011). This challenge would be especially pronounced in mountains where a wide variety of soils can often be found over a relatively short distance and altitudinal gradient (Körner, 2003).

A significant challenge is data. Currently, there are only a handful of empirical studies that have focused on the influence of anthropogenic stressors on mountain carbon stock dynamics. For example, Li *et al* (2007) showed that the unsustainable management of alpine pastureland in Western China reduces soil carbon significantly. Britton *et al* (2011) quantified substantial carbon stocks in wet oceanic alpine grasslands. Wangchuk *et al* (2013) demonstrated that prescribed burning is an effective tool in restoring degraded high-altitude shrub-dominated grasslands. Wang, Li,Wang, and Wu (2008) highlighted the influence of climate change and thawing permafrost on vegetation and soil carbon loss in the Qinghai Tibetan-Plateau. More data are needed if climate policy is to be robustly supported, and carbon market and climate finance opportunities are to be realized in accordance with vigorous measurement, reporting, and verification (MRV) requirements. Attracting research funding, however, may require greater efforts to get MGS on the political agenda, with a recommended initial focus on quantifying the climatic-economic benefits of these ecosystems at a broader geographical level. Unfortunately, existing studies are relatively limited in this respect and are mainly centred on forests and/or the European Alps (Grêt-Regamey, Walz, and Bebi, 2008; Körner, 2009; Odermatt, 2004).

6.5.3 Opportunities

Notwithstanding the aforementioned challenges, carbon offset projects offer considerable climate change mitigation and sustainable development opportunities, and under the right conditions present potentially attractive financial returns and enhanced socio-economic and environmental outcomes (Trumper *et al*, 2009). The Páramo of the Northern and Central Andes Mountains provides a good example. Threatened by agricultural intensification, overgrazing, and urbanization, the loss of Páramo has a significant negative impact on catchment hydrological function, biodiversity, and local communities (Buytaert *et al*, 2006). Moreover, its clearance also releases large volumes of CO_2 into the atmosphere – between 877 and 1,758 tonnes of CO_2e per hectare ($tCO_2e\ ha^{-1}$) (Hofstede, Segarra and Mena, 2003).

Hypothetically, for every tonne of CO_2e avoided (or sequestered), one carbon offset could be realized if the necessary methodologies, skills, data, and institutions were available. For example, and based on the $tCO_2e\ ha^{-1}$ estimates above, if the introduction of sustainable cropping practices on a potato farm in the Central Andes reduced business-as-usual (BAU) emissions by 25%, the carbon offset yield would be between 219 and 439 carbon offset units per hectare. Assuming the net carbon offset price paid to the landholder is US$4.9 per unit (World Bank, 2014b), the financial return to the landholder would be between US$1,073 and US$2,151 per hectare. This revenue can be considered an incentive to fund technologies and practices that would otherwise be cost-prohibitive for the landholder. However, a key issue here is whether carbon markets can provide enough of an incentive to compensate landholders for land-use opportunity cost (Isenberg and Potvin, 2010). In this regard, carbon offset projects undertaken in mountain grassland and shrublands of high carbon density (such as the Páramo) are more likely to overcome the land-use opportunity cost hurdles facing projects developed in areas where the carbon density per hectare is lower, such as the alpine meadows of the Rocky Mountains (Seastedt, 2001).

Another consideration is the preference for NRM activities that sequester carbon (e.g. revegetation) versus those that avoid GHG emissions (e.g. avoided ecosystem clearance, such as REDD +). Carbon uptake rates in MGS are relatively small compared to other ecosystems (Körner, 2003), whereas existing carbon stocks are relatively large (Ward *et al*, 2014). In the current carbon market it would only be financially viable to focus on NRM activities that avoid emissions in areas where carbon density stocks are relatively high, such as the Páramo. However, carbon prices are forecast to increase in the medium to long term, as demand increases with the introduction of new emissions trading schemes (e.g. in China) and as emissions caps are tightened under existing climate policies e.g. the EU Emissions Trading Scheme (Newell, Pizer and Raimi, 2014; World Bank, 2014b). Until this occurs there are better prospects of tapping into 'boutique' carbon markets where companies may voluntarily pay a price premium for carbon offsets that are corporate social responsibility goals. Biosequestration projects are highly valued in this regard and have in recent times yielded an average payment of US$4.9 per unit (World Bank, 2014b). It is also important to point out that the demand for CO_2 removal measures (i.e. biosequestration carbon offset projects that have a net reduction impact) is likely to become more urgent if we are to meet the Intergovernmental Panel on Climate Change (IPCC)'s minimum safe emissions reduction targets (Benson, 2014).

This study recommends a focus be put on the establishment of NAMAs as a way to strategically position developing countries to capture a proportion of climate finance investment flows. As discussed, NAMAs are generally large-scale emission reduction strategies that are important as a basis for accessing climate finance. According to the Würtenberger (2013), NAMAs that are attractive to donors will need to be cost-effective, strategically appropriate to the country, and have a long-term impact. Examples of NAMAs related to MGS might include, ‘Grassland management strategy for the Qinghai-Tibetan Plateau’ or ‘Sustainable agriculture management in the Páramo ecosystems of Ecuador’.

6.6 Conclusions and recommendations on a carbon finance policy framework

In this study, experts agreed that anthropogenic stressors are having an increasing impact MGS, predominantly in developing countries where formal protection is minimal. It was also found that the NRM actions required to deal with these stressors are widely underfunded. While it is important to seek recognition through the intergovernmental process (e.g. the inclusion of mountain-based metrics in the Sustainable Development Goals framework) policy makers should also consider existing mechanisms that could directly or indirectly help meet the same objectives.

Although the opportunity is limited for industrialized countries, climate finance represents a resource that could potentially help achieve this. In the medium to long term, a higher carbon market price may also provide an adequate incentive to discourage unsustainable land-use and land use change. Although there are many barriers to leveraging carbon incentives for NRM, progress is being made. Suitable methodologies have been approved and are also being developed under the Gold Standard, the Verified Carbon Standard, and the Chinese Certified Emissions Reduction scheme. Robust and cost-effective in situ methods (e.g. soil spectroscopy) for measuring soil carbon are also emerging, helping to reduce the barriers associated with conventional laboratory-based methods (Gehl and Rice, 2007). Valuable lessons can also be taken from the extensive studies and projects being undertaken on REDD+, including the determination of the systemic drivers for ecosystem degradation, establishing measurement, reporting, and verification frameworks, implementing pilot projects, determining land tenure, addressing technical skills shortages and the lack of institutional capacity and data availability.

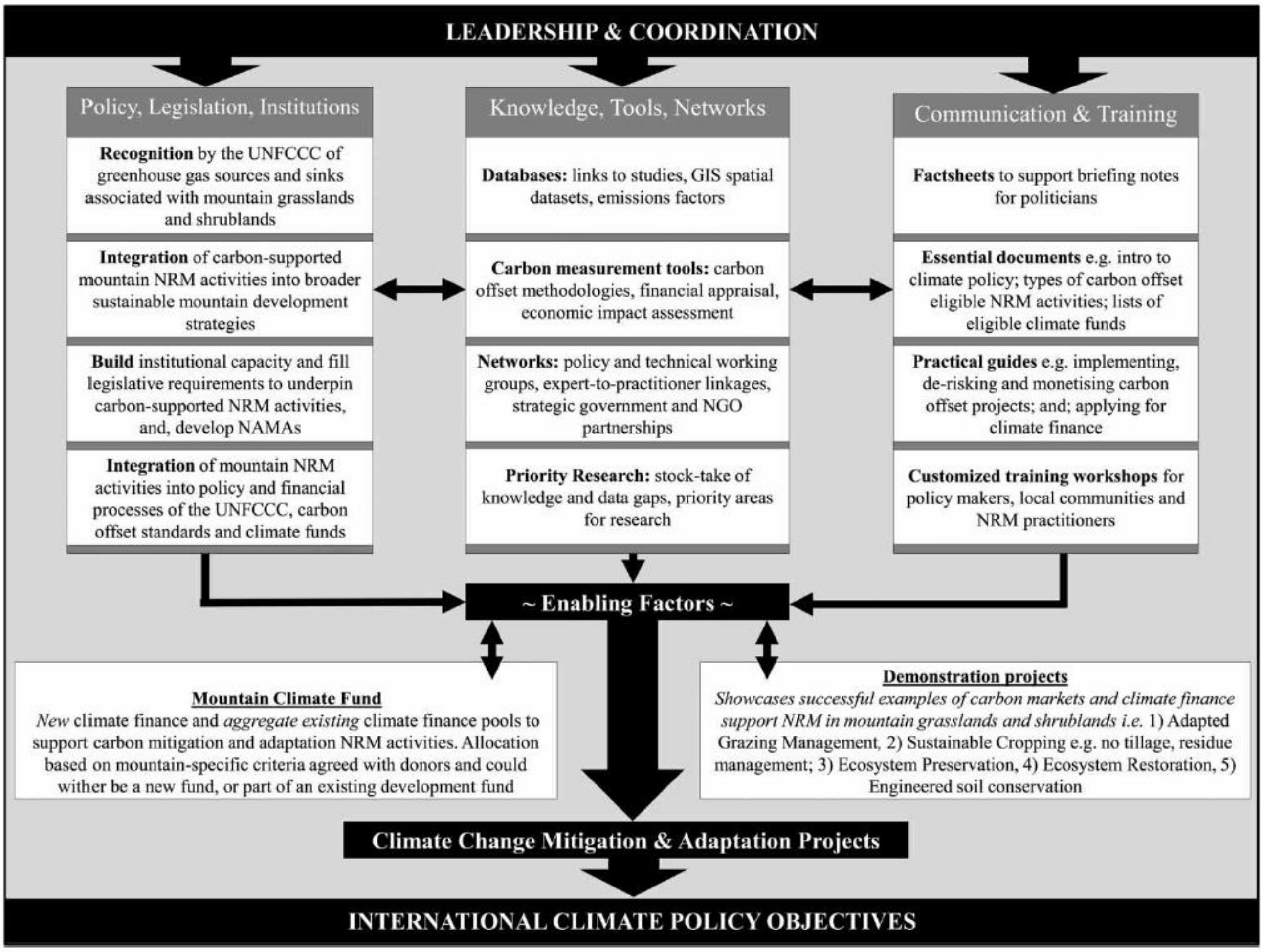
Despite this progress there are still many barriers to using carbon finance to support NRM in MGS. This study therefore recommends a systematic top-down approach be considered to address knowledge gaps and investigate innovative ways in which these mechanisms can be applied across MGS. This approach could consist of a collaboration of suitable organizations such as the Food and Agriculture Organisation, the Mountain Research Initiative, the International Centre for Integrated

Mountain Development, and the Consortium for Sustainable Development of the Andean Ecoregion. Figure 23 provides a top-down conceptual policy framework on how this might be achieved.

Central to this policy framework are five ‘enabling factors’ that would be needed to access climate finance and carbon markets for the NRM of MGS. Broadly, these consist of (1) policy, legislation, and institutions; (2) knowledge, tools, and networks; (3) communication and training; (4) a climate finance fund specific to mountains; and (5) successful demonstration projects. The first three factors focus on developing accurate measurements of MGS carbon into international climate policy, building technical skills, data and knowledge, and institutional capacity. Adapted Grazing Management, Sustainable Cropping, Ecosystem Preservation, Ecosystem Restoration, and Engineered Soil Conservation represent activities that are likely to achieve the duel objectives of climate mitigation and sustainable land-use. Demonstration projects should be developed to show stakeholders that these NRM actions can be successfully implemented in MGS, and to gain important economic, social and technical experience and data. It would also be important to assess the potential of these NRM activities in addressing the major stressors highlighted by the experts in this study.

The proposed *Mountain Climate Fund* would focus on channelling mitigation and adaptation finance to NRM activities. The fund could be created as a stand-alone mechanism or be designed to support existing funding mechanisms. Importantly, although the fund could seek new climate finance, it would be preferable to seek to aggregate existing climate finance pools based on a tailored set of criteria and MRV requirements established in collaboration with major donor organizations such as the World Bank or Global Environment Facility. Criteria could include (for example): the marginal cost of abatement (US$ per tCO_2e mitigated); compatibility with Nationally Appropriate Mitigation Actions; measurable social and environmental co-benefits, towards Sustainable Development Goals; scalability; and trans-boundary replication. The advantages of such a fund would lie in utilizing existing mountain-focused knowledge, skills, and institutions to disperse aggregated funds to areas where the greatest sustainable development outcomes could be gained. Spatial targeting using GIS could also play an important role here (Lin, Sills, and Cheshire, 2013). Ultimately, the aim would be to facilitate greater participation by simplifying many of the technical requirements that would otherwise be a barrier to using carbon finance, particularly within mountain communities, which may often lack technical knowledge and capacity (Körner *et al*, 2005).

Figure 23. Distribution top-down conceptual policy framework for enabling the usage of carbon markets and climate finance for mountain grasslands and shrublands



By putting in place these enabling factors, a new – and much needed – line of finance could be made available, forming part of the solution to the lack of funding for mountain NRM. Although this will be a challenge, the reasons to do so are compelling. There is an opportunity to leverage carbon markets and climate finance to support conservation, improve ecosystem services, and alleviate poverty through investment in NRM projects in MGS – much in the same way that has been done for forests and marine ecosystems. Such actions would also contribute towards the mitigation of climate change and towards international climate policy objectives. Accessing climate finance should not be the only motivation to act in this regard. As for other ecosystems, the carbon stored in MGS and shrublands needs to be considered as part of the 'global carbon budget'. Failing to address degradation and loss in these areas will likely offset gains in other areas, ultimately serving to undermine progress towards climate mitigation goals.

6.7 Supplementary information

Table 15 – Survey Questions

	Question		Options
1)	**Please tell us about yourself and your work in the mountains:**	a.	In what country do you currently live and work?
		b.	On what mountain range (in the country you live) do you primarily work?
		c.	What is the name/location of specific mountain area/watershed you work on?
		d.	What is the approximate land area (hectares) of this mountain area/watershed?
		e.	What is the approximate altitude range (m) of this mountain area?
		f.	What proportion (%) of this mountain area is above treeline but below the permanent snowline?
2)	**In what type of organisation do you work? (Choose one option)**	a.	Government
		b.	Non-government
		c.	Research & education
		d.	Other
3)	**What is the nature of your work? (Choose one option)**	a.	Natural resource management
		b.	Policy making
		c.	Research & education
		d.	Economic assessment
		e.	Modelling, technical or engineering
		f.	Other
4)	**How would you best describe the state of the mountain range which you work on? (Choose one option)**	a.	Isolated, relatively pristine, untouched
		b.	Isolated, landuse sustainably managed, minimal degradation
		c.	Isolated, land use managed sometimes, moderate to high degradation
		d.	Isolated, land use generally not managed, severe and widespread degradation
		e.	Close to major human settlement, relatively pristine, untouched
		f.	Close to major human settlement, land use sustainably managed, minimal degradation
		g.	Close to major human settlement, land use managed sometimes, moderate to high degradation
		h.	Close to major human settlement, land use generally not managed, severe and widespread degradation
5)	**How would you best describe the general level of protection of this mountain range and its watersheds?**	a.	A protected area that is well managed
		b.	A protected area that is poorly managed
		c.	An area with some protected areas that are well managed
		d.	An area with some protected areas that are poorly managed
		e.	An area that is largely unprotected but well managed
		f.	An area that is largely unprotected and poorly managed
6)	**What is the health of treeless alpine, subalpine and montane grassland and shrubland ecosystems located within the mountain range / country in which you work?**	a.	Improving overall
		b.	Improving, however some ecosystems stable
		c.	Improving, however some ecosystems declining
		d.	Relatively stable overall
		e.	Relatively stable, however some ecosystems improving
		f.	Relatively stable, however some ecosystems declining
		g.	Declining overall
		h.	Generally declining, however some ecosystems improving
		i.	Generally declining, however some ecosystems stable
7)	**With regards to treeless alpine, subalpine and montane grasslands and shrublands, please rank the following threats in order of most significant to least significance (with 1 being the most significant threat and 15 being the least significant):**	a.	Increasing frequency of wildfires
		b.	Increasing intensity of wildfires
		c.	Increasing number of introduced animal pests
		d.	Increasing number of introduced plant pests
		e.	Increasing conversion of land to intensive agriculture – grazing
		f.	Increasing conversion of land to intensive agriculture – cropping
		g.	Increasing incidence of metals and mineral extraction (mining)
		h.	Increasing illegal removal of native plants
		i.	Increasing tourist numbers in winter
		j.	Increasing tourist numbers in summer
		k.	Increasing infrastructure development (e.g. dams and roads)
		l.	Increasing urban encroachment
		m.	Increasing temperatures due to climate change
		n.	Changing rainfall patterns due to climate change
		o.	Other
8)	**What types of activities are currently being used to manage the threats identified in Question 7? Check all that apply.**	a.	Revegetation of catchment headwaters
		b.	Slope stabilisation using engineered solutions
		c.	Soil management (including fertiliser management)
		d.	Rangeland management
		e.	Fencing-off of streams from cattle and sheep
		f.	Buffer areas for streams

		g. Alternative livelihood programs h. Wildfire management i. Pest control (non-native plants) j. Pest control (non-native animals) k. Enforcement of illegal activities (e.g. firewood collection) l. Establishing protected areas
9)	**What are the greatest challenges in managing high-mountain ecosystems and upland watersheds in this mountain range? Please rate each challenge below in terms of its significance**	a. Lack of funding for management activities b. Lack of technical skills and/or knowledge c. Insufficient institutional capacity d. Community opposition e. Community poverty f. Legislative frameworks (or lack thereof) g. Remoteness and/or environmental conditions
10)	**Through the preservation and enhancement of carbon stocks in treeless high-mountain ecosystems, carbon markets and carbon financing could play a role in supporting the sustainable management upland watershed areas. Do you agree?**	a. Yes, it is important that carbon markets & financing be considered as a source of funding b. Possibly, but carbon markets and carbon finance is very complex to understand and potentially risky c. No, it is too complex and too risky for my organisation to consider d. I do not understand how carbon markets & financing could apply
11)	**Do you understand the difference between carbon markets and climate finance?**	a. Yes b. No
12)	**Please rate the following factors in the context of developing carbon offsets projects in your natural resource management area of responsibility (If you have not considered carbon offset projects before, please still answer this question while considering the potential factors)**	a. Lack of general knowledge and/or understanding about carbon offsets and carbon markets b. Lack of technical and/or operational capacity to develop carbon offset projects c. Inadequate partnerships and linkages required to develop carbon offset projects d. Lack of readily available carbon offset calculation methodologies e. Absence of required government institutions f. Absence of required legislation and/or regulations g. Inaccessibility to capital finance to kick-start the project h. Difficulties in establishing land tenure/ownership i. Community opposition j. Perceived risks associated with carbon markets k. The low market price of carbon l. There is no real opportunity to develop a carbon offset project in the area m. Unsure of the types of carbon offset projects that could be implemented
13)	**What is the main reason why you have not considered 'climate finance' to support natural resource management activities in your mountain grassland and shrubland area?**	a. Lack of general knowledge on climate finance and how it works b. Lack of specific information on what climate finance funds we could apply for c. Lack of technical capacity in meeting application requirements for climate finance d. Lack of institutional capacity in meeting application requirements for climate finance e. Lack of legislative/regulatory frameworks in meeting application requirements for climate finance f. There are no suitable climate finance funds for the area that I work in g. I did not know there was a difference between carbon markets and climate finance h. I have not heard of climate finance
14)	**Now that you have completed this survey, do you understand the difference between carbon markets and climate finance?**	a. Yes b. No, I still don't know the difference c. Not applicable
15)	**With regards to climate finance, do you understand the role that Nationally Appropriate Mitigation Actions (NAMAs) and National Adaptation Programmes of Action (NAPAs) could play in reducing emissions, adapting to climate change, achieving natural resource management and sustainability outcomes in the area/s where you work?**	a. Yes, we are currently investigating/have identified the specific role that NAMAs/NAPAs could play b. Yes, but we have NOT yet investigated the specific role that NAMAs/NAPAs could play c. No, NAMAs/NAPAs will have no role to play in the management of the area in which I work d. What are NAMAs and NAPAs? e. Not applicable

Chapter 7. Conclusion and recommendations

While science has progressively shone light on the relative importance of forest, lowland grassland and marine ecosystems to climate mitigation, it has seemingly sidestepped the role that alpine, subalpine and montane grasslands and shrublands might play. As discussed, this is not surprising given the general lack of interdisciplinary knowledge and data for mountains. As stated, the overall aim of this thesis was to answer this question by providing both a biochemical and economic estimate of C stored in MGS areas, and propose how this C pool might be factored into international climate policy frameworks and budgets when also considering the opportunities for using climate finance to address NRM stressors.

Chapter 2 of this thesis explored the environmental and socioeconomic factors relevant to this study. Although it did not contribute directly to the research questions, it provided context, aiding the reader in interpreting the results presented in *Chapters 4, 5* and *6*. Highlighting the biophysical factors influencing mountains, such as extreme climatic conditions, facilitates a better understanding as to why biomass C stocks are particularly low in MGS compared to forests. Likewise, looking at demographic factors such as poverty and illiteracy rates (which are more common at higher altitudes) helps one recognise the challenges that would accompany the on-ground application of climate finance in a MGS context. Furthermore, by knowing the evolution and current application of ecological economics and carbon markets we can get a better appreciation as to why this thesis is trying to estimate the value of climate regulation by MGS, and how and why this links with the use of climate finance to solve priority NRM issues. Importantly, this chapter highlighted the data and knowledge gaps that exist for C stored in MGS on a global scale.

Chapter 4 established a baseline measure of C stored in mountain areas, thus filling the aforementioned research gap. Using spatial analysis, this thesis was the first to estimate there to be between 60.5 Pg C and 82.8 Pg of C contained within the biomass and soils of the world's MGS. It then put these figures into perspective by comparing global C pools in tropical Savannas and grasslands, temperate forests and tropical peatlands as estimated by other studies to contain 326–330 Pg C, 159–292 Pg C and 88.6 Pg C respectively. In making such a comparison this thesis did not set out to prove which ecosystem is superior with regards to C storage and thus importance to climate policy. Rather, it attempts to draw attention to MGS C as an important natural capital stock, that like other ecosystems, should be considered intrinsically *invaluable*, as part of the emerging green economy and eventually given due regard in future climate policy decisions. With respect to this last point, *Chapter 5* also found that MGS C stocks are not reliably accounted for in international carbon budgets. These findings thus answer *Research Question 1* which asked "What is the spatial distribution and significance of carbon stored in the world's MGS? How is it accounted for in global carbon budgets and international carbon accounting frameworks?".

Chapter 5 builds on *Chapter 4* by modelling the impact of LULUC and the exchange of CO_2 between MGS and the atmosphere. This chapter clarified, both in biochemical and economic terms, how LULUC impacts MGS C stocks. It put the first global economic estimate for MGS CO_2 sequestration at between US$1.24 billion and US$11.8 billion per annum. It also pointed out that if land use was managed more sustainably, MGS ecosystems could sequester up to an additional 8.4 Mt CO_2 per annum while contributing US$0.093 billion - US$0.89 billion annually in added economic value to society. When considered for its total *in-situ* C stock, this chapter estimated that MGS ecosystems contain at least 252 ± 39 gigatonnes CO_2 (68 ± 11 petagrams C) as at 31 December 2015, with an equivalent ecological asset value of between US$2.5 (± 0.43) trillion and US$26.5 (± 4.1) trillion (2007 dollars). These figures represent the most comprehensive and up-to-date global biochemical and economic estimate provided for MGS C stores and associated CO_2 fluxes. With respect to ecological economics and the green economy, these estimates could potentially fill gaps in other respected studies considering the relative economic contribution of ecosystem services worldwide e.g. Costanza *et al*, 2014. This chapter thus answers *Research Question 2* which asked "To what extent is carbon globally exchanged between MGS and the atmosphere? How is this impacted by land use change? What is the *non-tradable* economic value of these exchanges when considering climate policy and broader sustainable perspectives?".

Noting the relatively significant C stocks found in MGS, and the associate economic value, *Chapter 6* investigated how climate finance might be used to support priority NRM actions and thus address LULUC derived stressors. In so doing, both C stocks and economic value can potential be enhanced, as is estimated in *Chapter 5* under the sustainable land management simulation scenario. Notably, *Chapter 6* also sought to uncover how experts understood the risks and opportunities of using climate change in this respect, and what methodologies, institutional arrangements and other enabling factors would be required to do so. Through the use of a survey, interviews and literature review the chapter concluded the priority anthropogenic stressors for MGS to be principally grazing and cropping intensification, and the proliferation of exotic plants. It also found the lack of NRM funding to be key constraint in dealing with these stressors, adding weight to the postulation that climate finance could provide an innovative and valuable source of financial support. Importantly, the results of this chapter noted that around half of SMD focused experts surveyed did not have a firm grasp on the opportunities, risks and barriers to using climate finance to solve MGS stressors. It also established that the required climate finance measurement methodologies, policy frameworks and data to be largely undeveloped. The chapter concluded by proposing a top-down conceptual policy framework that can be used to develop key ‘enabling factors’ with the view of extending the eligibility of carbon markets and climate finance to NRM activities undertaken in MGS in the same way that has been afforded to other ecosystems. This chapter thus answers *Research Question 3* which asked “What are the stressors, NRM challenges and priorities related to carbon stocks in MGS? Why has climate finance not been utilized in this context? What is required to position these NRM activities eligible for carbon finance incentives, and in so doing, ensuring that MGS are more sustainably used and the aforementioned economic value is maintained and/or improved?”.

This thesis has the following implications for policy making. Firstly, the results provide a sound first-step in developing (or improving) global environmental accounts for C stored in MGS ecoregions. As an environmental asset, these results provide an initial baseline estimate and methodology with which to monitor and manage C these C stocks. This baseline has direct application to improving the resolution of MGS carbon accounting modalities issued by the IPCC and UNFCCC, which as discussed in *Chapter 4* are currently inadequate to reflect the unique biophysical characteristics of these ecosystems. Any improvement in this respect will also improve the reliability of the science and thus progress towards the targets set by the UNFCCC *Paris Agreement*. Second, from an accounting perspective, this thesis could potentially provide input data into other global studies (e.g. Costanza *et al*, 1997; Costanza *et al*, 2014) which have excluded estimates for C in alpine areas which until now has not been available. Third, when combined, the estimates for C stocks, CO_2 sequestration and economic value provided herein justify further investigation of how carbon markets and climate finance might be used specifically to address anthropogenic stressors impacting MGS around the world. As noted, globally there is approximately US$390 billion in climate finance transacted annually. If just 0.1 percent of this amount could be set aside then this would open up around US$39 billion of NRM funding for MGS every year. To put this figure in context, the Australian Alps alone contribute circa US$7.7 billion in direct economic benefits every year due to the provision of water to users in the Murray-Darling Basin (Worboys and Good, 2011). For only US$0.05 billion invested annually in priority NRM actions, most of the major stressors effecting this valuable natural capital stock could be managed effectively. Sadly, like for many mountain areas around the world, even this relatively small sum has not been funded by government.

We acknowledge that there are limitations to this study. Mostly, these related to temporal and spatial resolution of input data. As described in the relevant *Materials and Methods* sections of this thesis the best data available has been used, though much of it is based on mean values for a particular ecoregion which can be thousands of square metres in area. Specifically, the HWSD dataset underpinning our estimates for soil C are generally aggregated. However, this is a global study that is seeking to provide an initial and high-level estimate for international climate policy. The spatial resolution of the HWSD, GLC2000 and other datasets used by this study are therefore more than adequate to meet this requirement. It was not the intention of this thesis to provide empirical measurements or local scale estimates. The intention was for the global scale estimate provided to be used as a starting point to determine where best to do this. Moreover, the estimates provided here improve on existing proxy values and methodologies for *lowland* grasslands, such as those provided by the IPCC, providing a more specific reference point for *upland* MGS that better reflect the native biophysical conditions.

This study sets the stage for future research, including in the following areas. First, the results could be used to support the selection of the most suitable MGS locations around the world for targeted and/or empirical research and trials. Second, the results could also be used to improve national and regional environmental accounts for MGS as required under UNFCCC and other mandatory and voluntary reporting regimes. Third, further development of the IBM (and underlying assumptions) presented in *Chapter 5* could produce more reliable results, providing greater confidence when forecasting LULUCF trends in MGS. Four, a more detailed and specific MGS climate policy roadmap, such as that which has been completed for marine ecosystems (i.e. *Blue Carbon Policy Framework 2.*0 IUCN, 2012), would assist in resolving the many design, implementation and monitoring issues that exist for using climate finance in MGS. Finally, investigating the economics of carbon mitigation projects in MGS could help government and industry gain confidence in the financial and non-financial benefits that such projects could generate e.g. Siikamäki *et al* (2012).

In conclusion, this thesis has estimated there to be substantial amounts of C stored in MGS around the world. It also found annual CO_2 sequestration rates to be of global importance, both in biochemical and economic terms. It is hoped that policy makers can use the results of this thesis to build a business case for integrating MGS into international climate policy discourse, and ultimately establish mechanisms and modalities to enable the use of climate finance in the MGS context.

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Appendix A – model dataset snapshot from ExcelTM

B2 ECO_NAME

	ECO_NAME	G200_REGIO	ECO_CODE	TC_HA_SOIL	ERR_HA	SOIL_TYPE	COUNTRY	CONTINENT	_H_MU_GLOB	AREA_HA	TC_SOIL	ERR_TC_SOIL	C_HA_BIOMAS	R_TC_HA_BIOMAS	TC_BIOMASS	ERR_TC_BIOMASS	TC_TOTAL	ERR_TC_TOTAL
3	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	7772	424371.95	54707.238	0.400	0.314	3108.952	976.211	427480.900	55683.449
4	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	34888	1904906.64	245568.023	0.400	0.314	13955.360	4381.983	1918862.000	249950.006
5	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	23233	1268510.88	163528.071	0.400	0.314	9293.120	2918.040	1277804.000	166446.111
6	Sulaiman Range alpine meadows		PA1018	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	115798	6322570.80	815064.202	0.400	0.314	46319.200	14544.229	6368890.000	829608.431
7	Ghorat-Hazarajat alpine meadow		PA1004	28.41	0.133693374	Calcisols	Afghanistan	Asia	3503	29883	848872.18	113488.586	0.400	0.314	11953.000	3753.242	860825.178	117241.828
8	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	6393	349061.08	44998.656	0.400	0.314	2557.224	802.968	351618.300	45801.624
9	Karakoram-West	Tibetan Plateau s	PA1006	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	103798	5667370.80	730600.132	0.400	0.314	41519.200	13037.029	5708890.000	743637.161
10	Karakoram-West	Tibetan Plateau s	PA1006	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	6446	351977.81	45374.662	0.400	0.314	2578.592	809.678	354556.400	46184.340
11	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	4080	222756.53	28716.306	0.400	0.314	1631.916	512.422	224388.450	29228.727
12	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	30111	1644082.44	211944.284	0.400	0.314	12044.560	3781.992	1656127.000	215726.276
13	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	5905800	322456680.00	41568992.265	0.400	0.314	2362320.000	741768.480	324819000.000	42310760.745
14	Northwestern Himalayan alpine shru		PA1012	54.60	0.128913416	Leptosols	Afghanistan	Asia	3503	150345	8208837.00	1058229.222	0.400	0.314	60138.000	18883.332	8268975.000	1077112.554
15	Sulaiman Range alpine meadows		PA1018	54.60	0.128913416	Leptosols	Afghanistan	Asia	3599	8011	437416.43	56388.847	0.400	0.314	3204.516	1006.218	440620.950	57395.065
16	Ghorat-Hazarajat alpine meadow		PA1004	54.60	0.128913416	Leptosols	Afghanistan	Asia	3599	150089	8194859.40	1056427.322	0.400	0.314	60035.600	18851.178	8254895.000	1075278.501
17	Sulaiman Range alpine meadows		PA1018	54.60	0.128913416	Leptosols	Afghanistan	Asia	3581	52174	2848684.02	367233.589	0.400	0.314	20869.480	6553.017	2869553.500	373786.606
18	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12212	13500	206908.37	26673.265	1.500	0.314	20249.400	6358.312	227157.769	33031.576
19	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12212	8843	135529.30	17471.546	1.500	0.314	13263.780	4164.827	148793.084	21636.373
20	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12210	112857	1729759.24	222989.173	1.500	0.314	169285.500	53155.647	1899044.739	276144.820
21	Angolan montane forest-grassland r		AT1001	29.16	0.128913416	Leptosols	Angola	Africa	12203	143297	4178540.52	538669.934	1.500	0.314	214945.500	67492.887	4393486.020	606162.821
22	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12187	22723	348275.42	44897.374	1.500	0.314	34084.500	10702.533	382359.921	55599.907
23	Angolan montane forest-grassland r		AT1001	126.88	0.128913416	Leptosols	Angola	Africa	12007	478	60709.23	7826.234	1.500	0.314	717.705	225.359	61426.936	8051.594
24	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12187	56049	859056.89	110743.959	1.500	0.314	84072.900	26398.891	943129.792	137142.849
25	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12187	152	2329.73	300.334	1.500	0.314	228.003	71.593	2557.738	371.927
26	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12187	181	2771.90	357.336	1.500	0.314	271.277	85.181	3043.180	442.516
27	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12187	4	56.18	7.243	1.500	0.314	5.498	1.726	61.680	8.969
28	Angolan montane forest-grassland r		AT1001	15.33	0.128913416	Leptosols	Angola	Africa	12187	378	5797.41	747.364	1.500	0.314	567.372	178.155	6364.779	925.518
29	Angolan montane forest-grassland r		AT1001	50.32	0.109986209	Ferralsols	Angola	Africa	12215	571	28747.24	3161.800	1.500	0.314	857.001	269.098	29604.243	3430.898
30	Angolan montane forest-grassland r		AT1001	44.99	0.109986209	Ferralsols	Angola	Africa	12217	33338	1499709.93	164947.410	1.500	0.314	50007.000	15702.198	1549716.930	180649.608
31	Angolan montane forest-grassland r		AT1001	44.99	0.109986209	Ferralsols	Angola	Africa	12213	194389	8744589.17	961784.212	1.500	0.314	291583.500	91557.219	9036172.665	1053341.431
32	Angolan montane forest-grassland r		AT1001	44.99	0.109986209	Ferralsols	Angola	Africa	12216	60008	2699441.89	296901.380	1.500	0.314	90011.400	28263.580	2789453.286	325164.959
33	Angolan montane forest-grassland r		AT1001	44.99	0.109986209	Ferralsols	Angola	Africa	12213	130768	5882598.48	647004.706	1.500	0.314	196152.000	61591.728	6078750.480	708596.434
34	Angolan montane forest-grassland r		AT1001	44.99	0.109986209	Ferralsols	Angola	Africa	12217	4045	181958.93	20012.973	1.500	0.314	6067.320	1905.138	188026.247	21918.111
35	Angolan montane forest-grassland r		AT1001	37.63	0.109986209	Ferralsols	Angola	Africa	12233	77057	2899346.68	318888.150	1.500	0.314	115585.500	36293.847	3014932.182	355181.997
36	Angolan montane forest-grassland r		AT1001	37.63	0.109986209	Ferralsols	Angola	Africa	12232	31632	1190196.92	130905.247	1.500	0.314	47448.450	14898.813	1237645.370	145804.061
37	Angolan montane forest-grassland r		AT1001	37.63	0.109986209	Ferralsols	Angola	Africa	12232	683	25708.87	2827.621	1.500	0.314	1024.911	321.822	26733.779	3149.443
38	Angolan montane forest-grassland r		AT1001	67.58	0.128913416	Leptosols	Angola	Africa	12209	71692	4844566.63	624529.635	1.500	0.314	107537.550	33766.791	4952104.178	658296.426
39	Angolan montane forest-grassland r		AT1001	67.58	0.128913416	Leptosols	Angola	Africa	12209	444886	30063171.45	3875546.140	1.500	0.314	667329.000	209541.306	30730500.450	4085087.446
40	Angolan montane forest-grassland r		AT1001	52.34	0.109986209	Ferralsols	Angola	Africa	12206	108629	5685098.72	625282.456	1.500	0.314	162943.500	51164.259	5848042.215	676446.715
41	Angolan montane forest-grassland r		AT1001	52.34	0.109986209	Ferralsols	Angola	Africa	12239	6007	314353.32	34574.530	1.500	0.314	9009.840	2829.090	323363.158	37403.619
42	Angolan montane forest-grassland r		AT1001	52.34	0.109986209	Ferralsols	Angola	Africa	12240	1597	83587.37	9193.458	1.500	0.314	2395.740	752.262	85983.109	9945.720
43	Angolan montane forest-grassland r		AT1001	52.34	0.109986209	Ferralsols	Angola	Africa	12240	82	4293.38	472.213	1.500	0.314	123.055	38.639	4416.435	510.852
44	Angolan montane forest-grassland r		AT1001	52.34	0.109986209	Ferralsols	Angola	Africa	12240	60	3136.30	344.950	1.500	0.314	89.891	28.226	3226.192	373.176
45	Angolan montane forest-grassland r		AT1001	67.58	0.109986209	Ferralsols	Angola	Africa	12201	2046	138226.01	15202.955	1.500	0.314	3068.280	963.440	141294.294	16166.395
46	Angolan montane forest-grassland r		AT1001	67.58	0.109986209	Ferralsols	Angola	Africa	12204	141797	9581932.28	1053880.407	1.500	0.314	212695.500	66786.387	9794627.775	1120666.794
47	Angolan montane forest-grassland r		AT1001	67.58	0.109986209	Ferralsols	Angola	Africa	12199	51486	3479166.45	382660.329	1.500	0.314	77229.000	24249.906	3556395.450	406910.235
48	Angolan montane forest-grassland r		AT1001	67.58	0.109986209	Ferralsols	Angola	Africa	12234	54582	3688371.89	405670.042	1.500	0.314	81872.850	25708.075	3770244.743	431378.117

Sheet1 Sheet2

Appendix B - Screenshot of model in AnyLogic™