# The socio-ecological dynamics of pastoralism and overstocking in the Dhofar Mountains of Oman

Lawrence Ball

The Durrell Institute of Conservation and Ecology (DICE)

School of Anthropology and Conservation

The University of Kent, Canterbury, UK

November 2018

Thesis submitted for the degree of Doctor of Philosophy in

Biodiversity Management

"With the help of basic knowledge such as this and from experience gained in other similar regions of the world, we will be better equipped to guide, for our own good, the natural continuous changes in the ecosystems of our country. Our development activities will also cause changes. Our ability to guide all these will depend on the depth of our knowledge. The skill with which we do it will determine the quality of the life to be lived by our people for generations to come – whether they will live, as now, among green trees and grass or in an ever widening desert."

Forward by His Majesty Sultan Qaboos bin Said in the Scientific Results of the Oman Flora and Fauna Survey 1977

## Acknowledgements

I wish to sincerely thank my supervisors, Joseph Tzanopoulos and Douglas MacMillan, for their advice, comments and support. I wish to thank my wife Alana, not only for her assistance during the fieldwork, but for her continued interest and support throughout my studies. I would like to thank my good friend Hadi Al Hikmani for introducing me to the Dhofar region and for looking out for me as I conduct my research. I wish to thank Ali Bawazir for his valuable assistance during the research period and Andrew Spalton for his comments on the chapters. In addition I would like to thank the Economic and Social Research Council, the Anglo-Omani Society, Dhofar University, the Oman Botanic Gardens and the German Aerospace Center for their logistical, financial, administrative and/or technical support of the project.

## Author's declaration

The research was conceived, designed and conducted by Lawrence Ball. All chapters of this thesis were written by Lawrence Ball. Dr Joseph Tzanopoulos (primary supervisor) provided comments and editorial suggestions on all chapters. Professor Douglas MacMillan (secondary supervisor) provided comments and editorial suggestions on Chapter 2.

# Abstract

Achieving sustainable use of natural resources is the greatest challenge facing humanity today. Rangelands, which cover one-third to one-half of the earth's ice-free surface, are frequently mismanaged, vulnerable to climate change, and in a degraded state, and their inhabitants are some of the poorest and most marginalized communities on earth. Despite over a century of scientific attention, we still lack an adequate understanding of how rangeland socio-ecological systems operate and how rangeland vegetation responds to abiotic and biotic variables.

The Dhofar Mountains represent a rather unique rangeland case study, with atypical social, cultural, political, economic and ecological situations, which could provide valuable insights for rangeland science. Moreover, the Dhofar mountain region is understudied, globally unique, supports a wealth of biodiversity and provides valuable ecosystem services to the local population, yet the threat of overstocking, despite being well-recognised, has received little scientific attention.

Therefore, this interdisciplinary research which utilises contemporary methods from the social, ecological and rangeland sciences, aims to firstly understand the social processes driving overstocking in rural Dhofar and secondly, assess the impacts of overstocking on vegetation communities. Data collection methods included interviews, questionnaires, participatory mapping exercises, vegetation sampling and remote sensing. Analytical procedures included qualitative coding, the application of a socio-ecological systems framework, multivariate analysis of vegetation communities and GIS spatial analysis.

The results provide the first detailed analysis of the socio-ecological system surrounding pastoralism in Dhofar. We find that livestock ownership is principally motivated by strong pastoral values rooted in cultural norms. But livestock ownership is expensive due to the requirement for daily feedstuff provisioning, which in turn makes local livestock prices uncompetitive against imported livestock. Few livestock are sold and the expense means some better-educated or wealthier individuals are losing interest. By applying a socio-ecological system framework we identify variables inhibiting self-organization, which can be summarised as too many resource users in an unproductive system with undervalued resources.

Feedstuff provision is found to be a critical variable which deems many rangeland concepts inapplicable and maintains livestock populations beyond the carrying capacity of the environment. Subsequently, the rangelands, which receive reliable precipitation, exhibit equilibrium properties. Several decades of overbrowsing has increased the frequency of unpalatable species, decreased plant density, reduced advanced growth, altered population age structures, and altered plant phytomorphology through the damaging effects of management practises, bark stripping and browsing.

We identify six new variants and a pre-described seventh variant of the *Anogeissus* forest. Our results suggest that two variants are the result of historical agricultural practises and deforestation, and long-term stocking rates are the primary driver of vegetation change across all variants. Finally, using a novel method, we calculate that seventeen percent of continuous-canopy forest has been lost in the study area and provide further evidence that unforested areas are the result of anthropogenic deforestation.

Our findings contribute valuable insights for rangeland science and demonstrate the need for new case studies, and synthesis of concepts and theories, specific to pastoralism in the Middle East. Our findings highlight a requirement for an intervention to reduce livestock pressure on the rangelands in Dhofar. We propose a shift away from the status quo of unmanaged and unproductive overstocking to an economically and environmentally sustainable rural livestock production system through certification, sustainable intensification and marketization.

# Contents

## List of figures

## List of tables

## List of appendices

# 1 Introduction

## 1.1 Global livestock production

Global calorie production will need to increase by 43% to meet the needs of the global human population by 2050 (Meyfroidt, 2018). Global annual demand for meat is estimated to increase by between 6 and 23 kg per person and global cattle numbers are estimated to increase from 1.5 billion to 2.6 billion (Robinson *et al.*, 2011). Global livestock production has already responded to this increasing demand, primarily through a shift from extensive, small-scale, livestock production systems to more intensive, large-scale, specialized production units. Further intensification leads to higher levels of mechanization at which point production becomes 'industrial' (Robinson *et al.*, 2011).

It is estimated that livestock contribute to food security and poverty reduction amongst 70 percent of the world's 1.4 billion extreme poor (Herrero *et al.*, 2013), but livestock sector growth can threaten this role of livestock, as smallholders are squeezed out of market participation (Robinson *et al.*, 2011). Livestock sector growth also increases greenhouse gas emissions, currently estimated at 14.5% of global emissions (Gerber *et al.*, 2013) and crop production for livestock feedstuffs is inefficient both in terms of the land required (one third of global cereal production) and because the conversion efficiency of plant-based feedstuffs into animal matter is 10% (Godfray *et al.*, 2010; Herrero *et al.*, 2013; Mottet *et al.*, 2017). Therefore, extensive livestock production systems, where feedstuff consumption is minimal, may be considered more sustainable, albeit less efficient, than industrialised livestock production (Godfray *et al.*, 2010; Herrero *et al.*, 2013).

Increasing global livestock production through sustainable means will be a global challenge (Robinson *et al.*, 2011; Nabarro & Wannous, 2014). Godfray et al. (2014) state the requirement for a radical overhaul in the way food is produced, stored, processed, distributed and accessed to match the changing demand of a larger and more affluent population, to abolish undernutrition, and to ensure food production is environmentally and socially sustainable. Interdisciplinary research at the local scale is therefore valuable to inform sustainable intensification of agricultural production systems (Petersen & Snapp, 2015).

## 1.2 Rangelands

Extensive livestock production systems predominantly occur in rangelands. They are areas that are too dry, too unreliable, too infertile or too remote to warrant intensive management (Stafford Smith, 1996). Rangelands encompass grasslands, savannahs, tundra, steppe, prairies, shrublands, deserts, woodlands and forests (Holechek, Pieper & Herbel, 2001) and cover one third to one half of the earth's ice-free terrestrial surface (Sayre, 2017). Livestock production is the dominant use, indeed the absence of other uses is intrinsic to the definition of rangelands (Reynolds *et al.*, 2007; Sayre *et al.*, 2013; Sayre, 2017). Rangelands offer provisioning services such as freshwater and forage, regulating services such as carbon sequestration, supporting services such as nutrient cycling, soil conservation and biodiversity, and cultural services such as spiritual and religious value, traditional knowledge and tourism (Briske, 2017). Although productivity is low at a global scale, rangeland pastoral systems support the nutritional security and incomes of 1-2 billion people (Sayre *et al.*, 2013), of which 250 million are estimated to be affected by rangeland degradation (Reynolds *et al.*, 2007).

## 1.3 Rangeland degradation

Estimates of global rangeland degradation vary between 10 and 80 percent (Millennium Ecosystem Assessment, 2005; Sayre *et al.*, 2013). Bias, inconsistencies, mapping limitations and a lack of precise definitions hinders accurate estimations (Gibbs & Salmon, 2015). The United Nations Convention to Combat Desertification (UNCCD) defined land degradation as 'a reduction or loss of biological or economic productivity and complexity'. Rangeland degradation in arid, semi-arid and dry sub-humid areas is popularly termed as desertification. The causes of rangeland degradation can be natural factors primarily associated with a variable climate or human-induced factors including overcultivation, deforestation, poor irrigation practises or overgrazing (Burns, 1995).

Overgrazing was defined by Wilson and MacLeod (1991) as 'a concomitant vegetation change and loss of animal productivity arising from herbivore grazing activity', which importantly considers both ecological ('vegetation change') and economic ('productivity') effects. However, the term is often used interchangeably by different stakeholders (Perevolotsky & Seligman, 1998; Mysterud, 2006). For example, conservationists may more readily assume overgrazing is taking place whilst livestock owners, landowners or governments with economic interests may not (Homewood and Rodgers, 1987).

The most commonly reported ecological impact of grazing in rangelands is woody plant encroachment which is a threat to the maintenance of savannah and grassland ecosystems. Woody plant encroachment coincided with a global intensification of livestock grazing which reduced herbaceous communities and thus the frequency and intensity of fires, facilitating woody plant encroachment (Briske, 2017). Three further shifts in vegetation community composition are frequently reported as a result of overgrazing. These are, shifts from palatable to unpalatable vegetation (Wardle, 2002), shifts in dominant grass species, and shifts between grass and forb dominance (Fernandez-Gimenez & Allen-Diaz, 1999).

Grazing also impacts the ecohydrology of rangelands, although research shows that only under heavy grazing is soil infiltrability significantly reduced (Wood & Blackburn, 1981; Hiernaux *et al.*, 1999; Savadogo, Sawadogo & Tiveau, 2007). Biological soil crusts are important, but often overlooked components of biogeochemical processes in rangelands, which are vulnerable to trampling from livestock (Belnap & Lange, 2003). In addition, the physical and chemical properties of soils can be altered due to changes in soil-plant relationships (Briske, 2017) and due to urination, defecation or compaction by livestock (Hiernaux *et al.*, 1999; Drewry, Cameron & Buchan, 2008), the latter of which can result in decreased soil stability and increased vulnerability to erosion (Eldridge, 1998).

In contrast to woody plant encroachment, browsing livestock such as goats and camels, or wild browsers such as deer, moose and elk, can reduce woody cover. The consumption of seedlings and saplings (by browsers and grazers) (Ripple *et al.*, 2001; Côté *et al.*, 2004; Staver *et al.*, 2009) and the removal of reproductive components from adults (Augustine & Decalesta, 2003) are considered the main processes by which browsers maintain open ecosystems. In addition, a range of human activities can facilitate loss of woody cover and inhibit woody plant regeneration. A loss of shrubs and trees can negatively impact pastoralists through a loss of high-quality browse, shade for animals and people, protein source from seedpods and through a loss of productive and nutrient-dense herbs from the tree understory (Robin & Ellis, 1995). It has been suggested loss of woody cover can also affect local climate through a sustained decrease in rainfall (Schlesinger *et al.*, 1990).

Much of the literature focuses on livestock overgrazing in grassland ecosystems, whereas much less focuses on overbrowsing in wooded environments. At the time of writing, an online literature search returned 514 articles with 'overgrazing' in the title and just 22 with 'overbrowsing' in the title. This may be due to the term overgrazing being used interchangeably, which is problematic due to the vastly different impacts that browsers and grazers can have on an ecosystem. For example, the most commonly reported effect of overgrazing on rangeland vegetation is woody plant encroachment, however, the most commonly reported effect of overbrowsing is reduced woody plant cover (Asner *et al.*, 2004). Studies on overbrowsing tend to focus on wild browsers, whilst few studies have addressed large browsing livestock such as camels, despite 80% of their diet comprising of woody plants (Dereje & Uden, 2005).

## 1.4 Advancements in rangeland science

Rangeland science is one of the oldest fields in conservation yet despite over a century of scientific attention, rangeland degradation still persists (Herrick *et al.*, 2010). This is in part due to the hampered progression of rangeland science itself, which for decades was founded upon two flawed theories (Sayre, 2017).

The first was Clementsian or successional theory (Clements, 1916, 1920), whereby plant communities are at equilibrium with abiotic factors unless disturbed by exogenous (usually anthropogenic) drivers (Behnke, 2000; Vetter, 2005; Sayre *et al.*, 2012). Terms such as 'carrying capacity', 'stocking rates', 'range condition' and 'rangeland degradation' were typical of this thinking, and overgrazing was famously linked to rangelands in the USA (Herbel, 1979), Australia (Curry & Hacker, 1990) and Africa (Lamprey, 1983). However, this theory was found ill-suited to explain trends in many arid rangelands, and was replaced in the 1980s by the theory of equilibrium and non-equilibrium rangelands based on state and transition models of vegetation dynamics (Wiens, 1984; Ellis & Swift, 1988; Sayre, 2017). Non-equilibrium models were found more applicable to arid rangelands, where climatic variability, rather than overstocking, was the principle driver of vegetation change (Scoones, 1995). Contemporary evidence now infers that many rangeland systems may encompass elements of both equilibrium and non-equilibrium models (Stafford Smith, 1996; Oba, Stenseth & Lusigi, 2000; Vetter, 2005).

The second was the theory of the tragedy of the commons (Hardin, 1968) which assumed that overstocking was inevitable in communally grazed systems (Herskovits, 1926; Lamprey, 1983) This has since been challenged by numerous case studies where pastoralist mobility, self-organisation and adaptive management have sustainably governed the use of open access lands (Ellis & Swift, 1988; Westoby, Walker & Noy-Meir, 1989; Ostrom, 1990; Scoones, 1995). This is thought to enable pastoralists to adapt to the spatio-temporal heterogeneity of forage resources which results from climatic variability in non-equilibrium rangelands (Scoones, 1995).

Despite these advancements and our improved capacity to identify patterns at greater spatial and temporal scales (Reichman, Jones & Schildhauer, 2011), we still lack an adequate understanding of rangeland functioning due to ecological variability between and within rangelands (Lynam & Stafford Smith, 2004). Furthermore, the recognition that rangelands are complex socio-ecological systems, although valuable, does not provide a robust framework to inform conservation practise (Sayre *et al.*, 2012). In addition, rangeland conservation has suffered from a research-implementation-research gap (Knight *et al.*, 2008) where practitioners have routinely failed to implement informed recommendations (Boyd & Svejcar, 2009) and both have failed to implement monitoring methodologies over time (Briske *et al.*, 2011).

As a result, rangeland conservation has yet to be effective (Lynam & Stafford Smith, 2004; Sayre *et al.*, 2012) but despite these shortfalls we now realise the importance of addressing rangelands on a case-by-case basis (Costanza *et al.*, 1998; Grice & Hodgkinson, 2002; Reynolds *et al.*, 2007). Furthermore, a substantial body of literature on rangeland and natural resource science has been synthesised by modern scholars into some promising theoretical frameworks enabling improved understanding and analysis of the sustainability of rangeland systems (e.g. Westoby, Walker & Noy-Meir, 1989; Ostrom, 1990, 2007; Scheffer *et al.*, 2001; Gunderson, 2002; Norberg & Cumming, 2008; Waltner-Toews, Kay & Lister, 2008).

## 1.5 Rangeland systems of the Arabian Peninsula

Pastoralism in the Arabian Peninsula is thought to have established amongst hunting communities in the southern mountain areas in the late seventh millennium BC with the introduction of herd animals from Africa (Petraglia & Rose, 2010). These traditional mobile pastoral systems, like those of the whole Middle East and North Africa (MENA) region, remained stable throughout most of their history (Galaty & Bonte, 1991), responding to short term variation in climate and forage availability (Schwartz, 2005). Today the MENA region is home to over 600 million people; about one third of the world's dryland population (Winslow & Thomas, 2007). Since the early twentieth century livestock numbers have increased alongside modernization, population growth and widespread national land reforms. Traditional land preservation systems such as the *harim* (preservation of natural environments) and *hema* (protection of resources for use) have broken down and open grazing systems have ensued (Blench, 1995; Gallacher & Hill, 2008; Louhaichi & Tastad, 2010). The increased use of four-wheel drive vehicles and supplementary feedstuffs has placed greater pressure on rangelands (Sidahmed, 1992; Blench, 1995; Thomson *et al.*, 2000; Louhaichi & Tastad, 2010).

It has been estimated that 85% of rangelands in the MENA region are degraded (Lal, 2002) and that over 90% of the total land area of the Arabian Peninsula suffers some form of desertification, and 44% is severely degraded (Peacock *et al.*, 2003; Breulmann *et al.*, 2007). Current pastoral systems in Arabia may be considered less sustainable to those of continental Africa due to longer drought periods, higher levels of water and soil salinity, smaller rangeland areas, unsustainable supplementary feed production, and reluctance amongst livestock owners to sell surplus unproductive animals (Peacock *et al.*, 2003). In many MENA countries, camels which are no longer required for transport, have been replaced by more profitable livestock like goats and sheep (Sidahmed, 1992; Blench, 1995). However in the Arabian Peninsula, particularly in the UAE, Saudi Arabia and Oman, camel ownership has increased as a secondary income or a hobby, and predominantly due to social and cultural reasons (Gallacher, 2010). To provide an idea of the popularity of camels in these Arab nations, we call attention to the 57 million USD of prize money won at the 2018 King Abdulaziz Camel Festival in Saudi Arabia. We have a poor understanding of the motivations for livestock ownership (which will vary between countries, regions and families) and of the socio-ecological systems surrounding modern pastoralism, including its ecological impacts, in the Arabian Peninsula.

## 1.6 Pastoralism in the Dhofar Mountains

In the Sultanate of Oman, the Al Hajar Mountains (near Muscat) in the north and the Dhofar Mountains (near Salalah) in the south comprise the two main rangeland areas, with the rest of the country dominated by gravel plains or desert dune systems. The Dhofar Mountains, on which this research is focused, are part of a mountain belt that lies on the southern coast of the Al Mahra Governorate in Yemen and the Dhofar governorate in Oman (Figure 1.1).

Much of the south coast of the Arabian Peninsula receives a mean annual precipitation of 100 mm but 250 mm is received in mountainous areas in Al Mahra and Dhofar (Ghazanfar & Fisher, 1998). Most of the precipitation is received during the southwest monsoon, known locally as the *Khareef*. Between June and September, south-western winds cause an upwelling of cold deep sea water off the coast, lowering the sea temperature to c. 18 degrees. The warmer moist winds blowing over it are subsequently cooled to dew point and a bank of dense fog forms against the south-facing mountain escarpments. It is prevented from moving northwards over the mountains due to a flow of warm dry air from the desert (Whitehead *et al.,* 1988).

The Khareef fog supports the *Hybantho durae-Anogeissetum dhofaricae* (Kürschner et al., 2004), a drought-deciduous forest community endemic to Al Mahra and Dhofar (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2007, 2008; Friesen *et al.*, 2018), which we refer to herein as the *Anogeissus* forest. It has been labelled a ‘cloud forest’ as the estimated quantity of water captured through horizontal precipitation by the endemic and dominant *Anogeissus dhofarica* tree (250% more than rainfall) is amongst the highest recorded for any tree species (Friesen *et al.*, 2018). The density of the Khareef fog is much higher a few meters above than close to the ground (Price, Al-Harthy and Whitcombe, 1988; 34-35 l/m2 per day at 4.2 m, 13 l/m2 at 0.9 m height) and thus trees capture substantially more water than grasses. Hildebrandt and Eltahir (2006) estimated net precipitation which reached the ground to be three times as high as rainfall.

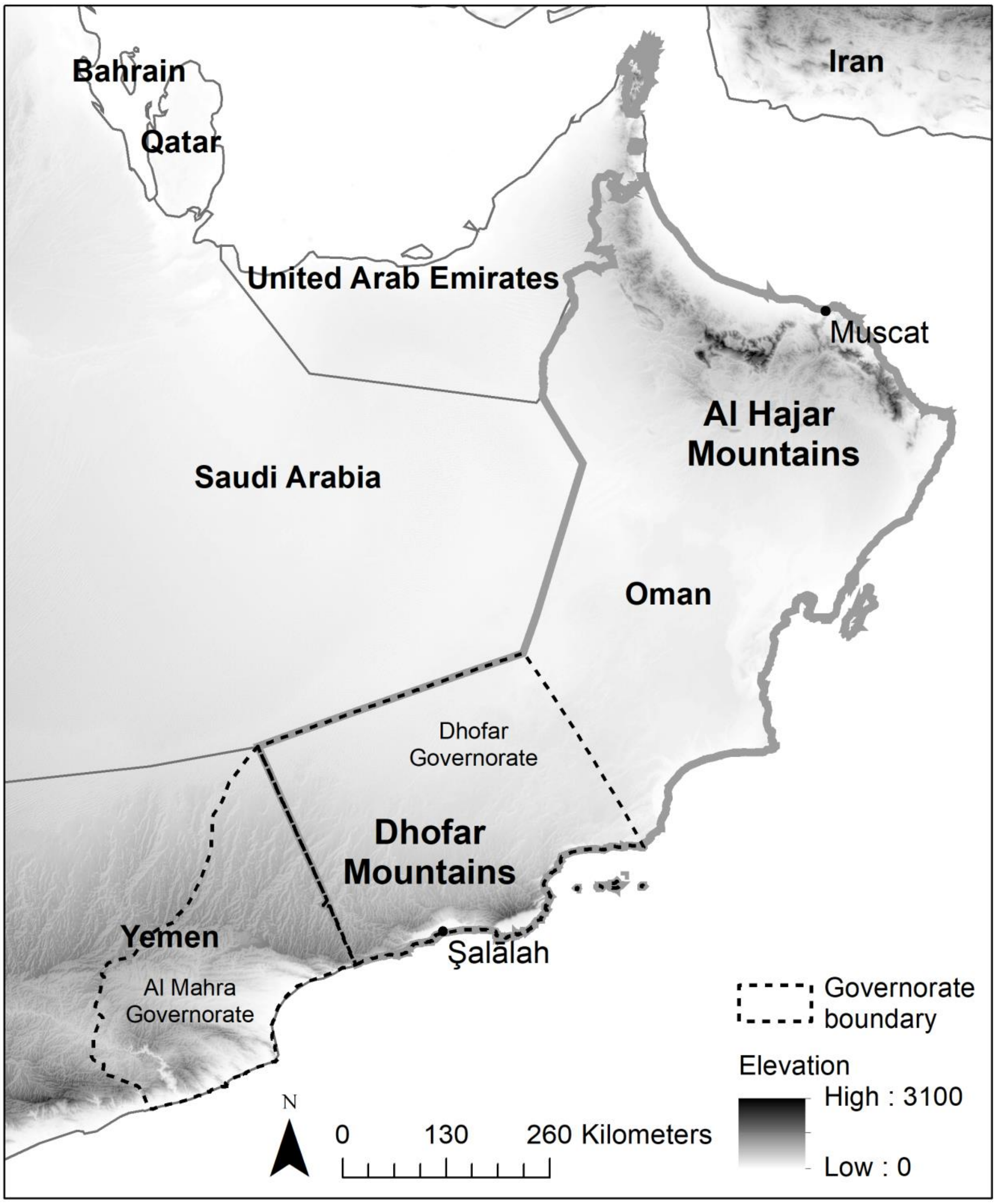



**Figure 1.1. Map of the south-east Arabian Peninsula showing the mountain regions of Oman. The Dhofar Mountains are part of a mountain belt that lies on the southern coast of the Al Mahra Governorate in Yemen and the Dhofar governorate in Oman.**

The Dhofar Mountains are part of the Horn of Africa biodiversity hotspot (Mittermeier *et al.*, 2004). At least 817 vascular plant species have been described from the Dhofar region, of which 145 are endemic, near endemic or regionally endemic (Patzelt, 2015). The critically endangered Arabian leopard *Panthera pardus nimr* has a global stronghold in Dhofar (Spalton *et al.*, 2006; Breitenmoser *et al.*, 2010). It is a flagship and umbrella species in Oman and has been the focus of substantial research effort (Spalton & Al Hikmani, 2014). In addition, regionally threatened Arabian subspecies of wolf *Canis lupus arabs*, hyena *Hyaena hyaena sultana* and caracal *Caracal caracal schmitzi*, and other smaller mammals, as well as globally important bird (Ball, Al Fazari & Borrell, 2015) and endemic reptile (Ball & Borrell, 2016) populations also persist.

This ecological richness, supported by the annual Khareef fog, has enabled pastoralism in the region for millennia. Prior to the 1960s, the pastoral communities of Dhofar lived traditional semi-nomadic lifestyles. They relied on their small herds of livestock for nutrition and survival. Dhofar at this time was an independent province from the Sultanate of Muscat and Oman and exploited under the rule of Sultan Said bin Taimur. In 1962 a rebellion began, led by the Dhofar Liberation Front (DLF). Over the following years, the rebellion grew stronger, supported by South Yemen and China, and by 1970 rebels controlled the entire Dhofar Mountains. On 23 July 1970 Said bin Taimur was deposed, and went in to exile in London. His son, Sultan Qaboos bin Said replaced him, and continues to rule today. With the help of the British Special Air Services (SAS) the rebellion was defeated in January 1976 (DeVore, 2012). One pivotal factor that ceased the rebellion was the offer of amnesty to the rebels and the promise of a job in the Sultan's 'Firqat' military forces. The Firqat centres remain operational and offer employment to many livestock keepers in rural areas today. This relatively recent political turbidity is practically undetectable in the fabric of today's society but the almost instantaneous nationwide development that occurred thereafter, driven by oil revenue, shaped modern Oman and had pertinent effects on the trajectory of pastoralism in the Dhofar region.

Since the 1970s livestock numbers have increased dramatically and it has been repeatedly reported that the Dhofar mountain ecosystems are becoming degraded due to unsustainable stocking rates of camels, cattle, and to a lesser extent goats, by rural pastoralists (Lamprey, 1976; Lawton, 1978; Oman Office of the Government Adviser for Conservation of the Environment, 1980; Wilson & MacLeod, 1991; Ghazanfar, 1998; Peacock *et al.*, 2003; Hedges & Lawson, 2006; Tardelli & Raffaelli, 2006; Directorate-General of Nature Conservation, 2010; El-Mahi, 2011b). There are concerns current stocking rates are reducing biological productivity (Peacock *et al.*, 2003) and the efficiency of ecosystem services (Kürschner *et al.*, 2004; Galletti, 2015), and undermining biodiversity conservation efforts (Spalton & Al Hikmani, 2014; Al Hikmani *et al.*, 2015; Ball, Al Fazari & Borrell, 2015). It has also been reported that the rangelands are becoming dominated by unpalatable species, as palatable species fail to regenerate (Ghazanfar, 1998; Peacock *et al.*, 2003; Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005; Patzelt, 2012). Hildebrandt & Eltahir (2008) state that camel and cattle browsing has led to loss of woody cover, and facilitated forest-grassland transitions. This is particularly concerning given the importance of horizontal precipitation capture to the mountain ecohydrology and the local water economy (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2008).

Data for the whole of Oman shows an increasing trend in cattle and camel populations (Figure 1.2), but there is no accurate long-term data on livestock numbers in Dhofar due to insufficient sampling and unreliable data collection. However, the increase in livestock populations in Dhofar since the 1970s has been described as dramatic and exponential (A Spalton 2014, personal communication, 10 August).

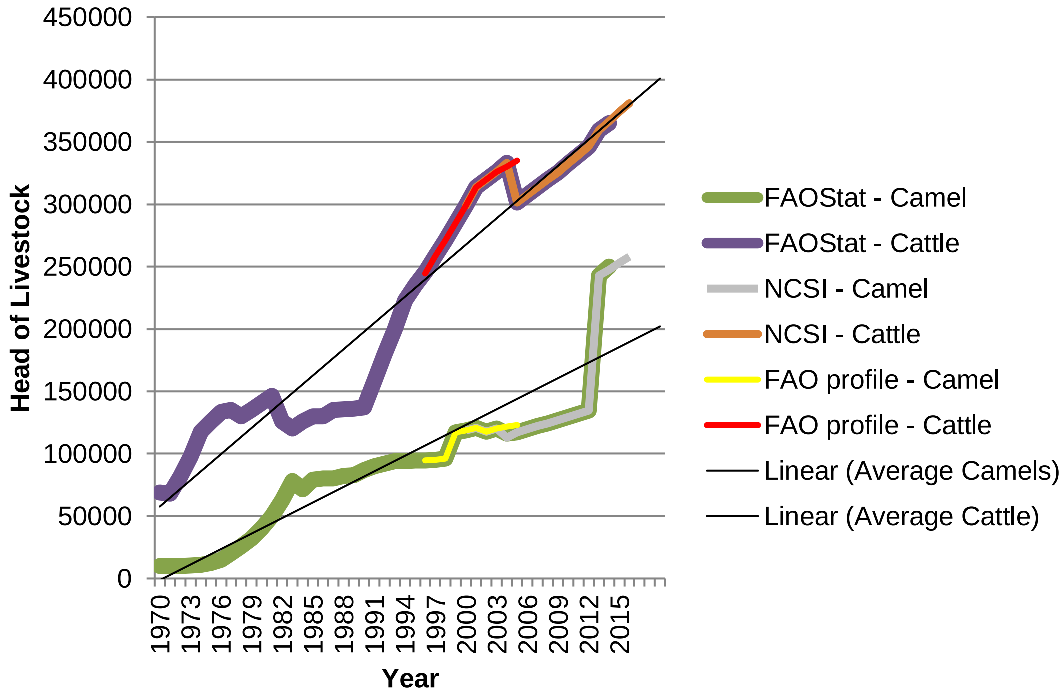



**Figure 1.2. Available long-term and short-term livestock numbers datasets for Oman. FAOStat: (FAO, 2013), NCSI: (Oman National Centre for Statistics and Information, 2017b), FAO profile: (Al-Mashaki & Koll, 2007). FAOStat data for the 1970-80s is mostly FAO estimates.**

Over the past decades, a number of studies, projects and working groups developed reports and action plans that included objectives to address overstocking in Dhofar. Notable projects include HTSL (1978), GRM International (1982), Janzen (1990), JICA (1990), Mott MacDonald International Ltd. (1991), Al-Kuthairi (1992), ESCWA, UNEP & FAO (1993) and Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD (2005). In the period 1984-1993, the 'Planning Committee for Development and Environment in the Southern Region (PCDESR)', made up of international planning, socio-economic and ecological specialists produced a number of valuable reports and technical working papers. Most notable of these are the Regional Development Plan (WS Atkins International, 1989) and Sub-Regional Land Use Plans (WS Atkins International, 1990) designed to support and coordinate sustainable development, including objectives to tackle overstocking (Whitcombe, 1998). In 1993 the PCDESR was subsumed within a national planning agency and the momentum towards sustainable livestock management was lost. Despite their value these reports were underutilised by the government.

The government of Oman has made some efforts to tackle overstocking. Destocking programs were conducted in 1983-1989 and 2000-2003, the latter in coincidence with the National Symposium on Desertification in Dhofar (Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005). The government bought 90% of each camel herd, but the majority of livestock owners gave false herd size information in order to minimise livestock loss. In the period 1986-1989 eighteen fenced exclosures were established in the Dhofar Mountains by the Rangeland Regeneration Project in the Southern Region of Oman (GRM International, 1989) to conduct a study to compare the biomass of forage within and outside the exclosures. They found the exclosures yielded 81% more forage. Unfortunately, these results did not motivate management actions and the exclosures have since fallen into disrepair. Overall, these projects and interventions have had a negligible impact on inhibiting livestock population growth in Dhofar.

## 1.7 The study area – Jabal Qamar

Three separate mountain ranges constitute the Dhofar Mountains (Figure 1.3 inset map). These are, from West to East, Jabal Qamar (highest altitude 1393 m), Jabal Qara (1277 m) and Jabal Samhan (1765 m). This research is focused on Jabal Qamar, where the first author has conducted research for the last seven years.

Jabal Qamar receives more precipitation than the other mountain ranges and boasts the highest botanical diversity (515 vascular species) of any area in Oman (Patzelt, 2015). Variants of the drought-deciduous *Anogeissus* forest (Kürschner *et al.*, 2004) are dominant on the south-facing escarpments, with sparse *A. dhofarica* woodland and grasslands (dominated by *Arthraxon* spp., *Apluda mutica* and *Themeda quadrivalvis*) in flatter areas. Grasslands in the monsoon-influenced zones of the Dhofar Mountains have been considered a result of historical forest clearance in favour of pastures for cattle (Kürschner *et al.*, 2004). At elevations over 1000 m a.s.l. the *Euphorbia balsamifera* cushion shrub community dominates (Patzelt, 2015). The main geologic formation in Jabal Qamar is limestone of tertiary origin. Layers of the Hadramout group are present. These are, from bottom to top, the Umm Er Radhuma (UER), the Rus (RUS) and the Dammam (DAM) formations (Friesen *et al.*, 2018).

Jabal Qamar comprises two administrative districts, the Wilayah of Dhalkut (west) and the Wilayah of Rakhyut (east). The coastal towns of Dhalkut and Rakhyut are located to the west and east, respectively, of a large biodiverse wadi (seasonal river valley) known as Wadi Sayq (Ball, 2014; Ball, Al Fazari & Borrell, 2015; Ball & Borrell, 2016). Our study focused on the mountaintop plateau and southern mountain slopes (the monsoon-influenced area south of the main Highway 47) between Sarfait at the Oman-Yemen border in the West, to the village of Sha'at at the eastern end of the mountain range. We did not study the northern desert slopes (*Nejd*) as few people and livestock reside there.

There are seventy-five permanently and ten seasonally (Khareef) inhabited villages in Jabal Qamar (Figure 1.3) with a total human population of 7,799 (Oman National Centre for Statistics and Information, 2017a). Livestock-owning households are present in all villages. The 2015 national livestock census recorded 15,164 camels in 802 holdings, 27,522 head of cattle in 1,060 holdings, and 14,217 goats in 439 holdings (Oman National Centre for Statistics and Information, 2017b). Based on these statistics livestock outnumber people 7 to 1. Camels are moved in a transhumance system to the mountain plateau during the Khareef to avoid soft mud and biting flies (El-Mahi, 2011a).

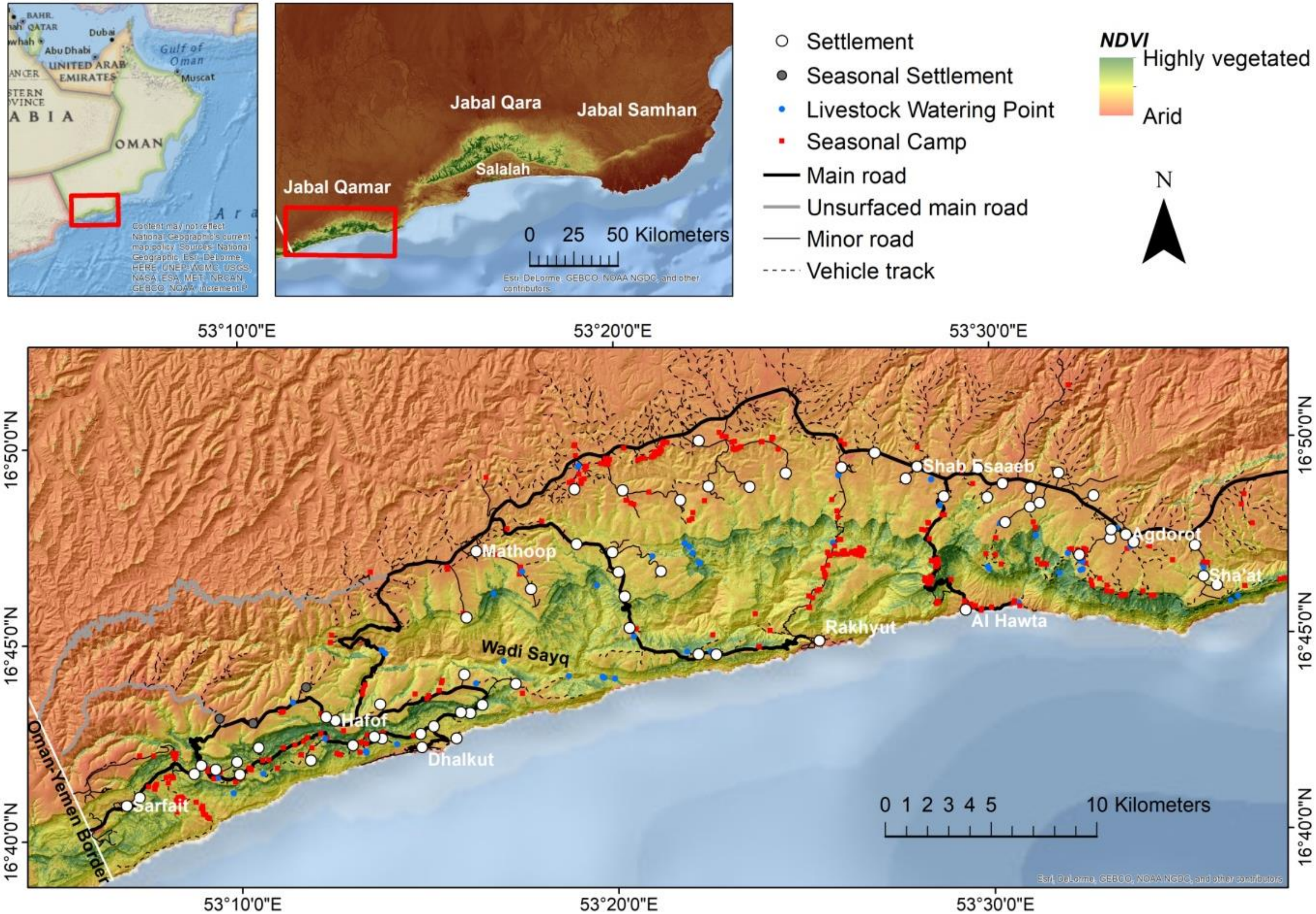



**Figure 1.3. Map of Jabal Qamar showing locations of settlements, watering points, seasonal camps, roads and vehicular trails, overlaid on a vegetation greenness (NDVI) base map. Two inset maps show the whole Dhofar Mountains and their location in Oman.**

Household sizes are often large with over ten members spanning several generations. Most women do not work but rather look after the children and take care of the household. Most poorly-educated males work at Firqat centres which have persisted since the Dhofar rebellion, or in other government-paid positions such as school bus drivers. Better educated males are generally in higher-earning employment such as high-ranking government positions, teachers in higher education, or owners of private businesses (H Al Hikmani 2018, personal communication, 10 September). Unemployment levels are high among young adults and jobs in public-facing roles in public sector services such as restaurants and garages are filled by expatriate workers. Regarding livestock husbandry, men usually tend to the camels and cattle and women usually tend to the goats and occasionally also cattle.

## 1.8 Thesis outline and objectives

The Dhofar Mountains represent a unique rangeland case study, with atypical social, cultural, political, economic and ecological situations, which could provide valuable insights for rangeland science. The rangelands provide valuable ecosystem services to the local population, yet the threat of overstocking, despite being well-recognised, has received little scientific attention. In addition, the *Anogeissus* forests are understudied, unique on a global scale and support a wealth of endemic and/or threatened biodiversity. Therefore, this thesis which utilises contemporary methods from the social, ecological and rangeland sciences, aims to (1) understand the social processes driving overstocking in rural Dhofar and (2) assess the impacts of overstocking on vegetation communities.

This study is interdisciplinary, combining a mixed-methods approach from the social sciences (Chapter 2) with multivariate analysis of vegetation communities (Chapter 3 and 4), and remote sensing and plant species distribution modelling (Chapter 5). It represents the first detailed analysis of overstocking in the region and therefore aims to provide evidence to inform local decision making and to provide a foundation upon which to build future research for regional development and conservation.

Chapter 2 uses a mixed-methods approach from the social sciences, and a socio-ecological systems framework, to identify variables driving overstocking in Dhofar. It also serves to provide a detailed description of the pastoral system and local attitudes and behaviours.

Chapter 3 quantifies vegetation responses to biotic and abiotic variables using ordination and mixed-effects models, with a focus on the effects of stocking rates on species composition and structure of the woody plant layer.

Chapter 4 identifies and describes six new habitat variants of the *Anogeissus* forest using cluster and indicator species analysis and reviews associated topoclimatic conditions, vegetation characteristics and disturbance factors.

Chapter 5 employs a novel method to quantify long-term deforestation in the study area. Species distribution models are stacked to provide a historical baseline range of the *Anogeissus* forest which is then analysed in relation to unforested areas. The cartographic outputs provide a means to visualise the probability of anthropogenic deforestation.

In the final discussion (Chapter 6) we summarise our key findings, discuss our contributions to rangeland science and examine the implications of our research for local conservation. We highlight avenues for future research and propose a concept for sustainable intensification of livestock production in Jabal Qamar.

## 1.9 Literature cited

Al-Kuthairi A. M. (1992) Forests and Pastoral Activities in Dhofar Mountains: Study of Strategy for their Rehabilitation. Arabian Gulf University, Bahrain.

Al-Mashaki M. S. A. & Koll E.-H. B. A. (2007) Country pasture/forage resource profiles: Oman. Rome: FAO.

Asner G. P., Elmore A. J., Olander L. P., Martin R. E. & Harris A. T. (2004) Grazing Systems, Ecosystem Responses, and Global Change. Annual Review of Environment and Resources, 29, 261–299.

Augustine D. J. & Decalesta D. (2003) Defining deer overabundance and threats to forest communities: From individual plants to landscape structure. Ecoscience, 10, 472–486.

Ball L. (2014) An investigation of odonate communities within Wadi Sayq, Dhofar Province, Oman (Insecta: Odonata). Check List, 10, 857–863.

Ball L. & Borrell J. (2016) An inventory of herpetofauna from Wadi Sayq, Dhofar, Oman. Journal of Threatened Taxa, 8, 9454–9460.

Ball L., Al Fazari W. & Borrell J. (2015) Birds of Wadi Sayq, Dhofar, Oman: British Exploring Society expeditions January–March 2012 and 2013. Sandgrouse, 37, 2–12.

Behnke R. H. (2000) Equilibrium and non-equilibrium models of livestock population dynamics in pastoral Africa: their relevance to Arctic grazing systems. Rangifer, 20, 141–152.

Belnap J. & Lange O. L. (2003) Biological Soil Crusts: Structure, Function, and Management. Berlin: Springer.

Blench R. (1995) Rangeland degradation and socio-economic changes among the Bedu of Jordan: results of the 1995 IFAD survey. London: Pastoral Development Network & Overseas Development Institute.

Boyd C. S. & Svejcar T. J. (2009) Forum Managing Complex Problems in Rangeland Ecosystems. Rangeland Ecology & Management, 62, 491–499.

Breitenmoser U., Breitenmoser C., Mallon D. & Edmonds J.-A. (2010) Strategy for the Conservation of the Leopard in the Arabian Peninsula. Sharjah: Environment and Protected Areas Authority & SSC Cat Specialist Group.

Breulmann M., Böer B., Wernery U., Wernery R., Shaer H. El, Alhadrami G., Gallacher D., Peacock J., Chaudhary S. A., Brown G., et al. (2007) The Camel from Tradition To Modern Times. Doha: UNESCO.

Briske D. D., Sayre N. F., Huntsinger L., Fernandez-Gimenez M., Budd B. & Derner J. D. (2011) Origin, persistence, and resolution of the rotational grazing debate: Integrating human dimensions into rangeland research. Rangeland Ecology and Management, 64, 325–334.

Briske D. D. (2017) Rangeland Systems: Processes, Management and Challenges. Switzerland: Springer Nature.

Burns W. C. (1995) The International Convention to Combat Desertification: Drawing a Line in the Sand? Michigan Journal of International Law, 16, 831–882.

Clements F. E. (1916) Plant Succession: An Analysis of the Development of Vegetation. Washington: Carnegie Institution of Washington (Carnegie Institution of Washington publication).

Clements F. E. (1920) Plant Indicators: The Relation of Plant Communities to Process and Practice. Washington: Carnegie Institution of Washington (Carnegie Institution of Washington publication).

Costanza R., D'Arge R., de Groot R., Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neill R. V., Paruelo J., et al. (1998) The value of the world's ecosystem services and natural capital. Nature, 387, 253–260.

Côté S. D., Rooney T. P., Tremblay J., Dussault C. & Waller D. M. (2004) Ecological Impacts of Deer Overabundance. Annual Review of Ecology, Evolution, and Systematics, 35, 113–147.

Curry P. J. & Hacker R. B. (1990) Can pastoral grazing management satisfy endorsed conservation objectives in arid Western Australia? Journal of Environmental Management, 30, 295–320.

Dereje M. & Uden P. (2005) The browsing dromedary camel: I. Behaviour, plant preference and quality of forage selected. Animal Feed Science and Technology, 121, 297–308.

DeVore M. R. (2012) A more complex and conventional victory: Revisiting the Dhofar counterinsurgency, 1963-1975. Small Wars and Insurgencies. Taylor & Francis Group, 23, 144–173.

Directorate-General of Nature Conservation (2010) Fourth National Report To the Convention on Biological Diversity. Muscat: Ministry of Environment and Climate Affairs.

Drewry J. J., Cameron K. C. & Buchan G. D. (2008) Pasture yield and soil physical property responses to soil compaction from treading and grazing - A review. Australian Journal of Soil Research, 46, 237–256.

El-Mahi A. T. (2011a) Hematophagous Flies in Dhofar (Sultanate of Oman): A Limiting Factor and a Potential Pathogenic Vector. Sultan Qaboos University Journal - Agricultural and Marine Sciences, 16, 65–73.

El-Mahi A. T. (2011b) Old ways in a changing space: the issue of camel pastoralism in Dhofar. Sultan Qaboos University Journal - Agricultural and Marine Sciences, 16, 51–64.

Eldridge D. J. (1998) Trampling of microphytic crusts on calcareous soils, and its impact on erosion under rain-impacted flow. Catena, 33, 221–239.

Ellis J. E. & Swift D. M. (1988) Stability of African Pastoral Ecosystems: Alternate Paradigms and Implications for Development. Journal of Range Management, 41, 450.

ESCWA, UNEP & FAO (1993) The national plan of action to combat desertification in the Sultanate of Oman. Amman: ESCWA, Manama: UNEP, Rome: FAO.

FAO (2013) FAOSTAT database collections. Rome. (Accessed: 9 August 2017).

Fernandez-Gimenez M. E. & Allen-Diaz B. (1999) Testing a non-equilibrium model of rangeland vegetation dynamics in Mongolia. Journal of Applied Ecology, 36, 871–885.

Friesen J., Zink M., Bawain A. & Müller T. (2018) Hydrometeorology of the Dhofar cloud forest and its implications for groundwater recharge. Journal of Hydrology: Regional Studies. Elsevier, 16, 54–66.

Galaty J. G. & Bonte P. (1991) Herders, Warriors, and Traders: Pastoralism in Africa. Michigan: Westview Press.

Gallacher D. J. (2010) Arid rangeland degradation in an oil-rich Gulf state; inertia of perceived heritage and pro-agricultural policies. in Veress, B. and Szigethy, J. (eds) Horizons in Earth Science Research. New York: Nova Science Publishers, 335–350.

Gallacher D. J. & Hill J. P. (2008) Intensification of rangeland grazing in an oil-rich state; causes, consequences and possible solutions. Dubai: Zayed University.

Galletti C. (2015) Understanding Environmental Change and Biodiversity in a Dryland Ecosystem through Quantification of Climate Variability and Land Modification: The Case of the Dhofar Cloud Forest, Oman. Arizona State University.

Gerber P. J., Steinfeld H., Henderson B., Mottet A., Opio C., Dijkman J., Falcucci A. & Tempio G. (2013) Tackling climate change through livestock – A global assessment of emissions and mitigation opportunities. Rome: FAO.

Ghazanfar S. A. (1998) Status of the flora and plant conservation in the sultanate of Oman. Biological Conservation, 85, 287–295.

Ghazanfar S. A. & Fisher M. (1998) Vegetation of the Arabian Peninsula. Dordrecht: Kluwer Academic Publishers.

Gibbs H. K. & Salmon J. M. (2015) Mapping the world's degraded lands. Applied Geography, 57, 12–21.

Godfray H. C. J., Beddington J. R., Crute I. R., Haddad L., Lawrence D., Muir J. F., Pretty J., Robinson S., Thomas S. M. & Toulmin C. (2010) Food Security: The Challenge of Feeding 9 Billion People. Science, 327, 812–818.

Grice A. C. & Hodgkinson K. C. (2002) Global Rangelands: Progress and Prospects. Wallingford: CABI Publishing.

GRM International (1982) Range and Livestock Survey. Brisbane: G.R.M. International Pty Ltd.

GRM International (1989) Rangeland Regeneration Project in the Southern Region. Brisbane: G.R.M. International Pty Ltd.

Gunderson L. H. (2002) Panarchy: Understanding Transformations in Human and Natural Systems. Washington: Island Press.

Hardin G. (1968) The Tragedy of the Commons. Science, 162, 1243–1248.

Hedges S. & Lawson D. (2006) Monitoring the Illegal Killing of Elephants: Dung Survey Standards for the Mike Programme. Nairobi: CITES MIKE Programme.

Herbel C. (1979) Utilization of Grass- and Shrublands of the South-Western United States. in Walker, B. (ed.) Management of Semi-Arid Ecosystems. New York: Elsevier, 161–203.

Herrero M., Havlik P., Valin H., Notenbaert A., Rufino M. C., Thornton P. K., Blummel M., Weiss F., Grace D. & Obersteiner M. (2013) Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. Proceedings of the National Academy of Sciences, 110, 20888–20893.

Herrick J. E., Lessard V. C., Spaeth K. E., Shaver P. L., Dayton R. S., Pyke D. A., Jolley L. & Goebel J. J. (2010) National ecosystem assessments supported by scientific and local knowledge. Frontiers in Ecology and the Environment, 8, 403–408.

Herskovits M. J. (1926) The Cattle Complex in East Africa. American Anthropologist, 28, 230–272.

Hiernaux P., Bielders C. L., Valentin C., Bationo A. & Fernandez-Rivera S. (1999) Effects of livestock grazing on physical and chemical properties of sandy soils in Sahelian rangelands. Journal of Arid Environments, 41, 231–245.

Al Hikmani H., Zabanoot S., Al Shahari T., Zabanoot N., Al Hikmani K. & Spalton A. (2015) Status of the Arabian Gazelle, Gazella arabica (Mammalia: Bovidae), in Dhofar, Oman. Zoology in the Middle East, 61, 295–299.

Hildebrandt A. & Eltahir E. A. B. (2006) Forest on the edge: Seasonal cloud forest in Oman creates its own ecological niche. Geophysical Research Letters, 33, 2–5.

Hildebrandt A. & Eltahir E. A. B. (2007) Ecohydrology of a seasonal cloud forest in Dhofar: 2. Role of clouds, soil type, and rooting depth in tree-grass competition. Water Resources Research, 43, 1–13.

Hildebrandt A. & Eltahir E. A. B. (2008) Using a horizontal precipitation model to investigate the role of turbulent cloud deposition in survival of a seasonal cloud forest in Dhofar. Journal of Geophysical Research, 113, 1–11.

Holechek J. L., Pieper R. D. & Herbel C. H. (2001) Range Management: Principles and Practices. Upper Saddle River: Prentice Hall.

HTSL (1978) Livestock Sector Review and Project Identification - Final Report, Volumes I and II. Hemel Hempstead: Hunting Technical Services Limited.

Janzen J. (1990) Land Tenure and Land Use in the Pastoral Living Areas of Dhofar, Sultanate of Oman – Tradition, Change and Development Perspectives. Muscat: FAO & Land Tenure and Land Use Consultancy.

JICA (1990) Ten year Master Plan for Agricultural Development. Tokyo: Japan International Co-operation Agency.

Knight A. T., Cowling R. M., Rouget M., Balmford A., Lombard A. T. & Campbell B. M. (2008) Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. Conservation Biology, 22, 610–617.

Kürschner H., Hein P., Kilian N. & Hubaishan M. A. (2004) The Hybantho durae-Anogeissetum dhofaricae ass. nova - Phytosociology, structure and ecology of an endemic South Arabian forest community. Phytocoenologia, 34, 569–612.

Lal R. (2002) Carbon sequestration in dryland ecosystems of West Asia and North Africa. Land Degradation & Development, 13, 45–59.

Lamprey H. F. (1976) Preliminary Proposals for the Conservation and Management of the Jebel Qara Region. Nairobi: UNESCO Regional Office of Science and Technology for Africa.

Lamprey H. F. (1983) Pastoralism yesterday and today: The overgrazing problem. in Bourlière, F. (ed.) Tropical Savannas (Ecosystems of the World). Amsterdam: Elsevier, 643–666.

Lawton R. M. (1978) A reconnaissance survey of the Jabal Qara grazing land ecosystem, with particular reference to the impact of development. London: Overseas Development Agency, Land Resources Development Centre.

Louhaichi M. & Tastad A. (2010) The Syrian steppe: Past trends, current status, and future priorities. Rangelands, 32, 2–7.

Lynam T. J. P. & Stafford Smith M. (2004) Monitoring in a complex world - Seeking slow variables, a scaled focus, and speedier learning. African Journal of Range and Forage Science, 21, 69–78.

Meyfroidt P. (2018) Trade-offs between environment and livelihoods: Bridging the global land use and food security discussions. Global Food Security, 16, 9–16.

Millennium Ecosystem Assessment (2005) Ecosystems and Human Well-being: Desertification Synthesis. Washington: World Resources Institute.

Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD (2005) National Action Programme to Combat Desertification in the Sultanate of Oman. Muscat: Ministry of Regional Municipalities, Environment, and Water Resources.

Mittermeier R. A., Gil P. R., Hoffman M., Pilgrim J., Brooks T., Mittermeier C. G., Fonseca G. A. B. da & Lamoreux J. (2004) Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions. Edited by R. A. Mittermeier. Rugby: CEMEX.

Mott MacDonald International Ltd. (1991) National Resources Master Plan. Cambridge.

Mottet A., de Haan C., Falcucci A., Tempio G., Opio C. & Gerber P. (2017) Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. Global Food Security, 14, 1–8.

Mysterud A. (2006) The concept of overgrazing and its role in management of large herbivores. Wildlife Biology, 12, 129–141.

Nabarro D. & Wannous C. (2014) The potential contribution of livestock to food and nutrition security: the application of the One Health approach in livestock policy and practice. Rev. sci. tech. Off. int. Epiz, 33, 475–485.

Norberg J. & Cumming G. S. (2008) Complexity Theory for a Sustainable Future. New York: Columbia University Press.

Oba G., Stenseth N. C. & Lusigi W. J. (2000) New perspectives on sustainable grazing management in arid zones of sub-Saharan Africa. BioScience, 50, 35–51.

Oman National Centre for Statistics and Information (2017a) 2017 Statistical Year Book. Muscat: National Centre for Statistics and Information.

Oman National Centre for Statistics and Information (2017b) National Centre for Statistics and Information Data Portal. (Accessed: 22 August 2017).

Oman Office of the Government Adviser for Conservation of the Environment (1980) The scientific results of the Oman flora and fauna survey 1977 (Dhofar), Journal of Oman Studies Special Report No. 2. Muscat: Ministry of Information and Culture.

Ostrom E. (1990) Governing the Commons: The Evolution of Institutions for Collective Action. 14th edn. Cambridge: Cambridge University Press.

Ostrom E. (2007) A diagnostic approach for going beyond panaceas. Proceedings of the National Academy of Sciences., 104, 15181–15187.

Patzelt A. (2012) Notes on the Flora and Vegetation of Wadi Kharfut on Jabal Qamar, Oman. in Borrell, J. et al. (eds) The British Exploring Society Empty Quarter Expedition 2012. London: British Exploring Society, 94–100.

Patzelt A. (2015) Synopsis of the Flora and Vegetation of Oman, with Special Emphasis on Patterns of Plant Endemism. Abhandlungen der Braunschweigischen Wissenschaftlichen Gesellschaft, 282–317.

Peacock J. ., Ferguson M. ., Alhadrami G. A., McCann I. R., Al Hajoj A., Saleh A. & Karnik R. (2003) Conservation through utilization: a case study of the indigenous forage grasses of the Arabian Peninsula. Journal of Arid Environments, 54, 15–28.

Perevolotsky A. & Seligman N. G. (1998) Role of Grazing in Mediterranean Rangeland Ecosystems. BioScience, 48, 1007–1017.

Petersen B. & Snapp S. (2015) What is sustainable intensification? Views from experts. Land Use Policy, 46, 1–10.

Petraglia M. D. & Rose J. I. (2010) The Evolution of Human Populations in Arabia. New York: Springer.

Reichman O. J., Jones M. B. & Schildhauer M. P. (2011) Challenges and opportunities of open data in ecology. Science, 331, 703–705.

Reynolds J. F., Stafford Smith D. M., Lambin E. F., Turner B. L., Mortimore M., Batterbury S. P. J., Downing T. E., Dowlatabadi H., Fernández R. J., Herrick J. E., et al. (2007) Ecology: Global desertification: Building a science for dryland development. Science, 316, 847–851.

Ripple W. J., Larsen E. J., Renkin R. A. & Smith D. W. (2001) Trophic cascades among wolves, elk and aspen on Yellowstone National Park's northern range. Biological Conservation, 102, 227–234.

Robin R. & Ellis J. (1995) Impacts of Pastoralists on Woodlands in South Turkana, Kenya: Livestock-Mediated Tree Recruitment. Ecological Applications, 5, 978–992.

Robinson T. P., Thornton P. K., Franceschini G., Kruska R. L., Chiozza F., Notenbaert A., Cecchi G., Herrero M., Epprecht M., Fritz S., et al. (2011) Global livestock production systems. Rome: FAO & ILRI.

Savadogo P., Sawadogo L. & Tiveau D. (2007) Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. Agriculture, Ecosystems and Environment, 118, 80–92.

Sayre N. F., Debuys W., Bestelmeyer B. T. & Havstad K. M. (2012) 'The range problem' after a century of rangeland science: New research themes for altered landscapes. Rangeland Ecology and Management, 65, 545–552.

Sayre N. F., McAllister R. R., Bestelmeyer B. T., Moritz M. & Turner M. D. (2013) Earth Stewardship of rangelands: Coping with ecological, economic, and political marginality. Frontiers in Ecology and the Environment, 11, 348–354.

Sayre N. F. (2017) The Politics of Scale: A History of Rangeland Science. Chicago: University of Chicago Press.

Scheffer M., Carpenter S., Foley J. A., Folke C. & Walker B. (2001) Catastrophic shifts in ecosystems. Nature, 413, 591–596.

Schlesinger W. H., Reynolds J. F., Cunningham G. L., Huenneke L. F., Jarrell W. M., Virginia R. A. & Whitford W. G. (1990) Biological feedbacks in global desertification. Science, 247, 1043–1048.

Schwartz H. J. (2005) Ecological and Economic Consequences of Reduced Mobility in Pastoral Livestock Production Systems. in Fratkin, E. and Roth, E. A. (eds) As Pastoralists Settle. Boston: Springer US, 69–86.

Scoones I. (1995) Living with Uncertainty: New Directions in Pastoral Development in Africa. Rugby: Intermediate Technology Publications.

Sidahmed A. E. (1992) Sustainable Rangelands in the Near East and North Africa. Rangelands, 14, 201–205.

Spalton A., Al Hikmani H. M., Willis D. & Said A. S. B. (2006) Critically Endangered Arabian leopards Panthera pardus nimr persist in the Jabal Samhan Nature Reserve, Oman. Oryx, 40, 287–294.

Spalton A. & Al Hikmani H. (2014) The Arabian Leopards of Oman. 1st edn. London: Stacey International.

Stafford Smith M. (1996) Management of Rangelands: Paradigms at their Limits. in Hodgson, J. and Illius, A. (eds) The Ecology and Management of Grazing Systems. Wallingford: CAB International, 325–357.

Staver A. C., Bond W. J., Stock W. D., Rensburg S. J. Van & Waldram M. S. (2009) Browsing and fire interact to suppress tree density in an African savanna. Ecological Applications, 19, 1909–1919.

Tardelli M. & Raffaelli M. (2006) Some aspects of the vegetation of Dhofar (Southern Oman). Bocconea, 19, 109–112.

Thomson E. F., von Kaufmann R., Li Pun H., Treacher T. & van Houten H. (2000) Global Agenda for Livestock Research. Nairobi: International Livestock Research Institute.

Vetter S. (2005) Rangelands at equilibrium and non-equilibrium: Recent developments in the debate. Journal of Arid Environments, 62, 321–341.

Waltner-Toews D., Kay J. J. & Lister N.-M. E. (2008) The Ecosystem Approach: Complexity, Uncertainty, and Managing for Sustainability. New York: Columbia University Press.

Wardle D. (2002) Communities and Ecosystems: Linking the Aboveground and Belowground Components. Princeton: Princeton University Press.

Westoby M., Walker B. & Noy-Meir N. (1989) Opportunistic management for rangelands not at equilibrium. Journal of Range Management, 42, 266–274.

Whitcombe R. P. (1998) Botany, environment and planning in drylands. in Prendergast, H, D, V. et al. (eds) Plants for Food and Medicine. London: Royal Botanic Gardens, Kew, 203–221.

Whitehead E. E., Hutchinson C. F., Timmermann T. M. & Varady R. G. (1988) Arid Lands: Today and Tomorrow. Colorado: Westview Press.

Wiens J. A. (1984) On Understanding a Non-Equilibrium World: Myth and Reality in Community Patterns and Processes. in Strong, D. R. et al. (eds) Ecological Communities: Conceptual Issues and the Evidence. Princeton: Princeton University Press, 439–457.

Wilson A. & MacLeod N. (1991) Overgrazing: Present or absent? Journal of Range Management, 44, 475–482.

Winslow M. & Thomas R. (2007) Desertification in the Middle East and North Africa: warning signs for a global future. Agriculture and Rural Development, 1, 10–12.

Wood M. K. & Blackburn W. H. (1981) Grazing Systems: Their Influence on Infiltration Rates in the Rolling Plains of Texas. Journal of Range Management, 34, 331–335.

WS Atkins International (1989) Regional development plan for the Southern Region, Final Report, August 1989. Epsom: WS Atkins International & Planning Committee for Development and Environment in the Southern Region (PCDESR).

WS Atkins International (1990) Subregional land use plans for the Southern Region, Final Report, March 1990. Epsom: WS Atkins International & Planning Committee for Development and Environment in the Southern Region (PCDESR).

# 2 Application of a socio-ecological systems framework to understand overstocking by modern livestock keepers in the Dhofar Mountains of Oman

## 2.1 Abstract

Livestock numbers in the Dhofar Mountains of Oman have increased substantially since the 1970s and there are concerns widespread overstocking is degrading the unique cloud forest ecosystem and the services it provides. We used a mixed-methods data collection approach with livestock keepers and applied a socio-ecological systems framework to understand the social processes motivating livestock ownership and the endogenous and exogenous forces leading to overstocking. Thirty-four framework variables were found to be relevant. Our results reveal how factors associated with the recent and rapid development of Oman have transformed the relationships between pastoralists, their livestock and the rangelands. Feedstuff provision for most of the year has decreased dependence on the rangelands, leaving little incentive for self-organization, collective action or sustainable use. We find livestock accumulation is primarily motivated by cultural values, despite the financial costs from feedstuff provision and poor market access. However, we find evidence of changing values and a disengagement with livestock keeping amongst the wealthy or better educated population. Some of these processes have been recognised in previous studies, whilst some are unique, but their amalgamation in Dhofar results in a novel, and ultimately destructive pastoral system – a situation which requires urgent attention from policy makers.

## 2.2 Introduction

It is widely accepted that overstocking of camels, cattle, and to a lesser extent goats since the 1970s is degrading the natural environment of the Dhofar Mountains in Oman (Lamprey, 1976; Lawton, 1978; Oman Office of the Government Adviser for Conservation of the Environment, 1980; Wilson & MacLeod, 1991; Ghazanfar, 1998; Peacock *et al.*, 2003; Kürschner *et al.*, 2004; Tardelli & Raffaelli, 2006; Hedges & Lawson, 2006; Directorate-General of Nature Conservation, 2010; El-Mahi, 2011b; Galletti, Turner & Myint, 2016). There are concerns a reduction in woody cover due to overbrowsing inhibits horizontal precipitation, a process critical to the survival of the cloud forests (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2007, 2008; Friesen *et al.*, 2018) and that overstocking is undermining biodiversity conservation efforts (Al Hikmani *et al.*, 2015), especially concerning the critically endangered Arabian leopard *Panthera pardus nimr* (Spalton & Al Hikmani, 2014).

Numerous reports and action plans have included objectives to tackle overstocking (e.g. HTSL, 1978; GRM International, 1982; WS Atkins International, 1989, 1990; Janzen, 1990; JICA, 1990; Mott MacDonald International Ltd., 1991; Al-Kuthairi, 1992; ESCWA, UNEP & FAO, 1993; Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005), yet recommendations on topics such as commodification, zonation and reforestation have not translated into national policy (Whitcombe, 1998). A requirement now exists for an updated evidence base to inform policy decisions which utilises modern developments in community-based and socio-ecological system (SES) research approaches.

In the past, scholars might have been quick to blame overstocking in Dhofar on a 'tragedy of the commons' scenario. However, in recent decades this theory has been challenged and replaced by an appreciation of distinct processes operating at multiple levels which govern the sustainability of rangeland use. For example, mobility and freedom of movement in open access rangelands can be critical to sustainable use (Fernández-Giménez, 2002; Moritz, Scholte, *et al.*, 2013) and local self-organization and collective action often successfully governs use of common-pool resources (McCabe, 1990; Ostrom, 1990; McPeak, 2005). Processes such as these, derived from the common components of decades of resource system studies, informed Ostrom's socio-ecological system (SES) framework (Ostrom, 2007; McGinnis & Ostrom, 2014).

The SES framework provides a general list of concepts that can be used to analyse a range of socio-ecological systems (SESs). It was designed to build a common vocabulary and structure for policymakers and scholars in varying disciplines to develop a coherent mode of analysis of complex SESs (Ostrom, 2007; Ostrom & Cox, 2010; Hinkel, Bots & Schlüter, 2014; McGinnis & Ostrom, 2014). One of the attractions of this framework is its flexibility when being applied to different systems where new processes, pathways, sub-categories and concepts can be appended (McGinnis & Ostrom, 2014). It encompasses the actors (A) who use resource units (RU) from a resource system (RS) according to rules and procedures determined by a governance system (GU), within the context of related ecological systems (ECO) and social-political-economic settings (S) (Figure 2.1). At the centre of the framework are the focal action situations where inputs are transformed by the actions of multiple actors through interactions (I) which produce outcomes (O). There is often feedback between action situations and the seven top tier categories, each of which contain multiple variables at the second and third tiers, the relevance of which depend on the study system and research question (McGinnis & Ostrom, 2014).

In this article we have applied Ostrom's SES framework to analyse the pastoral system in Jabal Qamar in western Dhofar which has been the primary focus of the first author's research over the last seven years. Our research question was: which social processes have led to overstocking and subsequent environmental degradation in Dhofar?

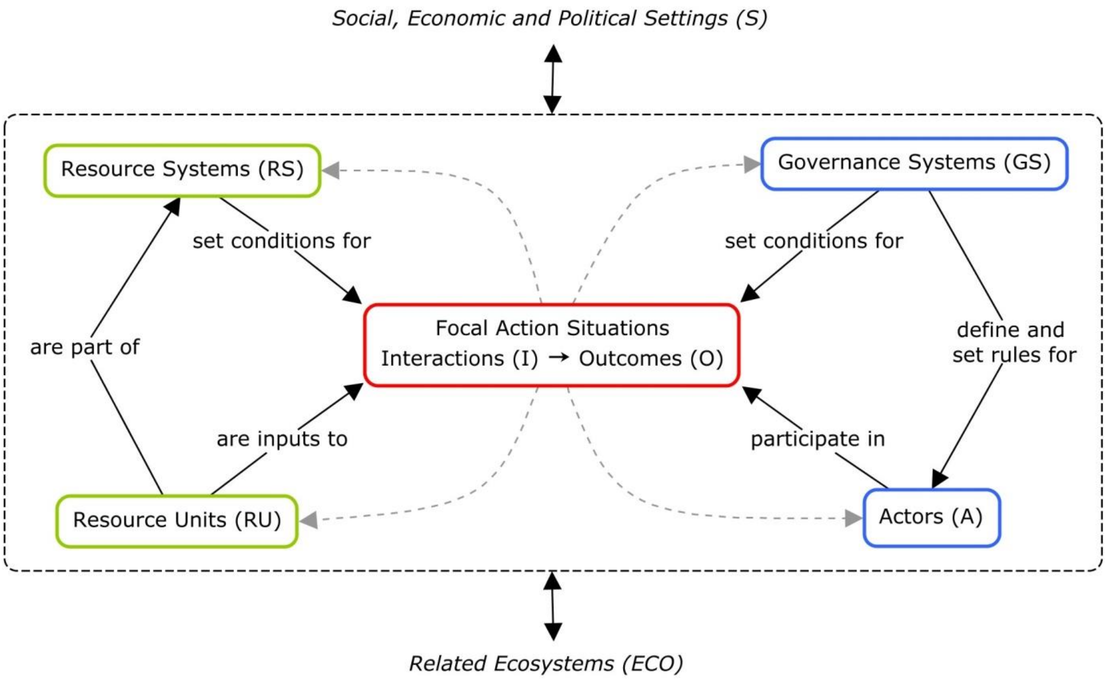



**Figure 2.1. The first-tier categories of Ostrom (2007) socio-ecological systems framework including refinements made by McGinnis and Ostrom (2014). Solid arrows represent direct links and dashed arrows represent feedback links.**

## 2.3 Study system

The Dhofar Mountains are part of a mountain belt that lies on the southern coast of Oman and eastern Yemen. A localised subtropical climate results from the annual influence of the Indian monsoon between June and September, known locally as the *Khareef*. During these months, thick fog inundates the southern mountain escarpments (Whitehead *et al*., 1988; Ghazanfar & Fisher, 1998), providing moisture for a globally unique south Arabian forest community with high levels of biological diversity and endemism (Ghazanfar, 1998; Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006; Mosti, Raffaelli & Tardelli, 2012; Ball & Borrell, 2016), which has provided forage resources for pastoralism for millennia (Petraglia & Rose, 2010; El-Mahi, 2011b).

This study was conducted in Jabal Qamar, the western-most mountain range in Dhofar (Figure 2.2), which comprises two administrative districts, the Wilayah of Dhalkut (west) and the Wilayah of Rakhyut (east). There are seventy-five permanently and ten seasonally (Khareef) inhabited villages in Jabal Qamar with a total human population of 7,799 (Oman National Centre for Statistics and Information, 2017a). Livestock-owning households are present in all villages. The 2015 national livestock census recorded 15,164 camels in 802 holdings, 27,522 head of cattle in 1,060 holdings, and 14,217 goats in 439 holdings in Jabal Qamar (Oman National Centre for Statistics and Information, 2017b). Based on these statistics livestock outnumber people 7 to 1.

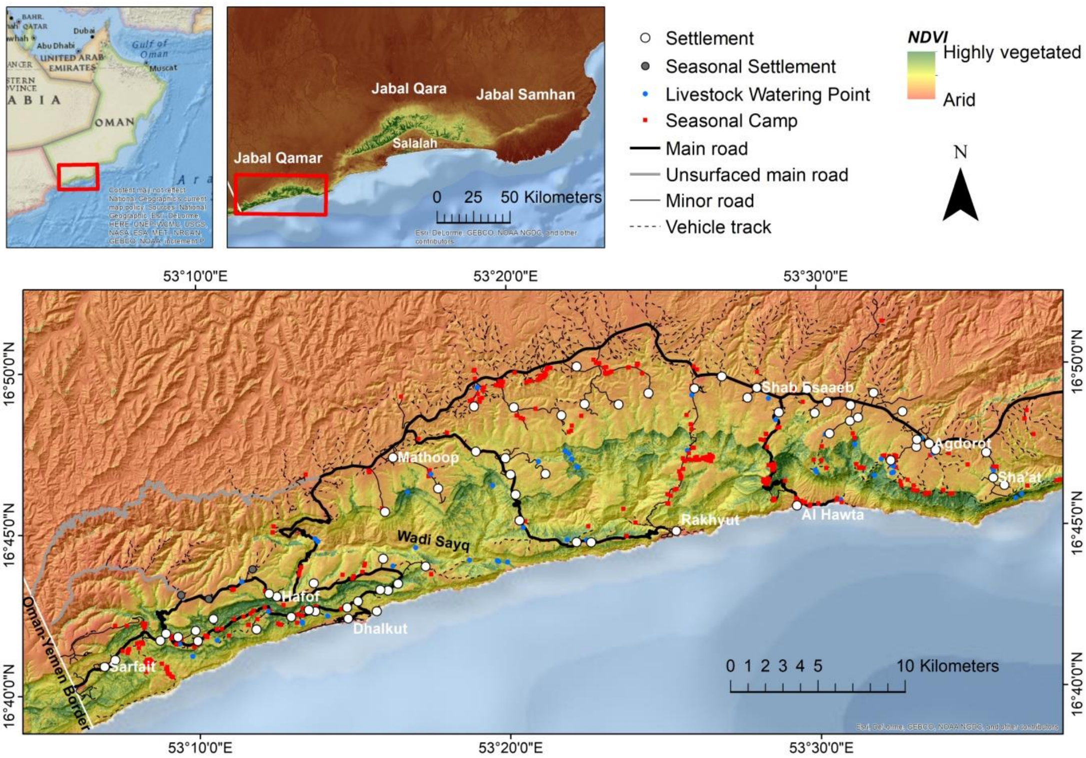



**Figure 2.2. Map of Jabal Qamar showing locations of settlements, watering points, seasonal camps, roads and vehicular trails, overlaid on a vegetation greenness (NDVI) base map. Two inset maps show the whole Dhofar Mountains and their location in Oman.**

Household sizes are often large with over ten members spanning several generations. Most women do not work but rather look after the children and take care of the household. Most poorly-educated males work at Firqat centres, formerly known as the 'Sultan's Firqat Military Forces', which have persisted since the Dhofar rebellion (1962-1976) (DeVore, 2012), or in other government-paid positions such as school bus drivers. Better educated males are generally in higher-earning employment such as high-ranking government positions, teachers in higher education, or owners of private businesses (H Al Hikmani 2018, personal communication, 10 September). Unemployment levels are high among young adults and jobs in public-facing roles in public sector services such as restaurants and garages are filled by expatriate workers. Regarding livestock husbandry, men usually tend to the camels and cattle and women usually tend to the goats.

## 2.4 Methods

We employed a mixed-methods approach involving semi-structured interviews, participatory mapping exercises and socio-economic and Likert questionnaires with livestock keepers in Jabal Qamar between April 2016 and September 2016. Three government officials and a feedstuff company manager were also interviewed and multiple in-depth interviews took place with an additional government official who worked in conservation and was formerly a livestock keeper.

A mixed-methods approach was chosen to ensure we had a range of tools to obtain a representative and holistic account of the SES (Shaffer, 2013). Topics such as socio-demographics, livestock ownership, and household economics suited quantitative data collection methods, whilst qualitative methods acquired detailed accounts of the socio-cultural processes and wider political and economic forces influencing livestock keeper attitudes and behaviours. In addition, the mixed-methods approach suited the unpredictability of the work timetable and logistics which arose from using a translator/facilitator, and enabled us to carry out within-subject, between-subject and cross method triangulation to ensure our findings were trustworthy (David & Sutton, 2011; Newing, 2011).

A British male primary investigator (lead author), a British female research assistant, and an Omani translator (and facilitator) conducted the data collection. The translator was from Jabal Qamar and spoke the local dialect of Jabali, and provided real-time translation. During the early stages of the fieldwork the relationship with the translator had to be carefully managed, as Omani culture dictates that visitors are treated as important guests and shown hospitality, with their needs met to ensure their happiness and wellbeing. Previous research by the author has found this can raise issues when working with a new translator or facilitator, wherein there is a risk that information could be changed in translation in order to minimise offence to the visiting researcher. Thus, for this study the researchers did not divulge their environmentalist views on the issue of overstocking in order to minimise the risk of biased translation. In addition, the translator was quickly incorporated into the research team and briefed on the purpose and objectives of the research to curtail the host-guest relationship. It transpired that the translator eventually comprehended the personal pro-conservation views of the researchers. However, by this stage researcher-translator rapport had been built and the reliability of the translator was no longer under question. Helpfully, the translator was familiar with principles of scientific rigour having studied as a medical technician. On occasions when an informant could speak comprehendible or fluent English, they were asked about topics that the research team had flagged as potentially unreliably translated. This was not a formal structured analysis and in all cases the translation was triangulated and found reliable.

Upon meeting with informants and following greetings and introductions, the aims of the study were explained, and informed consent was sought. Participants were assured that their responses would be confidential and anonymous. Ethical considerations followed the guidelines of the American Anthropological Association (AAA). Interviews with young people (ages 14–17) followed the guidelines of the World Association of Opinion and Marketing Research Professionals (ESOMAR). No children under the age of 14 were involved in the research.

In order to obtain a diversity of information, we defined livestock keepers (our study population) as any individuals from a household that had previously or currently kept livestock. We used cluster sampling, a probability sampling method, in which villages were randomly selected within each settlement area. Simple random sampling was then used to select households within the villages to conduct socio-economic questionnaires and interviews. Efforts were made to sample a greater number of households in large villages, known as probability proportional to size (Newing, 2011). Seven of the 84 interviews were referrals (snowball sampling) where an informant encouraged the research team to speak with friends or family and twenty-six were opportunistic when livestock keepers were encountered away from households; for example, when carrying out livestock husbandry. Most informants were males due to the cultural barriers of speaking with females but fortunately males are predominantly involved in livestock keeping.

**Table 2.1. A table outlining the types of data acquired by each survey method with details on units of measurement and the analysis methods used. The variables used in each analysis are indicated by superscript digits.**

| Method and sample size | Description of the data acquired | Unit | Analysis method |
|---|---|---|---|
| Socio-economic questionnaires | Age[2], gender[0] | Years, m/f | [0]Descriptive statistics |
| 72 households in 21 villages (25% of villages); 92% male, 8% female; aged 23-80 years (median = 40, interquartile range = 30-50, mean = 37.1 ± SE 2.08). | Village of residence[1] | Village of residence | |
| | Residential status[0] | Since village was founded/since birth/visiting family/new resident | [1]Mann-Whitney test of herd size vs. rurality (Wilayah of residence) (N=198) |
| | Household size[3] | Small (<4 residents), medium (4-8 residents), large (> 8 residents) | [2]Spearman's rho of age vs. herd size (N=198) |
| | Herd sizes[01234] | Head of livestock | [3]Kruskal-Wallis test of household size vs. herd size (N=198) |
| | Production and use of livestock products[0] | Livestock, meat/milk/ghee/hide, for sale/use at home | [4]Spearman's rho of livestock sales vs. herd size (N=72) |
| | Livestock sales[04] and prices | Animals sold per year, average sale price in OMR | |
| | Profit/loss from livestock keeping | OMR per year | Unreliable - omitted from analysis |
| | Do your children help now?[0] | Yes/No | |
| | Will your children help in the future?[0] | Yes/No | |
| Likert questionnaires | Age[2], gender[0] | Years, m/f | [5]Chi-squared test of household size vs. household income (N=126) |
| 126 households in 18 villages (22% of villages); 73% male, 27% female; aged 16-70 years (median = 35, interquartile range = 18-45, mean = 33.4 ± SE 1.28). | Village of residence[1] | Village of residence | [6]Kruskal-Wallis test of herd size vs. household income (N=126) |
| | Household size[35] | Small (<4 residents), medium (4-8 residents), large (> 8 residents) | |
| | Herd sizes[01236] | Head of livestock | |
| | Household income[0156] | High/medium/low | |
| | Test agreement with Likert items (Appendix 1) to elucidate prevalence of attitudes and behaviours. | | Classification and regression tree (CART) analysis. New tree produced with each statement and demographic data field appointed as response variable. |
| Semi-structured interviews<br>84 households in 37 villages (45% of villages); 82 male; 2 female; aged 14-80 years. | Rank reasons why keep livestock; rank problems associated with livestock keeping; why and how people keep livestock and overstocking within the contexts of governance, economics, culture and husbandry. | Qualitative notes | Coding and production of a dynamic conceptual framework (DCF). |
| Participatory mapping exercises<br>8 mapping exercises in 7 villages | Spatial and temporal arrangement of livestock activity and changes in regimes. | GIS feature layers with attributes | Cartographic representation in GIS. |

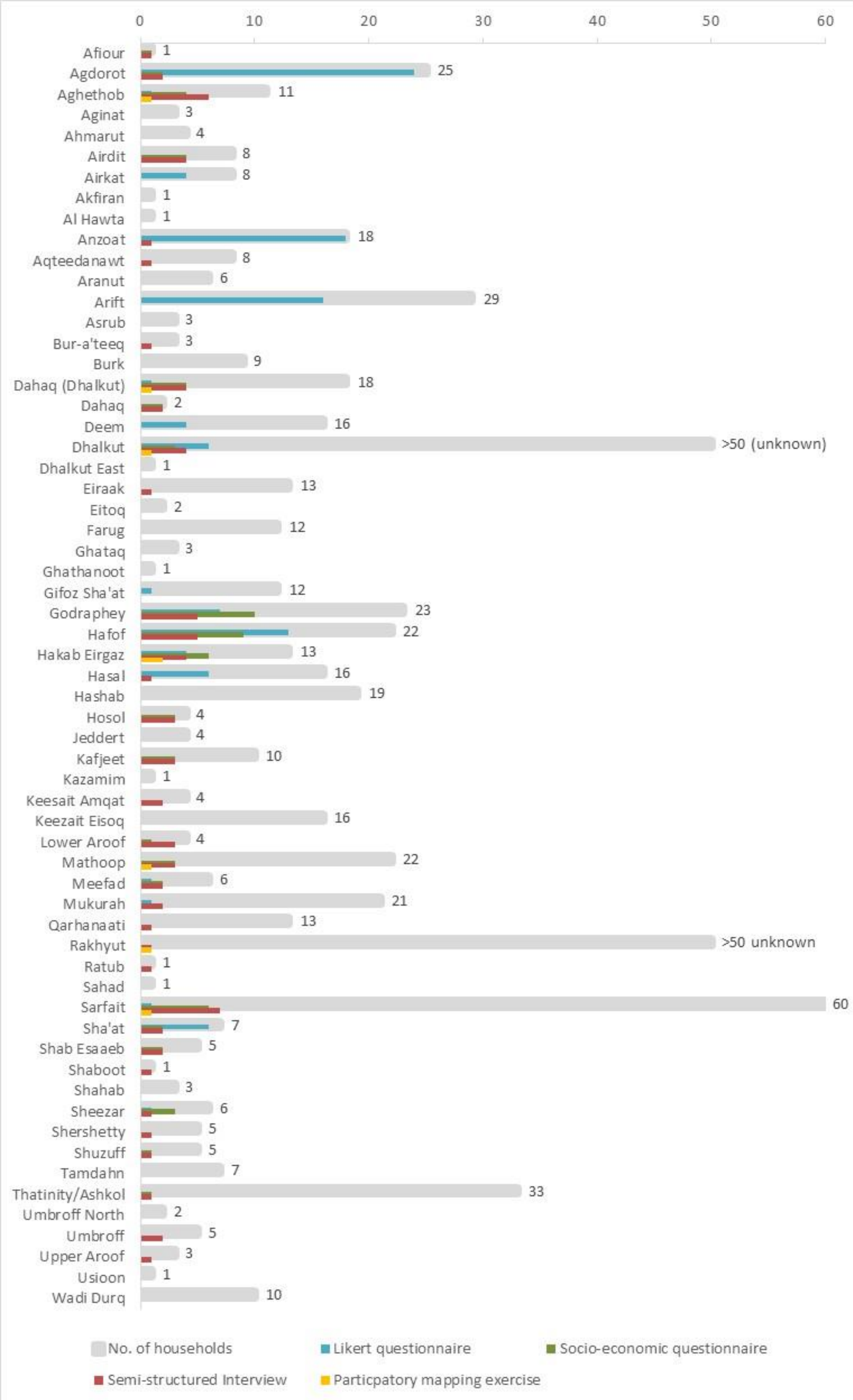



**Figure 2.3. The number of households in each settlement (shown in plot area) and the number of households sampled in each of the four survey methods.**

The data collection methods, sample sizes, types of data acquired and analysis methods are summarised in **Error! Reference source not found.**. **Error! Reference source not found.** illustrates the proportion of households sampled in each village and Figure 2.4 shows the spatial distribution of each data collection method. In Table 2.2 we provide generalised descriptions of the knowledge and behaviour of different age classes when interacting with the research team.

Socio-economic questionnaires were administered during face-to-face meetings with livestock keepers and prior to any interviews. The first section of the questionnaire collected basic socio-demographic information and the second and third sections sought information related to husbandry, economics and livelihoods (**Error! Reference source not found.**). The questionnaires were deemed to have little or no influence on interview responses.

Approximately ten percent of the interviews were conducted with groups of up to ten people whilst the remainder involved a single interviewee. The interviews followed a loose framework to allow freedom to informant narratives to identify unexpected social processes and peculiarities (Newing, 2011). The basic structure of the interview involved ranking of motivations for livestock keeping and the problems faced (**Error! Reference source not found.**). This template was sufficient to stimulate rich discussion on why and how people keep livestock and the issue of overstocking within the contexts of policy, economics, culture and husbandry. The research assistant transcribed qualitative responses from the interviews real-time, whilst the interviewer took targeted notes.

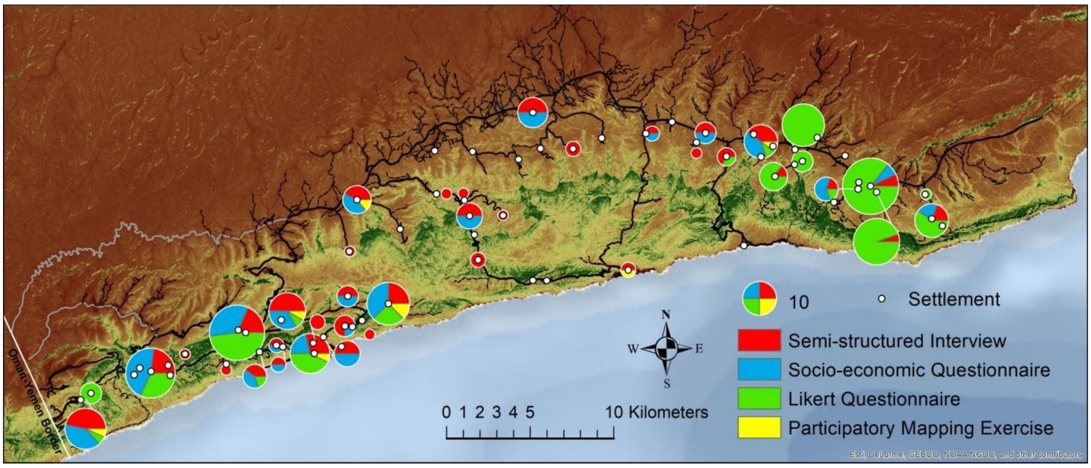


**Figure 2.4. A map showing pie charts of the spatial distribution and relative number of participants involved in semi-structured interviews, socio-economic questionnaires, Likert questionnaires and participatory mapping exercises.**

**Table 2.2. Informant age classes with generalized descriptions of their knowledge and behaviour when interacting with the research team. Reliability was judged based on triangulation of data with key informants.**

| Age bracket | Knowledge and behaviour |
|---|---|
| Young people (age <16) | Reliable information; Enthusiastic about livestock as help elders with husbandry; Less involved in teenage years; Readily provide information but possess limited historical or wider knowledge. |
| Young adult (age 16–35) | Less reliable information; Less time spent with livestock; Other interests or in college or employment (in Salalah); Good knowledge of social politics; Aim to please with lots of (often irrelevant) information; Difficult to get straight answers. |
| Adult (age 35–50) | Mostly reliable information; More time spent with livestock than young adults; Good understanding of livestock management; Good understanding of plant species; Knowledge varies greatly depending on employment and wealth class; More suspicious of research project; Provide information in anticipation of it being heard by government. |
| Older generation (> 50) | Very reliable information; Best knowledge of plant species; Best knowledge of traditional livestock management practises; Sometimes provide information in the context of traditional management; Shy, but willing to trust after some discussion; Realise the seriousness of overgrazing as they have observed degradation; Sometimes glorify the truth to exaggerate culture. |

Eight participatory mapping exercises, which sought to understand the spatial and temporal arrangement of grazing regimes (**Error! Reference source not found.**) were conducted with individuals or groups of participants. Contrary to conventional principles of participatory research, participants were not fully aware of the research aims and views of the researchers, in order to ensure reliable spatial data was provided. Participants made annotations on A0-size laminated satellite imagery maps or plotted features in ArcGIS Collector Application on an Apple iPad Air 2, depending on the preference of the participant. DigitalGlobe satellite imagery at a resolution of < 1 m provided sufficient detail on the basemap to enable participants to identify landscape features to orientate themselves and apply their information. Assistance was provided to older or less educated individuals to ensure the participants were correctly orientated with the map but the younger generation were more comfortable with the process, being accustomed to using touch-screen devices. Having the informant map their spatial information directly into GIS software cuts out the process of digitising paper maps where accuracy of spatial information can be lost (Hall & Close, 2007).

Likert questionnaires were designed towards the latter stage of the research period and distributed by teachers to one child from each household across all schools in the study area. The questionnaires instructed the adult member of the household most involved in livestock keeping to complete the questionnaire. Of the 1400 distributed, 199 were returned and 73 were omitted from analysis as they were incomplete, poorly completed or completed by an individual under the age of 16. The topics in the questionnaire were informed by the findings of the qualitative research and used to test the extent to which pastoralists agreed or disagreed with specific statements. This enabled us to elucidate the prevalence of particular attitudes and behaviours across a larger sample size. A response option of 'neutral' was not included to force respondents to commit to an agreement position. Socio-demographic data was also collected (**Error! Reference source not found.**). Socio-demographic and herd size data was pooled from both questionnaires and analysed in R Studio (R Core team, 2013) using the methods summarised in **Error! Reference source not found.**.

Classification and regression tree (CART) analysis was conducted on the Likert questionnaire data (Appendix 1) in R Studio (R Core team, 2013) using the 'rpart' package (Therneau, Atkinson & Ripley, 2017) to determine significant groupings of respondents based on their level of agreement with the Likert scale items. Prior to CART analysis the level of agreement scale was grouped into two responses of agree or disagree to increase the robustness of the relatively small sample size and facilitate interpretation of the CART results. Each statement in the Likert scale and the demographic data components were appointed as the response variable and the analysis repeated for each using the default parameters. Several trees failed and others provided no sensible results. For the latter instances, the explanatory variables obscuring logical interpretation of the trees were removed until sensible results were achieved or the tree failed. For example, the analysis for 'Wilayah' (local administrative zone) as the response variable, found age and gender as the most significant predictor variables as a result of our sampling strategy through single-gender schools.

The SES framework can be used to inform research questions or data collection or to organize or analyse findings or any combination of these (Ostrom, 2009). We applied the SES framework predominantly to discuss our findings in relation to existing resource-use concepts and to understand the complex and multi-level aspects of our study SES (Hinkel, Bots & Schlüter, 2014). Given the importance of avoiding a one-size-fits-all approach to SES research (particularly pertinent for studies on pastoralism) and to avoid manifestation of the framework variables in our inductive data collection approach, we applied the SES framework post hoc. Additionally, we believe this enables better testing of the applicability of the SES framework to new systems.

Consequently our analytical procedure was as follows. The qualitative data from interviews was routinely digitised and top level, secondary and tertiary codes were developed and assigned to the themes as the research period progressed (Newing et al., 2011). Concurrently, a dynamic conceptual framework (DCF) was constructed to map the themes and their interrelatedness (Appendix 2). Themes of interest, with conflicting responses or with unexplained phenomena were revisited with future informants until saturation was reached and the DCF proved robust. Upon completion of the fieldwork, the coding system was reviewed and revised and the DCF augmented with the results of the questionnaire analysis. Finally, the SES framework variable codes were assigned to our themes or new codes were created for unclassifiable themes.

**All framework variables were considered potentially applicable before specific variables were selected based on themes in the DCF. Thirty-four second-tier framework variables were identified to be important properties in our SES related to overstocking and environmental degradation (**

Table 2.3). Twenty-three variables were either absent from or not pertinent to the study SES or not relevant to the research question. Three third-tier variables were developed for ECO3 to classify provisioning, regulating and cultural ecosystem services which represent flows into and out of the focal SES. Two third-tier variables were developed for A2 to differentiate wealth and education as socioeconomic attributes of the resource users. An additional second-tier interaction variable termed 'reinforcement activities' was created to account for the cultural reinforcement effects of camel competitions. Here, we present and discuss our results under three central umbrella themes which were core components of the DCF: governance, culture and economics. The framework variable codes are cited in-text, as in Nagendra & Ostrom (2014).

**Table 2.3. Description of the 57 variables in the social-ecological system (SES) framework and justification for the inclusion (34) or exclusion (23) of the variables in our study. Boldface font indicates variables included in our analysis.**

| Category<br>Variable Code | Variable Name | | |
|---|---|---|---|
| Social, Economic, and Political Settings (S) | | Used | Reason for inclusion/exclusion |
| **S1** | **Economic development** | **Yes** | **Rapid economic development has transformed pastoralism.** |
| **S2** | **Demographic trends** | **Yes** | **Attitudes and actions vary within the population.** |
| S3 | Political stability | No | No notable political instability affects pastoralism. |
| S4 | Other governance systems | No | Other governance systems not relevant. |
| **S5** | **Markets** | **Yes** | **Cheaper imported livestock outcompetes local livestock in national food market. Consumer demand for cheap produce. Local taste for local meat drives small market system.** |
| S6 | Media organizations | No | Media not interested in livestock activities. |
| **S7** | **Technology** | **Yes** | **Feedstuff manufacturing and distribution. Four-wheel drive vehicles have affected livestock management techniques.** |
| Related Ecosystems (ECO) | | | |
| **ECO1** | **Climate patterns** | **Yes** | **Climate affects growth rate and spatial and temporal distribution of RU (RU2 and RU7).** |
| ECO2 | Pollution patterns | No | Pollution not pertinent to overstocking. |
| **ECO3** | **Flows into and out of focal SES** | **Yes** | **Ecosystem services.** |
| **ECO3a** | **Provisioning services** | **Yes** | **Water, fire wood, frankincense, non-timber forest products (e.g. honey, mushrooms, fruits and seeds) and livestock forage resources.** |
| **ECO3b** | **Regulatory services** | **Yes** | **Capture and storage of water, erosion and flood control, carbon** |

| | | | |
|---|---|---|---|
| | | | **sequestration, decomposition and pest control.** |
| **ECO3c** | **Cultural services** | **Yes** | **Maintenance of traditional pastoralist culture, scientific research, tourism and recreation.** |
| Resource Systems (RS) | | | |
| RS1 | Sector (e.g., water, forests, pasture, fish) | No | A rangeland system. |
| RS2 | Clarity of system boundaries | No | System boundaries are relatively clear, although could be interpreted differently by different analysts. |
| **RS3** | **Size of resource system** | **Yes** | **Resource system size varies by location and topography.** |
| **RS4** | **Human-constructed facilities** | **Yes** | **Livestock management features (e.g. water troughs, camps) have affected management practises and distribution of livestock.** |
| **RS5** | **Productivity of the system** | **Yes** | **Productivity of the resource system is low.** |
| **RS6** | **Equilibrium properties** | **Yes** | **Equilibrium rangeland system where livestock, sustained on feedstuffs, are primary drivers of vegetation change.** |
| RS7 | Predictability of system dynamics | No | System dynamics such as climate are predictable, an attribute of equilibrium rangelands. |
| RS8 | Storage characteristics | No | Resources are not stored. |
| RS9 | Location | No | Jabal Qamar, Dhofar, Oman. |
| Resource Units (RU) | | | |
| RU1 | Resource unit mobility | No | Forage resources not mobile. |
| **RU2** | **Growth or replacement rate** | **Yes** | **Long term growth and replacement rate is low or negative.** |
| RU3 | Interaction between resource units | No | Interaction may exist between herbaceous (grazing) and woody (browsing) forage resources, but not pertinent to this discussion. |
| **RU4** | **Economic Value** | **Yes** | **Forage resources have a low economic value.** |
| **RU5** | **Number of Units** | **Yes** | **Number of units is insufficient for current stocking rates.** |
| **RU6** | **Distinctive characteristics** | **Yes** | **RU can be made available through tree branch management practises.** |
| **RU7** | **Spatial or temporal distribution** | **Yes** | **The spatial and temporal distribution of RU is highly variable.** |
| Actors (A) | | | |
| **A1** | **Number of relevant actors** | **Yes** | **The number of resource users and the diversity of stakeholders are highly relevant.** |
| **A2** | **Socioeconomic attributes** | **Yes** | **Socioeconomic attributes of the resource users affects their attitudes and actions.** |

| | | | |
|---|---|---|---|
| **A2a** | **Wealth** | **Yes** | **Individual or household wealth affects attitudes and actions.** |
| **A2b** | **Education** | **Yes** | **Levels of education affect attitudes and actions.** |
| **A3** | **History or past experiences** | **Yes** | **Recent history has shaped aspects of livestock keeper's attitudes and actions.** |
| **A4** | **Location** | **Yes** | **The location of the resource user and livestock (e.g. house or camp) influences overstocking.** |
| **A5** | **Leadership/ entrepreneurship** | **Yes** | **Some are giving up pastoralism. Few make an income from livestock keeping.** |
| **A6** | **Norms (trust-reciprocity)/ social capital** | **Yes** | **Social norms are fundamental drivers of livestock ownership.** |
| **A7** | **Knowledge of SES/mental models** | **Yes** | **Knowledge varies with age and affects attitudes and actions.** |
| **A8** | **Importance of resource (dependence)** | **Yes** | **Dependence on resource has changed following increased livestock populations and feedstuff provision.** |
| A9 | Technologies available | No | Included under S7. |
| Governance Systems (GS) | | | |
| **GS1** | **Government organizations** | **Yes** | **Ministry of Environment and Climate Affairs, Ministry of Agriculture and Fisheries, Office for Conservation of the Environment, Ministry of Heritage and Culture.** |
| **GS2** | **Nongovernmental organizations** | **Yes** | **Feedstuff production companies (e.g. Dhofar Cattle Feed Co.) and other livestock-related companies. A number of consultancies have produced action plans to tackle overstocking.** |
| **GS3** | **Network structure** | **Yes** | **Unrecognised responsibilities and ineffective state governance system.** |
| **GS4** | **Property-rights systems** | **Yes** | **State ownership. Transformation from informal tribal territories to unrestricted access.** |
| **GS5** | **Operational-choice rules** | **No** | **Several gates limit movement of livestock to southern escarpments following the Khareef.** |
| GS6 | Collective-choice rules | No | No formal rules determining collective-choice outcomes. |
| GS7 | Constitutional-choice rules | No | No formal rules determining constitutional-choice outcomes. |
| **GS8** | **Monitoring and sanctioning rules** | **Yes** | **National laws sanctioning damage to biodiversity are not enforced. Two unsuccessful destocking programs occurred in 1983-1989 and 2000-2003** (Ministry of Regional Municipalities Environment and Water |

| | | | |
|---|---|---|---|
| | | | Resources & UNEP & UNCCD, 2005) **when livestock owners gave false hard size information to minimise livestock loss.** |
| Interactions (I) | | | |
| **I1** | **Harvesting** | **Yes** | **Spatio-temporal variation in browsing and/or grazing activity.** |
| I2 | Information sharing | No | Not a significant variable in our study. |
| I3 | Deliberation processes | No | Not a significant variable in our study. |
| I4 | Conflicts | No | Conflicts occurred in the past, some enduring tribal values persist but no conflicts occur. |
| I5 | Investment activities | No | No investment activities at current. Plans exist for future investment in dairy produce. |
| I6 | Lobbying activities | No | Livestock keepers would like the price of feedstuffs to be reduced but no lobbying occurs. |
| I7 | Self-organizing activities | No | No adaptive livestock management strategies or other self-organizing activities occur. |
| I8 | Networking activities | No | No regular networking activities. In 2016 government sent message to livestock keepers via instant messaging to delay movement of livestock. |
| I9 | Monitoring activities | No | No monitoring activities occur. |
| I10 | Evaluative activities | No | No evaluative activities occur. |
| **I11** | **Reinforcement activities** | **Yes** | **Camel competitions.** |
| Outcomes (O) | | | |
| **O1** | **Social performance measures (e.g., efficiency, equity, accountability, sustainability)** | **Yes** | **Household income/loss from livestock production. Loss of ecosystem services. Preservation of pastoral culture.** |
| **O2** | **Ecological performance measures (e.g., overharvested, resilience, biodiversity, sustainability)** | **Yes** | **Loss of ecosystem services. See ECO3.** |
| **O3** | **Externalities to other SESs** | **Yes** | **Water economy and feedstuff production.** |

## 2.5 Results and discussion

### 2.5.1 Descriptive results

The gender and ages of interviewees and questionnaire respondents are summarised in **Error! Reference source not found.**. Socio-economic questionnaire respondents were either a resident since the founding of the village (34%), a resident in their village from birth (52%), living away from home and visiting their family (8%) or had moved into a village (6%). Pooling the data from both questionnaires, out of livestock-owning households in 28 villages (34% of villages), 81% owned camels, 91% owned cattle and 36% owned goats. Camel herd sizes ranged from 1 to 200 (median = 27, interquartile range = 15-40, mean = 31.7 ± SE 2.29), cattle herd sizes ranged from 3 to 250 (median = 35, interquartile range = 20-50, mean = 41.1 ± SE 2.5) and goat herd sizes ranged from 1 to 300 (median = 25, interquartile range = 15-41.5, mean = 38.85 ± SE 5.9). Herd sizes did not significantly differ with household size for individual livestock types (camel: ($H(2) = 4.8$, $p = 0.089$) ; cattle: ($H(2) = 5.2$, $p = 0.073$); goats: ($H(2) = 4.3$, $p = 0.114$), however for total livestock, small households had significantly less livestock then medium and large households ($H(2) = 14.3$, $p = < 0.001$). There was no significant correlation between age ($r_s = -0.11$, $p = 0.133$) or rurality ($W = 4690.5$, $p = 0.795$) and herd sizes. Some questionnaire data on profit or loss from livestock keeping was found to be exaggerated in anticipation of greater financial support from the government and thus was excluded from further analysis.

Interviewees were involved in livestock-related activities to varying degrees, and spent varying amounts of time at their family home. Motivations for keeping livestock, in order of importance, were: (1) inherited from parents; (2) financial security; and (3) produce for the household. Problems associated with livestock keeping ranked by livestock keepers, in order of importance, were: (1) bark stripping behaviour; (2) lack of grazing resources; (3) expensive feedstuffs; (4) weakening Khareef; (5) the construction of buildings and roads; and (6) vehicle damage caused by off-road driving.

### 2.5.2 Governance

Prior to Oman’s renaissance in 1970, pastoralist families in Dhofar were subsistent goat or cattle herders that lived in primitive stone and wood huts or caves. No piped water or veterinary care and high disease prevalence prevented them from keeping significant numbers of livestock (Janzen, 1983). They practised a semi-nomadic transhumance system based on seasonal variation in climatic and habitat conditions **(ECO1, RU7)** (Al-Mashaki & Koll, 2007). Traditional tribal land tenure institutions regulated the use of water and forage resources (Al-Mashaki & Koll, 2007).

Following the Dhofar Rebellion (1962-1975) (DeVore, 2012) the region rapidly developed. The government installed a water supply network, improved road infrastructure, constructed high capacity livestock watering troughs, built dams at springs **(RS4)** and established a system of manufacturing and distributing subsidised feedstuffs **(S7)**. This led to a reduction in pastoralist mobility as family's sedentarized in villages close to the new amenities. Similar reduced mobility has been common in North Africa and the Middle East (Blench, 1995; Masri, 2001; Nedjraoui, 2001; Louhaichi & Tastad, 2010). The traditional tribal land tenure institutions broke down and were replaced by an open access system in the state owned rangelands **(GS4)** (Rouchiche *et al.*, 2003; Spalton *et al.*, 2006; Al-Mashaki & Koll, 2007; El-Mahi, 2011b). The increasing use of four-wheel drive vehicles promoted a more opportunistic stocking strategy in previously inaccessible areas **(S7)** (Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005; Victor, 2012). Such a scenario has also been reported from Jordan where the Bedu replaced their traditional land tenure institutions in favour of a more opportunistic system using trucks to transport feedstuff, water and livestock (Blench, 1995; Masri, 2001).

State control of previously participant controlled resources tends to be less effective (Blench, 1995; Ostrom, 1999; Louhaichi & Tastad, 2010), although open access does not necessarily result in overexploitation (Ostrom, 1990, 1999; Moritz, Scholte, *et al.*, 2013; Moritz, 2016). Contrary to Hardin's theory of the tragedy of the commons (Hardin, 1968), local rules **(GS5, GS6)**, information sharing **(I2)**, deliberation processes **(I3)**, and self-organising **(I7)**, monitoring **(I9)** and evaluative **(I10)** activities have since been reported to govern sustainable use of rangeland resources in open access systems (Oba & Lusigi, 1987; Kohler-Rollefson, 1992; Moritz, Catherine, *et al.*, 2013; Moritz, Scholte, *et al.*, 2013; Harris *et al.*, 2016). However, there is little evidence of such activities **(I)** taking place in Dhofar over the last five decades **(GS4)**, which we attribute to three key properties.

Firstly, a steadfast and upward trend in livestock ownership **(A1)** which rapidly exceeded a naturally low baseline carrying capacity **(RU2, RU5, ECO1, RS6)**, deemed self-organising activities inscrutable and ineffectual. Indeed, Ostrom (2009) identified the 'number of users' **(U1)** and 'productivity of the system' **(RS5)** as two key variables influencing self-organization. The system shifted suddenly from high resource abundance to low resource abundance, with only a short window during which livestock keepers could have perceived a need to manage for the future **(RS5)** (Ostrom, 2009). Secondly, unlike pastoral SESs in developing nations (Moritz, Scholte, *et al.*, 2013), where livelihoods and wellbeing depend on rangeland-based livestock production, livestock keepers in Dhofar have not been compelled to manage for the future **(A8)**. Finally, daily provisioning of feedstuff for the majority of the year substantially reduced the dependence of livestock and livestock keepers on forage resources **(RU4, S7, A8)**, and masked a requirement for mutual agreement on resource use and self-organising activities (Ostrom, 1990). This has been described among the Bedu of Jordan (Blench, 1995) and can be pinpointed as a key factor in the status quo of overstocking in Dhofar. A low dependence leaves little or no motivation for conservation of the resource. Accordingly, Ostrom (2009) identified the 'importance of the resource' **(U8)** as another key variable influencing self-organization.

Livestock keepers in Dhofar today still follow a transhumance regime (Figure 2.5) to avoid the adverse conditions within the monsoon-influenced zone during the Khareef (El-Mahi, 2011a; Patzelt, 2015), which are uncomfortable for people and considered fatal for camels. Interviewees stated that the period of abundant forage availability following the Khareef has shortened considerably in recent decades **(RU2)** and explained that feedstuff provision in the morning and evening has led to low livestock dispersal and localised overstocking and habitat degradation close to villages, farmsteads and seasonal camps **(A4)**. In these areas livestock frequently strip the bark from trees (Appendix 3), and livestock keepers bend, break and cut tree branches for their livestock (Appendix 4), rather than herding their animals deeper into areas with accessible forage. Furthermore, the enthusiastic older generation are becoming less mobile and expatriate workers and wealthy and educated livestock keepers **(A2a, A2b)** do not want to herd livestock deep into the rangelands **(S2, S7)**, preferring to rely on vehicular access to grazing locations. One informant explained:

"*People should take the camels and cows down for many months to spread the grazing pressure. Instead they hang around the area close to the house. People are lazy, not taking the livestock far enough.*"

This information is suggestive of a 'grazing piosphere', which is a zone of ecological impact around a watering point (Andrew, 1988). In Dhofar, the definition extends to villages, farmsteads and camps, and implies that remote areas may be less degraded.

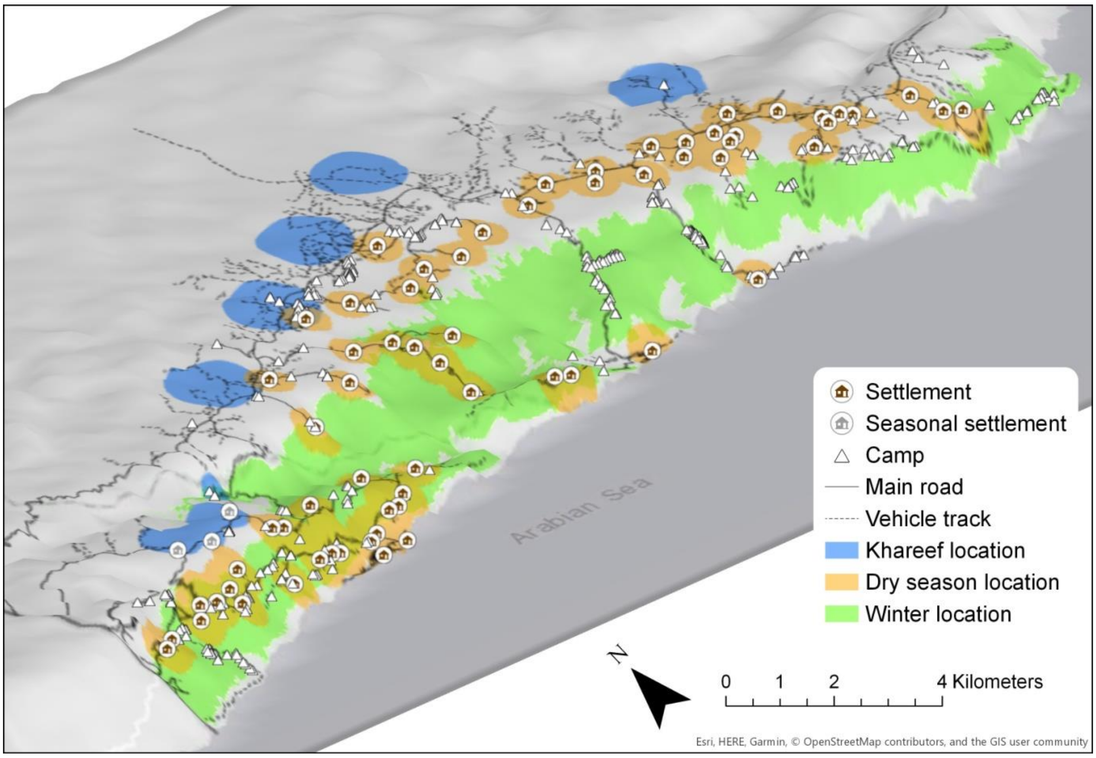



**Figure 2.5. A three dimensional map of Jabal Qamar showing the three locations of the transhumance management regime. Khareef location (July-September): The abundant moisture stimulates high rates of vegetative growth, mould invades property, soils become saturated and hematophagous flies are abundant so keepers move with their herds to the drier mountaintop plateau (c. 1000 m a.s.l.) where livestock are sustained on feedstuff. Winter location (October-January): livestock are moved down into the monsoon-influenced zone to utilise the abundant vegetation. Dry season location (February-June): livestock are kept close to villages or camps and sustained on feedstuff.**

In 2003 several gates were built by local people to stop livestock and people accessing the lower escarpments before the 25th September when the ground becomes dry enough for camels **(GS5).** The government has attempted to decree a later date, to allow vegetation to set seed to aid vegetation recovery, but few livestock keepers have adhered to this **(GS8)**. In 2015, another government effort to manage overstocking in Jabal Qamar saw the erection of roadside bollards to reduce damage from off road driving, although some have since been removed by local people to regain vehicular access. State governance of pastoral activities is further hindered by confusion amongst existing environmental departments over responsibility, ineffective networking chains to procure the evidence required to formulate new policy, and an avoidance to formulate and enforce new policy that leads to conflicts of interest **(GS3)**.

### 2.5.3 Culture

The culture surrounding pastoralism in Dhofar has transformed over the last five decades as the relationship between pastoralists and their livestock has transitioned from essential to extraneous. Prior to the 1970s, camels were highly valued for transport and their rich yield of milk and meat, but camel ownership was expensive and a luxury that few families could afford. As Oman developed, non-livestock employment fostered wealth amongst pastoralists in Dhofar **(A2a)**, a scenario also recognised amongst the Jordanian Bedouin (Abu-Rabia, 2000). Camel ownership became a possibility for many more families **(A3)** who were quick to start trading their goats for camels, and camel numbers rapidly increased. A cheap expatriate workforce to carry out livestock husbandry made livestock ownership a relatively easy venture and cattle numbers also increased. Rather than fading in the face of modernisation, a 'camel culture' evolved through cultural transmission and social reproduction, which is now deeply embedded in the identity of Dhofar's rural pastoral societies. Over seventy percent of Likert questionnaire respondents agreed or strongly agreed that they would like to keep more livestock (Figure 2.6). At the individual level this culture portrays itself as an overwhelming fondness for livestock keeping, which we refer to herein as 'pastoral values' **(A3, A6)**. These pastoral values were clearly apparent in the narratives of many livestock keepers **(S2)**:

*"I spend 70% of my time with my animals and the other 30% of my time thinking about my animals. From sunrise to sunset, 12 hours, I am with my animals."*

And:

*"I want my camels more than a massive company. I have forgotten about women and children, camels are my family."*

All informants stated that the primary reason for keeping livestock was due to the inheritance of livestock from parents. Only 2-5% are said to sell inherited livestock following their parents decease. Many livestock keepers have internalized their parents' values and are extremely passionate and actively engaged in livestock husbandry. These individuals are likely to be represented in our CART analysis

(Figure 2.6) by the 53% of respondents who would like to own more livestock, do not think they own too many, and actively seek forage resources. Most of the older generation fall into this category and many possess a detailed botanical knowledge **(A3, A7)**, which would have been fundamental to survival and wellbeing **(A8)** before the 1970s (Miller, Morris & Stuart-Smith, 1988). Interestingly, a number of younger informants stated that there is a misconception amongst the older generation, who in the past relied on camels for mobility and sustenance, that livestock still have an important use today **(A3)**. One informant explained:

*"Before 40 years the animals were useful. This idea is still in people's minds, despite modernisation of the country."*

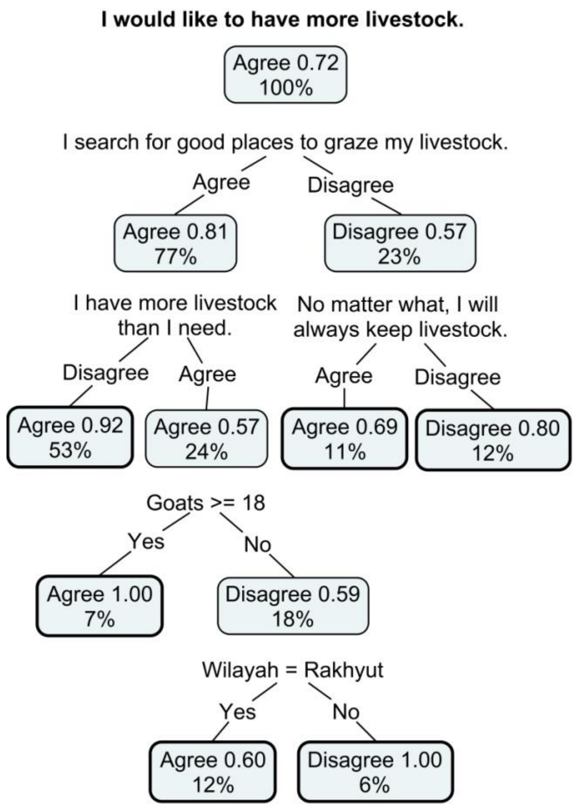



**Figure 2.6. CART analysis of Likert data showing groupings of respondents based on their agreement or disagreement with the statement 'I would like to have more livestock'. They are grouped based on their responses to the other Likert scale items including socio-demographic data. Each node shows the predicted class (agree or disagree), the predicted probability of the class and the percentage of observations in the node.**

Despite the current widespread popularity in livestock keeping, we identified that pastoral values are not homogenous within the population and levels of interest in, knowledge of and engagement with livestock keeping vary based on an individual's demography and socio-economic status. Most children are very passionate about livestock keeping, particularly boys who have helped their elders with livestock husbandry at home **(S2)**. Accordingly, 87% (n = 53 households) of questionnaire respondents stated their children help with livestock husbandry. However, only 68% (n = 56 households) of questionnaire respondents believed their children would continue to keep livestock in the future **(S2)**. Interviewees explained that some young adults who spend time away from home to attend college or university in Salalah, or well-educated adults in busy job roles are less interested in livestock husbandry **(A2b)**. It was said they tend to keep inherited livestock primarily out of respect for their family's values. These individuals may represent the twelve percent of Likert questionnaire respondents who stated they did not want more livestock and may stop keeping livestock in the future (Figure 2.6).

Several interviewees admitted they would prefer not to keep livestock at all, but are reluctant to sell their livestock out of fear of being perceived by others as weak and disrespectful of their family's pastoral values **(A6)**. Our translator's family had recently sold all their livestock, and he admitted that although people do talk, the financial and time benefits outweigh the 'loss of face' **(A5)**. It appears this is the first time such a 'peer pressure' culture has been described from a pastoral society and it was acutely clear in informant's narratives:

*"Every year it is getting more and more expensive. […] People lose more money than they are making. Only thing that is good is milk. People keep them just to respect parents and grandparents."*

And:

*"Young people hate animals, they don't want to have them. But if they sell them, then people will talk. For example, sell 10 camels for 10,000 OMR, people then talking, so buy back for double the price. Some people don't care about people talking but others do. Some sell up and move to Salalah."*

And:

*"In the last 10 years the old people have been dying in Eirkab and with the old people gone, people have been selling their livestock."*

A loss of interest in pastoral activities among the younger generation has been described from Borana pastoralists in Ethiopia (Gemedo-Dalle, Isselstein & Maass, 2006) and generational losses of traditional ecological knowledge (TEK) commonly occur (Aswani, Lemahieu & Sauer, 2018). In Dhofar, traditional knowledge has not been passed on to the younger generation, and a lack of environmental education in schools, and a lack of environmental policy that instils notions of the intrinsic value of biodiversity to human wellbeing, means the younger generation care little for the natural environment **(GS3)**. One young informant explained; “I am not bothered by desertification as I have roads, houses, cars and the internet”. This viewpoint is understandable given the successful development of many arid regions in the Arabian Peninsula, such as Dubai and Abu Dhabi.

The final demographic that we identified were wealthy livestock keepers for whom pastoralism could be considered a hobby (Gallacher, 2010). The extent to which they possess pastoral values is questionable as for many, owning large herds (> 100), or high-quality competition camels, is for social status **(A2a)**. They have available income for feedstuff provision and expatriate worker salaries **(A2a),** and are often less involved in livestock husbandry. Accordingly, we found that respondents who agreed that they spend less time with their livestock than their fathers, owned more camels. They may partly comprise the 24% of respondents that stated that they want more livestock despite agreeing that they already own more than they need (Figure 2.6). In 2012, local people with help from the private sector established a camel milking competition in Jabal Qamar, which was said to be the main driver of increasing camel numbers over the last five years **(I11)**. Camel competitions are known to facilitate preservation or evolution of rangeland culture in Gulf countries (Khalaf, 1999; Gallacher, 2010).

Almost all interviewees were aware of recent declines in vegetation abundance. The older generation remember the difference between the past and current vegetation structure of the rangelands **(A3)** in statements such as:

*“Before 40 years it was like a jungle, you had to climb a tree to see from here to over there.”*

And:

*Xfot (Blepharispermum hirtum) used to be everywhere here. Now it's very sparse. Before you could not pass through this area, it was like a fence. Further West there is more but it is still very damaged.*

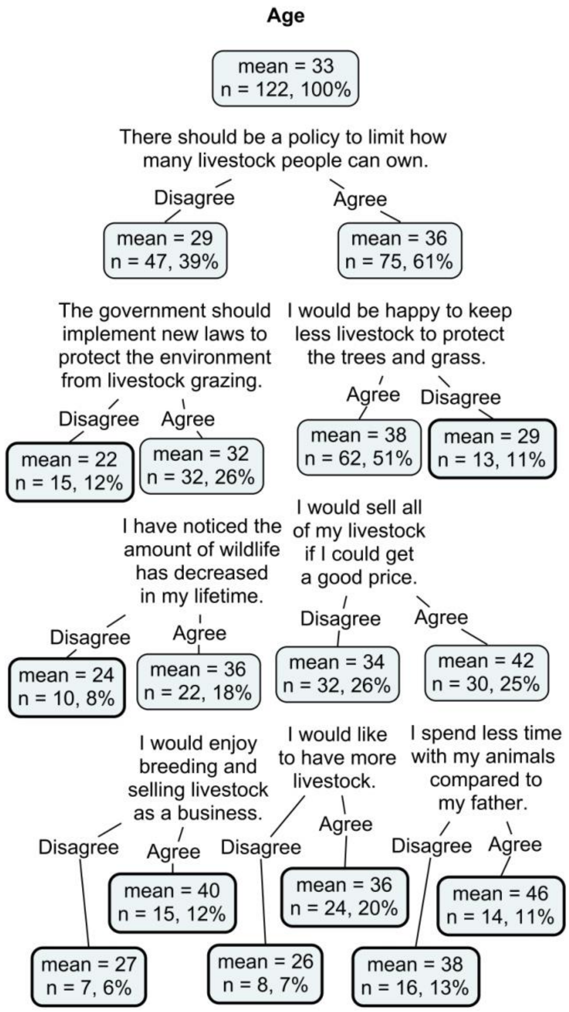



**Figure 2.7. CART analysis of Likert data showing groupings of respondents based on their age. They are grouped based on their responses to the other Likert scale items including socio-demographic data. Each node shows the mean predicted age and the number and percentage of observations in the node.**

CART analysis of the Likert questionnaire data with age as the response variable (Figure 2.7) showed that older respondents tended to possess pro-conservation views and would be willing to change their livestock management practises for the benefit of the environment, while younger respondents had opposite views **(S2)**. However, triangulation of these results with key informants revealed a different interpretation. They suggested that younger informants had given more reliable information, whilst older informants had given less reliable pro-conservation responses in anticipation of the results being seen by the government. They want to be perceived as having the 'correct' attitude towards environmental conservation, when in reality they would prefer ‘business as usual’. This may help to explain why 86.5% of respondents agreed that more roads should be built for livestock access, despite the obvious impacts this would have on the environment, and why more livestock keepers agreed that the environment should be protected for wildlife rather than their children’s future livestock. This is not to say there is a complete absence of consideration for the environment, rather environmental conservation comes second to livestock keeping, as one informant explained:

*“We care about the environment and realise the solution is to keep less animals. But we want to keep animals.”*

Furthermore, we cannot ignore the narratives of many older interviewees who frequently stated that the current situation needs to change and that they are awaiting an intervention from the government. Several added that they do not have a lot of faith in the government finding a solution.

### 2.5.4 Economics

Economic and market drivers have often stimulated transitions to unsustainable livestock production systems (Chang, 1994; Chatty & Colchester, 2002; Steinfeld, 2010; Robinson *et al.*, 2011). However, no substantial market exists for rural livestock in Dhofar. During the nineteenth century there was a rapid expansion of ranching in the grasslands of the Americas, Australia and Africa; termed the “the child of the industrial revolution” (Lessa, 1965). However, Oman bypassed the industrial revolution when primary sectors developed, and instead modernised during a time of globalisation and modern transport logistics **(S1)**. As Oman began to modernise, Dhofari livestock keepers increased their herd sizes and forage resources quickly depleted. A cycle of decline developed, whereby decreasing forage resources forced owners to purchase feedstuff, which in turn increased the price of livestock beyond a viable limit to compete in the food market against imported livestock from Africa and Oceania **(S5)**. Furthermore, due to a growing expatriate population, the demand for cheap meat from supermarkets and restaurants has increased. For example, there was a 1775% increase in live cattle imports to Oman between 2008 and 2013 (FAO, 2013). One informant explained:

*"Somalia to Salalah is easy, two days. People like cheap meat, some like Australian, local people like Dhofari, in Muscat they like cheap meat. Restaurants buy cheap meat to get more profit, and customers want a cheap price, they do not ask if it is local."*

In Dhofar, only a small-scale rural market system has developed due to a local taste for local meat. Three quarters (74%) of Likert questionnaire respondents agreed or strongly agreed that households prefer the taste, texture and nutritional value of local meat. In Jabal Qamar, this market accounts for an off-take of approximately two head of livestock per day. Each morning in Dhalkut a Pakistani butcher slaughters an animal and arranges the meat in to piles to be sold for 20 OMR (52 USD) for 5 kg or 4.5 OMR (12 USD) for 1 kg, and in Rakhyut local livestock owners sell meat out of the back of their vehicles in the town of Shab Esaaeb. Local meat is also sold in a number of traditional Omani restaurants and at the meat market in Salalah.

The socio-economic questionnaire results provide an estimate of the number of households involved in this local market, with approximately one third of respondents stating they sell camel or cattle meat, and fewer stating they sell milk (Figure 2.8). The production and consumption within the household, of meat and milk from all livestock types was the third and final reason given for keeping livestock. Accordingly, our questionnaire data shows household consumption of meat and milk as the most popular use of livestock products (Figure 2.8). All camel-owning households in our questionnaire sample produced milk from their camels for consumption in the household. Interestingly, purchasing meat and milk is in fact much cheaper than owning livestock, however, key informants explained the quality of milk varies and people prefer to consume milk from their own livestock. Indeed, many stressed the health benefits of camel milk afforded to their family and growing children. One particular individual explained:

*"If you go to hospital and have to have an anaesthetic, it is harder to get the needle into someone who has drunk camel milk every day because the muscle is firmer."*

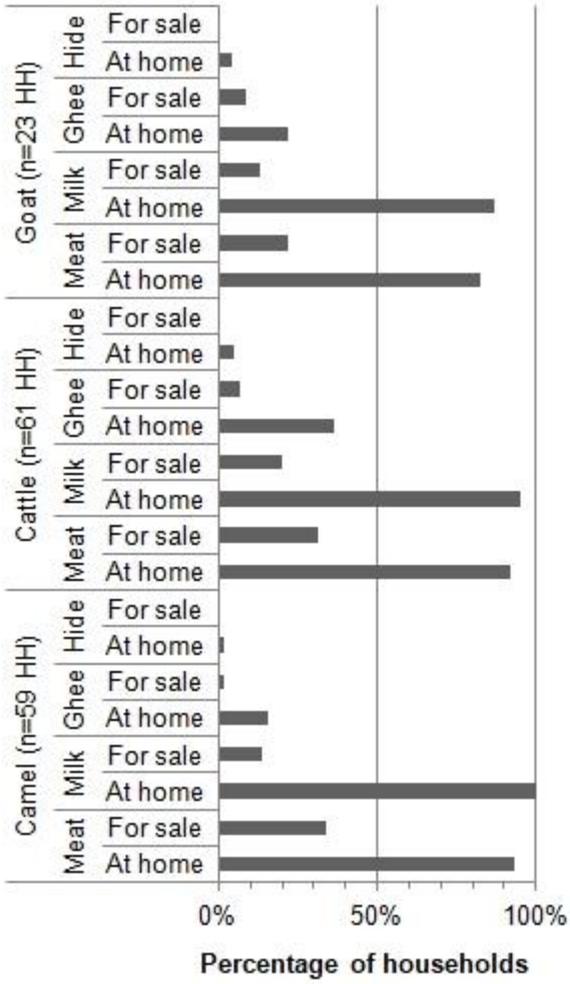



**Figure 2.8. Percentages of households in the questionnaire survey that produce livestock products for household consumption and sale.**

The average sale price of a Dhofari camel in the year 2000 was 300 OMR (779 USD) and today (2017) it is 1000 OMR (2596 USD). Due to the high price of livestock there has been no investment in livestock production in rural Dhofar **(I5)**. Moreover, entrepreneurship or collective action by livestock keepers to increase market participation has not occurred **(A5)**. For poor livestock producers around the world, market access influences risk management, income, food security and poverty reduction (Markelova & Mwangi, 2010) but the absence or low severity of these risks in Dhofar means market access and participation has been a low priority for livestock keepers. Nonetheless, sixty-six percent of Likert questionnaire respondents agreed or strongly agreed they would enjoy breeding and selling livestock as a business (Appendix 1).

In general, livestock keepers in Dhofar show a reluctance to sell surplus animals, as reported for Arabia as a whole (Peacock *et al.*, 2003). Our questionnaire results showed that annual camel sales ranged between 0-35 (mean = 5) animals and annual cattle sales ranged between 0-45 (mean = 8) animals. Informants ranked financial security as the second most important reason for keeping livestock. When asked for further details two themes emerged. The first was associated with financial security should there be an unpredictable event, of which the most commonly stated was loss of government employment and salary **(S3)**. This was exacerbated, at the time of writing, by the economic crisis in Oman associated with the drop in oil prices. The second was associated with the sale of multiple animals in one transaction for a sudden cash injection, if for example, a family member requires expensive healthcare or when purchasing or building a house.

These reasons are symptomatic of livestock accumulation strategies commonly seen in pastoral systems where livestock are accumulated as insurance against unpredictable events. Rather uniquely however in Dhofar, livestock keepers accumulate animals at a cost. Ninety-seven percent (n = 57 households) of questionnaire respondents stated making a net financial loss from owning livestock **(O1)**, and some livestock keepers are in debt to feedstuff retailers, often repaying the debt in livestock. Our data on annual profit or loss was unreliable however key informants explained annual losses of up to 5,000 OMR (12,988 USD) are not uncommon. Some livestock keepers spend all, or in excess of, their salary on livestock husbandry. If a family member's salary does not cover livestock expenses then higher-earning family members will contribute **(A2a)**. The culture of sharing wealth is strong in Oman and routed in Sunni Islamic culture **(A6)**. A young geologist from Dhalkut explained:

*"I give my Father money to cover the costs. My father has to spend 800 OMR on livestock each month, but his income is only 400 OMR, so I help to cover the difference. He is spending more than his salary on a hobby. His salary is small, unlike mine as a geologist, which is three times his. I can go to the bank, a livestock owner cannot."*

The greatest cost comes from purchasing feedstuff for 11 months of the year (4.3 OMR (11.16 USD) per 40 kg pellet feed, 2.9 OMR (7.53 USD) per 30 kg powder feed), and other costs include vehicle fuel, water tanks, feed troughs, veterinary care and expatriate worker salaries. Blench (1998) and Thomson (1997) identified feedstuff as the single biggest expenditure in livestock households in Jordan and Egypt, but unlike in Dhofar, a growing market sustained economic viability.

A significant positive relationship was found between household size and household income ($X^2$ (4, N = 73) = 15.536, $p$ = 0.004) and herd size and number of livestock sold per year (camels: $R_S = 0.61$, $p < 0.001$, cattle: $R_S = 0.35$, $p = 0.004$, goats: $R_S = 0.65$, $p < 0.001$). However, no significant relationship was found between household income and herd sizes which makes sense given that ownership is expensive. But this also suggests that less wealthy households do not have smaller herds to lessen husbandry costs, perhaps because they accumulate livestock as a financial reserve **(A2a)**. Furthermore, households have varying amounts of non-livestock income which may have little correlation with herd sizes due to other factors such as individual or household attitudes towards livestock keeping. Indeed, we have already discussed that some wealthier keepers want more livestock whilst others want fewer (Figure 2.6).

Unlike in Africa, it appears that herd accumulation in Dhofar is not a response to the highly variable nature of keeping livestock in arid environments (Sandford, 1983; McPeak, 2005) nor to the common property nature of tenure arrangements (Hardin, 1968; Jarvis, 1980). It is accumulation primarily due to cultural norms (Herskovits, 1926; Abu-Rabia, 1994) but rather uniquely this does not occur in parallel with income generation (Doran, Low & Kemp, 1979). Simultaneously, livestock offer insurance against unpredictable socio-economic events and the household benefits from meat and dairy produce.

## 2.6 Conclusion

We have presented a detailed account of modern pastoralism in Dhofar and applied Ostrom's socio-ecological systems framework (Ostrom, 2007) to understand the multiple historical and current socio-ecological variables driving overstocking. Our evidence shows that forces linked to the recent development of Oman have influenced a normative structure of deeply-embedded pastoral values which has motivated livestock accumulation despite significant household expenditure and poor market access. Our study represents a rare example of a pastoral system which has expanded primarily due to cultural traditions in the face of economic losses for pastoralists.

Although interesting for scholars, this unique systemic evolutionary path has resulted in an ecologically damaging resource use system, with few interactions **(I)** due to a lack of decision making at the individual and collective level. We have linked this lack of decision making to three variables identified by Ostrom (2009) as important for self-organization, which can be summarised for Dhofar as too many resource users **(U1)** in an unproductive system **(RS5)** with undervalued resources **(U8)**.

A detailed analysis of social and ecological performance measures **(O1, O2)** were outside the scope of this research. However, evidence shows that overstocking affects provisioning **(ECO3a)**, regulating **(ECO3b)** and cultural services **(ECO3c)** which influence a wide-range of stakeholders. For example, each year over half a million Arab tourists visit Dhofar during the Khareef to escape high summer temperatures elsewhere in the Arabian Peninsula **(O1)** and horizontal precipitation capture by woody vegetation has been found critical to support the cloud forest and the water economy of the region **(O2)** (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006; Friesen *et al.*, 2018).

We faced some minor issues when applying the framework to our study system. Firstly, we found the inclusion of some variables superfluous. For example the requirement for both technology (S7) and technologies available (A9) is unclear. Furthermore, we felt there was excessive overlap between RS variables and RU variables (such as location (RS9), size (RS3), productivity (RS5), distribution (RU7), number of units (RU5) and growth/replacement rate (RU2)), although this is probably due to the difficulties of defining spatial and temporal boundaries in rangeland systems. Despite these minor drawbacks, our research demonstrates the effectiveness of applying the SES framework to an atypical resource use system. It provided a concise way to present our findings and identify key variables driving degradation based on a wide range of empirically-derived concepts (Ostrom, 2007; McGinnis & Ostrom, 2014) – a marked improvement from conventional analysis approaches where variables driving degradation are identified from a limited literature review or conventional wisdom.

Our results illustrate the need for a transition away from unmanaged, unproductive, uneconomical and environmentally damaging pastoral practises and towards sustainable intensification (Tilman *et al.*, 2011). Efforts should be made to 'even the playing field' amongst livestock keepers and monopolise on the opportunity to allow those who no longer want to keep livestock to sell their livestock without 'loss of face'. Conversely, those who hold strong pastoral values should be allowed to participate in a livestock production system which is financially rewarding. They should be recognised as licensed sustainable producers who are incentivised to conform to a number of responsible production techniques. Sustainable intensification of livestock production in Dhofar could boost the rural economy and contribute to the national and export food markets (Al-Mashaki & Koll, 2007; El-Mahi, 2011b).

When Elinor Ostrom first proposed her SES framework (Ostrom, 2007) she called attention to the "perverse and extensive uses of policy panaceas in misguided efforts to make socio-ecological systems sustainable over time". She echoed the warnings of Korten (1980) to the danger of "blueprint approaches to the governance of tough social-ecological problems and urged that policy makers adopt a learning process rather than imposing final solutions". As a visiting international researcher with no local linguistic abilities, the lead author is aware of the inherent limitations to the depth and breadth of information in this paper, in comparison to what could be achieved via a well-organised collaboration between local stakeholders. Thus, for policy-makers to adopt an effective learning process, greater efforts must be made to establish a strong collaboration with livestock keepers to exchange information to inform decisions which address not only the issue of overstocking but target regional sustainable development objectives with substantial consideration for the present and future value of the biological diversity in Dhofar.

## 2.7 Literature cited

Abu-Rabia A. (1994) The Negev bedouin and livestock rearing : social, economic, and political aspects. Oxford: Berg Publishers.

Abu-Rabia A. (2000) Employment and unemployment among the Negev Bedouin. Nomadic Peoples, 4, 84–93.

Al-Kuthairi A. M. (1992) Forests and Pastoral Activities in Dhofar Mountains: Study of Strategy for their Rehabilitation. Arabian Gulf University, Bahrain.

Al-Mashaki M. S. A. & Koll E.-H. B. A. (2007) Country pasture/forage resource profiles: Oman. Rome: FAO.

Andrew M. H. (1988) Grazing impact in relation to livestock watering points. Trends in Ecology and Evolution, 3, 336–339.

Aswani S., Lemahieu A. & Sauer W. H. H. (2018) Global trends of local ecological knowledge and future implications. PLoS ONE, 13, e0195440.

Ball L. & Borrell J. (2016) An inventory of herpetofauna from Wadi Sayq, Dhofar, Oman. Journal of Threatened Taxa, 8, 9454–9460.

Blench R. (1995) Rangeland degradation and socio-economic changes among the Bedu of Jordan: results of the 1995 IFAD survey. London: Pastoral Development Network & Overseas Development Institute.

Chang C. (1994) Pastoralists at the Periphery: Herders in a Capitalist World. Tucson: University of Arizona Press.

Chatty D. & Colchester M. (2002) Conservation and Mobile Indigenous Peoples: Displacement, Forced Settlement, and Sustainable Development. New York: Berghahn Books.

David M. & Sutton C. D. (2011) Social research : An Introduction. London: SAGE Publications.

DeVore M. R. (2012) A more complex and conventional victory: Revisiting the Dhofar counterinsurgency, 1963-1975. Small Wars and Insurgencies. Taylor & Francis Group, 23, 144–173.

Directorate-General of Nature Conservation (2010) Fourth National Report To the Convention on Biological Diversity. Muscat: Ministry of Environment and Climate Affairs.

Doran M. H., Low A. R. C. & Kemp R. L. (1979) Cattle as a store of wealth in Swaziland: implications for livestock development and overgrazing in eastern and southern Africa. American Journal of Agricultural Economics, 61, 41–47.

El-Mahi A. T. (2011a) Hematophagous Flies in Dhofar (Sultanate of Oman): A Limiting Factor and a Potential Pathogenic Vector. Sultan Qaboos University Journal - Agricultural and Marine Sciences, 16, 65–73.

El-Mahi A. T. (2011b) Old ways in a changing space: the issue of camel pastoralism in Dhofar. Sultan Qaboos University Journal - Agricultural and Marine Sciences, 16, 51–64.

ESCWA, UNEP & FAO (1993) The national plan of action to combat desertification in the Sultanate of Oman. Amman: ESCWA, Manama: UNEP, Rome: FAO.

FAO (2013) FAOSTAT database collections. Rome. (Accessed: 9 August 2017).

Fernández-Giménez M. E. (2002) Spatial and social boundaries and the paradox of pastoral land tenure: A case study from postsocialist Mongolia. Human Ecology, 30, 49–78.

Friesen J., Zink M., Bawain A. & Müller T. (2018) Hydrometeorology of the Dhofar cloud forest and its implications for groundwater recharge. Journal of Hydrology: Regional Studies. Elsevier, 16, 54–66.

Gallacher D. J. (2010) Arid rangeland degradation in an oil-rich Gulf state; inertia of perceived heritage and pro-agricultural policies. in Veress, B. and Szigethy, J. (eds) Horizons in Earth Science Research. New York: Nova Science Publishers, 335–350.

Galletti C. S., Turner B. L. & Myint S. W. (2016) Land changes and their drivers in the cloud forest and coastal zone of Dhofar, Oman, between 1988 and 2013. Regional Environmental Change. Springer Berlin Heidelberg, 16, 2141–2153.

Gemedo-Dalle, Isselstein J. & Maass B. L. (2006) Indigenous ecological knowledge of Borana pastoralists in southern Ethiopia and current challenges. International Journal of Sustainable Development and World Ecology, 13, 113–130.

Ghazanfar S. A. (1998) Status of the flora and plant conservation in the sultanate of Oman. Biological Conservation, 85, 287–295.

Ghazanfar S. A. & Fisher M. (1998) Vegetation of the Arabian Peninsula. Dordrecht: Kluwer Academic Publishers.

GRM International (1982) Range and Livestock Survey. Brisbane: G.R.M. International Pty Ltd.

Hall G. B. & Close C. H. (2007) Local knowledge assessment for a small-scale fishery using geographic information systems. Fisheries Research, 83, 11–22.

Hardin G. (1968) The Tragedy of the Commons. Science, 162, 1243–1248.

Harris R. B., Samberg L. H., Yeh E. T., Smith A. T., Wenying W., Junbang W., Gaerrang & Bedunah T. L. D. J. (2016) Rangeland responses to pastoralists' grazing management on a Tibetan steppe grassland, Qinghai Province, China. Rangeland Journal, 38, 1–15.

Hedges S. & Lawson D. (2006) Monitoring the Illegal Killing of Elephants: Dung Survey Standards for the Mike Programme. Nairobi: CITES MIKE Programme.

Herskovits M. J. (1926) The Cattle Complex in East Africa. American Anthropologist, 28, 230–272.

Al Hikmani H., Zabanoot S., Al Shahari T., Zabanoot N., Al Hikmani K. & Spalton A. (2015) Status of the Arabian Gazelle, Gazella arabica (Mammalia: Bovidae), in Dhofar, Oman. Zoology in the Middle East, 61, 295–299.

Hildebrandt A. & Eltahir E. A. B. (2006) Forest on the edge: Seasonal cloud forest in Oman creates its own ecological niche. Geophysical Research Letters, 33, 2–5.

Hildebrandt A. & Eltahir E. A. B. (2007) Ecohydrology of a seasonal cloud forest in Dhofar: 2. Role of clouds, soil type, and rooting depth in tree-grass competition. Water Resources Research, 43, 1–13.

Hildebrandt A. & Eltahir E. A. B. (2008) Using a horizontal precipitation model to investigate the role of turbulent cloud deposition in survival of a seasonal cloud forest in Dhofar. Journal of Geophysical Research, 113, 1–11.

Hinkel J., Bots P. W. G. & Schlüter M. (2014) Enhancing the Ostrom social-ecological system framework through formalization. Ecology and Society, 19, 51.

HTSL (1978) Livestock Sector Review and Project Identification - Final Report, Volumes I and II. Hemel Hempstead: Hunting Technical Services Limited.

Janzen J. (1983) The modern development of nomadic living space in Southeast Arabia - the case of Dhofar (Sultanate of Oman). Geoforum, 14, 289–309.

Janzen J. (1990) Land Tenure and Land Use in the Pastoral Living Areas of Dhofar, Sultanate of Oman – Tradition, Change and Development Perspectives. Muscat: FAO & Land Tenure and Land Use Consultancy.

Jarvis L. S. (1980) Cattle as a Store of Wealth in Swaziland: Comment. American Journal of Agricultural Economics, 62, 606–613.

JICA (1990) Ten year Master Plan for Agricultural Development. Tokyo: Japan International Co-operation Agency.

Khalaf S. (1999) Camel racing in the Gulf. Notes on the Evolution of a Traditional Cultural Sport. Anthropos, 94, 85–106.

Kohler-Rollefson I. (1992) The Raika dromedary breeders of Rajasthan: a pastoral system in crisis. Nomadic Peoples, 30, 74–83.

Korten D. C. (1980) Community Organization and Rural Development: A Learning Process Approach. Public Administration Review, 40, 480–511.

Kürschner H., Hein P., Kilian N. & Hubaishan M. A. (2004) The Hybantho durae-Anogeissetum dhofaricae ass. nova - Phytosociology, structure and ecology of an endemic South Arabian forest community. Phytocoenologia, 34, 569–612.

Lamprey H. F. (1976) Preliminary Proposals for the Conservation and Management of the Jebel Qara Region. Nairobi: UNESCO Regional Office of Science and Technology for Africa.

Lawton R. M. (1978) A reconnaissance survey of the Jabal Qara grazing land ecosystem, with particular reference to the impact of development. London: Overseas Development Agency, Land Resources Development Centre.

Lessa W. A. (1965) Man, Culture, and Animals. The role of animals in human ecological adjustments. Edited by A. Leeds and A. P. Vayda. Washington: American Association for the Advancement of Science.

Louhaichi M. & Tastad A. (2010) The Syrian steppe: Past trends, current status, and future priorities. Rangelands, 32, 2–7.

Markelova H. & Mwangi E. (2010) Collective action for Smallholder Market Access: Evidence and Implications for Africa. Review of Policy Research, 27, 621–640.

Masri A. (2001) Country pasture/forage resource profiles: Syria. Rome: FAO.

McCabe J. T. (1990) Turkana Pastoralism: A Case Against the Tragedy of the Commons. Human Ecology, 18, 81–103.

McGinnis M. D. & Ostrom E. (2014) Social-ecological system framework: Initial changes and continuing challenges. Ecology and Society, 19, 30.

McPeak J. (2005) Individual and collective rationality in pastoral production: Evidence from northern Kenya. Human Ecology, 33, 171–197.

Miller A. G., Morris M. & Stuart-Smith S. (1988) Plants of Dhofar, The Southern Region of Oman: Traditional, Economic, and Medicinal Uses. Muscat: Office of the Adviser for Conservation of the Environment, Diwan of Royal Court.

Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD (2005) National Action Programme to Combat Desertification in the Sultanate of Oman. Muscat: Ministry of Regional Municipalities, Environment, and Water Resources.

Moritz M., Scholte P., Hamilton I. M. & Kari S. (2013) Open Access, Open Systems: Pastoral Management of Common-Pool Resources in the Chad Basin. Human Ecology, 41, 351–365.

Moritz M., Catherine L. B., Drent A. K., Kari S., Mouhaman A. & Scholte P. (2013) Rangeland governance in an open system: Protecting transhumance corridors in the Far North Province of Cameroon. Pastoralism, 3, 1–10.

Moritz M. (2016) Open property regimes. International Journal of the Commons, 10, 1–22.

Mosti S., Raffaelli M. & Tardelli M. (2012) Contribution to the Flora of Central-Southern Dhofar (Sultanate of Oman). Webbia, 67, 65–91.

Mott MacDonald International Ltd. (1991) National Resources Master Plan. Cambridge.

Nagendra H. & Ostrom E. (2014) Applying the social-ecological system framework to the diagnosis of urban lake commons in Bangalore, India. Ecology and Society, 19, 67.

Nedjraoui D. (2001) Profil fourrager de l'Algérie. Rome: FAO.

Newing H. (2011) Conducting Research in Conservation: A Social Science Perspective. 1st edn. Abingdon: Routledge.

Oba G. & Lusigi W. J. (1987) An Overview of Drought Strategies and Land Use in African Pastoral Systems. Marsabit: Kenya Aridlands Research Station.

Oman National Centre for Statistics and Information (2017a) 2017 Statistical Year Book. Muscat: National Centre for Statistics and Information.

Oman National Centre for Statistics and Information (2017b) National Centre for Statistics and Information Data Portal. (Accessed: 22 August 2017).

Oman Office of the Government Adviser for Conservation of the Environment (1980) The scientific results of the Oman flora and fauna survey 1977 (Dhofar), Journal of Oman Studies Special Report No. 2. Muscat: Ministry of Information and Culture.

Ostrom E. (1990) Governing the Commons: The Evolution of Institutions for Collective Action. 14th edn. Cambridge: Cambridge University Press.

Ostrom E. (1999) Coping With Tragedies of the Commons. Annual Review of Political Science, 2, 493–535.

Ostrom E. (2007) A diagnostic approach for going beyond panaceas. Proceedings of the National Academy of Sciences., 104, 15181–15187.

Ostrom E. (2009) A General Framework for Analyzing Sustainability of Social-Ecological Systems. Science, 325, 419–422.

Ostrom E. & Cox M. (2010) Moving beyond panaceas: A multi-tiered diagnostic approach for social-ecological analysis. Environmental Conservation, 37, 451–463.

Patzelt A. (2015) Synopsis of the Flora and Vegetation of Oman, with Special Emphasis on Patterns of Plant Endemism. Abhandlungen der Braunschweigischen Wissenschaftlichen Gesellschaft, 282–317.

Peacock J. ., Ferguson M. ., Alhadrami G. A., McCann I. R., Al Hajoj A., Saleh A. & Karnik R. (2003) Conservation through utilization: a case study of the indigenous forage grasses of the Arabian Peninsula. Journal of Arid Environments, 54, 15–28.

Petraglia M. D. & Rose J. I. (2010) The Evolution of Human Populations in Arabia. New York: Springer.

R Core team (2013) R: A Language and Environment for Statistical Computing. Vienna.

Robinson T. P., Thornton P. K., Franceschini G., Kruska R. L., Chiozza F., Notenbaert A., Cecchi G., Herrero M., Epprecht M., Fritz S., et al. (2011) Global livestock production systems. Rome: FAO & ILRI.

Rouchiche S., Al-Masheikhi M. S. A., Ahmed E. H. B. & Agieb S. E. A. M. (2003) Role of Planted Forests and Trees Outside Forests in Sustainable Forest Management in the Sultanate of Oman (Planted Forests and Trees Working Papers). Rome: FAO.

Sandford S. (1983) Management of pastoral development in the Third World. Chichester: Wiley.

Shaffer P. (2013) Q-Squared Combining Qualitative and Quantitative Approaches in Poverty Analysis. Oxford: Oxford University Press.

Spalton A., Al Hikmani H. M., Willis D. & Said A. S. B. (2006) Critically Endangered Arabian leopards Panthera pardus nimr persist in the Jabal Samhan Nature Reserve, Oman. Oryx, 40, 287–294.

Spalton A. & Al Hikmani H. (2014) The Arabian Leopards of Oman. 1st edn. London: Stacey International.

Steinfeld H. (2010) Livestock in a Changing Landscape, Volume 1: Drivers, Consequences, and Responses. 1st edn. Washington: Island Press.

Tardelli M. & Raffaelli M. (2006) Some aspects of the vegetation of Dhofar (Southern Oman). Bocconea, 19, 109–112.

Therneau T., Atkinson B. & Ripley B. (2017) Package ‘rpart’ (version 4.1-11). R Package Vignettes.

Thomson E. F., von Kaufmann R., Li Pun H., Treacher T. & van Houten H. (2000) Global Agenda for Livestock Research. Nairobi: International Livestock Research Institute.

Tilman D., Balzer C., Hill J. & Befort B. L. (2011) Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences, 108, 20260–20264.

Victor R. (2012) Sustainable Mountain Development Middle East and North Africa (MENA) from 1992 to 2012 and beyond. Muscat: Sultan Qaboos University.

Whitcombe R. P. (1998) Botany, environment and planning in drylands. in Prendergast, H, D, V. et al. (eds) Plants for Food and Medicine. London: Royal Botanic Gardens, Kew, 203–221.

Whitehead E. E., Hutchinson C. F., Timmermann T. M. & Varady R. G. (1988) Arid Lands: Today and Tomorrow. Colorado: Westview Press.

Wilson A. & MacLeod N. (1991) Overgrazing: Present or absent? Journal of Range Management, 44, 475–482.

WS Atkins International (1989) Regional development plan for the Southern Region, Final Report, August 1989. Epsom: WS Atkins International & Planning Committee for Development and Environment in the Southern Region (PCDESR).

WS Atkins International (1990) Subregional land use plans for the Southern Region, Final Report, March 1990. Epsom: WS Atkins International & Planning Committee for Development and Environment in the Southern Region (PCDESR).

# 3 Quantifying the impacts of livestock browsing on a drought deciduous cloud forest community in the Dhofar Mountains of Oman

## 3.1 Abstract

The Dhofar Mountains in southern Oman represent a unique drought-deciduous cloud forest rangeland system. It is suspected that overstocking of camels, cattle and goats has lead to rangeland degradation and a loss of woody cover, with negative effects on the ecosystem's ability to capture fog moisture through horizontal precipitation. Here we perform the first detailed analysis of the impacts of livestock browsing on the composition and structure of the woody vegetation of the *Anogeissus* forest. We analyse the effects of browsing relative to other abiotic and biotic factors using multivariate statistical analysis. Local spatial variability in the monsoon fog is found to be the primary driver of woody species composition whilst long-term stocking rates have increased the frequency of unpalatable species, decreased plant density, reduced advanced growth, and led to stunting, altered population age structures and plant damage through management practises, bark stripping and browsing. With livestock as the principle driver of vegetation change we conclude that the rangelands tend towards equilibrium.

## 3.2 Introduction

Rangelands are the most extensive anthropogenic biome, covering between one-third to one-half of the earth's ice-free terrestrial surface (Sayre, 2017). They are considered too arid and remote to warrant intensive management, and thus livestock production is the dominant use (Stafford Smith, 1996; Grice & Hodgkinson, 2002). Three hundred and twenty million people inhabit rangelands of which 250 million in the developing world are estimated to be directly affected by rangeland degradation (Reynolds *et al.*, 2007; Ellis & Ramankutty, 2008).

Rangeland degradation is synonymous with desertification and can be caused by both natural factors associated with a variable climate, or human-induced factors including poor irrigation practises, deforestation, overcultivation and overgrazing (Burns, 1995). Overgrazing has long been considered the primary anthropogenic driver of rangeland degradation (Sayre *et al.*, 2012), particularly in arid and semi-arid rangelands, and has been most famously linked to rangeland degradation in Australia (Curry & Hacker, 1990), Africa (Lamprey, 1983) and the USA (Herbel, 1979). Yet our understanding of rangeland ecology remains relatively poor, as there is no unifying set of general principles in rangeland ecology due to high variability between rangeland environments. We also lack a proficient understanding of how other drivers of degradation interact with grazing activity (Grice & Hodgkinson, 2002; Sayre *et al.*, 2012), which has in some instances resulted in pastoralists being incorrectly blamed for degradation, as conventional wisdom equates pastoralist regimes with overgrazing (Anderson & Grove, 1989; Moritz, 2017). Thus, there is a requirement to improve our understanding of the effects of overgrazing relative to other factors.

Rangelands encompass grasslands, savannahs, tundra, steppe, prairies, shrublands, deserts, woodlands and forests at any latitude (Holechek, Pieper & Herbel, 2001; Sayre, 2017). Much of the literature addresses the issue of livestock overgrazing in grassland ecosystems, whereas much fewer address the issue of overbrowsing in wooded environments. At the time of writing, an online literature search returned 514 articles with 'overgrazing' in the title and just 22 with 'overbrowsing' in the title. Often the term overgrazing is used interchangeably, despite the vastly different impacts that browsers and grazers can have on the ecosystem. For example, the most commonly reported effect of overgrazing on rangeland vegetation is woody plant encroachment, however, the most commonly reported effect of overbrowsing is decreased woody plant cover (Asner *et al.*, 2004). Subsequently, these processes can affect soil properties and hydrology (Briske, 2017). Studies on overbrowsing tend to focus on wild browsers, with the exception of goats, and few studies have addressed large livestock such as camels, despite 80% of their diet comprising of woody plants (Dereje & Uden, 2005).

Since Oman's renaissance in 1970, populations of camels and cattle have increased dramatically in the southern region of Dhofar. It has been repeatedly reported that the local mountain ecosystems are becoming degraded due to overstocking by rural pastoralists (Lamprey, 1976; Lawton, 1978; Oman Office of the Government Adviser for Conservation of the Environment, 1980; Wilson & MacLeod, 1991; Ghazanfar, 1998; Peacock *et al.*, 2003; Hedges & Lawson, 2006; Tardelli & Raffaelli, 2006; Directorate-General of Nature Conservation, 2010; El-Mahi, 2011b). There are concerns current stocking rates are reducing biological productivity (Peacock *et al.*, 2003), suppressing palatable plant growth and encouraging unpalatable species (Ghazanfar, 1998; Peacock *et al.*, 2003; Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005; Patzelt, 2012), reducing the efficiency of ecosystem services (Kürschner *et al.*, 2004; Galletti, 2015) and undermining biodiversity conservation efforts (Spalton & Al Hikmani, 2014; Al Hikmani *et al.*, 2015). The critically endangered Arabian leopard *Panthera pardus nimr* has a global stronghold in Dhofar and has been the focus of substantial research effort (Spalton *et al.*, 2006; Breitenmoser *et al.*, 2010; Spalton & Al Hikmani, 2014).

The Dhofar Mountains are part of a mountain belt that lies on the southern coast of the Arabian Peninsula, in the Al Mahra region of Yemen and the Dhofar region of Oman. The mountains receive a mean annual precipitation of 250 mm whilst neighbouring areas receive 100 mm. Most of this precipitation is received during the summer monsoon (known locally as the *Khareef*) between mid-June and mid-September, when thick fog inundates the southern mountain escarpments (Whitehead *et al.*, 1988; Ghazanfar & Fisher, 1998). Outside the Khareef season the climate is hot and dry, yet managed or wild fires rarely occur.

The Khareef fog supports the *Hybantho durae-Anogeissetum dhofaricae* (Kürschner *et al.*, 2004), a drought deciduous cloud forest community (262 floral species) which is the dominant habitat of the southern mountain escarpments and endemic to the region. The forest captures fog moisture through horizontal precipitation. The estimated quantity of water captured by *Anogeissus dhofarica* trees in Dhofar (250% more than rainfall) is amongst the highest recorded for any tree species (Friesen *et al.*, 2018). Furthermore, fog density is much higher a few meters above than close to the ground (Price, Al-Harthy and Whitcombe, 1988; 34-35 litres/m2 per day at 4.2 m, 13 litres/m2 per day at 0.9 m) and thus trees capture more water than grasses. For most of the year the Dhofar Mountains represent a moisture-limited environment with temperatures in excess of 30 °C and thus horizontal precipitation during the Khareef is considered critical to the survival of the *Anogeissus* forests (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2007, 2008; Friesen *et al.*, 2018).

Consequently, deforestation over past millennia for grazing pastures for cattle (Oman Office of the Government Adviser for Conservation of the Environment, 1980; Kürschner *et al.*, 2004; Patzelt, 2011), and a loss of woody cover due to livestock browsing (Ghazanfar, 1998; Hildebrandt & Eltahir, 2008) are considered to significantly reduce the quantity of moisture that enters the ecosystem through horizontal precipitation during the Khareef, with subsequent effects on the mountain ecohydrology and the local water economy (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2008).

Given the threat posed by overstocking on vegetation in Dhofar, and that numerous reports and action plans have included objectives to tackle overstocking (see Hunting Technical Services Limited (HTSL), 1978; G.R.M. Pty Ltd., 1982; Janzen, 1990; Japan International Co-operation Agency (JICA), 1990; Mott MacDonald International Ltd., 1991; Al-Kuthairi, 1992; ESCWA, UNEP and FAO, 1993; UNEP, 2005) but few studies have attempted to quantify its ecological impacts, the aim of this research was to quantify the impacts of overbrowsing by camels, cattle and goats on woody plants in Dhofar by addressing three objectives. Firstly, identify the biotic and abiotic variables that influence woody vegetation. Secondly, understand the effects of browsing on woody vegetation species composition. Finally, understand the effects of browsing on the structure of the woody plant layer, specifically plant density, age structure, and phytomorphology.

## 3.3 Materials and Methods

### 3.3.1 Study area

Our study was conducted in Jabal Qamar, the westernmost of the three mountain ranges in Dhofar (Figure 3.1). Jabal Qamar experiences a higher precipitation than the other two mountain ranges and boasts the highest botanical diversity (515 vascular species) of any area in Oman (Patzelt, 2015). Variants of the drought-deciduous *Anogeissus* forest (Kürschner *et al.*, 2004) are dominant on the south-facing escarpments, with sparse *A. dhofarica* woodland and grasslands (dominated by *Arthraxon* sp., *Apluda mutica* and *Themeda quadrivalvis*) in flatter areas. At elevations > 1000 m a.s.l. the *Euphorbia balsamifera* cushion shrub community dominates (Patzelt, 2015). The main geologic formation in Jabal Qamar is limestone of tertiary origin. Layers of the Hadramout group are present. These are, from bottom to top, the Umm Er Radhuma (UER), the Rus (RUS) and the Dammam (DAM) formations (Friesen *et al.*, 2018).

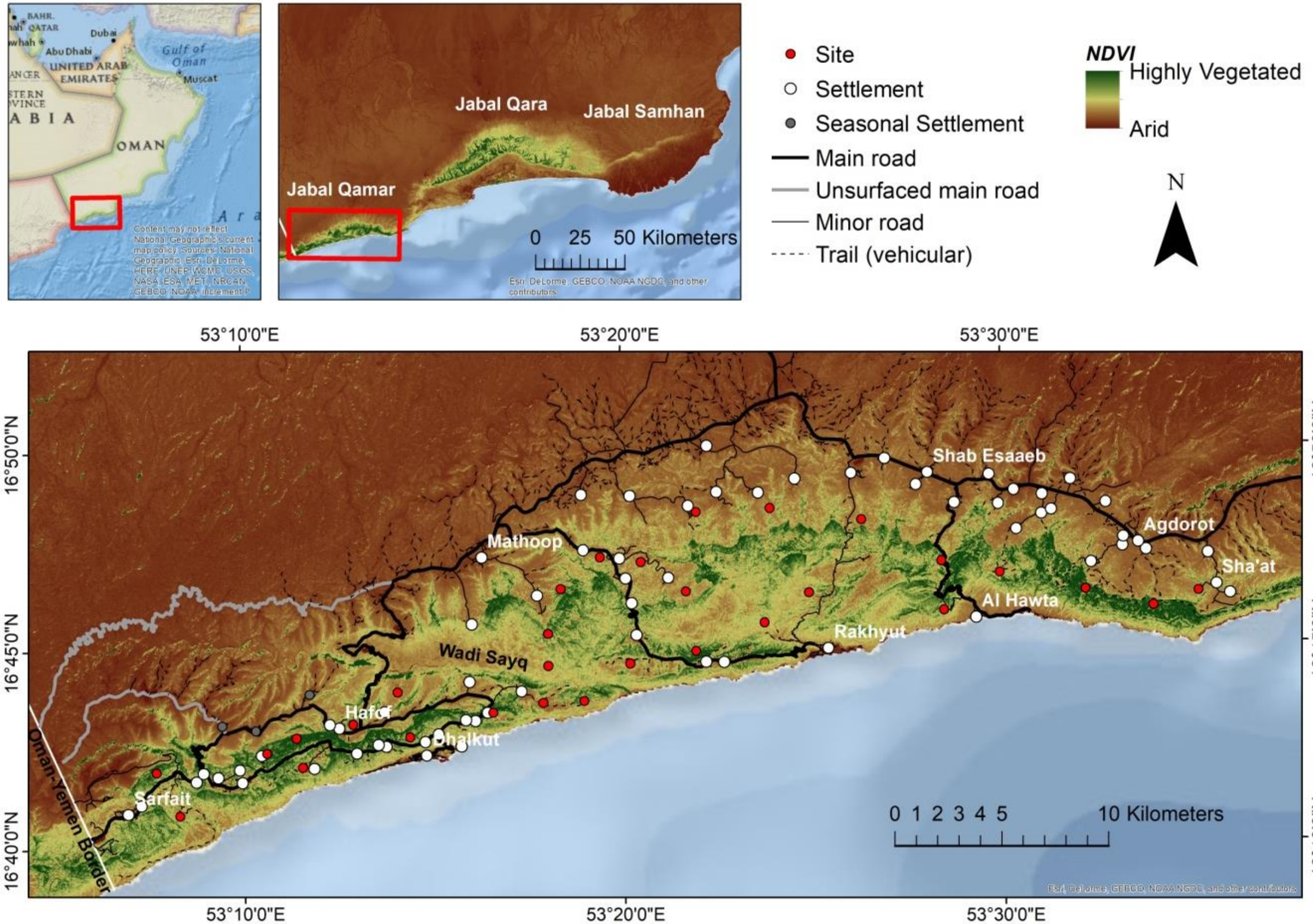



**Figure 3.1. Map of Jabal Qamar showing locations of the sampling sites, settlements, roads and vehicular tracks, overlaid on a vegetation greenness (NDVI) base map. Two inset maps show the whole Dhofar Mountains and their location in Oman.**

There are seventy-five permanently inhabited and ten seasonally (Khareef) inhabited villages in Jabal Qamar with a total human population of 7,799 (Oman National Centre for Statistics and Information, 2017a). The 2015 national livestock census recorded 15,164 dromedary camels in 802 holdings, 27,522 head of cattle in 1,060 holdings, and 14,217 goats in 439 holdings (Oman National Centre for Statistics and Information, 2017b). Based on these statistics livestock outnumber people 7 to 1. Livestock-owning households are present in all villages. Livestock are moved in a transhumance regime. During the Khareef (July-Sep) camels are moved to the plateau (c. 1000 m a.s.l.) to avoid the biting flies and soft mud on the southern escarpments, the latter of which can be fatal for camels if they slip. Following the Khareef (Oct-Jan), livestock are moved down into the monsoon-influenced zone to utilise the abundant vegetation. During the dry season (Feb-June) livestock are kept close to villages or camps. Livestock receive feedstuffs for ten to eleven months of the year.

### 3.3.2 Data collection

Stratified sampling was used to select sites for analysis. The landscape was firstly stratified into two altitudinal ranges (300–500 m and 700–900 m a.s.l.). These ranges encompassed representative proportions of the 100–1000 m a.s.l. altitudinal range of the *Anogeissus* forest (Kürschner *et al.*, 2004) – the study area – and incorporated many flat rangeland areas suitable for surveying. Within these ranges, land with a slope gradient greater than 30 degrees was omitted so sites were safely accessible to the research team and comparable in terms of vegetation communities, the composition of which can change substantially on steep slopes and cliffs (Patzelt, 2015). Thirty sites with varying stocking histories were then identified by livestock keepers during interviews and participatory mapping exercises. More precise estimates of long-term stocking rates were determined by a different method which is described later in the methods section. Sites covered the breadth of Jabal Qamar and were visited on four occasions every two months between September 2016 and March 2017.

The point-centered quarter (PCQ) method (Cottam and Curtis, 1956) was used to sample the composition, density and structure of woody vegetation at each site. In this method, density estimates are derived from distance measures between points and the closest plants, which are subsequently studied to estimate composition and structure. It is a plotless method making it more efficient than standard plot-based techniques. It has been shown to give more accurate density estimates per unit of sampling effort than other plotless methods (Cottam & Curtis, 1956; Beasom & Haucke, 1975). Although the PCQ method was initially designed for forestry studies it has been widely applied to natural systems too (Dickhoefer *et al.*, 2010; Dias *et al.*, 2017; Pereira *et al.*, 2018). We incorporated some recommendations by Dahdouh-Guebas and Koedam (2006) to address ambiguous field situations associated with multiple-stemmed trees.

Ten points were carried out during the first visit, ten during the second visit, five during the third visit and five during the fourth visit, resulting in a total of 30 points at each site. Consecutively rotating site visits controlled for intra-seasonal vegetation change such as senescence and livestock browsing to improve comparability between sites, and allowed for minor adjustments to be made to the methodology between each round of visits. PCQ point locations were generated in ArcGIS and randomly dispersed over an area of approximately 1 $km^2$ at each site as camels are highly mobile, do not eat for long periods from a single plant and spread out during browsing (Dereje & Uden, 2005). Conversely, cattle browsing can be patchy. Thus, small sampling sites may not have provided a representative average of the effects of browsing and may have overemphasised vegetation responses (Briske, Fuhlendorf & Smeins, 2003). At each sample point, the distances to the closest adult and the closest juvenile woody plant were recorded in each of four quarters, resulting in a total of 120 adult and 120 juvenile records per site. Measurements were taken from the point to the centre of the individual, rather than the closest plant component (Dahdouh-Guebas & Koedam, 2006).

For each individual the diameter at route collar (DRC), and where applicable, diameter at breast height at 130 cm above the ground ($DBH_{130}$) were measured using a diameter tape or callipers. For multi-stemmed plants all stems were measured, however thin suckers growing from large trees and shrubs were ignored. Stem status was recorded as alive, dead, broken or missing and stems that had been cut by a machete or chainsaw were noted. Very old or deteriorated cut or missing stems were ignored. If a plant only had dead stems at $DBH_{130}$ but additional live stems were present it was recorded as alive and stem statuses recorded accordingly. $DBH_{130}$ was not recorded for juveniles. An adult individual was recorded as dead when more than 80 percent of the plant was dead and a $DBH_{130}$ was present. Individuals with only dead stems below $DBH_{130}$ were classed as stumps and were ignored. Sprouting stumps were recorded as juveniles to recognise woody regrowth and because they couldn't be classified as alive or dead adults. In retrospect, they should have been classed as stumps and ignored to avoid overrepresentation of juvenile abundance, although few were encountered and thus their effect on the results is negligible.

Due to the diversity of woody plant species and their varying phytomorphology, preliminary work was carried out to determine parameters for the DRC measurements to distinguish adult individuals from juveniles for each species. A table of measurements are shown in Appendix 5. Existing methods to differentiate between adult and juvenile plants, that utilise diameter and height measurements, were deemed inappropriate as many plants exhibited altered morphology due to browsing activity. Adult and juvenile height was measured from the ground to the top of the plant unless the plant had fallen horizontally then the trunk length was measured.

For all individuals, browsing intensity was estimated by five classes according to the percentage of browsed branches below the browse line (~3 m). For adults, the proportion of broken branches was estimated on a five class scale according to the percentage of broken branches. To assess the prevalence of tree management practises, the proportion of bent or cut branches was estimated on a five class scale. The classes were defined as: (1) ~0%, (2) 1% – 33.3%, (3) 33.3% – 66.6%, (4) 66.6% – 99%, (5) ~100%. Therefore, the scale recognised undamaged (1) and entirely damaged (5) plants which are vital indicators of stocking rates and impacts, whilst intermediate classes could be quickly approximated as low, medium and high. Areas of stripped bark were also measured and additional relevant information was recorded. At each PCQ point a 1.2 m quadrat was deployed to sample the relative cover of herbs, grasses, rock and bare ground, and it also served to guide the PCQ quarters. Canopy cover was also recorded.

The dependent variables (vegetation responses) and independent variables (environmental variables) used in the analysis are shown in Table 3.2, along with results of univariate tests between them (see Appendix 7 for an exhaustive list of variables for each site). Independent variables related to core drivers of vegetation dynamics in rangelands, such as climate, topography, soils, geomorphology, herbivory and anthropogenic disturbance (Scholes & Archer, 1997).

A precise ranking of sites based on long-term stocking rates could not be quantified through discussions with livestock keepers. Instead, we used evidence of plant damage and a GIS-based adaptation of the piosphere model (Andrew, 1988) as a proxy for long-term stocking rates. This was expressed using a discrete scale with sites taking a value between 1 (lowest stocking rate) and 30 (highest stocking rate). This scale was calculated by ranking the sites for each of five measured variables (Table 3.1) and then summing the rankings and ordering these values. This ranking method was favoured over an ordination approach (such as principal component analysis) due to heterogeneity in the variables' units of measurements and probability distributions. Moreover, our intention was not to understand variance of the original variables or their relative contributions as in ordination, but to collapse the variables into usable numerical values for analysis. Factor analysis, in which a large number of variables are collapsed into a few interpretable underlying factors, could also have been used (Crawley, 2007).

Browsing intensity provides information about recent stocking rates whilst proportion of bent *A. dhofarica* branches and proportion of broken branches of all adults provide longer-term evidence (up to several decades) of stocking rates. It is reasonable to assume no issues with circularity in our analyses of density and basal area as these are unrelated to the damage indicators used to define stocking rates. Furthermore, by limiting average browsing intensity to seven key forage species and the proportion of bent branches to *A. dhofarica* we minimise circularity associated with damage varying by species and maximise comparability between sites.

**Table 3.1. Site ranks for each of the five variables used to quantify long-term stocking rates with the sum of ranks for each site in the final column. Site 22, with the third lowest long-term stocking rate, is highlighted as an example.**

| Site | Average browsing intensity of seven key forage species | Rank | Site | Average proportion of bent *A. dhofarica* branches | Rank | Site | Average proportion of broken branches | Rank | Site | Path distance to house | Rank | Site | Path distance to road or track | Rank | Site | Sum of ranks | **Long-term stocking rate (low/1 – high/30)** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 22 | 2.404 | 1 | 23 | 1.000 | 1 | 22 | 1.683 | 1 | 17 | 8.192 | 1 | 26 | 5.486 | 1 | 26 | 17 | **1** |
| 26 | 2.659 | 2 | 26 | 1.000 | 1 | 21 | 2.011 | 2 | 26 | 6.312 | 2 | 22 | 2.642 | 2 | 25 | 19 | **2** |
| 25 | 2.869 | 3 | 17 | 1.071 | 2 | 27 | 2.125 | 3 | 29 | 5.745 | 3 | 25 | 2.533 | 3 | 22 | 23 | **3** |
| 27 | 3.000 | 4 | 25 | 1.182 | 3 | 24 | 2.247 | 4 | 25 | 5.386 | 4 | 29 | 2.330 | 4 | 23 | 30 | **4** |
| 23 | 3.539 | 5 | 11 | 1.286 | 4 | 23 | 2.256 | 5 | 16 | 5.112 | 5 | 17 | 2.262 | 5 | 17 | 39 | **5** |
| 21 | 3.633 | 6 | 22 | 1.375 | 5 | 25 | 2.270 | 6 | 23 | 4.651 | 6 | 20 | 2.188 | 6 | 27 | 41 | **6** |
| 29 | 3.912 | 7 | 16 | 1.459 | 6 | 18 | 2.356 | 7 | 15 | 4.559 | 7 | 19 | 2.011 | 7 | 21 | 48 | **7** |
| 30 | 4.088 | 8 | 12 | 1.500 | 7 | 15 | 2.371 | 8 | 27 | 4.275 | 8 | 24 | 1.718 | 8 | 29 | 49 | **8** |
| 10 | 4.407 | 9 | 24 | 1.500 | 7 | 30 | 2.392 | 9 | 21 | 4.136 | 9 | 2 | 1.234 | 9 | 15 | 56 | **9** |
| 11 | 4.437 | 10 | 1 | 1.650 | 8 | 9 | 2.433 | 10 | 14 | 3.960 | 10 | 28 | 1.225 | 10 | 30 | 56 | **10** |
| 17 | 4.493 | 11 | 30 | 1.786 | 9 | 26 | 2.453 | 11 | 18 | 3.384 | 11 | 27 | 1.140 | 11 | 24 | 57 | **11** |
| 15 | 4.500 | 12 | 4 | 1.857 | 10 | 2 | 2.505 | 12 | 20 | 3.031 | 12 | 30 | 1.126 | 12 | 16 | 60 | **12** |
| 28 | 4.533 | 13 | 5 | 2.000 | 11 | 12 | 2.512 | 13 | 12 | 2.932 | 13 | 23 | 1.058 | 13 | 12 | 62 | **13** |
| 20 | 4.576 | 14 | 21 | 2.000 | 11 | 29 | 2.525 | 14 | 22 | 2.753 | 14 | 6 | 1.016 | 14 | 20 | 72 | **14** |
| 16 | 4.634 | 15 | 3 | 2.036 | 12 | 28 | 2.602 | 15 | 19 | 2.654 | 15 | 7 | 0.909 | 15 | 11 | 81 | **15** |
| 6 | 4.646 | 16 | 15 | 2.071 | 13 | 20 | 2.659 | 16 | 1 | 2.318 | 16 | 15 | 0.791 | 16 | 28 | 81 | **16** |
| 12 | 4.681 | 17 | 7 | 2.167 | 14 | 16 | 2.667 | 17 | 28 | 2.152 | 17 | 16 | 0.642 | 17 | 2 | 86 | **17** |
| 7 | 4.750 | 18 | 27 | 2.250 | 15 | 3 | 2.675 | 18 | 30 | 1.796 | 18 | 1 | 0.613 | 18 | 18 | 92 | **18** |
| 24 | 4.771 | 19 | 6 | 2.327 | 16 | 11 | 2.694 | 19 | 24 | 1.723 | 19 | 14 | 0.574 | 19 | 1 | 93 | **19** |
| 14 | 4.793 | 20 | 9 | 2.333 | 17 | 17 | 2.824 | 20 | 8 | 1.612 | 20 | 21 | 0.534 | 20 | 19 | 95 | **20** |
| 8 | 4.806 | 21 | 10 | 2.333 | 18 | 19 | 2.913 | 21 | 2 | 1.219 | 21 | 12 | 0.516 | 21 | 7 | 96 | **21** |
| 5 | 4.810 | 22 | 2 | 2.368 | 19 | 10 | 2.951 | 22 | 6 | 1.134 | 22 | 4 | 0.479 | 22 | 6 | 97 | **22** |
| 18 | 4.811 | 23 | 14 | 2.474 | 20 | 1 | 2.952 | 23 | 11 | 0.958 | 23 | 10 | 0.442 | 23 | 14 | 97 | **23** |
| 4 | 4.845 | 24 | 29 | 2.615 | 21 | 8 | 2.989 | 24 | 7 | 0.902 | 24 | 3 | 0.434 | 24 | 10 | 100 | **24** |
| 2 | 4.865 | 25 | 8 | 3.000 | 22 | 7 | 3.024 | 25 | 5 | 0.863 | 25 | 11 | 0.428 | 25 | 3 | 107 | **25** |
| 3 | 4.925 | 26 | 18 | 3.000 | 22 | 5 | 3.110 | 26 | 13 | 0.777 | 26 | 5 | 0.330 | 26 | 5 | 110 | **26** |
| 19 | 4.938 | 27 | 13 | 3.526 | 23 | 13 | 3.156 | 27 | 3 | 0.750 | 27 | 9 | 0.283 | 27 | 9 | 112 | **27** |
| 1 | 4.954 | 28 | 20 | 3.889 | 24 | 14 | 3.202 | 28 | 10 | 0.708 | 28 | 13 | 0.194 | 28 | 4 | 116 | **28** |
| 9 | 5.000 | 29 | 19 | 3.900 | 25 | 6 | 3.271 | 29 | 9 | 0.577 | 29 | 18 | 0.168 | 29 | 8 | 117 | **29** |
| 13 | 5.000 | 29 | 28 | 3.947 | 26 | 4 | 3.614 | 30 | 4 | 0.559 | 30 | 8 | 0.158 | 30 | 13 | 118 | **30** |

The piosphere model is an estimate of stocking rate based on the Euclidian distance of a given rangeland area to the closest waterpoint and is frequently used in studies of arid grazing systems (Lange, 1983; Andrew, 1988; Wilson & MacLeod, 1991; Turner & Hiernaux, 2002). It is applicable to Dhofar as livestock receive twice-daily provisioning of feedstuffs and water, usually returning to the house of their own accord each evening. Thus, stocking rates tend to be higher closer to houses, camps, roads and vehicle tracks. We calculated a path distance layer to these features, rather than Euclidian distance, to account for topographic effects on livestock mobility. First, a cost surface was created which accounted for the relationship between slope and access routes such as roads, vehicle tracks and livestock trails. Where slopes had an incline of ≤ 10 degrees, access routes were considered no less costly than other areas, but on slopes > 10 degrees, access routes were considered less costly. Distances were calculated using symmetric inverse linear vertical factor, which results in exponentially increasing resistance with slope steepness. All slopes with > 50 degree inclines and no access routes were considered inaccessible to livestock. The thirty least and most costly points were confirmed to be accurate by the lead author, who conducted the point sampling. The path distances were averaged across the points to give a value for each site.

In addition to long-term stocking rate, current stocking rates of camels and cattle were estimated using dung transects and the Faecal Accumulation Rate (FAR) method (Putman, 1984). Two transects were deployed during the first visit, two during the second, and one during the third. Transects were checked and cleared during each site visit, resulting in a total of eleven transect accumulation periods per site. Transects were one meter wide by fifty meters in length and the average accumulation period was 54 days. Due to the slow decomposition rate of dung as a result of the hot and dry climate, a long accumulation period was preferred. Twenty camels were followed and the time between defecation events was measured. Most camels were followed until three between-defecation periods had been timed. The resultant mean defecation rate was 40.36 events per day (min 26.5, max 71.88, median 35.11, SD 13.58, SE 3.04). Livestock densities were estimated for each site using the following equation: Dung piles per $km^2$ / accumulation period * 40 (defecation rate). See Appendix 7 for the results.

Topographic-related variables were calculated for each PCQ point from TanDEM-X 12 m global digital elevation model (DEM), in ArcGIS Desktop 10.5 (ESRI, 2016). Because horizontal precipitation is crucial to sustaining woody vegetation in Dhofar (Hildebrandt & Eltahir, 2008) the spatial variability in Khareef fog density was derived from the near-infrared (NIR) bands of thirteen Landsat 5 and four Landsat 7 products (Welch & Wielicki, 1986), and the ultra blue bands of twenty Landsat 8 products, acquired during Khareef seasons between 1990 and 2017 (Appendix 8). Through visual inspection a minimum threshold reflectance value was defined for each image to distinguish only the highly reflective fog layer, and the background values set to NULL. The images were then rescaled to a 0-1 range, stacked, and the mean calculated. Areas with higher reflectance values were interpreted as denser and more moisture-laden fog as the fogs upper altitude (cloud top) is limited to the altitude of the plateau due to warmer northerly winds from the desert (Kürschner *et al.*, 2004). As a measure of exposure to Khareef fog, we calculated slope aspect (Stage, 1976) which has lowest values on steep north-facing slopes and highest values on steep south-facing slopes.

Information on the geology of the sites was georeferenced from scanned 1:100,000 geological maps (Ministry of Petroleum and Minerals, 1986) of the research area and soil pH levels were tested from four composite soil samples collected from each site. The samples were crushed, passed through a 2mm sieve, mixed with distilled water (10cc of soil to 30ml of distilled water), and tested using a calibrated pH meter electrode. Underlying geology is known to affect soil properties including acidity (Miller, Singer & Nielsen, 1988; Barnes *et al.*, 1997), with subsequent effects on vegetation composition, however a one-way ANOVA found soil pH did not differ significantly by bedrock type ($F(4,25) = 0.846$, $p = 0.509$). This is most likely because the bedrock is just variants of limestone (Ministry of Petroleum and Minerals, 1986; Friesen *et al.*, 2018). It was also apparent that geology was serving as a proxy for topographic factors in the models, and thus to preserve degrees of freedom in the models and facilitate interpretation, geology was removed (Dubuis *et al.*, 2011).

### 3.3.3 Data analysis

Each of the vegetation responses and independent variables were tested against one another using the appropriate univariate statistical test depending on the type of variable and the distribution of the data (Table 3.2). Collinearity between independent variables was also tested using Pearson's correlation and box plots.

To quantify the effect of long-term stocking rates on the species composition of woody plants, constrained correspondence analysis (CCA) was carried out on the count data of adult and juvenile woody species separately. Ordination was conducted using the Vegan package (Oksanen *et al.*, 2018) in R studio (R Core team, 2013). The CCA models were built following a manual stepwise procedure, where the variables were gradually added based on their suspected influence, founded on our ecological understanding. Canopy cover was not considered as a constraining variable for adult woody species as it is a product of species composition. Variance Inflation Factor (VIF) values were calculated after each model as a diagnostic tool to identify collinear constraints (VIF > 3). Permutation tests for the joint and separate effects of constraining variables, as well as for marginal (Type III) effects, were performed to test the significance of each constraint.

Partial constrained correspondence analysis (pCCA) was used to examine the independent and dependent contributions of each variable to the explained inertia (Borcard, Legendre & Drapeau, 1992; Volis *et al.*, 2011; Paliy & Shankar, 2016). The independently explained inertia is the variation explained by each variable alone, whereas the dependently explained inertia is the variation explained by each variable after accounting for the effects of the other variables.

In addition to the ordination, we analysed the effects of the independent variables on a number of univariate vegetation measures using linear and ordinal mixed-effects models, with the lme4 package (Bates *et al.*, 2015) in R Studio (R Core team, 2013). Whereas in the ordination we were looking at site-level trends, in these models we were analysing point-level and individual-level trends and by including sites or points as random effects we could control for the spatial autocorrelation nested within our data. Firstly, we analysed limb damage of *A. dhofarica* trees as a function of distance from settlements, vehicle trails and waterpoints using ordinal mixed effects regression. Secondly, we analysed the effects of the independent variables on both adult and juvenile palatable woody plant densities, using 2591 and 2531 point-plant distances for adults and juveniles, respectively. The unpalatable species removed from the analysis (and from other analyses of palatable species only) were *Acridocarpus* *orientalis*, *Jatropha dhofarica*, *Dodonaea angustifolia*, *Solanum incanum*, *Adenium obesum*, *Cadia purpurea*, *Calotropis procera* and *Gomphocarpus fruticosus*. Site eight was excluded as it was a grassland site with low woody plant density. The point-plant distances were log-transformed which improved the normality of the residuals. Finally, we analysed the effects of the independent variables on the DRC measurements of 534 adult *A. dhofarica* trees.

To investigate stunting, we plotted tree height against DRC measurements for *Commiphora habessinica, Commiphora gileadensis, A. dhofarica, E. smithii* and *Z. dhofarense* in R studio (R Core team, 2013), and fitted a linearised version of Curtis's height-diameter function (Curtis, 1967) in the 'lmfor' package, which has been found as a satisfactory fit for most datasets (Mehtätalo, de-Miguel & Gregoire, 2015).

## 3.4 Results

Forty-two adult and forty-three juvenile woody plant species (total 47) were recorded (Appendix 9, Appendix 10). *Commiphora habessinica*, *Jatropha dhofarica,* and *A. dhofarica* were the most abundant species accounting for 12.2%, 10.2% and 9.2% of the 7,200 measured woody individuals, respectively. For adults alone, the order changed to *A. dhofarica* (14.8%), *C. habessinica* (13.2%) and then *J. dhofarica* (10.3%). Adults of 57% of the species and juveniles of 70% of the species were recorded at both altitudinal ranges. Of the most abundant species in the survey (> 10% frequency at a site) only *Adenium obesum* juveniles and *Flueggea virosa* adults were restricted to high and low altitudinal ranges, respectively. For a number of characteristic tree species, few juveniles were recorded, indicating low advanced growth. Advanced growth is the ratio of juveniles to adults, expressed here as a percentage (Appendix 9). Low advanced growth can indicate poor forest regeneration. Species with advanced growth < 20% were *Delonix elata* (15%), *Ficus vasta* (16%) and *Olea europaea* subsp. *cuspidata* (7%). A number of species have relatively small populations which should be monitored. *Delonix elata, Boscia arabica, Acridocarpus orientalis, Azima tetracantha, Grewia villosa, Rhamnus staddo, Hildebrandtia africana, Cordia perrottetii, Lawsonia inermis, Caesalpinia erianthera, Calotropis procera, Searsia pyroides, Ficus sycomorus* and *Ehretia obtusifolia* occurred at a frequency of less than 0.1% across the whole study area.

Eleven of the fifteen independent variables had significant effects on vegetation responses in the univariate tests (Table 3.2). Those that did not were elevation range, rock cover, soil pH and current cattle stocking rates. Long-term stocking rate showed significant correlations with thirteen vegetation responses indicating its importance. Notably, species diversity and plant density were significantly lower, and tree limb damage (bent, broken and browsed) and bark stripping prevalence significantly higher, in areas under higher stocking rates. Adult woody species diversity and canopy cover positively correlated with fog density.

Tests between dependent variables found plant density did not significantly correlate with species diversity (adult: $F(1, 28) = 3.613$, $p = 0.068$; juvenile: $F(1, 28) = 2.906$, $p = 0.099$). Trees had on average more bent ($r_s = -0.73$, $p < 0.001$), broken ($r_s = -0.63$, $p < 0.001$) and browsed ($r_s = -0.72$, $p < 0.001$) branches at sites with lower tree densities. Tests between independent variables found fog density did not significantly correlate with slope ($F(1, 28) = 2.112$, $p = 0.157$), but did significantly positively correlate with slope aspect ($F(1, 28) = 6.001$, $p = 0.021$). ANOVA tests found percentage rock cover was significantly greater at high elevations ($F(1,28) = 15.32$, $p < 0.001$), soil pH was significantly lower at high elevations ($F(1,28) = 18.75$, $p < 0.001$) and solar radiation was significantly higher at high elevations ($F(1,28) = 20.71$, $p < 0.001$).

**Table 3.2. Table of statistical results for each independent variable tested against each dependent variable. Significant (< 0.05) results are highlighted in bold. Abbreviations in brackets indicate the statistical test used. (LR) = Linear Regression: values are the unstandardized beta coefficient (*B*) and *p*-value. (ANOVA) = One-way analysis of variance: values are the F statistic (*F*) and *p*-value. (SR) = Spearman's rho: values are the correlation coefficient ($r_s$) and *p*-value. (K-W) = Kruskal-Wallis H test: values are the Chi-square value (*H*) and *p*-value. (MEOR) = Mixed effects ordinal regression: utilised to test individual-level dependent variables nested within sites and points, and values are the coefficient (or likelihood ratio (LR) when > 2 independent variable factor levels) and *p*-value based on the Wald statistic. Only palatable species were considered for tests on proportion of browsed branches. The total number of significant tests is included in the last row.**

| | | | Site-level independent variables N = 30 | | | | | | | | Point-level independent variables (averaged for each site) N = 30 | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Adult or Juvenile (A/J) | Elevation range (high, low) | Rock cover (%) | Geology (bedrock type) | Soil pH | Long-term stocking rate (ranked 1-30) | Current camel stocking rate (animals/hectare) | Current cattle stocking rate (animals/hectare) | Fog density (arb. 0-1) | Slope (°) | Slope aspect (0-180) | Compound Topographic Index | Solar radiation (∛kWH/m2) | Distance to vehicular access (km) | Distance to house (km) | Distance to waterpoint (km) |
| Site-level dependent variables N = 30 | Diversity (Shannon) | A | (ANOVA) $F_{1,28}$=0.562, $p$=0.460 | (LR) – $B_{1,28}$=0.010, $p$=0.333 | (ANOVA) $F_{4,25}$=1.825, $p$=0.155 | (LR) - $B_{1,28}$=0.009, $p$=0.984 | (LR) - $B_{1,28}$=0.026, $p$=**0.001** | (SR) $r_{s1,28}$=-0.120, $p$=0.526 | (SR) $r_{s1,28}$=0.063, $p$=0.739 | (LR) $B_{1,28}$=2.367, $p$=**0.022** | (LR) $B_{1,28}$=0.056, $p$=**<0.001** | (LR) $B_{1,28}$=0.011, $p$=**0.001** | (LR) $B_{1,28}$=-0.452, $p$=**<0.001** | (LR) $B_{1,28}$=-0.080, $p$=**0.017** | (LR) $B_{1,28}$=0.136, $p$=0.113 | (LR) $B_{1,28}$=0.086, $p$=**0.020** | (LR) $B_{1,28}$=0.227, $p$=**0.032** |
| | | J | (ANOVA) $F_{1,28}$=3.051, $p$=0.092 | (LR) - $B_{1,28}$=0.005, $p$=0.447 | (ANOVA) $F_{4,25}$=4.515, $p$=**0.007** | (LR) $B_{1,28}$=0.359, $p$=0.221 | (LR) - $B_{1,28}$=0.020, $p$=**<0.001** | (SR) $r_{s1,28}$=-0.025, $p$=0.895 | (SR) $r_{s1,28}$=-0.012, $p$=0.950 | (LR) $B_{1,28}$=1.191, $p$=0.101 | (LR) $B_{1,28}$=0.038, $p$=**<0.001** | (LR) $B_{1,28}$=0.007, $p$=**0.002** | (LR) $B_{1,28}$=-0.293, $p$=**0.002** | (LR) $B_{1,28}$=-0.059, $p$=**0.010** | (LR) $B_{1,28}$=0.156, $p$=**0.006** | (SR) $r_{s1,28}$=0.700, $p$=**<0.001** | (LR) $B_{1,28}$=0.148, $p$=**0.044** |
| | Density (plants/hectare) | A | (K-W) $H_{1,28}$=0.115, $p$=0.735 | (SR) $r_{s1,28}$=-0.025, $p$=0.895 | (K-W) $H_{4,25}$=5.518, $p$=0.238 | (SR) $r_{s1,28}$=0.280, $p$=0.135 | (SR) $r_{s1,28}$=-0.760, **$p$=<0.001** | (SR) $r_{s1,28}$=-0.390, $p$=**0.034** | (SR) $r_{s1,28}$=-0.367, $p$=0.046 | (SR) $r_{s1,28}$=-0.058, $p$=0.760 | (SR) $r_{s1,28}$=0.623, $p$=**<0.001** | (SR) $r_{s1,28}$=0.331, $p$=0.075 | (SR) $r_{s1,28}$=-0.573, $p$=**0.001** | (SR) $r_{s1,28}$=-0.360, $p$=0.051 | (SR) $r_{s1,28}$=0.596, $p$=**<0.001** | (SR) $r_{s1,28}$=0.562, $p$=**0.001** | (SR) $r_{s1,28}$=0.459, $p$=**0.012** |
| | | J | (K-W) $H_{1,28}$=0.065, $p$=0.800 | (SR) $r_{s1,28}$=-0.003, $p$=0.989 | (K-W) $H_{4,25}$=2.829, $p$=0.587 | (SR) $r_{s1,28}$=0.217, $p$=0.249 | (SR) $r_{s1,28}$=-0.722, $p$=**<0.001** | (SR) $r_{s1,28}$=-0.428, $p$=**0.019** | (SR) $r_{s1,28}$=-0.448, $p$=0.013 | (SR) $r_{s1,28}$=-0.225, $p$=0.230 | (SR) $r_{s1,28}$=0.572, $p$=**0.001** | (SR) $r_{s1,28}$=0.174, $p$=0.356 | (SR) $r_{s1,28}$=-0.530, $p$=**0.003** | (SR) $r_{s1,28}$=-0.345, $p$=0.063 | (SR) $r_{s1,28}$=0.627, $p$=**<0.001** | (SR) $r_{s1,28}$=0.395, $p$=**0.032** | (SR) $r_{s1,28}$=0.492, $p$=**0.006** |
| | Canopy cover (%) | - | (ANOVA) $F_{1,28}$=1.517, $p$=0.228 | (LR) $B_{1,28}$=-0.983, $p$=0.052 | (ANOVA) $F_{4,25}$=1.707, $p$=0.180 | (LR) $B_{1,28}$=22.340, $p$=0.304 | (LR) $B_{1,28}$=-1.485, $p$=**<0.001** | (LR) $B_{1,28}$=-0.007, $p$=0.900 | (SR) $r_{s1,28}$=-0.150, $p$=0.430 | (LR) $B_{1,28}$=127.200, $p$=**0.015** | (SR) $r_{s1,28}$=2.504, $p$=**<0.001** | (LR) $B_{1,28}$=0.547, $p$=**0.001** | (LR) $B_{1,28}$=-25.788, $p$=**<0.001** | (LR) $B_{1,28}$=-4.273, $p$=**0.012** | (LR) $B_{1,28}$=12.961, $p$=**0.002** | (LR) $B_{1,28}$=5.167, $p$=**0.005** | (LR) $B_{1,28}$=4.295, $p$=**0.008** |
| | Proportion of dead individuals (%) | A | (ANOVA) $F_{1,28}$=0.139, $p$=0.712 | (LR) $B_{1,28}$=0.001, $p$=0.504 | (ANOVA) $F_{4,25}$=4.309, $p$=**0.009** | (LR) $B_{1,28}$=-0.063, $p$=0.201 | (LR) $B_{1,28}$=0.002, $p$=0.117 | (LR) $B_{1,28}$=0.020, $p$=**0.024** | (LR) $B_{1,28}$=-0.000003, $p$=0.848 | (LR) $B_{1,28}$=0.062, $p$=0.624 | (LR) $B_{1,28}$=-0.002, $p$=0.123 | (LR) $B_{1,28}$=-0.0001, $p$=0.804 | (LR) $B_{1,28}$=0.018, $p$=0.294 | (LR) $B_{1,28}$=0.006, $p$=0.137 | (LR) $B_{1,28}$=-0.006, $p$=0.483 | (LR) $B_{1,28}$=-0.004, $p$=0.413 | (LR) $B_{1,28}$=0.004, $p$=0.348 |

| | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Proportion of bark-stripped individuals (%) | A | (K-W) $H_{1,28}$=0.022, $p$=0.882 | (SR) $r_{s1,28}$=0.071, $p$=0.708 | (K-W) $H_{4,25}$=3.795, $p$=0.435 | (SR) $r_{s1,28}$=-0.077, $p$=0.687 | (SR) $r_{s1,28}$=0.603, $p$=**<0.001** | (SR) $r_{s1,28}$=0.339, $p$=0.067 | (SR) $r_{s1,28}$=-0.010, $p$=0.960 | (SR) $r_{s1,28}$=-0.165, $p$=0.384 | (SR) $r_{s1,28}$=-0.678, $p$=**<0.001** | (SR) $r_{s1,28}$=-0.220, $p$=0.243 | (SR) $r_{s1,28}$=0.458, $p$=**0.011** | (SR) $r_{s1,28}$=0.429, $p$=**0.018** | (SR) $r_{s1,28}$=-0.266, $p$=0.155 | (SR) $r_{s1,28}$=-0.382, $p$=**0.037** | (SR) $r_{s1,28}$=-0.250, $p$=0.183 |
| | Total basal area (cm2) | A | (ANOVA) $F_{1,28}$=1.998, $p$=0.169 | (LR) $B_{1,28}$=-0.031, $p$=0.053 | (ANOVA) $F_{4,25}$=2.857, $p$=**0.045** | (LR) $B_{1,28}$=-0.097, $p$=0.889 | (SR) $r_{s1,28}$=0.390, **$p$=0.034** | (LR) $B_{1,28}$=0.004, $p$=**0.034** | (LR) $B_{1,28}$=0.003, $p$=0.329 | (LR) $B_{1,28}$=3.170, $p$=0.058 | (LR) $B_{1,28}$=-0.002, $p$=0.908 | (LR) $B_{1,28}$=0.005, $p$=0.394 | (LR) $B_{1,28}$=-0.062, $p$=0.791 | (LR) $B_{1,28}$=-0.015, $p$=0.796 | (LR) $B_{1,28}$=-0.088, $p$=0.429 | (LR) $B_{1,28}$=-0.030, $p$=0.624 | (LR) $B_{1,28}$=0.019, $p$=0.723 |
| | | J | (ANOVA) $F_{1,28}$=1.028, $p$=0.319 | (LR) $B_{1,28}$=5.000, $p$=0.280 | (ANOVA) $F_{4,25}$=1.284, $p$=0.303 | (LR) $B_{1,28}$=-176.9, $p$=0.363 | (SR) $r_{s1,28}$=0.662, $p$=**<0.001** | (LR) $B_{1,28}$=1.027, $p$=**0.029** | (SR) $r_{s1,28}$=0.286, $p$=0.125 | (LR) $B_{1,28}$=-605.8, $p$=0.211 | (LR) $B_{1,28}$=-23.742, $p$=**<0.001** | (LR) $B_{1,28}$=-4.779, $p$=**0.002** | (LR) $B_{1,28}$=222.39, $p$=**<0.001** | (LR) $B_{1,28}$=37.920, $p$=**0.013** | (LR) $B_{1,28}$=-56.16, $p$=0.069 | (LR) $B_{1,28}$=-34.73, $p$=**0.041** | (LR) $B_{1,28}$=-33.37, $p$=**0.024** |
| | Average height (cm) | A | (K-W) $H_{1,28}$=0.948, $p$=0.330 | (SR) $r_{s1,28}$=-0.178, $p$=0.345 | (K-W) $H_{4,25}$=3.780, $p$=0.437 | (SR) $r_{s1,28}$=0.153, $p$=0.419 | (SR) $r_{s1,28}$=-0.252, $p$=0.179 | (SR) $r_{s1,28}$=0.209, $p$=0.267 | (SR) $r_{s1,28}$=-0.152, $p$=0.423 | (SR) $r_{s1,28}$=0.176, $p$=0.351 | (SR) $r_{s1,28}$=0.325, $p$=0.080 | (SR) $r_{s1,28}$=0.257, $p$=0.169 | (SR) $r_{s1,28}$=-0.463, $p$=**0.011** | (SR) $r_{s1,28}$=-0.292, $p$=0.117 | (SR) $r_{s1,28}$=0.346, $p$=0.061 | (SR) $r_{s1,28}$=0.247, $p$=0.188 | (SR) $r_{s1,28}$=0.510, $p$=**0.004** |
| | | J | (K-W) $H_{1,28}$=0.145, $p$=0.703 | (SR) $r_{s1,28}$=-0.200, $p$=0.289 | (K-W) $H_{4,25}$=3.017, $p$=0.555 | (SR) $r_{s1,28}$=0.146, $p$=0.441 | (SR) $r_{s1,28}$=-0.364, $p$=**0.049** | (SR) $r_{s1,28}$=-0.428, $p$=**0.019** | (SR) $r_{s1,28}$=-0.448, $p$=0.013 | (SR) $r_{s1,28}$=0.329, $p$=0.076 | (SR) $r_{s1,28}$=0.229, $p$=0.223 | (SR) $r_{s1,28}$=0.188, $p$=0.317 | (SR) $r_{s1,28}$=-0.289, $p$=0.121 | (SR) $r_{s1,28}$=-0.236, $p$=0.208 | (SR) $r_{s1,28}$=0.211, $p$=0.263 | (SR) $r_{s1,28}$=0.504, $p$=**0.005** | (SR) $r_{s1,28}$=0.077, $p$=0.686 |
| | Proportion of broken branches (1-5 scale) N = 3600 | A | (MEOR) $B_{4,2764}$=0.093, $p$=0.728 | (MEOR) $B_{4,2764}$=0.010, $p$=0.569 | (MEOR) $LR_{8,2760}$=17.456, $p$=**0.015** | (MEOR) $B_{4,2764}$=-0.125, $p$=0.866 | (MEOR) $B_{4,2764}$=0.031, $p$=**<0.001** | (MEOR) $B_{4,2764}$=0.005, $p$=**0.002** | (MEOR) $B_{4,2764}$=-0.001, $p$=0.797 | (MEOR) $B_{4,2764}$=-0.330, $p$=0.822 | (MEOR) $B_{4,2764}$=-0.027, $p$=**0.006** | (MEOR) $B_{4,2764}$=-0.009, $p$=**<0.001** | (MEOR) $B_{4,2764}$=0.004, $p$=0.910 | (MEOR) $B_{4,2764}$=-0.031, $p$=0.300 | (MEOR) $B_{4,2764}$=0.039, $p$=0.663 | (MEOR) $B_{4,2764}$=0.011, $p$=0.859 | (MEOR) $B_{4,2764}$=-0.006, $p$=0.901 |
| Individual-level dependent variables | Proportion of bent branches (1-5 scale) N = 3600 | A | (MEOR) $B_{4,3247}$=-0.079, $p$=0.864 | (MEOR) $B_{4,3247}$=-0.007, $p$=0.828 | (MEOR) $LR_{8,3243}$=18.567, $p$=**0.010** | (MEOR) $B_{4,3247}$=-1.791, $p$=0.163 | (MEOR) $B_{4,3247}$=0.042, $p$=**<0.001** | (MEOR) $B_{4,3247}$=0.008, $p$=**0.002** | (MEOR) $B_{4,3247}$=0.004, $p$=0.481 | (MEOR) $B_{4,3247}$=1.904, $p$=0.402 | (MEOR) $B_{4,3247}$=-0.009, $p$=0.547 | (MEOR) $B_{4,3247}$=-<0.0001, $p$=0.998 | (MEOR) $B_{4,3247}$=-0.008, $p$=0.872 | (MEOR) $B_{4,3247}$=0.022, $p$=0.653 | (MEOR) $B_{4,3247}$=-0.249, $p$=0.077 | (MEOR) $B_{4,3247}$=-0.144, $p$=0.119 | (MEOR) $B_{4,3247}$=-0.030, $p$=0.721 |
| | Proportion of browsed branches (1-5 scale) | A N = 2681 | (MEOR) $B_{4,2520}$=-0.557, $p$=0.415 | (MEOR) $B_{4,2520}$=0.049, $p$=0.266 | (MEOR) $LR_{8,2515}$=21.349, $p$=**0.003** | (MEOR) $B_{4,2520}$=-2.220, $p$=0.254 | (MEOR) $B_{4,2520}$=0.086, $p$=**<0.001** | (MEOR) $B_{4,2520}$=0.011, $p$=**0.008** | (MEOR) $B_{4,2520}$=0.005, $p$=0.536 | (MEOR) $B_{4,2520}$=-6.553, $p$=**0.015** | (MEOR) $B_{4,2520}$=-0.070, $p$=**<0.001** | (MEOR) $B_{4,2520}$=-0.011, $p$=**<0.001** | (MEOR) $B_{4,2520}$=0.035, $p$=0.475 | (MEOR) $B_{4,2520}$=0.120, $p$=**<0.001** | (MEOR) $B_{4,2520}$=-0.568, $p$=**<0.001** | (MEOR) $B_{4,2520}$=-0.420, $p$=**<0.001** | (MEOR) $B_{4,2520}$=-0.287, $p$=**0.004** |
| | | J N = 2608 | (MEOR) $B_{4,2604}$=-0.154, $p$=0.687 | (MEOR) $B_{4,2604}$=0.015, $p$=0.561 | (MEOR) $LR_{8,2600}$=20.196, $p$=**0.005** | (MEOR) $B_{4,2604}$=-0.416, $p$=0.690 | (MEOR) $B_{4,2604}$=0.050, $p$=**<0.001** | (MEOR) $B_{4,2604}$=0.008, $p$=**<0.001** | (MEOR) $B_{4,2604}$=0.003, $p$=0.547 | (MEOR) $B_{4,2604}$=-2.970, $p$=0.122 | (MEOR) $B_{4,2604}$=-0.051, $p$=**<0.001** | (MEOR) $B_{4,2604}$=-0.009, $p$=**<0.001** | (MEOR) $B_{4,2604}$=-0.016, $p$=0.703 | (MEOR) $B_{4,2604}$=0.121, $p$=**<0.001** | (MEOR) $B_{4,2604}$=-0.413, $p$=**<0.001** | (MEOR) $B_{4,2604}$=-0.242, $p$=**<0.001** | (MEOR) $B_{4,2604}$=-0.187, $p$=**0.004** |
| Number of significant tests | | | 0 | 0 | 7 | 0 | 13 | 10 | 0 | 3 | 10 | 7 | 8 | 7 | 6 | 10 | 9 |

Long-term stocking rate (1-30 scale) decreased significantly by 0.711 for each one degree increase in slope incline (Figure 3.2). Long-term stocking rate did not differ significantly by elevation range. Current stocking rates (dung transects) of camels and cattle at the thirty sites, showed no significant correlation with slope gradient (cattle: $r(28) = 0.052$, $p = 0.784$; camels: $r(28) = 0.225$, $p = 0.231$) or with any other environmental variables, apart from fog density which significantly correlated with cattle density ($r(28) = 0.366$, $p = 0.047$). No significant correlation was observed between current and long-term stocking rates.

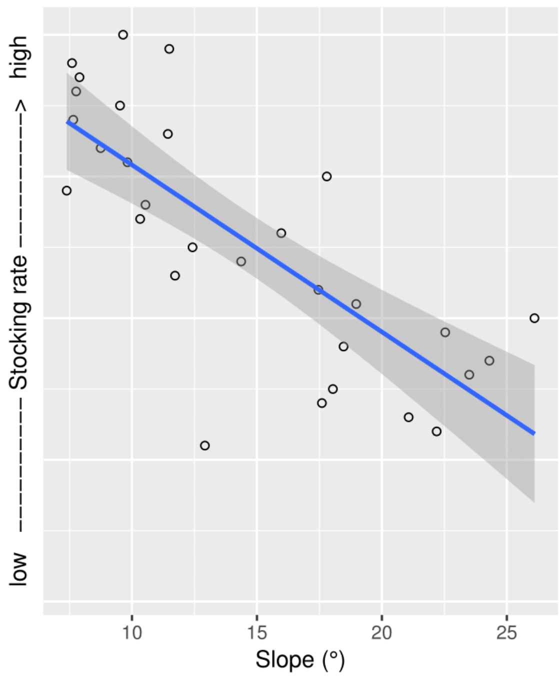



**Figure 3.2. Scatter plot of correlation between long-term stocking rate and slope with linear regression line and 95% confidence interval.**

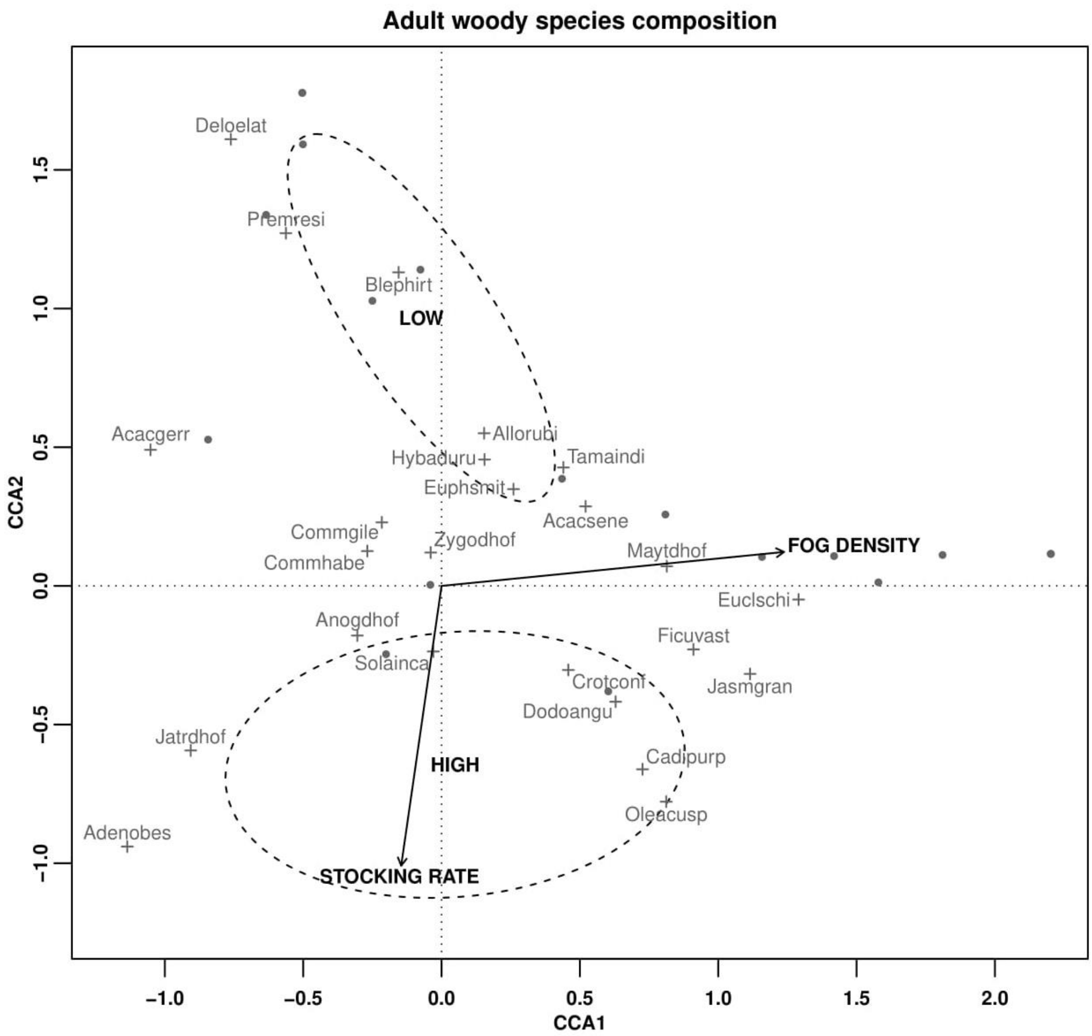
The CCA analysis identified fog density, long-term stocking rate, and elevation range as the most powerful constraining variables for both adult and juvenile woody species community composition, based on the proportion of explained inertia and permutation tests. The CCA models explained approximately 28% of the inertia in woody vegetation species composition. The pCCA (Table 3.3) showed that, after accounting for the effects of the other variables (dependently explained inertia), long-term stocking rate was the third most important factor affecting adult woody species composition, after fog density and elevation, and the second most important factor affecting juvenile woody species composition, after fog density.

In our CCA model building procedure, long-term stocking rate was found to be more powerful and significant than slope. When added to the model, slope was not significant but resulted in a 4.2% increase in absolute explained inertia, suggesting slope may have a slight independent effect on species composition. However, collinearity (VIF = 3) between slope and long-term stocking rate (Figure 3.2) confounded interpretation of the model, thus slope was removed from the model and degrees of freedom were preserved.

**Table 3.3. Dependently and independently explained inertia for each constraining variable and absolute explained inertia, for both adult and juvenile woody species communities.**

| Constrained Correspondence Analysis (CCA) | | | | | |
|---|---|---|---|---|---|
| Vegetation abundance measure | Constraining variable (inertia explained) | | | | Absolute inertia explained |
| | | Fog Density | Long-term stocking rate | Elevation range | |
| Adult woody species composition | Dependently | 12.3% | 7.5% | 7.7% | 28.1% |
| | Independently | 12.2% | 8.4% | 8.3% | |
| Juvenile woody species composition | Dependently | 9.8% | 9.7% | 8.5% | 28.4% |
| | Independently | 9.7% | 10.1% | 8.9% | |

The CCA biplots of species scores enable interpretation of the effects of the constraining variables on adult and juvenile woody species composition (Figure 3.3). In both biplots we see a higher diversity of species, particularly uncommon species, associated with lower stocking rates and higher fog densities. All unpalatable or unfavoured species have their optimum in areas with above-average stocking rates. *Adenium obesum* (unpalatable) and *J. dhofarica* (unfavoured) are very strongly associated with high stocking rates in drier areas with lower fog densities. *Dodonaea viscosa* subsp. *angustifolia* (unpalatable) and *Cadia purpurea* (unpalatable) are associated with high stocking rates in areas with higher fog densities. *Solanum incanum* (unfavoured) is a more generalist species, associated with average fog densities but higher stocking rates. Juveniles of *A. dhofarica* appear to prefer areas with higher fog densities, whilst adults are more abundant under lower fog densities and higher stocking rates. Juveniles of the palatable species, *Zygocarpum dhofarensis*, *Maytenus dhofarensis* and *Allophylus rubifolius*, are more abundant under higher stocking rates than adults of these species.



Adult woody species composition
Deloelat
Premresi
Blephirt
LOW
Acacgerr
Allorubi
Hybaduru
Tamaindi
Euphsmit
Acacsene
Commgile
Zygodhof
Commhabe
Maytdhof
FOG DENSITY
Euclschi
Anogdhof
Ficuvast
Solainca
Crotconf
Jasmgran
Dodoangu
Jatrdhof
HIGH
Cadipurp
Oleacusp
Adenobes
STOCKING RATE
CCA1
CCA2

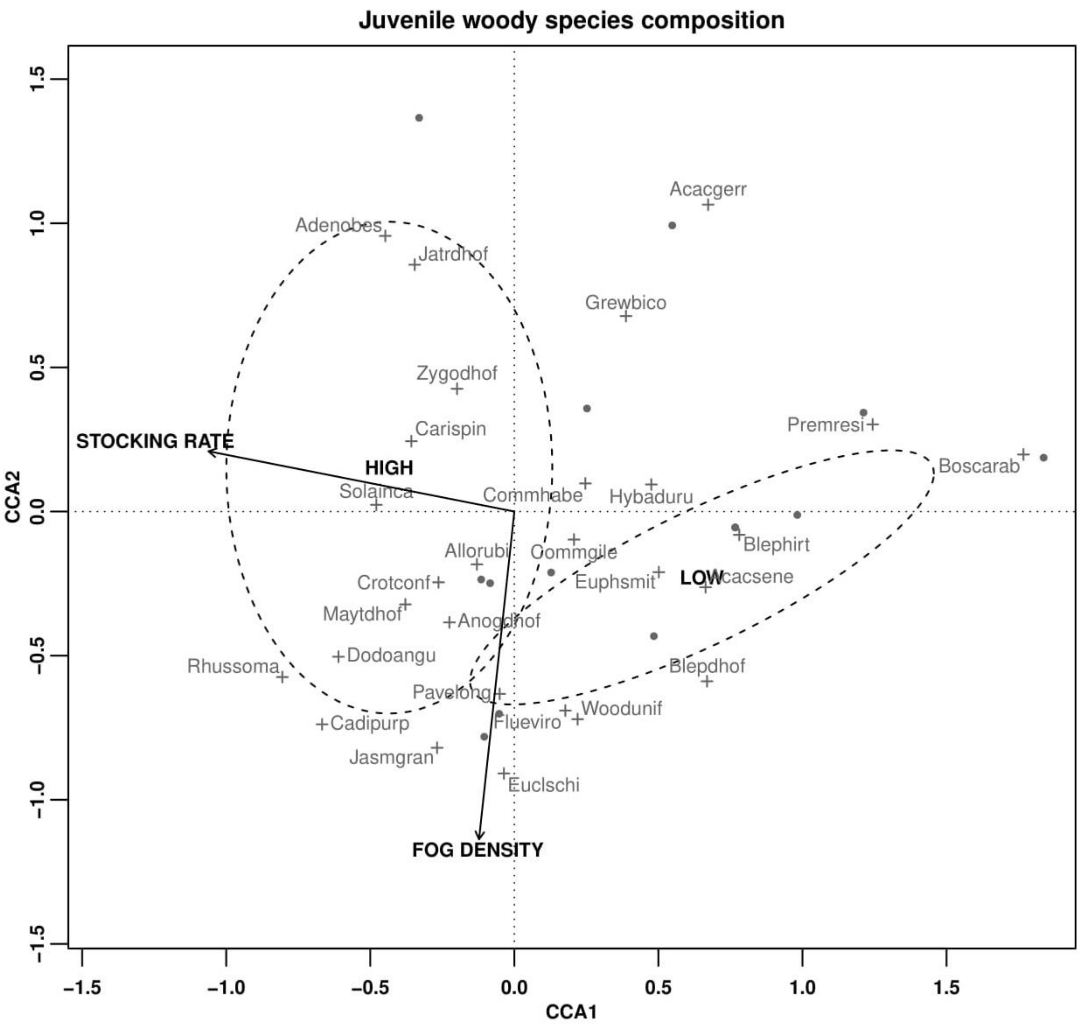



**Figure 3.3. Constrained correspondence analysis (CCA) biplots of adult and juvenile woody species composition. Common species (overall frequency > 10) are labelled using Cornell Ecology Programs (CEP) names and represented by a + symbol, and other species are represented by a point. Fog density and long-term stocking rate are continuous variables shown as arrows and elevation range is a categorical variable shown as centroids of low and high elevation ranges. The length of the arrow indicates the strength of the variable and the ellipse shows the standard error (0.95) of the weighted average of scores and the weighted correlation defines the direction of the principal axis of the ellipse.**

Ten percent of adult woody plants (4% total adult basal area) were dead. Of the adult trees and large shrubs (Table 3.4), 85% had broken limbs, 13% had been subject to branch bending management practises, and ten percent had exposed cambium due to bark stripping by livestock. *Dodonaea viscosa* subsp. *angustifolia, C. gileadensis* and *Olea europaea* subsp. *cuspidata* were the most frequently dead species and the majority of adults of all species had broken branches to some extent, either naturally or due to disturbance. Over half (57%) of *A. dhofarica* trees had been subject to branch management practises, and other managed species included *Ficus vasta, Tamarindus indica*, *Croton confertus*, *A. rubifolius* and *O. europaea* subsp. *cuspidata*. The most frequently bark-stripped species were *A. dhofarica*, *Blepharispermum hirtum*, *J. dhofarica*, *F. vasta* and *Ficus sycomorus*, and *O. europaea* subsp. *schimperi* had on average the largest areas of stripped bark.

**Table 3.4. Proportions of individuals of adult trees and large shrubs (n=2949) that were dead, had broken or bent limbs, or were bark stripped.**

| n= | Species | % Dead | % Broken limbs | % Bent limbs | % Bark stripped | Avg. area of stripped bark ($cm^2$) |
|---|---|---|---|---|---|---|
| 534 | *Anogeissus dhofarica* | 9% | 97% | 57% | 19% | 1490 |
| 474 | *Commiphora habessinica* | 5% | 81% | 0% | 5% | 245 |
| 371 | *Jatropha dhofarica* | 12% | 76% | 1% | 11% | 322 |
| 274 | *Dodonaea viscosa* subsp. *angustifolia* | 25% | 79% | 1% | 5% | 188 |
| 223 | *Euphorbia smithii* | 4% | 87% | 2% | 1% | 61 |
| 214 | *Blepharispermum hirtum* | 11% | 89% | 5% | 38% | 336 |
| 194 | *Commiphora gileadensis* | 13% | 74% | 2% | 9% | 997 |
| 155 | *Allophylus rubifolius* | 12% | 89% | 18% | 8% | 266 |
| 153 | *Maytenus dhofarensis* | 5% | 80% | 5% | 0% | 0 |
| 89 | *Croton confertus* | 3% | 86% | 19% | 5% | 69 |
| 86 | *Olea europaea* subsp. *cuspidata* | 13% | 98% | 16% | 7% | 2930 |
| 60 | *Euclea racemosa* subsp. *schimperi* | 7% | 76% | 0% | 0% | 0 |
| 44 | *Acacia senegal* | 11% | 80% | 2% | 2% | 20 |
| 19 | *Ficus vasta* | 0% | 80% | 24% | 16% | 1897 |
| 18 | *Acacia gerrardii* | 0% | 80% | 0% | 0% | 0 |
| 13 | *Delonix elata* | 8% | 100% | 0% | 0% | 0 |
| 11 | *Tamarindus indica* | 0% | 88% | 20% | 0% | 0 |
| 10 | *Rhus somalensis* | 10% | 86% | 11% | 0% | 0 |
| 5 | *Ficus sycomorus* | 0% | 100% | 0% | 40% | 1378 |
| 2 | *Boscia arabica* | 0% | 100% | 0% | 0% | 0 |
| 2949 | Grand total | 10% | 85% | 13% | 10% | 802.43 |

Our results in Table 3.2 show that across all palatable species the proportion of browsed branches of adults and juveniles decreased significantly with increasing distance from houses, vehicular access routes and water points, indicative of the piosphere model. The same scenario was observed for the proportion of broken and bent branches of *A. dhofarica* adults in relation to houses, but not for adults of all palatable species. For *A. dhofarica*, an increase in distance (km) from houses was associated with a decrease in the odds of a tree having more broken branches, with an odds ratio of - 0.325 (SE: 0.092), Wald $p$-value < 0.001. Likewise, an increase in distance (km) was associated with a decrease in the odds of a tree having more bent branches, with an odds ratio of - 0.300 (SE: 0.143), Wald $p$-value < 0.036.

Figure 3.4 shows the results of the linear mixed-effects regression analysis of palatable woody plant density. The results show that very high long-term stocking rates significantly reduce adult palatable woody plant densities and high or very high long-term stocking rates significantly reduce juvenile palatable woody plant densities. Palatable woody plant densities are significantly lower at lower elevations, and increases in slope gradient and rock cover result in significantly higher palatable woody plant densities. Only adult density significantly increased with increasing distance from settlements. Fog density did not significantly influence palatable woody plant densities.

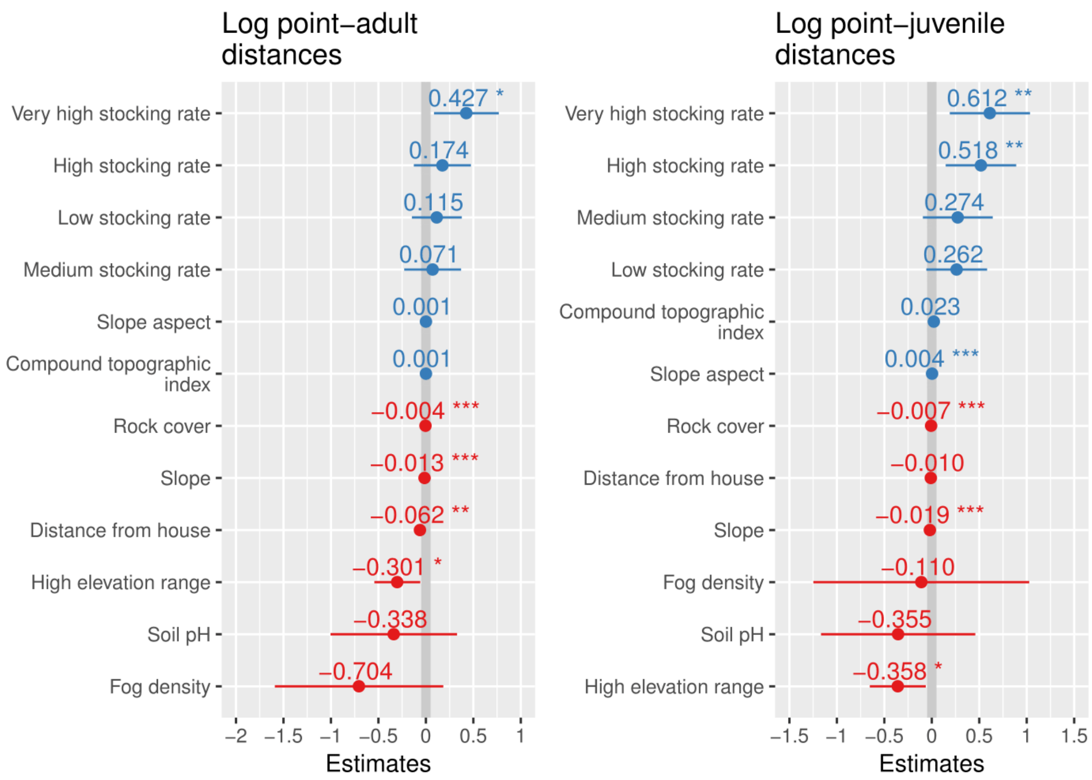



**Figure 3.4. Coefficient estimates, confidence intervals (lines) and significance of log adult and juvenile point-plant distances (palatable species only) for each variable in a linear mixed-effects regression model with sites, points and species as random group effects. A positive coefficient indicates lower plant densities and a negative coefficient indicates higher plant densities. Very low stocking rate and low elevation range are the reference classes for categorical variables. A grassland site with very low woody plant density has been excluded from the analysis.**

Figure 3.5 shows the results of the linear mixed-effects regression analysis of *A. dhofarica* basal area. The model showed fog density had the greatest significant positive effect on basal area. Low, high and very high long term stocking rates (relative to very low stocking rates) also had a significant positive effect on basal area. Rock cover had a slight but significant negative effect on basal area.

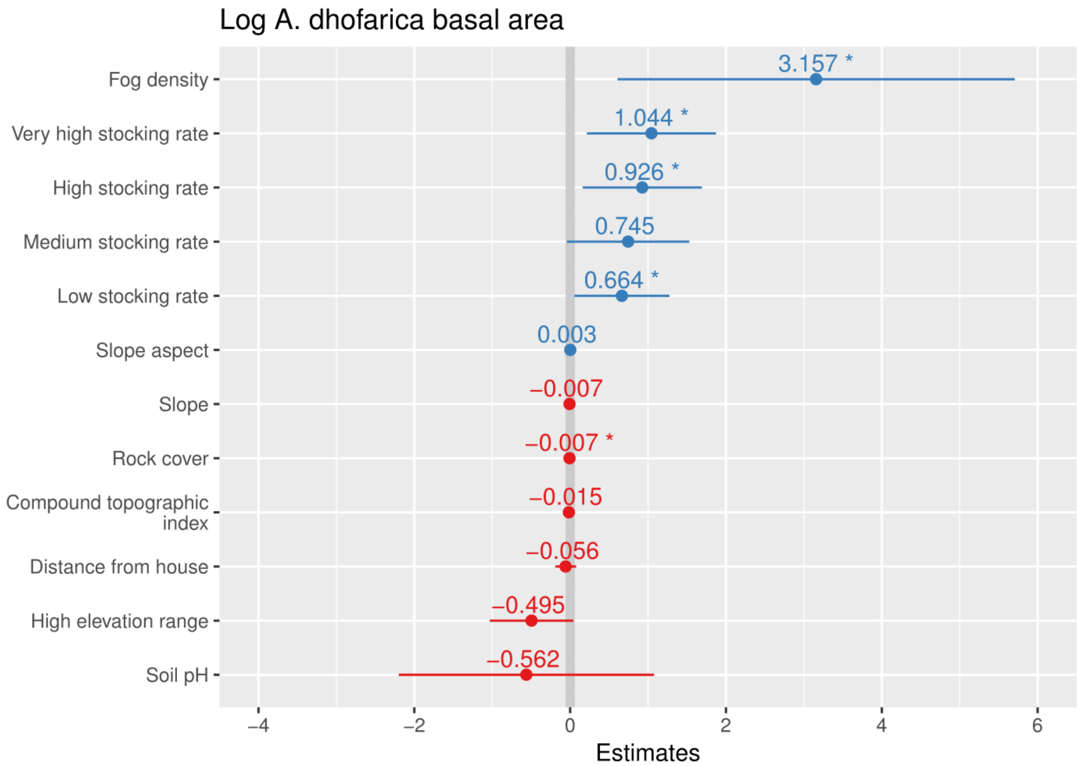



**Figure 3.5. Coefficient estimates, confidence intervals (lines) and significance of log adult *Anogeissus dhofarica* basal areas for each variable in a linear mixed-effects regression model with sites and points as random group effects. A positive coefficient indicates larger basal area and a negative coefficient indicates smaller basal area. Very low stocking rate and low elevation range are the reference classes for categorical variables.**

We plotted the DRC-height relationships of five widespread and abundant woody species (Figure 3.6). *Commiphora habessinica* showed the most pronounced trend towards shorter and thinner individuals under high stocking rates, whilst under low stocking rates a much broader range of tree sizes were recorded. Under very low stocking rates a similarly broad range of sizes of *C. gileadensis* were recorded but as stocking rates increased the data appears to split into two groups of small and large trees with few medium-sized trees between 2-4m in height and 10-15 cm in diameter. The frequency of large mature *A. dhofarica* trees increased under higher stocking rates. *Euphorbia smithii* and *Z. dhofarense* showed a trend towards shorter individuals under high stocking rates. Young adults, with smaller DRC's, for all five species tended towards stunted forms under high stocking rates, showing shorter heights for equivalent DRC measurements as stocking rates increased.

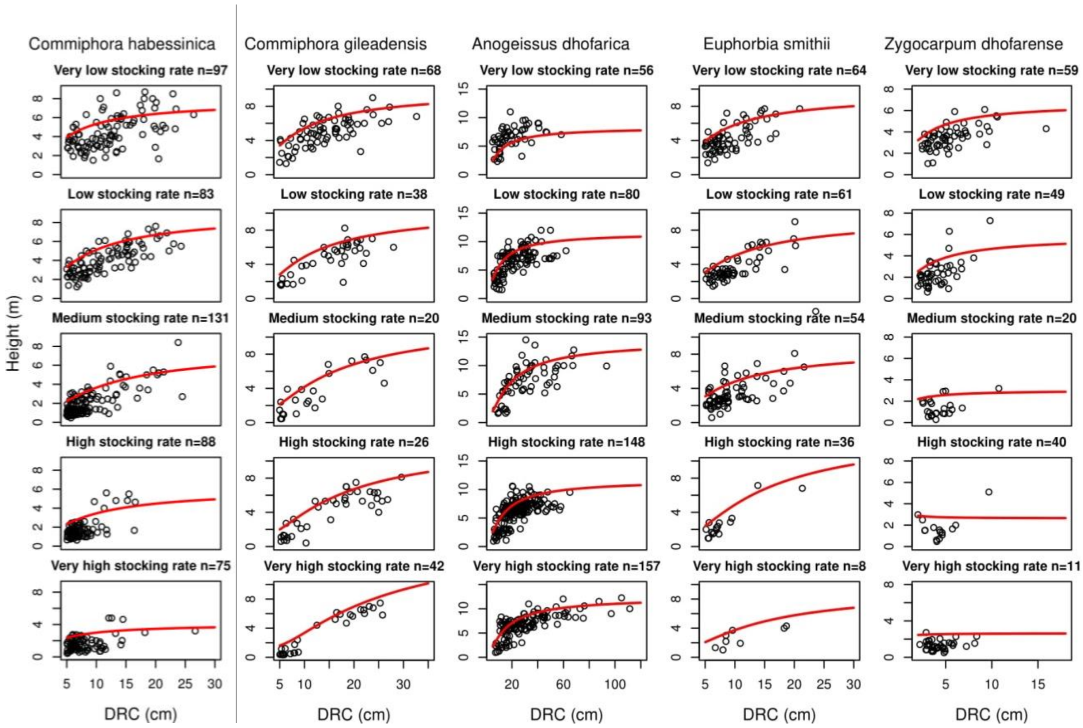



**Figure 3.6. Adult diameter at root collar (DRC) and height relationships of five widespread and abundant woody species. We have fitted a linearised version of Curtis's height-diameter function indicated by the red line.**

## 3.5 Discussion

### 3.5.1 Biotic and abiotic variables that influence woody vegetation

Our research corroborates existing evidence that the Khareef fog is critical to supporting woody vegetation in Dhofar (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2007; Friesen *et al.*, 2018). Moreover, our research identifies for the first time, that small-scale spatial variability in fog density is the primary factor influencing woody vegetation species composition. Spatial variability in fog density is closely linked to topography (Appendix 8). For example, fog density is highest in areas where steep topography limits its movement inland (such as at the top of tributaries and below cliffs). In these areas we see higher species diversity (Table 3.2, Figure 3.3) and larger trees (Figure 3.5). We also see greater canopy cover (Table 3.2) which conforms to findings from African savannahs, where in areas receiving a mean annual precipitation of less than 650 mm, woody canopy cover is constrained by and increases with precipitation (Sankaran *et al.*, 2005). Despite this, adult woody plant density did not differ significantly with fog density (Figure 3.4). We attribute this primarily to the presence of larger trees at lower densities (Scholes & Archer, 1997) but also large rocks and boulders in high-moisture areas (e.g. at the base of cliffs) may limit woody plant density. Interestingly, in Dhofar, evidence suggests that a complex tree canopy structure, with varying canopy heights and gaps, is likely to capture greater quantities of fog than a continuous smooth canopy (Hildebrandt & Eltahir, 2008). The fog-vegetation relationship is important in other arid cloud forests where rainfall is limited and horizontal precipitation provides for a large part of the water balance (Hildebrandt & Eltahir, 2008). For example, fog has been found to correlate with distributions of *Dracaena cinnabari* on the island of Socotra, where quantities of horizontal precipitation are comparable with Dhofar (Scholte & De Geest, 2010). In addition, fog and aspect differentially influenced vegetation in the Negev desert (Kidron, 2005) and areas of fog accumulation affected vegetation in the arid mountains of Mexico (Martorell & Ezcurra, 2002).

Aspect, being closely linked to temperature and light availability, is often an important constraint affecting plant communities (Stage, 1976), however linear aspect showed no significant relationship with vegetation measures in our study because most sites were south-facing, thus it was excluded from analysis. Our aspect-derived variables such as slope aspect and solar radiation were significant, but were acting as proxies for fog exposure (although fog density was a more powerful variable), rather than representing incoming solar radiation as both are southerly-oriented processes which vary with slope gradient. Indeed, Holland and Steyn (1975) suggested that the role of aspect at lower latitudes is less related to sunlight and temperature, and more explicable by reference to the directions of the principal rain-bearing winds.

Elevation range showed no significant relationship with our univariate vegetation measures such as plant density, species diversity and tree heights (Table 3.2) but was the third and second most important factor affecting juvenile and adult woody plant species composition, respectively. Elevation can be a key driver of vegetation patterns in topographically varied landscapes in arid regions (Shreve, 1922), depending on the elevation range in question. In this study, the effects of elevation were limited because our sampling was restricted to the 300–500m and 700–900m a.s.l. zones of the 100–1000m a.s.l. altitudinal range of the *Anogeissus* forest (Kürschner *et al.*, 2004). Consequently, we suggest future descriptive or classification studies of the *Anogeissus* forest should account for local variability in fog, and strictly stocking rates too, in addition to altitude.

Palatable woody plant density was found to be greater at the high elevation range despite no significant altitudinal variation in long-term stocking rates. However, ground cover of small rocks was greater at high elevations and plant density increases with rock cover (Figure 3.4). High densities of *Commiphora* shrubs occurred at rocky sites at high elevations near Sha'at and Aghethob whilst several flat lowland sites had sparse understories. It is likely high rock cover reduces accessibility to livestock and protects saplings (Appendix 11) in a similar way that Carson *et al.* (2005) found boulders protected woody species from deer browsing in Pennsylvania. In addition, low herbaceous cover in rocky areas may allow woody plants to quickly achieve vertical dominance (Scholes & Archer, 1997).

In Jabal Qamar, akin to other rangeland systems, as slope gradient increases, stocking rate (and browsing intensity) decreases (Figure 3.2), as livestock prefer to feed on flat ground (Mueggler, 1965; Mwendera, Mohamed Saleem & Dibabe, 1997). Furthermore, in Dhofar browsing intensity is higher near settlements (Table 3.2) where livestock are watered and receive feedstuffs, leading to localised vegetation degradation. The addition of slope to the CCA models revealed a slight independent effect on species composition, possibly due to slope-vegetation relationships pertaining to radiant energy income, hydrology and soils (Holland & Steyn, 1975; Nearing, 1997). Current stocking rates (prevailing at our sites during the fieldwork period) did not vary by slope or distance from settlements. This suggests that although slope gradient has influenced stocking rates in the past, at present, livestock may be occupying steeper and less accessible areas. Indeed, cattle were observed navigating rocky slopes approaching 45 degree inclines to access grazing areas; a behaviour seen in small cattle breeds (Howery, Bailey & Laca, 1999).

### 3.5.2 Effect of browsing on woody vegetation species composition

Long-term stocking rate was the third most influential variable affecting adult woody vegetation species composition. This indicates that the prevailing species composition of the mature woody layer has been influenced by browsing activity. For juveniles, it was the second most influential variable. Seedlings and saplings are more susceptible than adults due to their smaller size (Scholes & Archer, 1997; Ripple *et al.*, 2001; Côté *et al.*, 2004; Smit *et al.*, 2007; Staver *et al.*, 2009) and the composition of juveniles can be affected by removal of adult reproductive components (Augustine & Decalesta, 2003). Thus, we can expect to see greater shifts in species composition as the forest regenerates under current conditions. Camel browsing has been found to affect plant species composition in deserts in the UAE where exclusion of camels resulted in increased plant density and species richness (El-Keblawy, Ksiksi & El Alqamy, 2009).

Decreased botanical diversity and increased frequencies of unpalatable species are well-known symptoms of overstocked rangelands (Thalen, 1979; Tzanopoulos, Mitchley & Pantis, 2005; Seymour *et al.*, 2010; Briske, 2017), which have been previously described from Dhofar (Ghazanfar, 1998; Peacock *et al.*, 2003; Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005; Patzelt, 2012). Here we provide further evidence to support these claims. In particular, we find all unpalatable or unfavoured woody species have their optimums at above-average stocking rates, although the succulents, *A. obesum* and *J. dhofarica*, naturally prefer the more xeric conditions of the plateau, where coincidently stocking rates are amongst the highest due to the close proximity to settlements.

From model simulations, Hildebrandt and Eltahir (2008) concluded that degraded forests in Dhofar may not recover due to reduced horizontal precipitation following degradation. Promisingly, we found juveniles of *Maytenus dhofarensis*, *Allophylus rubifolius* and *Zygocarpum dhofarensis* were more abundant under higher stocking rates than adults, with the latter also favouring drier conditions which prevail in sparse woodlands and grasslands (Miller, Morris & Stuart-Smith, 1988). These may represent important species for restoration of degraded areas. The reasons for their survivability are unclear, as all three species are important browsing fodder species (although *M. dhofarensis* is unfavoured by cattle) (Miller, Morris & Stuart-Smith, 1988). They may be encroaching due to low herbaceous biomass and the absence of fires (Scholes & Archer, 1997), exhibiting higher seed production under stress (Huntley & Walker, 1982), hold a competitive phenological advantage due to early bud burst (Scholes & Archer, 1997), or are protected by an unpalatable herbaceous layer (Scholes & Archer, 1997; Smit *et al.*, 2007). It is also plausible that the strength and hardness of the wood of these species, which was traditionally favoured for construction and to make weapons such as fishing spears (Miller, Morris & Stuart-Smith, 1988) makes them too indigestible to be consumed in their entirety by livestock.

### 3.5.3 Effect of browsing on the structure of the woody plant layer

We found palatable woody species density decreases significantly, and quite substantially, under very high and high stocking rates, respectively for adults and juveniles. These results, particularly for adults, may be indicative of an ecological threshold (Friedel, 1991), where forest recovery is inhibited by a reduced capacity for horizontal precipitation capture. Once tree density is low and horizontal precipitation is reduced, less moisture penetrates deep into the soil (exacerbated further by compaction from trampling), giving grasses a competitive advantage (Casper & Jackson, 1997). The restoration of woody cover may well require significant intervention (Vetter, 2005).

In a global study using data from 236 locations, Milchunas and Lauenroth (1993) found above-ground net primary productivity decreased with grazing and Al-Rowaily *et al.* (2015) reported decreased woody species density with increased stocking rates in Saudi Arabia. Akin to our results for species composition, our results for density, suggest juveniles are more susceptible to browsing than adults. The susceptibility of seedlings and saplings, prevents their maturation into adults, which is generally considered the main process by which browsers maintain open ecosystems (Ripple *et al.*, 2001; Côté *et al.*, 2004; Staver *et al.*, 2009), in addition to the removal of reproductive components from adults (Augustine & Decalesta, 2003). Camels select the freshest, most nutritious plant parts (Iqbal & Khan, 2001), and reduced seedling recruitment under high camel stocking rates has been reported from the United Arab Emirates (Gallacher & Hill, 2006a, 2006b). Cattle have also been reported to reduce seedling frequencies in Iran (Pour *et al.*, 2012). In Dhofar, it is likely seedlings and saplings are targeted by cattle once the grass and herb layer has senesced or been consumed, particularly as the foliage of adult woody plants is out of reach.

We observed altered age structure in the populations of woody species due to stocking rates. For example, we found that *A. dhofarica* trees are older (Figure 3.5) and total basal area of juveniles of all species is greater (Table 3.2) in areas under higher long-term stocking rates, most probably because larger plants are more resilient to browsing (Smit *et al.*, 2007). Furthermore, several species exhibited shorter heights at equivalent DRC measurements under higher stocking rates (Figure 3.6). Such stunted plant forms occur naturally in Dhofar and are usually associated with low moisture availability (Patzelt, 2015), such as the stunted *J. dhofarica* and *C. habessinica* communities on the plateau. However stunting due to severe browsing pressure was clearly observable in the field (Appendix 12) affected numerous species and occurred in areas with high fog densities. It has also been reported before in Dhofar (Oman Office of the Government Adviser for Conservation of the Environment, 1980; Miller, Morris & Stuart-Smith, 1988; Al-Mashaki & Koll, 2007) and elsewhere (Pour *et al.*, 2012; Hoppe-Speer & Adams, 2015; Box *et al.*, 2016).

*Anogeissus dhofarica* is the most abundant tree, and from an ecological and ethno-botanical perspective, the most important woody species in Dhofar. It was traditionally used as a building material and has a long history of use as a fuel wood and fodder plant (Miller, Morris & Stuart-Smith, 1988). This long history of use was apparent in our data. Ninety-seven percent of adult *A. dhofarica* trees in the study (n=534) had broken limbs and 57 percent had been subject to branch bending management practises. Branches are bent to bring the tree foliage in reach of livestock but its impact on tree survivability remains unknown and should be a priority for future research.

Adult *A. dhofarica* trees were over four times as abundant as juveniles indicating low advanced growth and subsequently poor forest regeneration. Only nine percent of *A. dhofarica* trees were dead indicating their resilience to heavy browsing pressure and the amount of standing dead adults of all species was surprisingly low compared to other studies (Angelstam, 1997: 30% - 40% of individuals compared to 10% in our study; Tritton and Siccama, 2014: 3% - 43% of basal area compared to 4% in our study). It is possible the number of standing dead trees is low because weak, dying and dead biomass is rapidly harvested for firewood and livestock bomas.

Bark stripping by livestock is commonly ranked by pastoralists as one of the greatest threats to the rangelands and the teeth of camels are often removed, yet we found only 10 percent of trees and large shrubs had an average 800 $cm^2$ area of stripped bark (Table 3.4). The cause of this behaviour in camels and cattle in Dhofar is unknown. Based on a review by Nicodemo and Porfírio-da-Silva (2018), it is most likely associated with diet and pasture quality, although behaviour and parasite control factors could be relevant. Camels have been found to selectively browse species and plant parts to balance the chemical composition of their diet (Amin, Abdoun & Abdelatif, 2007; Desalegn & Mohammed, 2012) or to obtain pharmaceutically active compounds (Villalba *et al.*, 2014). Hence, bark which contains polysaccharides, pectic substances, phenolic polymers and cross-linked polyesters (Feng *et al.*, 2013; Nicodemo & Porfírio-da-Silva, 2018), could potentially contribute beneficial nutrients and medicines to their diet. An assessment of the chemical composition of the most bark-stripped tree species in Dhofar (Table 3.4) and a review of the nutrient composition of livestock feedstuffs should be a priority for future research.

In summary, our results demonstrate that livestock in Dhofar have had substantial impacts on woody vegetation. As one would expect for an arid rangeland, moisture availability from the Khareef is the principle driver of woody plant community composition, however livestock browsing has affected the compositional and structural characteristics of these communities. Thus, Dhofar exhibits the widely cited properties of an equilibrium system because livestock browsing is the principal driver of woody vegetation change. Unlike in non-equilibrium rangelands, livestock populations are not density-dependent due to widespread feedstuff provisioning and the predictable annual monsoon supports vegetation communities in a relatively stable state (Ellis & Swift, 1988; Behnke, Scoones & Kerven, 1993; Sullivan & Rohde, 2002; Vetter, 2005). Moreover, past accounts suggest that livestock have been the principle driver of vegetation change for several decades (Lamprey, 1976; Lawton, 1978; Oman Office of the Government Adviser for Conservation of the Environment, 1980; Wilson & MacLeod, 1991; Ghazanfar, 1998).

## 3.6 Conclusion

We performed the first detailed analysis of the impact of camel, cattle and goat browsing on woody species of the drought-deciduous *Anogeissus* forests in Dhofar. Few studies have addressed the impacts of camel browsing on vegetation, particularly in an arid cloud forest rangeland at equilibrium. Thus we are contributing a novel study to the rangeland sciences. We found the Khareef fog strongly influences woody species composition and that browsing pressure has increased the frequency of unpalatable species, decreased plant density, reduced advanced growth, and led to stunting, altered population age structures and plant damage through management practises, bark stripping and browsing. Three key limitations of this study include the absence of data on the physical and chemical properties of soils, the differential impacts of cattle, camel and goat browsing, and forest-grassland interactions.

The functional integrity of vegetation communities is usually a good indicator of the health of wider biodiversity (Noss, 1990; Ferris & Humphrey, 1999), particularly in arid environments where botanical biomass is limited by constraining climatic conditions (Ludwig *et al.*, 2004). In Dhofar, woody plants play an even more vital role in supporting wider biodiversity, much of which is range-restricted, endemic and threatened, by irrigating the ecosystem through horizontal precipitation during the Khareef season (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006). We have provided empirical evidence of vegetation degradation due to overstocking in Dhofar and conclude that an intervention is urgently required to reduce browsing pressure and increase conservation efforts in the Dhofar Mountains.

## 3.7 Literature cited


Al-Kuthairi A. M. (1992) Forests and Pastoral Activities in Dhofar Mountains: Study of Strategy for their Rehabilitation. Arabian Gulf University, Bahrain.

Al-Mashaki M. S. A. & Koll E.-H. B. A. (2007) Country pasture/forage resource profiles: Oman. Rome: FAO.

Al-Rowaily S. L., El-Bana M. I., Al-Bakre D. A., Assaeed A. M., Hegazy A. K. & Ali M. B. (2015) Effects of open grazing and livestock exclusion on floristic composition and diversity in natural ecosystem of Western Saudi Arabia. Saudi Journal of Biological Sciences. King Saud University, 22, 430–437.

Amin A. S. A., Abdoun K. A. & Abdelatif A. M. (2007) Seasonal variation in botanical and chemical composition of plants selected by one-humped camel (Camelus dromedarius). Pakistan Journal of Biological Sciences, 10, 932–935.

Anderson D. & Grove R. H. (1989) Conservation in Africa: peoples, policies and practice. Cambridge: Cambridge University Press.

Andrew M. H. (1988) Grazing impact in relation to livestock watering points. Trends in Ecology and Evolution, 3, 336–339.

Angelstam P. (1997) Landscape Analysis as a Tool for the Scientific Management of Biodiversity. Ecological Bulletins, 46, 140–170.

Asner G. P., Elmore A. J., Olander L. P., Martin R. E. & Harris A. T. (2004) Grazing Systems, Ecosystem Responses, and Global Change. Annual Review of Environment and Resources, 29, 261–299.

Augustine D. J. & Decalesta D. (2003) Defining deer overabundance and threats to forest communities: From individual plants to landscape structure. Ecoscience, 10, 472–486.

Barnes B. V, Zak D. R., Denton S. R. & Spurr S. H. (1997) Forest Ecology. New York: John Wiley and Sons.

Bates D., Maechler M., Bolker B. & Walker S. (2015) Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software, 67, 1–48.

Beasom S. L. & Haucke H. H. (1975) A Comparison of Four Distance Sampling Techniques in South Texas Live Oak Mottes. Management, 28, 142–144.

Behnke R. H., Scoones I. & Kerven C. (1993) Range Ecology at Disequilibrium: New Models of Natural Variability and Pastoral Adaptation in African Savannas. London: Overseas Development Institute.

Borcard D., Legendre P. & Drapeau P. (1992) Partialling out the Spatial Component of Ecological Variation. Ecology, 73, 1045–1055.

Box J. B., Nano C. E. M., McBurnie G., Waller D. M., McConnell K., Brock C., Paltridge R., McGilvray A., Bubb A. & Edwards G. P. (2016) The impact of feral camels (Camelus dromedarius) on woody vegetation in arid Australia. The Rangeland Journal, 38, 181–190.

Breitenmoser U., Breitenmoser C., Mallon D. & Edmonds J.-A. (2010) Strategy for the Conservation of the Leopard in the Arabian Peninsula. Sharjah: Environment and Protected Areas Authority & SSC Cat Specialist Group.

Briske D. D. (2017) Rangeland Systems: Processes, Management and Challenges. Switzerland: Springer Nature.

Briske D. D., Fuhlendorf S. D. & Smeins F. E. (2003) Vegetation dynamics on rangelands: A critique of the current paradigms. Journal of Applied Ecology, 40, 601–614.

Burns W. C. (1995) The International Convention to Combat Desertification: Drawing a Line in the Sand? Michigan Journal of International Law, 16, 831–882.

Carson W. P., Banta J. a, Royo a a & Kirschbaum C. (2005) Plant communities growing on boulders in the Allegheny National Forest: Evidence for boulders as refugia from deer and as a bioassay of overbrowsing. Natural Areas Journal, 25, 10–18.

Casper B. B. & Jackson R. B. (1997) Plant Competition Underground. Annual Review of Ecology and Systematics, 28, 545–570.

Côté S. D., Rooney T. P., Tremblay J., Dussault C. & Waller D. M. (2004) Ecological Impacts of Deer Overabundance. Annual Review of Ecology, Evolution, and Systematics, 35, 113–147.

Cottam G. & Curtis J. T. (1956) The Use of Distance Measures in Phytosociological Sampling. Ecology, 37, 451–460.

Curry P. J. & Hacker R. B. (1990) Can pastoral grazing management satisfy endorsed conservation objectives in arid Western Australia? Journal of Environmental Management, 30, 295–320.

Curtis R. O. (1967) Height-Diameter and Height-Diameter-Age Equations for second-growth Douglas Fir. Forest Science. Society of American Foresters, 13, 365–375. (Accessed: 15 March 2018).

Dahdouh-Guebas F. & Koedam N. (2006) Empirical estimate of the reliability of the use of the Point-Centred Quarter Method (PCQM): Solutions to ambiguous field situations and description of the PCQM+ protocol. Forest Ecology and Management, 228, 1–18.

Dereje M. & Uden P. (2005) The browsing dromedary camel: I. Behaviour, plant preference and quality of forage selected. Animal Feed Science and Technology, 121, 297–308.

Desalegn T. & Mohammed Y. K. (2012) Preferably browsed forage species by camels (Camelus dromedarius) and their mineral contents in Jijiga district, eastern Ethiopia. Livestock Research for Rural Development, 24.

Dias A. S., dos Santos K., dos Santos F. A. M. & Martins F. R. (2017) How liana loads alter tree allometry in tropical forests. Plant Ecology, 218, 119–125.

Dickhoefer U., Buerkert A., Brinkmann K. & Schlecht E. (2010) The role of pasture management for sustainable livestock production in semi-arid subtropical mountain regions. Journal of Arid Environments. Elsevier Ltd, 74, 962–972.

Directorate-General of Nature Conservation (2010) Fourth National Report To the Convention on Biological Diversity. Muscat: Ministry of Environment and Climate Affairs.

Dubuis A., Pottier J., Rion V., Pellissier L., Theurillat J. P. & Guisan A. (2011) Predicting spatial patterns of plant species richness: A comparison of direct macroecological and species stacking modelling approaches. Diversity and Distributions, 17, 1122–1131.

El-Keblawy A., Ksiksi T. & El Alqamy H. (2009) Camel grazing affects species diversity and community structure in the deserts of the UAE. Journal of Arid Environments. Elsevier Ltd, 73, 347–354.

El-Mahi A. T. (2011) Old ways in a changing space: the issue of camel pastoralism in Dhofar. Sultan Qaboos University Journal - Agricultural and Marine Sciences, 16, 51–64.

Ellis E. C. & Ramankutty N. (2008) Putting people in the map: Anthropogenic biomes of the world. Frontiers in Ecology and the Environment, 6, 439–447.

Ellis J. E. & Swift D. M. (1988) Stability of African Pastoral Ecosystems: Alternate Paradigms and Implications for Development. Journal of Range Management, 41, 450.

ESCWA, UNEP & FAO (1993) The national plan of action to combat desertification in the Sultanate of Oman. Amman: ESCWA, Manama: UNEP, Rome: FAO.

ESRI (2016) ArcGIS Desktop: Release 10.5. Redlands, CA: Environmental Systems Research Institute.

Feng S., Cheng S., Yuan Z., Leitch M. & Xu C. (2013) Valorization of bark for chemicals and materials: A review. Renewable and Sustainable Energy Reviews, 26, 560–578.

Ferris R. & Humphrey J. W. (1999) A review of potential biodiversity indicators for application in British forests. Forestry, 72, 313–328.

Friedel M. H. (1991) Range Condition Assessment and the Concept of Thresholds: A Viewpoint. Journal of Range Management, 44, 422.

Friesen J., Zink M., Bawain A. & Müller T. (2018) Hydrometeorology of the Dhofar cloud forest and its implications for groundwater recharge. Journal of Hydrology: Regional Studies. Elsevier, 16, 54–66.

Gallacher D. & Hill J. (2006) Effects of camel vs oryx and gazelle grazing on the plant ecology of the Dubai Desert Conservation Reserve. Journal of arid environments, 66, 85–95.

Gallacher D. J. & Hill J. P. (2006) Effects of camel grazing on the ecology of small perennial plants in the Dubai (UAE) inland desert. Journal of Arid Environments, 66, 738–750.

Galletti C. (2015) Understanding Environmental Change and Biodiversity in a Dryland Ecosystem through Quantification of Climate Variability and Land Modification: The Case of the Dhofar Cloud Forest, Oman. Arizona State University.

Ghazanfar S. A. (1998) Status of the flora and plant conservation in the sultanate of Oman. Biological Conservation, 85, 287–295.

Ghazanfar S. A. & Fisher M. (1998) Vegetation of the Arabian Peninsula. Dordrecht: Kluwer Academic Publishers.

Grice A. C. & Hodgkinson K. C. (2002) Global Rangelands: Progress and Prospects. Wallingford: CABI Publishing.

GRM International (1982) Range and Livestock Survey. Brisbane: G.R.M. International Pty Ltd.

Hedges S. & Lawson D. (2006) Monitoring the Illegal Killing of Elephants: Dung Survey Standards for the Mike Programme. Nairobi: CITES MIKE Programme.

Herbel C. (1979) Utilization of Grass- and Shrublands of the South-Western United States. in Walker, B. (ed.) Management of Semi-Arid Ecosystems. New York: Elsevier, 161–203.

Al Hikmani H., Zabanoot S., Al Shahari T., Zabanoot N., Al Hikmani K. & Spalton A. (2015) Status of the Arabian Gazelle, Gazella arabica (Mammalia: Bovidae), in Dhofar, Oman. Zoology in the Middle East, 61, 295–299.

Hildebrandt A. & Eltahir E. A. B. (2006) Forest on the edge: Seasonal cloud forest in Oman creates its own ecological niche. Geophysical Research Letters, 33, 2–5.

Hildebrandt A. & Eltahir E. A. B. (2007) Ecohydrology of a seasonal cloud forest in Dhofar: 2. Role of clouds, soil type, and rooting depth in tree-grass competition. Water Resources Research, 43, 1–13.

Hildebrandt A. & Eltahir E. A. B. (2008) Using a horizontal precipitation model to investigate the role of turbulent cloud deposition in survival of a seasonal cloud forest in Dhofar. Journal of Geophysical Research, 113, 1–11.

Holechek J. L., Pieper R. D. & Herbel C. H. (2001) Range Management: Principles and Practices. Upper Saddle River: Prentice Hall.

Holland P. G. & Steyn D. G. (1975) Vegetational Responses to Latitudinal Variations in Slope Angle and Aspect. Journal of Biogeography, 2, 179–183.

Hoppe-Speer S. C. L. & Adams J. B. (2015) Cattle browsing impacts on stunted Avicennia marina mangrove trees. Aquatic Botany, 121, 9–15.

Howery L. D., Bailey D. W. & Laca E. A. (1999) Impact of spatial memory on habitat use. Grazing Behavior of Livestock and Wildlife, 70, 91–100.

HTSL (1978) Livestock Sector Review and Project Identification - Final Report, Volumes I and II. Hemel Hempstead: Hunting Technical Services Limited.

Huntley B. & Walker B. (1982) Ecology of Tropical Savannas. New York: Springer.

Iqbal A. & Khan B. B. (2001) Feeding Behaviour of Camel: A Review. Pakistan Journal of Agricultural Sciences, 38, 58–63.

Janzen J. (1990) Land Tenure and Land Use in the Pastoral Living Areas of Dhofar, Sultanate of Oman – Tradition, Change and Development Perspectives. Muscat: FAO & Land Tenure and Land Use Consultancy.

JICA (1990) Ten year Master Plan for Agricultural Development. Tokyo: Japan International Co-operation Agency.

Kidron G. J. (2005) Angle and aspect dependent dew and fog precipitation in the Negev desert. Journal of Hydrology, 301, 66–74.

Kürschner H., Hein P., Kilian N. & Hubaishan M. A. (2004) The Hybantho durae-Anogeissetum dhofaricae ass. nova - Phytosociology, structure and ecology of an endemic South Arabian forest community. Phytocoenologia, 34, 569–612.

Lamprey H. F. (1976) Preliminary Proposals for the Conservation and Management of the Jebel Qara Region. Nairobi: UNESCO Regional Office of Science and Technology for Africa.

Lamprey H. F. (1983) Pastoralism yesterday and today: The overgrazing problem. in Bourlière, F. (ed.) Tropical Savannas (Ecosystems of the World). Amsterdam: Elsevier, 643–666.

Lange R. T. (1983) Estimation of sheep stocking intensity at any location in arid zone paddocks. Transactions of the Royal Society of South Australia, 107, 137–137.

Lawton R. M. (1978) A reconnaissance survey of the Jabal Qara grazing land ecosystem, with particular reference to the impact of development. London: Overseas Development Agency, Land Resources Development Centre.

Ludwig J. A., Tongway D. J., Bastin G. N. & James C. D. (2004) Monitoring ecological indicators of rangeland functional integrity and their relation to biodiversity at local to regional scales. Austral Ecology, 29, 108–120.

Martorell C. & Ezcurra E. (2002) Rosette scrub occurrence and fog availability in arid mountains of Mexico. Journal of Vegetation Science, 13, 651–662.

Mehtätalo L., de-Miguel S. & Gregoire T. G. (2015) Modeling height-diameter curves for prediction. Canadian Journal of Forest Research, 45, 826–837.

Milchunas D. G. & Lauenroth W. K. (1993) Quantitative effects of grazing on vegetation and soils over a global range of environmnents. Ecological Monographs, 63, 327–366.

Miller A. G., Morris M. & Stuart-Smith S. (1988) Plants of Dhofar, The Southern Region of Oman: Traditional, Economic, and Medicinal Uses. Muscat: Office of the Adviser for Conservation of the Environment, Diwan of Royal Court.

Miller M. P., Singer M. J. & Nielsen D. R. (1988) Spatial variability of wheat yield and soil properties on complex hills. Soil Science Society of America Journal, 52, 1133–1141.

Ministry of Petroleum and Minerals (1986) Geological map, Oman 1:100,000. London: Colourmap Scanning Ltd.

Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD (2005) National Action Programme to Combat Desertification in the Sultanate of Oman. Muscat: Ministry of Regional Municipalities, Environment, and Water Resources.

Moritz M. (2017) Misreading a pastoral property regime in the logone floodplain, Cameroon. Ecology and Society, 22, 13.

Mott MacDonald International Ltd. (1991) National Resources Master Plan. Cambridge.

Mueggler W. F. (1965) Cattle Distribution on Steep Slopes. Journal of Range Management, 18, 255.

Mwendera E. J., Mohamed Saleem M. A. & Dibabe A. (1997) The effect of livestock grazing on surface runoff and soil erosion from sloping pasture lands in the Ethiopian highlands. Australian Journal of Experimental Agriculture, 37, 421–430.

Nearing M. A. (1997) A Single, Continuous Function for Slope Steepness Influence on Soil Loss. Soil Science Society of America Journal, 61, 917.

Nicodemo M. L. F. & Porfírio-da-Silva V. (2018) Bark stripping by cattle in silvopastoral systems. Agroforestry Systems, 1–11.

Noss R. (1990) Indicators for monitoring biodiversity: A hierarchical approach. Conservation Biology, 4, 355–364.

Oksanen A. J., Blanchet F. G., Kindt R., Legendre P., Minchin P. R., Hara R. B. O., Simpson G. L., Solymos P., Stevens M. H. H. & Wagner H. (2018) Package ‘vegan’ (Version 2.6). R Package Vignettes, 1–292.

Oman National Centre for Statistics and Information (2017a) 2017 Statistical Year Book. Muscat: National Centre for Statistics and Information.

Oman National Centre for Statistics and Information (2017b) National Centre for Statistics and Information Data Portal. (Accessed: 22 August 2017).

Oman Office of the Government Adviser for Conservation of the Environment (1980) The scientific results of the Oman flora and fauna survey 1977 (Dhofar), Journal of Oman Studies Special Report No. 2. Muscat: Ministry of Information and Culture.

Paliy O. & Shankar V. (2016) Application of multivariate statistical techniques in microbial ecology. Molecular Ecology, 25, 1032–1057.

Patzelt A. (2011) The Themeda Quadrivalvis Tall-Grass Savannah of Oman at the Crossroad Between Africa and Asia. Edinburgh Journal of Botany, 68, 301–319.

Patzelt A. (2012) Notes on the Flora and Vegetation of Wadi Kharfut on Jabal Qamar, Oman. in Borrell, J. et al. (eds) The British Exploring Society Empty Quarter Expedition 2012. London: British Exploring Society, 94–100.

Patzelt A. (2015) Synopsis of the Flora and Vegetation of Oman, with Special Emphasis on Patterns of Plant Endemism. Abhandlungen der Braunschweigischen Wissenschaftlichen Gesellschaft, 282–317.

Peacock J. ., Ferguson M. ., Alhadrami G. A., McCann I. R., Al Hajoj A., Saleh A. & Karnik R. (2003) Conservation through utilization: a case study of the indigenous forage grasses of the Arabian Peninsula. Journal of Arid Environments, 54, 15–28.

Pereira I. M., Rios J. M., do Vale V. S., Gomes-Klein V. L., Coneglian A. & Rocha E. C. (2018) Forest Structure and the Species Composition of the Parque Estadual Mata Atlântica, Located in Goiás State, Brazil. International Journal of Ecology, Article ID 1219374.

Pour M. J., Mohadjer M. R. M., Etemad V. & Zobeiri M. (2012) Effects of grazing on natural regeneration of tree and herb species of Kheyroud forest in northern Iran. Journal of Forestry Research, 23, 299–304.

Putman R. J. (1984) Facts from faeces. Mammal Review, 14, 79–97.

R Core team (2013) R: A Language and Environment for Statistical Computing. Vienna.

Reynolds J. F., Stafford Smith D. M., Lambin E. F., Turner B. L., Mortimore M., Batterbury S. P. J., Downing T. E., Dowlatabadi H., Fernández R. J., Herrick J. E., et al. (2007) Ecology: Global desertification: Building a science for dryland development. Science, 316, 847–851.

Ripple W. J., Larsen E. J., Renkin R. A. & Smith D. W. (2001) Trophic cascades among wolves, elk and aspen on Yellowstone National Park’s northern range. Biological Conservation, 102, 227–234.

Sankaran M., Hanan N. P., Scholes R. J., Ratnam J., Augustine D. J., Cade B. S., Gignoux J., Higgins S. I., Roux X. Le, Ludwig F., et al. (2005) Determinants of woody encroachment and cover in African savannas. Nature, 438, 846–849.

Sayre N. F., Debuys W., Bestelmeyer B. T. & Havstad K. M. (2012) ‘The range problem’ after a century of rangeland science: New research themes for altered landscapes. Rangeland Ecology and Management, 65, 545–552.

Sayre N. F. (2017) The Politics of Scale: A History of Rangeland Science. Chicago: University of Chicago Press.

Scholes R. J. & Archer S. R. (1997) Tree-Grass Interactions in Savannas. Annual Review of Ecology and Systematics, 28, 517–544.

Scholte P. & De Geest P. (2010) The climate of Socotra Island (Yemen): A first-time assessment of the timing of the monsoon wind reversal and its influence on precipitation and vegetation patterns. Journal of Arid Environments, 74, 1507–1515.

Seymour C. L., Milton S. J., Joseph G. S., Dean W. R. J., Ditlhobolo T. & Cumming G. S. (2010) Twenty years of rest returns grazing potential, but not palatable plant diversity, to Karoo rangeland, South Africa. Journal of Applied Ecology, 47, 859–867.

Shreve F. (1922) Conditions indirectly affecting vertical distribution on desert mountains. Ecology, 3, 269–274.

Smit C., Vandenberghe C., Den Ouden J. & Müller-Schärer H. (2007) Nurse plants, tree saplings and grazing pressure: Changes in facilitation along a biotic environmental gradient. Oecologia, 152, 265–273.

Spalton A., Al Hikmani H. M., Willis D. & Said A. S. B. (2006) Critically Endangered Arabian leopards Panthera pardus nimr persist in the Jabal Samhan Nature Reserve, Oman. Oryx, 40, 287–294.

Spalton A. & Al Hikmani H. (2014) The Arabian Leopards of Oman. 1st edn. London: Stacey International.

Stafford Smith M. (1996) Management of Rangelands: Paradigms at their Limits. in Hodgson, J. and Illius, A. (eds) The Ecology and Management of Grazing Systems. Wallingford: CAB International, 325–357.

Stage A. R. (1976) An expression of the effects of aspect, slope, and habitat type on tree growth. Forest Science, 22, 457–460.

Staver A. C., Bond W. J., Stock W. D., Rensburg S. J. Van & Waldram M. S. (2009) Browsing and fire interact to suppress tree density in an African savanna. Ecological Applications, 19, 1909–1919.

Sullivan S. & Rohde R. (2002) On non-eqilibrium in arid and semi-arid grazing systems. Journal of Biogeography, 29, 1595–1618.

Tardelli M. & Raffaelli M. (2006) Some aspects of the vegetation of Dhofar (Southern Oman). Bocconea, 19, 109–112.

Thalen D. C. P. (1979) Ecology and utilization of desert shrub rangelands in Iraq. Dordrecht: Springer.

Tritton L. M. & Siccama T. G. (1990) What Proportion of Standing Trees in Forests of the Northeast are Dead? Bulletin of the Torrey Botanical Club, 117, 163–166.

Turner M. D. & Hiernaux P. (2002) The use of herders' accounts to map livestock activities across agropastoral landscapes in Semi-Arid Africa. Landscape Ecology, 17, 367–385.

Tzanopoulos J., Mitchley J. & Pantis J. (2005) Modelling the effects of human activity on the vegetation of a northeast Mediterranean island. Applied Vegetation Science, 8, 27–38.

Vetter S. (2005) Rangelands at equilibrium and non-equilibrium: Recent developments in the debate. Journal of Arid Environments, 62, 321–341.

Villalba J. J., Miller J., Ungar E. D., Landau S. Y. & Glendinning J. (2014) Ruminant self-medication against gastrointestinal nematodes: evidence, mechanism, and origins. Parasite, 21, 31.

Volis S., Dorman M., Blecher M., Sapir Y. & Burdeniy L. (2011) Variation partitioning in canonical ordination reveals no effect of soil but an effect of co-occurring species on translocation success in Iris atrofusca. Journal of Applied Ecology, 48, 265–273.

Welch R. M. & Wielicki B. A. (1986) The stratocumulus nature of fog. Journal of Climate and Applied Meteorology, 25, 101–111.

Whitehead E. E., Hutchinson C. F., Timmermann T. M. & Varady R. G. (1988) Arid Lands: Today and Tomorrow. Colorado: Westview Press.

Wilson A. & MacLeod N. (1991) Overgrazing: Present or absent? Journal of Range Management, 44, 475–482.

# 4 Six new variants of the *Hybantho durae-Anogeissetum dhofaricae* ass. in Jabal Qamar, Dhofar, Oman

## 4.1 Abstract

A phytosociological study of the *Anogeissus* forest in Jabal Qamar, Dhofar is carried out. Six new habitat variants are identified and described using hierarchical cluster analysis and indicator species analysis. These are the *Dodonaea viscosa* subsp. *angustifolia* shrubland variant, *Cadia purpurea-Olea europaea* forest variant, *Euclea racemosa-Jasminum grandiflorum* shrubland variant, *Maytenus dhofarensis-Ficus sycomorus* woodland variant, *Jatropha dhofarica-Zygocarpum dhofarense* sparse woodland variant and the *Premna resinosa-Hybanthus durus* forest variant. A seventh, the broad-leaved *Blepharispermum hirtum* variant, was previously described by Kürschner *et al.* (2004). Associated topoclimatic factors, vegetation characteristics and disturbance factors are discussed. The interplay between the complex topography and the monsoon fog as well as stocking rates are identified as key variables affecting vegetation community composition. A review of the literature suggests that the *Dodonaea viscosa* subs. *angustifolia* shrubland variant persists as a result of historical agricultural practises whilst the *Maytenus dhofarensis-Ficus sycomorus* sparse woodland persists as a result of historical deforestation.

## 4.2 Introduction

The *Hybantho durae-Anogeissetum dhofaricae* (Kürschner *et al.*, 2004), a drought deciduous cloud forest community, is endemic to the south-facing escarpments of a limestone mountain chain on the central south coast of the Arabian Peninsula, in the governorate of Al-Mahra in Yemen and the governorate of Dhofar in Oman. It is the remnants of a once continuous tropical flora which spanned Africa and Arabia, the remainder of which disappeared with Arabia's increasing aridity in the late Tertiary (Kürschner *et al.*, 2004; Meister *et al.*, 2005).

The *Anogeissus* forest is supported by a mean annual precipitation of 250mm, whilst the surrounding semi-deserts and deserts receive less than 100mm. Most of the precipitation is received in the monsoon season, known locally as the *Khareef*. From mid-June to mid-September south-western winds cause an upwelling of cold deep sea water off the coast, lowering the sea temperature to c. 18 degrees. The warmer moist winds blowing over it are subsequently cooled to dew point and a bank of dense fog forms against the south-facing mountain escarpments (Whitehead *et al.*, 1988; Ghazanfar & Fisher, 1998). The Khareef fog, which is denser a few meters above than close to the ground (Price, Al-Harthy and Whitcombe, 1988; 34-35 l/m2 per day at 4.2 m, 13 l/m2 at 0.9 m height), is captured by tree canopies, through a process known as horizontal precipitation (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2007, 2008; Friesen *et al.*, 2018). Net precipitation which reaches the ground is estimated to be three times as high as rainfall (Hildebrandt & Eltahir, 2006).

An altitudinal gradient of vegetation communities has long been recognised in Dhofar (Radcliffe-Smith, 1980; Raffaelli & Tardelli, 2006), and these communities have been the subject of botanical inventories for several decades (Radcliffe-Smith, 1980; Miller, Morris & Stuart-Smith, 1988; Miller & Cope, 1996; Ghazanfar & Fisher, 1998; Mosti, Raffaelli & Tardelli, 2012). Many of these communities have been described qualitatively (Al-Zidjali, 1995; Kilian, Hein & Hubaishan, 2002; Raffaelli & Tardelli, 2006; Knees *et al.*, 2007; Patzelt, 2015) whilst others, such as the coastal vegetation (Ghazanfar, 1996, 1999) and a mid-altitude plateau grassland (Patzelt, 2011), have been subject to quantitative phytosociological sampling.

Inventorial research has recorded 262 vascular plant species (19% of the total flora of Oman) from the *Anogeissus* forest (Patzelt, 2015). Miller *et al.* (1988) generated a valuable reference material and ethnobotanical knowledge for many woody species, yet a detailed phytosociological study was not completed until more recently by Kürschner *et al.* (2004). This description of the association used data from 38 releves in the Hawf and Fartak mountains in Yemen (Kürschner *et al.*, 2004), with supplementary data from two research trips to Dhofar in October 1998 by P. Hein & N. Kilian and in December 2002 by N. Kilian, and the results of Radcliffe-Smith (1980).

This paper aims to compliment those results through a phytosociological study of the *Anogeissus* forest in Jabal Qamar in Dhofar. We discuss associated topoclimatic factors, vegetation characteristics and disturbance factors, with a focus on livestock activity, which has posed a threat to the vegetation of Dhofar for several decades (Lamprey, 1976; Lawton, 1978; Wilson & MacLeod, 1991; Ghazanfar, 1998; Peacock *et al.*, 2003; Kürschner *et al.*, 2004; Tardelli & Raffaelli, 2006; El-Mahi, 2011b).

## 4.3 Methods

### 4.3.1 Study area

Jabal Qamar is the westernmost of the three mountain ranges in Dhofar (Figure 4.1). It experiences a higher precipitation than the other two mountain ranges and boasts the highest botanical diversity (515 vascular species) of any area in Oman (Patzelt, 2015). The drought-deciduous *Anogeissus* forest (Kürschner *et al.*, 2004) is dominant on the south-facing escarpments, with sparse woodland and grasslands in flatter areas. The main geologic formation in Jabal Qamar is limestone of tertiary origin. Layers of the Hadramout group are present. These are, from bottom to top, the Umm Er Radhuma (UER), the Rus (RUS) and the Dammam (DAM) formations (Friesen *et al.*, 2018).

There are seventy-five permanently inhabited and ten seasonally (Khareef) inhabited villages in Jabal Qamar with a total human population of 7,799 (Oman National Centre for Statistics and Information, 2017a). Livestock-owning households are present in all villages. The 2015 national livestock census recorded 15,164 dromedary camels in 802 holdings, 27,522 head of cattle in 1,060 holdings, and 14,217 goats in 439 holdings (Oman National Centre for Statistics and Information, 2017b).

### 4.3.2 Field sampling

Data collection took place between September 2016 and May 2017. The point-centered quarter (PCQ) method (Cottam and Curtis, 1956) was used to sample the composition, density and structure of woody vegetation at thirty sites (Figure 4.1). In this method, density estimates are derived from distance measures between points and the closest plants, which are subsequently studied to estimate composition and structure. It is a plotless method making it more efficient than standard plot-based techniques (Cottam & Curtis, 1956; Beasom & Haucke, 1975). Although the PCQ method was initially designed for forestry studies it has been widely applied to more natural systems (Dickhoefer *et al.*, 2010; Dias *et al.*, 2017; Pereira *et al.*, 2018). Dahdouh-Guebas and Koedam (2006) addressed several ambiguous field situations which we incorporated in our study.

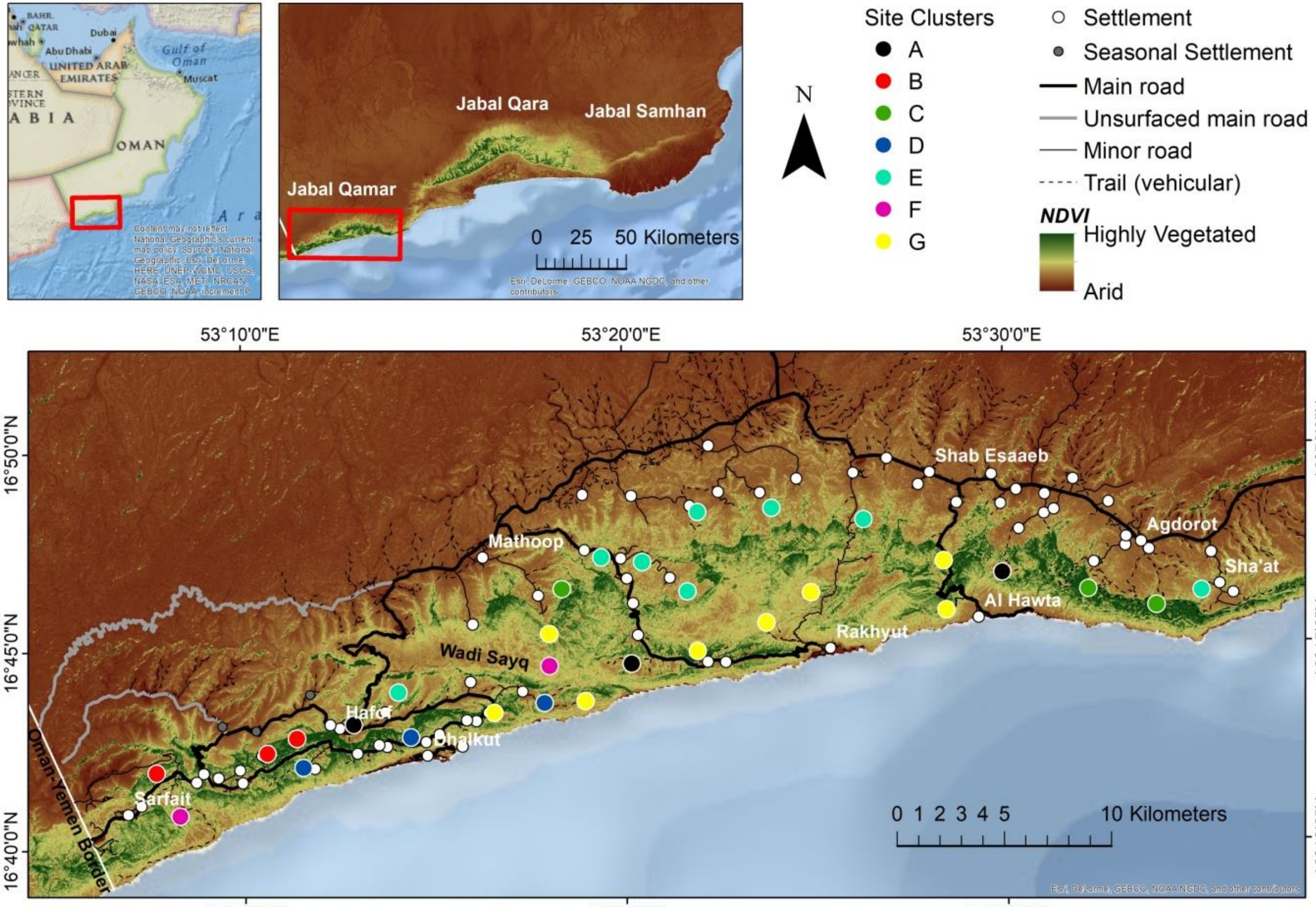



**Figure 4.1. Map showing the location of Jabal Qamar in the Dhofar Mountains and the locations of thirty sampling sites. The coloured site markers represent the seven habitat variants identified using hierarchical cluster analysis.**

Sites were visited on four occasions. Ten points were carried out during the first visit, ten during the second visit, five during the third visit and five during the fourth visit, resulting in a total of 30 points at each site. Consecutively rotating site visits controlled for intra-seasonal vegetation change such as senescence and livestock grazing and browsing to improve comparability between sites, and allowed for minor adjustments to be made to the methodology between each round of visits. Points were randomly dispersed over an area of approximately 1 $km^2$ as camel browsing can be dispersed while cattle grazing can be patchy (Dereje & Uden, 2005), and thus small sampling sites may not have provided a representative average of the effects of livestock disturbance and may have overemphasised vegetation responses (Briske, Fuhlendorf & Smeins, 2003). At each sample point, the distances to the closest adult and the closest juvenile woody plant were recorded in each of four quarters. At each PCQ point a 1.2 m x 1.2 m quadrat was deployed to sample the ground vegetation which also served to guide the PCQ quarters. Grass and herb species richness and the percentage cover of grasses, herbs, rock and bare ground were recorded. Additionally, the percentage cover of the three most abundant species was estimated.

To summarise, we sampled 30 sites, each with 30 points, with four adults and four juveniles measured per point, resulting in a total of 120 adult and 120 juvenile records per site, resulting in a total of 3600 adult and 3600 juvenile woody plant records.

### 4.3.3 Classification

Hierarchical cluster analysis (HCA) using Bray-Curtis dissimilarly indices and average linkage clustering was carried out on the woody (combined juvenile and mature) species count data in R studio (R Core team, 2013). Unlike in non-nested clustering algorithms such as K-means clustering, the number of clusters of interest is not pre-specified in HCA. Rather, a single nested hierarchy of clusters is produced and visualised in a dendrogram enabling comparison among clusters. Selection of clusters representing biologically important subpopulations is left to the researcher which in some studies can be problematic, although various approaches exist for assessing the statistical significance of clustering. In this study, no such issues were faced as the hierarchy of clusters was partitioned at seven groups ($K$ = 7) on the basis that the seventh group conformed to the pre-described *Blepharispermum hirtum* variant of the *Anogeissus* forest (Kürschner *et al.*, 2004). TWINSPAN (Hill, 1979) is another widely used classification technique, although it has received much criticism (Belbin &

McDonald, 1993; McCune, Urban & Grace, 2002; Dufrene & Legendre, 2013). We discounted this method as it assumes a single strong gradient dominating the data structure, while we suspected multiple gradients in Dhofar (e.g. fog and stocking rates).

The Bray-Curtis dissimilarity measure was used as it is appropriate for raw count data including zeros (Legendre & Legendre, 1998; Greenacre & Primicerio, 2013) and is particularly suited to uniform sample sizes which resulted from our PCQ sampling method (Chao *et al.*, 2006). Average linkage clustering was used as it is a natural compromise between single and complete linkage clustering and provides a more accurate reflection of the distance between clusters as it incorporates information about variance (Yim & Ramdeen, 2015). Data on the herbaceous species was not included in the cluster analysis as we recorded only the three most abundant species and due to the short growing season and rapid senescence of the herbaceous layer it does not 'characterise' habitats per se (Appendix 13).

The clusters were visualised in a CCA biplot of woody species composition (combined juvenile and mature) to observe the relationship between the proposed variants and three important environmental constraints. The CCA models were conducted in R studio (R Core team, 2013) using the Vegan package (Oksanen *et al.*, 2018) and built following a manual stepwise procedure, where the variables were gradually added based on their suspected influence, founded on our ecological understanding. Permutation tests for the joint and separate effects of constraining variables, as well as for marginal (Type III) effects, were performed to test the significance of each constraint. One-way ANOVA tests and Tukey HSD post-hoc tests were used to compare topoclimatic parameters, vegetation characteristics and disturbance factors between variants.

### 4.3.4 Indicator species analysis

Diagnostic species of the proposed variants were determined by calculating group-equalized individual-based correlation indices (r.ind.g) for each species within each cluster (De Cáceres & Legendre, 2009). This index was preferred over others by the same authors for two reasons. Firstly, it is suited to our sampling design where an equal number of individuals were sampled at each site (De Cáceres & Legendre, 2009). Secondly, it equalizes the relative sizes of all clusters allowing for comparisons between values corresponding to clusters of different sizes (Tichý & Milan, 2006; De Cáceres & Legendre, 2009). This method reduces the candidacy of rare species. The same method was used for herbaceous species, however due to unequal cover totals we transformed the data set by dividing each abundance value by the sum of the abundances at each site (De Cáceres & Legendre, 2009). The transgressive character species included in the variant descriptions were selected based on a simple average abundance threshold of > 5% within variants. Classification of the variants as shrubland, woodland or forests follows the classification scheme by Ellenberg and Mueller Dombois (1967).

The significance of species-variant memberships for diagnostic species was tested using permutation tests under the null hypothesis that the abundance of a given species in sites belonging to the variant was not higher than its abundance in sites not in the variant (Bakker, 2008). Diagnostic species were assigned to the variant for which they had the lowest significant *p*-value after Sidak's correction for multiple testing. The analysis was conducted on woody plants and herbs separately, under the same site groupings, using the indicspecies package (De Cáceres & Jansen, 2016) in R studio (R Core team, 2013).

## 4.4 Results

We recorded 47 woody species (42 adult and 43 juvenile) (Appendix 9, Appendix 10) and 110 herbaceous species across the thirty sites (Appendix 14, Appendix 15). The colours of the clusters in the hierarchical cluster analysis (HCA) dendrogram (Figure 4.2) and the CCA biplot of clusters (Figure 4.3) match the coloured site map markers in Figure 4.1. Based on our field observations, the resultant clusters appear to accurately distinguish between the key habitats in the monsoon-influenced zone of Jabal Qamar.

The CCA found fog density, long-term stocking rate and elevation range explained 12%, 9% and 8.3% of the inertia in woody species composition, respectively. This is the dependently explained inertia after accounting for the effects of the other variables. In the biplot of clusters (Figure 4.3) one can observe three entirely distinct 'meta-clusters' of sites spread around the ordination space. Firstly, the high altitude plateau woodlands of cluster E (top left of the ordination space), under high stocking rates (most likely due to proximity to settlements) and on the edge of the monsoon influenced zone. Second, clusters F and G (bottom-left of the ordination space) at low altitudes, with average fog densities and low stocking rates. And finally, clusters A, B, C and D (right side of the ordination space) at higher altitudes, associated with above-average fog densities but varying stocking rates. Within this last 'meta-cluster' we can distinguish clusters B and C as being more closely associated with high fog densities, and cluster A more closely linked to high stocking rates.

Beyond this level we assume that the distribution of the clusters in the ordination space is attributable to other variables not included in the CCA. All the measured environmental parameters, along with the results of ANOVA tests for statistical differences between variants, are shown in Table 4.1. In the following section we describe the variants using the synoptic table (Table 4.2) and include data on other variables from Table 4.1.

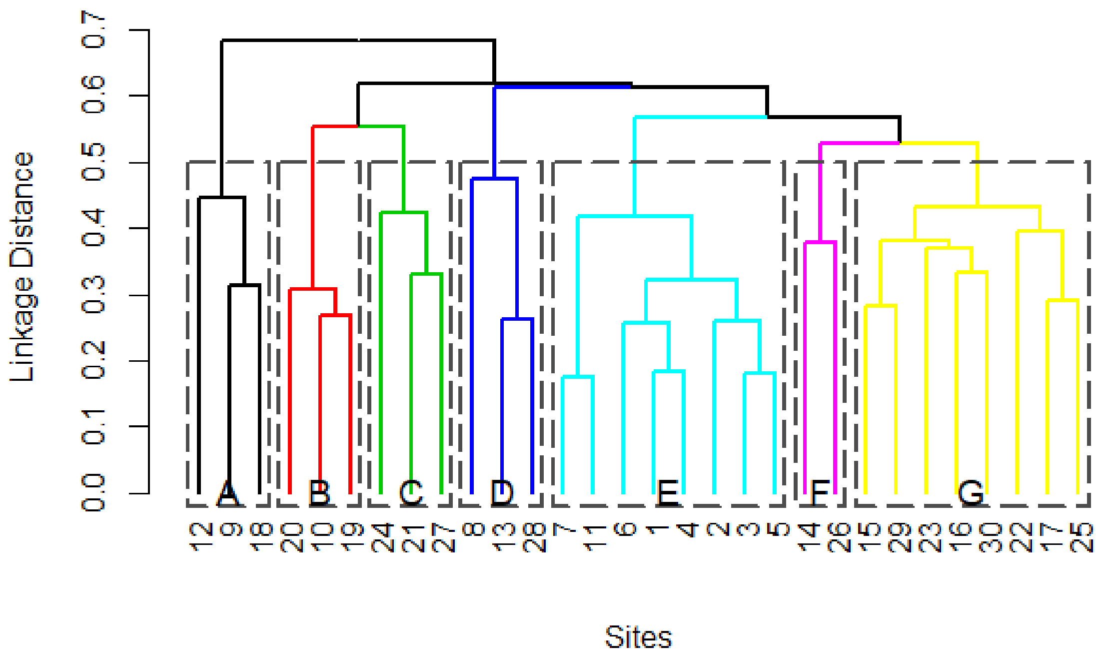



**Figure 4.2. Dendrogram of seven clusters of thirty sites using Bray-Curtis dissimilarity indices and average linkage clustering.**

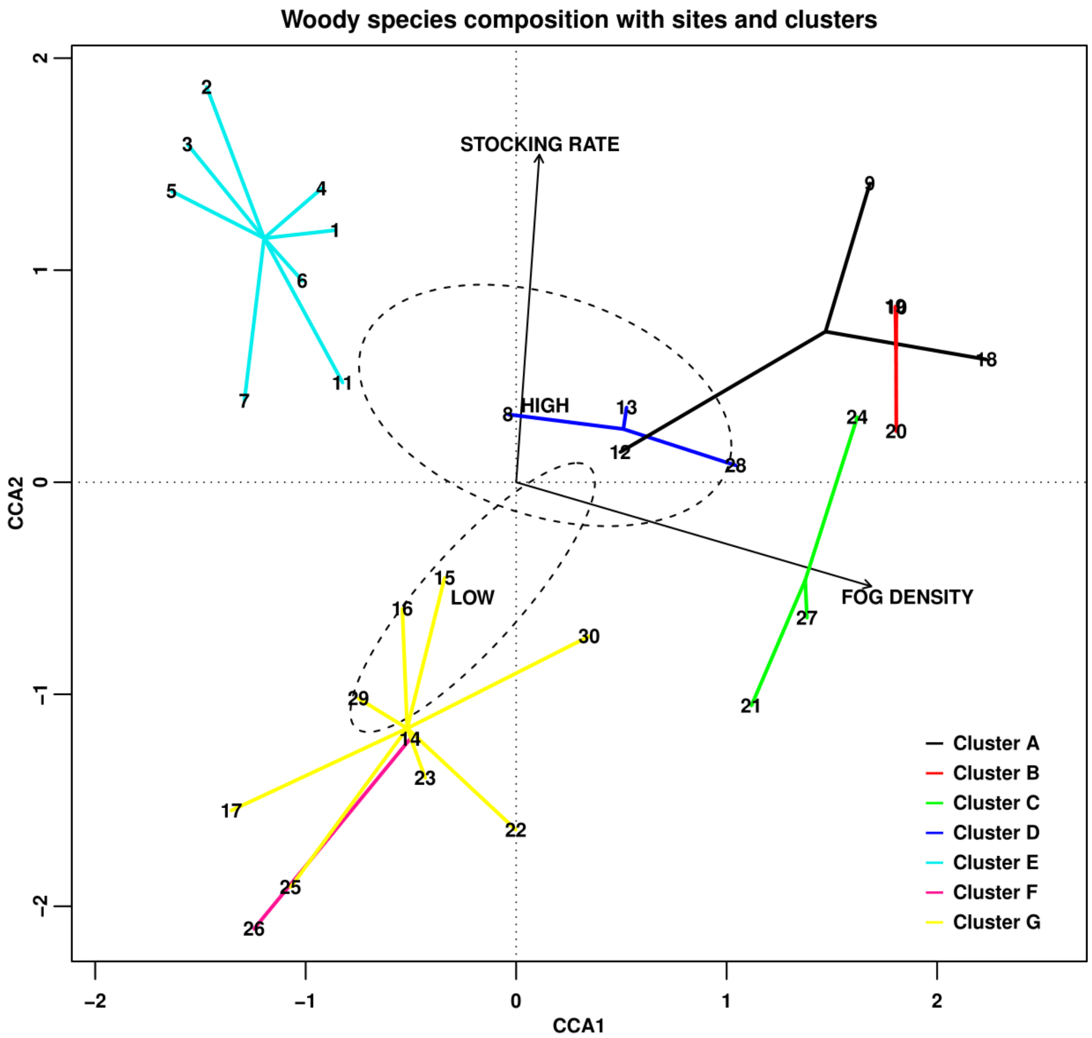



**Figure 4.3. Constrained correspondence analysis (CCA) biplot of site scores with clusters shown as spider plots, where colours match with those in the dendrogram (Figure 4.2) and the map (Figure 4.1). Fog density and stocking rate are continuous constraining variables shown as arrows, and elevation range is a categorical constraining variable shown as centroids of low and high elevation ranges (300–500 m a.s.l. and 700–900 m a.s.l.). The length of the arrow indicates the strength of the variable and the ellipse shows the standard error (0.999) of the weighted average of scores and the weighted correlation defines the direction of the principal axis of the ellipse.**

**Table 4.1. Mean values (Mean) and standard deviation (SD) for measured environmental parameters. Several topclimatic parameters were measured from 200 points per site to increase site-level precision. Other parameters were measured at the individual (e.g. adult height), point (e.g. adult density) or a different level (e.g. soil pH) based on the sampling procedure. Differences between variants were tested with one-way ANOVA's. Letters signify the results of Tukey HSD post-hoc tests and indicate pairwise non-significant results. The last column shows overall significance: < 0.001 '***'; < 0.01 '**'; < 0.05 '*'.**

| **Parameters** | ***A: Dodonaea viscosa* subsp. *angustifolia* shrubland variant** | | | | ***B: Cadia purpurea-Olea europaea* forest variant** | | | | **C: *Euclea racemosa-Jasminum grandiflorum* shrubland variant** | | | | **D: *Maytenus dhofarensis-Ficus sycomorus* woodland variant** | | | | **E: *Jatropha dhofarica-Zygocarpum dhofarense* sparse woodland variant** | | | | **F: Broad-leaved *Blepharispermum hirtum* variant (Kürschner et al., 2004)** | | | | **G: *Premna resinosa-Hybanthus durus* forest variant** | | | | **P** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Topoclimatic factors** | N | Mean | SD | No signf. | N | Mean | SD | No signf. | N | Mean | SD | No signf. | N | Mean | SD | No signf. | N | Mean | SD | No signf. | N | Mean | SD | No signf. | N | Mean | SD | No signf. | |
| Elevation (m) (n/site = 200) | 600 | 636 | 137 | | 600 | 750 | 57 | C | 573 | 764 | 47 | B | 600 | 367 | 59 | | 1600 | 790 | 52 | F | 400 | 357 | 41 | D | 1465 | 423 | 152 | | *** |
| Aspect (°) (n/site = 200) | 600 | 196 | 86 | C | 600 | 144 | 41 | DF | 573 | 187 | 67 | A | 600 | 143 | 72 | BF | 1600 | 170 | 85 | | 400 | 149 | 56 | BD | 1465 | 227 | 93 | | *** |
| Slope (°) (n/site = 200) | 600 | 10.06 | 3.78 | E | 600 | 13.27 | 7.74 | DF | 573 | 22.16 | 9.56 | | 600 | 12.37 | 7.04 | BF | 1600 | 9.20 | 4.47 | A | 400 | 12.18 | 5.27 | BD | 1465 | 20.34 | 6.88 | | *** |
| Fog density (arb. 0-1) (n/site = 200) | 600 | 0.55 | 0.03 | | 600 | 0.57 | 0.05 | DC | 573 | 0.57 | 0.06 | DB | 600 | 0.57 | 0.06 | CB | 1600 | 0.42 | 0.05 | | 400 | 0.52 | 0.04 | | 1465 | 0.49 | 0.04 | | *** |
| Solar radiation (kWH/m2) (n/site = 30) | 90 | 1776 | 40 | BC | 90 | 1790 | 35 | AE | 90 | 1750 | 77 | ADF | 90 | 1717 | 53 | CF | 240 | 1795 | 54 | B | 60 | 1732 | 35 | CD | 240 | 1633 | 135 | | *** |
| Heat Load Index (McCune and Keon) (n/site = 200) | 600 | 1.07 | 0.02 | DF | 600 | 1.08 | 0.03 | | 573 | 1.10 | 0.04 | | 600 | 1.07 | 0.03 | AEF | 1600 | 1.07 | 0.02 | D | 400 | 1.07 | 0.03 | AD | 1465 | 1.09 | 0.04 | | *** |
| Terrain Roughness (n/site = 200) | 600 | 20.03 | 14.16 | EF | 600 | 43.79 | 47.39 | D | 573 | 129.11 | 123.51 | | 600 | 35.52 | 40.84 | BF | 1600 | 18.32 | 26.87 | A | 400 | 28.97 | 22.71 | AD | 1465 | 88.04 | 55.16 | | *** |
| Soil pH (n/site = 4) | 12 | 8.06 | 0.10 | BCDE | 12 | 7.99 | 0.08 | ACDE | 12 | 7.90 | 0.20 | ABE | 12 | 8.12 | 0.09 | ABEFG | 32 | 8.00 | 0.17 | ABCD | 8 | 8.35 | 0.09 | DG | 32 | 8.25 | 0.09 | DF | *** |
| Rock cover (%) (n/site = 30) | 90 | 7.5 | 15.0 | BDFG | 90 | 13.6 | 17.8 | ACF | 90 | 17.1 | 20.1 | BE | 90 | 5.0 | 12.7 | AFG | 240 | 20.2 | 19.5 | C | 58 | 6.4 | 15.7 | ABDG | 240 | 6.4 | 11.6 | ADF | *** |
| **Vegetation characteristics** | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Proportion of dead adults (%) (n/site = 1) | 3 | 12.50 | 3.82 | | 3 | 16.94 | 1.27 | | 3 | 9.16 | 5.77 | | 3 | 10.28 | 3.36 | | 8 | 10.21 | 3.69 | | 2 | 5.42 | 4.12 | | 8 | 8.33 | 5.23 | | . |
| Adult density (plants/hectare) (n/site = 30) | 90 | 1214 | 995 | BCEF | 90 | 1107 | 617 | ACEF | 90 | 1485 | 1032 | ABG | 90 | 322 | 440 | | 240 | 956 | 941 | ABF | 60 | 1002 | 883 | ABE | 240 | 1706 | 1213 | C | *** |
| Juvenile density (plants/hectare) (n/site = 30) | 90 | 2059 | 2282 | BCDEF | 90 | 5527 | 6975 | ACDEF | 90 | 5610 | 6173 | ABDEF | 90 | 1271 | 1897 | ABCEF | 240 | 4968 | 6981 | ABCDF | 60 | 2510 | 2345 | ABCDE | 240 | 11231 | 19983 | | *** |
| Adult basal area (cm2) (n/site = 120) | 360 | 105.9 | 380.6 | CEG | 360 | 370.4 | 838.1 | CDFG | 360 | 197.9 | 444.9 | ABEFG | 360 | 560.1 | 1724.2 | BF | 960 | 178.4 | 338.0 | ACG | 240 | 224.9 | 427.1 | BCDG | 960 | 169.5 | 317.0 | ABCEF | *** |
| Adult height (cm) (n/site = 120) | 360 | 235.7 | 126.8 | | 360 | 464.9 | 294.9 | | 360 | 374.3 | 214.3 | DFG | 360 | 354.8 | 300.1 | CF | 960 | 304.7 | 228.2 | | 240 | 359.3 | 215.4 | CD | 960 | 414.1 | 220.0 | C | *** |
| Juvenile basal area (cm2) | 360 | 5.6 | 9.6 | E | 360 | 3.6 | 6.8 | DF | 360 | 2.1 | 4.1 | G | 360 | 3.2 | 5.1 | BF | 960 | 4.7 | 9.1 | A | 240 | 4.9 | 8.4 | BD | 960 | 2.4 | 5.2 | C | *** |

| | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| (n/site = 120) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Juvenile height (cm) (n/site = 120) | 360 | 85.4 | 60.4 | BCF | 360 | 77.4 | 75.9 | ACFG | 360 | 73.5 | 82.4 | ABFG | 360 | 46.1 | 42.8 | E | 960 | 57.0 | 43.0 | D | 240 | 80.3 | 74.6 | ABCG | 960 | 70.4 | 76.1 | BCF | *** |
| **Disturbance or livestock factors** | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Long-term stocking rate (arb. scale 1-30) (n/site = 1) | 3 | 19.33 | 7.09 | BCDEF | 3 | 19.33 | 5.03 | ACDEF | 3 | 8.00 | 2.65 | ABFG | 3 | 25.00 | 7.81 | ABEF | 8 | 21.63 | 4.53 | ABDF | 2 | 12.00 | 15.56 | ABCDEG | 8 | 6.63 | 3.62 | CF | *** |
| Distance to road (km) (n/site = 30) | 90 | 1.86 | 1.33 | | 90 | 2.03 | 0.42 | CG | 90 | 2.88 | 1.04 | B | 90 | 1.07 | 0.61 | | 240 | 0.73 | 0.46 | | 60 | 4.47 | 1.80 | | 240 | 2.40 | 1.41 | B | *** |
| Distance to vehicular access (km) (n/site = 30) | 90 | 0.32 | 0.30 | D | 90 | 1.55 | 0.88 | CFG | 90 | 1.13 | 0.65 | BFG | 90 | 0.53 | 0.55 | A | 240 | 0.68 | 0.47 | | 60 | 3.03 | 2.53 | BCG | 240 | 1.67 | 0.98 | BCF | *** |
| Distance to house (km) (n/site = 30) | 90 | 2.30 | 1.28 | D | 90 | 2.13 | 1.09 | D | 90 | 3.38 | 1.25 | | 90 | 1.51 | 0.72 | AB | 240 | 1.09 | 0.63 | | 60 | 5.14 | 1.24 | | 240 | 6.04 | 3.77 | | *** |
| Distance to camp (km) (n/site = 30) | 90 | 1.34 | 1.12 | CDE | 90 | 1.82 | 0.88 | CFG | 90 | 1.54 | 0.66 | ABFG | 90 | 1.20 | 0.73 | AE | 240 | 0.86 | 0.45 | AD | 60 | 3.40 | 2.96 | BCG | 240 | 2.15 | 1.58 | BCF | *** |
| Distance to waterpoint (km) (n/site = 30) | 90 | 1.04 | 0.75 | | 90 | 4.51 | 0.82 | G | 90 | 3.15 | 0.83 | D | 90 | 3.16 | 1.07 | C | 240 | 2.78 | 1.64 | | 60 | 1.89 | 1.07 | | 240 | 5.50 | 2.79 | B | *** |
| Adult broken (avg. 1-5 classes) (n/site = 120) | 360 | 2.43 | 1.21 | | 360 | 2.83 | 1.21 | | 360 | 2.13 | 1.06 | | 360 | 2.89 | 1.33 | | 960 | 2.97 | 1.36 | | 240 | 2.83 | 1.20 | | 960 | 2.36 | 1.05 | | - |
| Adult bent (avg. 1-5 classes) (n/site = 120) | 360 | 1.06 | 0.39 | | 360 | 1.33 | 0.90 | | 360 | 1.10 | 0.47 | | 360 | 1.53 | 1.10 | | 960 | 1.25 | 0.72 | | 240 | 1.19 | 0.66 | | 960 | 1.12 | 0.48 | | - |
| Bark stripping ($cm^2$) (n/site = no. of bark-stripped individuals) | 22 | 228.86 | 448.68 | CDFG | 21 | 985.33 | 1336.44 | CDEFG | 3 | 147.33 | 121.99 | ABDEFG | 22 | 867.09 | 1211.50 | ABCEFG | 121 | 1216.39 | 2209.76 | BCDFG | 17 | 1415.59 | 4795.85 | ABCDEG | 16 | 478.00 | 967.71 | ABCDEF | * |
| Juvenile browsing intensity (avg. 1-5 classes) (n/site = no. of palatable individuals) | 144 | 3.83 | 1.54 | | 257 | 3.49 | 1.69 | | 300 | 3.43 | 1.69 | | 279 | 4.19 | 1.43 | | 610 | 3.78 | 1.60 | | 202 | 3.78 | 1.55 | | 890 | 3.76 | 1.69 | | - |
| Adult browsing intensity (avg. 1-5 classes) (n/site = no. of palatable individuals) | 138 | 4.73 | 0.55 | | 237 | 4.50 | 0.97 | | 288 | 3.45 | 1.35 | | 273 | 4.81 | 0.54 | | 562 | 4.76 | 0.59 | | 196 | 3.71 | 1.49 | | 831 | 3.80 | 1.34 | | - |

**Table 4.2. Synoptic table of the proposed habitat variants. Numbers represent percentage frequencies and modified correlation indices multiplied by 100 (superscript). The correlation indices refer to the strength of association between species and variants. Significance of correlation indices was tested using 999 permutations and all species with p = < 0.1 following Sidak's correction are included under their associated variant. The most significant species-variant association is highlighted in grey. Diagnostic species are indicated by significance asterisks and were selected based on the significance of the association (p = < 0.05). Herbaceous species with p = > 0.1 are not included.**

| **Variant (cluster)** | A | B | C | D | E | F | G | | |
|---|---|---|---|---|---|---|---|---|---|
| **Number of sites** | 3 | 3 | 3 | 3 | 8 | 2 | 8 | | |
| **Mean woody species richness** | 15 | 18 | 25 | 19 | 16 | 17 | 20 | | |
| **Mean herbaceous species richness** | 14 | 16 | 20 | 18 | 17 | 14 | 18 | | |
| **Shannon Diversity Index** | 2.15 | 2.56 | 3.10 | 2.47 | 2.26 | 2.37 | 2.87 | | |
| **Number of diagnostic species** | 1 | 4 | 3 | 5 | 7 | 1 | 10 | p-value | |
| **Cluster A: *Dodonaea viscosa* subsp. *angustifolia* shrubland variant** | | | | | | | | | |
| **Woody plants** | | | | | | | | | |
| *Dodonaea viscosa* subsp. *angustifolia* | 43.2 $^{51.8}$ | - $^{-12.3}$ | 6.5 $^{-2.6}$ | 7.5 $^{-1.1}$ | 0.1 $^{-12.2}$ | - $^{-12.3}$ | 0.6 $^{-11.3}$ | 0.007 | ** |
| *Rhus somalensis* | 2.1 $^{9.3}$ | - $^{-2.9}$ | 1.3 $^{4.4}$ | - $^{-2.9}$ | 0.1 $^{-2.6}$ | - $^{-2.9}$ | 0.1 $^{-2.6}$ | 0.088 | |
| **Cluster B: *Cadia purpurea-Olea europaea* forest variant** | | | | | | | | | |
| **Woody plants** | | | | | | | | | |
| *Cadia purpurea* | 0.6 $^{-8.4}$ | 28.6 $^{43.7}$ | 1.7 $^{-6.4}$ | 0.3 $^{-8.9}$ | 0.1 $^{-9.4}$ | 3.3 $^{-3.3}$ | 1.1 $^{-7.4}$ | 0.007 | ** |
| *Olea europaea* subsp. *cuspidata* | 0.7 $^{-3.0}$ | 7.8 $^{20.1}$ | 1.4 $^{-0.7}$ | 0.4 $^{-3.9}$ | 0.8 $^{-2.7}$ | - $^{-5.2}$ | 0.2 $^{-4.7}$ | 0.007 | ** |
| *Croton confertus* | - $^{-6.3}$ | 9.6 $^{19.8}$ | 0.8 $^{-4.0}$ | 1.7 $^{-1.7}$ | 0.4 $^{-5.3}$ | 2.5 $^{0.5}$ | 1.1 $^{-3.1}$ | 0.007 | ** |
| **Herbs** | | | | | | | | | |
| *Oplismenus burmanni* | 0.4 $^{-10.1}$ | 19.5 $^{28.8}$ | 6.8 $^{4.3}$ | 3.2 $^{0.4}$ | - $^{-10.9}$ | - $^{-10.9}$ | 2.7 $^{-1.7}$ | 0.028 | * |
| **Cluster C: *Euclea racemosa-Jasminum grandiflorum* shrubland variant** | | | | | | | | | |
| **Woody plants** | | | | | | | | | |
| *Euclea racemosa* subsp. *schimperi* | 2.4 $^{0.9}$ | 1.3 $^{-2.3}$ | 10.1 $^{23.3}$ | 0.3 $^{-5.1}$ | - $^{-5.9}$ | - $^{-5.9}$ | 0.3 $^{-5.0}$ | 0.007 | ** |
| *Jasminum grandiflorum* | 3.5 $^{5.2}$ | 2.6 $^{2.7}$ | 5.4 $^{11.2}$ | 0.6 $^{-3.8}$ | 0.2 $^{-4.8}$ | - $^{-5.5}$ | 0.2 $^{-5.0}$ | 0.048 | * |
| *Pavetta longiflora* | - $^{-2.5}$ | 0.7 $^{2.0}$ | 1.9 $^{10.3}$ | - $^{-2.5}$ | 0.1 $^{-2.2}$ | - $^{-2.5}$ | - $^{-2.5}$ | 0.007 | ** |
| *Gomphocarpus fruticosus* subsp. *setosus* | 0.4 $^{1.5}$ | - $^{-2.0}$ | 1.3 $^{8.5}$ | - $^{-2.0}$ | - $^{-2.0}$ | - $^{-2.0}$ | - $^{-2.0}$ | 0.061 | |
| *Ficus vasta* | - $^{-2.5}$ | 0.6 $^{1.1}$ | 1.5 $^{7.5}$ | 0.4 $^{0.2}$ | 0.2 $^{-1.5}$ | - $^{-2.5}$ | 0.1 $^{-2.2}$ | 0.055 | |
| *Woodfordia uniflora* | 0.6 $^{1.0}$ | - $^{-2.6}$ | 1.5 $^{7.3}$ | 0.3 $^{-0.8}$ | - $^{-2.6}$ | 0.4 $^{0.1}$ | - $^{-2.6}$ | 0.061 | |
| *Azima tetracantha* | - $^{-0.8}$ | - $^{-0.8}$ | 0.3 $^{4.9}$ | - $^{-0.8}$ | - $^{-0.8}$ | - $^{-0.8}$ | - $^{-0.8}$ | 0.055 | |
| **Herbs** | | | | | | | | | |
| *Dichanthium annulatum* | 2.4 $^{4.6}$ | - $^{-4.3}$ | 3.7 $^{14.4}$ | 0.3 $^{-2.6}$ | 0.1 $^{-3.6}$ | - $^{-4.3}$ | - $^{-4.3}$ | 0.055 | |
| *Arundinella pumila* | 1.1 $^{-7.2}$ | 5.1 $^{3.4}$ | 8.8 $^{13.0}$ | 1.1 $^{-4.6}$ | 1.5 $^{-5.6}$ | 0.7 $^{-4.7}$ | 5.4 $^{5.7}$ | 0.061 | |
| *Enteropogon dolichostachyus* | - $^{-2.3}$ | 0.7 $^{4.5}$ | 1.0 $^{6.9}$ | - $^{-2.3}$ | - $^{-2.3}$ | - $^{-2.3}$ | - $^{-2.3}$ | 0.094 | |
| **Cluster D: *Maytenus dhofarensis-Ficus sycomorus* woodland variant** | | | | | | | | | |
| **Woody plants** | | | | | | | | | |
| *Maytenus dhofarensis* | 3.1 $^{-5.2}$ | 6.4 $^{0.5}$ | 5.1 $^{-1.6}$ | 21.8 $^{26.9}$ | 3.2 $^{-5.0}$ | 1.3 $^{-8.3}$ | 1.8 $^{-7.4}$ | 0.014 | * |
| *Ficus sycomorus* | - $^{-1.3}$ | - $^{-1.3}$ | - $^{-1.3}$ | 0.7 $^{7.7}$ | - $^{-1.3}$ | - $^{-1.3}$ | - $^{-1.3}$ | 0.048 | * |
| *Calotropis procera* | - $^{-1.0}$ | - $^{-1.0}$ | - $^{-1.0}$ | 0.4 $^{6.0}$ | - $^{-1.0}$ | - $^{-1.0}$ | - $^{-1.0}$ | 0.021 | * |
| **Herbs** | | | | | | | | | |
| *Brachiaria eruciformis* | - $^{-2.2}$ | - $^{-2.2}$ | - $^{-2.2}$ | 0.9 $^{10.1}$ | 0.3 $^{-0.1}$ | - $^{-2.2}$ | 0.1 $^{-1.2}$ | 0.041 | * |
| *Blumea lacera* | - $^{-0.8}$ | - $^{-0.8}$ | - $^{-0.8}$ | 0.2 $^{4.7}$ | - $^{-0.8}$ | - $^{-0.8}$ | - $^{-0.8}$ | 0.048 | * |
| *Blumea axillaris* | 1.1 $^{3.2}$ | 0.2 $^{-2.0}$ | 0.1 $^{-2.8}$ | 2.1 $^{11.7}$ | - $^{-3.4}$ | - $^{-3.4}$ | - $^{-3.4}$ | 0.081 | |
| **Cluster E: *Jatropha dhofarica-Zygocarpum dhofarense* sparse woodland variant** | | | | | | | | | |
| **Woody plants** | | | | | | | | | |
| *Jatropha dhofarica* | 3.8 $^{-5.6}$ | 1.7 $^{-8.8}$ | 1.0 $^{-9.9}$ | 3.2 $^{-6.4}$ | 27.9 $^{32.3}$ | 9.2 $^{2.9}$ | 4.5 $^{-4.4}$ | 0.007 | ** |
| *Zygocarpum dhofarense* | 2.5 $^{-6.3}$ | 6.8 $^{1.0}$ | 7.4 $^{1.9}$ | 1.4 $^{-8.2}$ | 15.2 $^{15.1}$ | 1.0 $^{-8.8}$ | 9.3 $^{5.2}$ | 0.007 | ** |
| *Adenium obesum* | - $^{-2.7}$ | - $^{-2.7}$ | 0.4 $^{-0.1}$ | - $^{-2.7}$ | 2.6 $^{13.5}$ | - $^{-2.7}$ | - $^{-2.7}$ | 0.007 | ** |

| | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Herbs** | | | | | | | | | | | | | | | | |
| *Arthraxon junnarensis* | 52.2 | 13.5 | 26.6 | -2.0 | 17.8 | -11.1 | 38.4 | 15.0 | 51.1 | 14.6 | 7.2 | -16.8 | 16.3 | -13.1 | 0.007 | ** |
| *Setaria* sp. | 2.6 | 4.3 | - | -5.0 | - | -5.0 | 1.8 | 1.6 | 5.2 | 12.2 | 0.0 | -4.6 | 0.4 | -3.6 | 0.048 | * |
| *Dactyloctenium aegyptium* | - | -1.2 | - | -1.2 | - | -1.2 | - | -1.2 | 0.6 | 7.2 | - | -1.2 | - | -1.2 | 0.041 | * |
| *Orobanche dhofarensis* | 0.0 | -0.4 | - | -0.6 | - | -0.6 | - | -0.6 | 0.1 | 3.2 | - | -0.6 | - | -0.6 | 0.021 | * |
| *Dyschoriste dalyi* | - | -1.0 | - | -1.0 | - | -1.0 | - | -1.0 | 0.3 | 5.9 | - | -1.0 | - | -1.0 | 0.088 | |
| **Cluster F: Broad-leaved *Blepharispermum hirtum* variant (Kürschner et al., 2004)** | | | | | | | | | | | | | | | | |
| **Woody plants** | | | | | | | | | | | | | | | | |
| *Blepharispermum hirtum* | 0.4 | -9.9 | - | -10.6 | 0.1 | -10.3 | 0.6 | -9.6 | 0.2 | -10.3 | 34.6 | 47.7 | 8.0 | 3.0 | 0.014 | * |
| **Herbs** | | | | | | | | | | | | | | | | |
| *Apluda mutica* | 29.9 | 2.5 | 6.6 | -13.4 | 15.1 | -2.2 | 6.6 | -10.3 | 26.7 | 3.3 | 35.8 | 20.3 | 18.9 | -0.3 | 0.075 | |
| **Cluster G: *Premna resinosa-Hybanthus durus* forest variant** | | | | | | | | | | | | | | | | |
| **Woody plants** | | | | | | | | | | | | | | | | |
| *Premna resinosa* | 0.1 | -2.9 | - | -3.5 | 0.6 | -0.9 | 0.3 | -2.2 | 0.3 | -2.0 | 0.4 | -1.5 | 3.5 | 13.1 | 0.007 | ** |
| *Hybanthus durus* | - | -7.8 | 5.6 | 4.4 | 3.6 | 0.1 | - | -7.8 | 3.6 | 0.1 | 3.1 | -0.9 | 9.0 | 11.9 | 0.021 | * |
| *Euphorbia smithii* | 5.8 | 3.4 | 4.7 | 1.2 | 5.4 | 2.6 | 1.3 | -5.9 | 1.6 | -5.2 | 2.3 | -3.8 | 7.9 | 7.7 | 0.021 | * |
| *Acacia senegal* | 0.6 | -7.0 | 3.8 | -0.3 | 4.3 | 0.9 | 3.1 | -1.7 | 0.5 | -7.1 | 8.1 | 9.0 | 6.8 | 6.2 | 0.048 | * |
| *Delonix elata* | - | -1.6 | - | -1.6 | - | -1.6 | - | -1.6 | - | -1.6 | 0.4 | 2.7 | 0.7 | 5.4 | 0.081 | |
| **Herbs** | | | | | | | | | | | | | | | | |
| *Launaea crassifolia* | 0.1 | -3.5 | 0.2 | -2.7 | 0.1 | -3.5 | - | -3.9 | - | -3.9 | 0.4 | -0.9 | 4.4 | 18.3 | 0.007 | ** |
| *Rungia pectinata* | 4.3 | -3.8 | 11.9 | 13.6 | 5.6 | 1.4 | 1.0 | -8.0 | 0.6 | -9.8 | 0.9 | -5.7 | 11.3 | 12.3 | 0.021 | * |
| *Lepidagathis calycina* | - | -2.3 | - | -2.3 | - | -2.3 | - | -2.3 | - | -2.3 | 0.1 | -0.7 | 1.4 | 12.0 | 0.028 | * |
| *Megalochlamys violacea* | - | -2.2 | 0.2 | -0.6 | - | -2.2 | - | -2.2 | - | -2.2 | - | -2.2 | 1.2 | 11.6 | 0.007 | ** |
| *Barleria hochstetteri* | - | -1.1 | - | -1.1 | - | -1.1 | - | -1.1 | - | -1.1 | - | -1.1 | 0.4 | 6.9 | 0.007 | ** |
| *Ruttya fruticosa* | - | -1.6 | 0.2 | 1.3 | 0.1 | -0.3 | - | -1.6 | 0.0 | -1.3 | - | -1.6 | 0.4 | 5.2 | 0.048 | * |
| *Ruellia grandiflora* | - | -3.0 | 0.2 | -1.8 | - | -3.0 | - | -3.0 | - | -3.0 | 0.1 | -2.6 | 2.2 | 16.2 | 0.061 | |
| **Companion woody plants** | | | | | | | | | | | | | | | | |
| *Acacia gerrardii* | - | -2.9 | - | -2.9 | - | -2.9 | - | -2.9 | 1.0 | 3.1 | 1.5 | 5.6 | 1.0 | 2.8 | 0.738 | |
| *Acridocarpus orientalis* | - | -0.4 | - | -0.4 | - | -0.4 | - | -0.4 | - | -0.4 | - | -0.4 | 0.1 | 2.1 | 0.903 | |
| *Allophylus rubifolius* | 1.1 | -6.3 | 3.6 | -1.1 | 2.5 | -3.4 | 8.6 | 9.1 | 2.2 | -3.9 | 4.4 | 0.4 | 6.8 | 5.3 | 0.126 | |
| *Anogeissus dhofarica* | 2.5 | -8.6 | 4.4 | -5.8 | 4.0 | -6.4 | 16.5 | 12.1 | 12.9 | 6.7 | 9.2 | 1.2 | 8.9 | 0.8 | 0.204 | |
| *Blepharis dhofarensis* | - | -4.7 | 0.8 | -1.6 | 5.4 | 15.0 | - | -4.7 | - | -4.7 | - | -4.7 | 2.8 | 5.3 | 0.463 | |
| *Boscia arabica* | - | -2.1 | - | -2.1 | - | -2.1 | - | -2.1 | - | -2.1 | 1.3 | 8.1 | 0.5 | 2.2 | 0.463 | |
| *Caesalpinia erianthera* | - | -0.6 | - | -0.6 | - | -0.6 | - | -0.6 | - | -0.6 | - | -0.6 | 0.2 | 3.7 | 0.871 | |
| *Carissa spinarum* | 0.6 | 1.7 | - | -2.3 | 0.8 | 3.7 | 0.1 | -1.3 | 0.6 | 1.8 | - | -2.3 | 0.2 | -1.2 | 0.690 | |
| *Commiphora gileadensis* | 3.5 | -1.6 | 1.4 | -5.8 | 6.5 | 4.6 | 5.7 | 2.9 | 3.9 | -0.7 | 1.9 | -4.8 | 6.9 | 5.4 | 0.276 | |
| *Commiphora habessinica* | 13.9 | 3.8 | 3.5 | -9.8 | 11.0 | 0.0 | 10.0 | -1.3 | 16.0 | 6.6 | 9.8 | -1.6 | 12.8 | 2.4 | 0.446 | |
| *Cordia ovalis* | - | -1.3 | - | -1.3 | 0.1 | 0.4 | 0.3 | 2.2 | 0.1 | 0.0 | - | -1.3 | 0.2 | 1.3 | 0.823 | |
| *Cordia perrottetii* | - | -0.5 | - | -0.5 | - | -0.5 | - | -0.5 | 0.1 | 1.2 | - | -0.5 | 0.1 | 1.2 | 0.987 | |
| *Ehretia obtusifolia* | - | -0.8 | - | -0.8 | - | -0.8 | - | -0.8 | - | -0.8 | - | -0.8 | 0.3 | 4.7 | 0.897 | |
| *Flueggea virosa* | 0.1 | -2.6 | 0.4 | -1.2 | 0.1 | -2.6 | 2.8 | 10.9 | 0.1 | -3.0 | - | -3.3 | 1.0 | 1.8 | 0.132 | |
| *Grewia bicolor* | 0.3 | 1.0 | - | -1.7 | 0.4 | 2.4 | - | -1.7 | 0.3 | 0.9 | - | -1.7 | 0.3 | 0.9 | 0.826 | |
| *Grewia villosa* | - | -0.6 | - | -0.6 | 0.1 | 3.5 | - | -0.6 | - | -0.6 | - | -0.6 | - | -0.6 | 0.536 | |
| *Hildebrandtia africana* | - | -0.5 | - | -0.5 | - | -0.5 | - | -0.5 | 0.1 | 1.2 | - | -0.5 | 0.1 | 1.2 | 0.983 | |
| *Lawsonia inermis* | - | -0.9 | 0.1 | 1.7 | 0.1 | 1.7 | - | -0.9 | 0.1 | 0.1 | - | -0.9 | - | -0.9 | 0.908 | |
| *Rhamnus staddo* | - | -0.6 | - | -0.6 | 0.1 | 3.5 | - | -0.6 | - | -0.6 | - | -0.6 | - | -0.6 | 0.568 | |
| *Searsia pyroides* | - | -0.8 | - | -0.8 | 0.1 | 1.9 | - | -0.8 | 0.1 | 0.2 | - | -0.8 | 0.1 | 1.2 | 0.925 | |
| *Solanum incanum* | 8.5 | 3.3 | 5.7 | -1.3 | 5.8 | -1.1 | 11.8 | 8.8 | 6.0 | -0.8 | 4.8 | -2.8 | 2.8 | -6.1 | 0.132 | |
| *Tamarindus indica* | - | -2.1 | - | -2.1 | 1.0 | 5.5 | 0.1 | -1.0 | - | -2.1 | 0.6 | 2.8 | 0.2 | -0.9 | 0.571 | |

### 4.4.1 Cluster A: *Dodonaea viscosa* subsp. *angustifolia* shrubland variant

Diagnostic species: *Dodonaea viscosa* subsp. *angustifolia*

Transgressive character species: *Apluda mutica, Arthraxon junnarensis, Impatiens balsamina, Themeda quadrivalvis, Solanum incanum, Commiphora habessinica, Euphorbia smithii*

**Figure 4.4. *Dodonaea viscosa* subsp. *angustifolia* shrubland**

*Dodonaea viscosa* subsp. *angustifolia* can form an almost continuous shrub layer, or a shrubland-grassland mosaic, with few accompanying shrub and tree species, and thus we propose shrublands dominated by this species such as those around Hafof, Hakab Eirgaz and south of Hasal, as a distinct habitat variant. The absence of forest cover is indicated by a low species diversity ($H$ = 2.15) and the mean adult tree height (236 cm), which was the lowest of all habitat variants. No diagnostic herbaceous species were present in this variant. The results of the CCA and the ANOVA indicate this shrubland occupies areas with gentle gradients (mean of 10°), low terrain roughness, and above average fog moisture levels. Accordingly, it occurs within the monsoon-influenced zone between 400 and 800 m above sea level. The sites of this variant that we surveyed were heavily stocked with cattle soon after the Khareef, leading to soil compaction and desiccation cracks. Almost all *Anogeissus dhofarica* trees had been subject to branch bending management practises, perhaps due to the close proximity to vehicular access routes.

### 4.4.2 Cluster B: *Cadia purpurea-Olea europaea* forest variant

Diagnostic species: *Cadia purpurea, Olea europaea subsp. cuspidata, Croton confertus, Oplismenus burmanni*

Transgressive character species: *Rungia pectinata, Arthaxon junnarensis, Apluda mutica, Arundinella pumila, Maytenus dhofarensis, Zygocarpum dhofarense, Hybanthus durus, Solanum incanum*

**Figure 4.5.** ***Cadia purpurea-Olea europaea*** **forest variant**

This tall mixed forest community attains an average height of 4.6 m and particularly tall individuals of *Euphorbia smithii, Croton confertus* and *Olea europaea* subsp. *cuspidata* are present, the latter growing to heights of 15 m. Sapling *O. europaea* subsp. *cuspidata* were rarely seen or recorded and *Cadia purpurea* is highly abundant. The proportion of standing dead adult woody plants is significantly higher than the other variants and mosses and lichens are abundant. This forest occupies rugged, rocky terrain with a naturally terraced topography, on medium-gradient slopes (mean of 13°) between 600 and 900 m a.s.l. in areas of high fog density. The forest grows on and around huge fallen boulders, some of which are over 20 m in height. It is the dominant habitat at the base of the plateau cliff in western Jabal Qamar, from the Oman-Yemen border in the west, to Shuzuff in the east (approx. 1000 hectares). Here, the cliff acts as a barrier to the incoming fog during the Khareef, depositing considerable levels of moisture on this forest below. Elsewhere in Jabal Qamar similar species assemblages occur in isolated fragments along gullies and drainage channels at equivalent altitudes. Stocking rates were high across much of this habitat due to its close proximity to the large villages of Sarfait and Godraphey. Browsing damage was clearly visible on many species and branch management practises had been carried out extensively on *A. dhofarica* and *O. europaea* subsp. *cuspidata*. Cattle were more abundant than camels due to their ability to navigate the rugged terrain.

### 4.4.3 Cluster C: *Euclea racemosa-Jasminum grandiflorum* shrubland variant

Diagnostic species: *Euclea racemosa* subsp. *schimperi, Jasminum grandiflorum, Pavetta longiflora*

Transgressive character species: *Dodonaea viscosa* subsp. *angustifolia*, *Maytenus dhofarensis*, *Zygocarpum dhofarense, Euphorbia smithii*, *Blepharis dhofarensis*, *Commiphora habessinica*, *Commiphora gileadensis*, *Solanum incanum, Apluda mutica, Arthraxon junnarensis, Arundinella pumila, Heteropogon contortus, Oplismenus burmanni*, *Rungia pectinata*

**Figure 4.6. *Euclea racemosa-Jasminum grandiflorum* shrubland variant.**

Species richness (31 of 42 recorded adult woody species) and diversity ($H$ = 3.10) was highest in this variant which exclusively harboured large individuals of uncommon species such as *Pavetta longiflora* and *Rhus somalensis*, and large individuals of usually stunted species such as *Hybanthus durus* and *Zygocarpum dhofarense*. The large adult shrubs and trees were often clustered amongst boulders which neighboured small grassland patches, which together repeated over terraced slopes, forming a rugged, terraced grassland-shrubland mosaic. The density of adult plants was high (mean of 1485 plants/hectare) and in some flatter areas *D. viscosa* subsp. *angustifolia* was dominant. This species-rich shrubland occupied rugged, rocky and steep slopes (mean of 22°) between 600 and 900 m a.s.l. in areas which receive high levels of fog moisture during the Khareef season. Such areas include at the heads of wadis and tributaries and at the base of cliffs, where fog accumulates. Examples of this variant can be found in the large tributaries of Wadi Sayq such as south of Mathoop and Shershetty and beneath the cliffs to the south of Sha'at, Agdorot and Hasal. At the latter sites (21 and 27) stocking rates were low due to inaccessibility and only cattle were taken here late in the season (January), however, with the recent construction of a vehicular access route below the cliff more livestock have been brought here in recent years.

### 4.4.4 Cluster D: *Maytenus dhofarensis-Ficus sycomorus* sparse woodland variant

Diagnostic species: *Maytenus dhofarensis, Ficus sycomorus, Calotropis procera, Brachiaria eruciformis, Blumea lacera*

Transgressive character species: *Dodonaea viscosa* subsp. *angustifolia, Allophylus rubifolius, Anogeissus dhofarica, Commiphora gileadensis, Commiphora habessinica, Solanum incanum, Apluda mutica, Arthraxon junnarensis, Themeda quadrivalvis*

**Figure 4.7. *Maytenus dhofarensis-Ficus sycomorus* sparse woodland variant.**

Woody plant density can vary substantially but is lower than all other woodland and forest variants (mean of 322 adult plants per hectare). A common feature is the existence of isolated mature individuals of *A. dhofarica*, *Ficus vasta*, *Ficus sycomorus* and *Maytenus dhofarensis*, which results in the highest mean adult basal area of all variants. The unpalatable species *Calotropis procera* and *Blumea lacera* are diagnostic. This variant occurs at elevations between 200 and 800 m a.s.l. in gently sloping areas with high fog moisture levels which can be easily accessed from nearby settlements, such as the areas surrounding Hakab Eirgaz, Bur-a'teeq and north of Dhalkut. Stocking rates and the browsing intensity of woody plants was the highest of all variants. Most saplings were growing under the protection of rocks, soils were heavily compacted with desiccation cracks and there were numerous, often corrugated, livestock trails. Palatable ground vegetation cover was mostly absent at six months after the Khareef.

### 4.4.5 Cluster E: *Jatropha dhofarica-Zygocarpum dhofarense* sparse woodland variant

Diagnostic species: *Jatropha dhofarica, Zygocarpum dhofarense, Adenium obesum, Arthraxon junnarensis, Setaria* sp.*, Dactyloctenium aegyptium, Orobanche dhofarensis*

Transgressive character species: *Anogeissus dhofarica, Commiphora habessinica, Solanum incanum, Apluda mutica*

**Figure 4.8. *Jatropha dhofarica-Zygocarpum dhofarense* sparse woodland variant.**

The succulent *Jatropha dhofarica* and the legume *Z. dhofarense* are widespread species in Jabal Qamar and were recorded from all habitat variants. Nonetheless, they were markedly abundant and diagnostic for this variant. *Adenium obesum* was almost exclusive to this variant and *A. dhofarica* was the dominant large tree species. This sparse woodland habitat with high rock cover (mean of 20%) occupies the gentle-gradient plateau hills between 700 and 900 m a.s.l. throughout Rakhyut and eastern Dhalkut, covering an area of approximately 17,000 hectares. Extremely rocky areas, such as the hills around Sha'at and Aghethob, have higher tree densities and a more diverse herbaceous layer. Fog density is the lowest of all variants as the intensity of the Khareef diminishes on the plateau and solar radiation is high. Stocking rates were the second highest of all variants and the majority of woody individuals showed damage from browsing activity. Stocking rates are high because the plateau is easily accessed by livestock from the numerous villages nearby (mean distance to house = 1.1 km).

### 4.4.6 Cluster F: Broad-leaved *Blepharispermum hirtum* variant (Kürschner et al., 2004)

Diagnostic species: *Blepharispermum hirtum*

Transgressive character species: *Jatropha dhofarica, Acacia senegal, Anogeissus dhofarica, Commiphora habessinica, Apluda mutica, Arthraxon junnarensis, Mitreola petiolata*

**Figure 4.9. Broad-leaved *Blepharispermum hirtum* variant.**

In gentle-medium gradient lowland areas (300–500 m a.s.l.) with below-average fog moisture levels, *Blepharispermum hirtum* can form an almost continuous shrub layer. Soils are markedly alkaline (pH of 8.4) and rock cover is low (6%). The proportion of standing dead adults (5.42%) is the lowest of all variants. In Jabal Qamar, substantial areas of this habitat occur in the foothills below Sarfait and on the northern slopes of Wadi Sayq. The former is severely degraded due to exceptionally high stocking rates during the winter months whilst the latter is far from human settlements and subject to low stocking rates. This variant has been previously identified and described by Kürschner *et al.* (2004).

### 4.4.7 Cluster G: *Premna resinosa-Hybanthus durus* forest variant

Diagnostic species: *Premna resinosa, Hybanthus durus, Euphorbia smithii, Acacia senegal, Launaea crassifolia, Rungia pectinata, Lepidagathis calycina, Megalochlamys violacea, Barleria hochstetteri, Ruttya fruticosa*

Transgressive character species: *Zygocarpum dhofarense, Blepharispermum hirtum, Allophylus rubifolius, Anogeissus dhofarica, Commiphora gileadensis, Commiphora* *habessinica, Apluda mutica, Arthraxon junnarensis, Arundinella pumila, Rungia pectinata*

**Figure 4.10. *Premna resinosa-Hybanthus durus* forest variant.**

This forest variant had the second highest species diversity (*H* = 2.87) of all variants and can be easily identified from its high woody plant density (1706 adult plants/hectare) and diversity of diagnostic herbaceous and subshrub species. It is associated with relatively steep slopes (mean of 20°) in dry areas at elevations between 300 and 800 m a.s.l. and is widespread in Jabal Qamar. Herbs tend to outnumber grasses and steeper areas are characterised by scattered herbs, bare red soils and a shallow leaf litter. Large and reasonably intact examples can be found in the mid-altitude escarpments of Rakhyut, and on the northern slopes of Wadi Sayq. Rock cover is low (6%), soils are markedly alkaline (pH of 8.25) and terrain roughness can be high, although the terrain is not rocky and rugged, rather steeply undulating with numerous small wadis. Stocking rates are low as this habitat generally occurs a substantial distance from human settlements (mean distance to house = 6.04 km). Six species are transgressive with the *Broad-leaved Blepharispermum hirtum* variant, reflecting the similar environmental characteristics and the close syntaxonomical relation between both communities.

## 4.5 Discussion

The interplay between the Khareef fog and the topography of the coastal mountain chain in Yemen and Oman underlies the existence of the *Anogeissus* forest. Throughout much of these mountains, *Acacia-Commiphora* woodlands which are typical for most semi-arid escarpments of the south-western and southern mountains of the Arabian Peninsula are largely absent, replaced instead by variants of the *Anogeissus* forest due to higher precipitation levels (Kürschner *et al.*, 2004). As they share the same mountain chain and biogeographical zone, one might have intuitively expected our classification to reach a similar conclusion to Kürschner *et al.* (2004), with a typical variant and two additional variants. However, our results identified the *Blepharispermum hirtum* shrubland variant as the seventh group in the clustering hierarchy, thus resulting in six variants with greater dissimilarities. This is most likely because a more complex topography in Jabal Qamar, which is differentially influenced by a more precipitous Khareef fog as well as anthropogenic and livestock disturbances, results in a higher spatio-temporal heterogeneity of vegetation communities. The greater topographic complexity of Jabal Qamar in comparison to the Hawf Mountains is clearly observable on topographic maps. Jabal Qamar is also more topographically complex than Jabal Qara, which would explain the differences between the distributions of vegetation communities we observed in Jabal Qamar compared to those summarised as an altitudinal gradient in Jabal Qara (Raffaelli & Tardelli, 2006).

Our proposed *Dodonaea* shrubland variant conforms to the qualitative descriptions made by Kürschner *et al.* (2004) and a number of other authors (Kilian, Hein & Hubaishan, 2002; Kürschner *et al.*, 2004; Al Khulaidi, 2006). *Dodonaea viscosa* subsp. *angustifolia* was traditionally used as firewood, snuff, building material and fertiliser for seasonal rain-fed agricultural plots. For the latter purpose, the branches of this shrub were cut, distributed across the field, burnt, and then the ash mixed with the soil before the Khareef (Miller, Morris & Stuart-Smith, 1988). The *Dodonaea viscosa* subsp. *angustifolia* shrubland in Jabal Qamar, which grows close to settlements in areas with high fog moisture levels, is most likely the remnants of these agricultural plots, which may have since expanded as the shrub is fast-growing and unattractive to livestock (Miller, Morris & Stuart-Smith, 1988). Shrubland dominated by *Dodonaea viscosa* subsp. *angustifolia* also occurs close to settlements in Jabal al Akhdar in northern Oman (Brinkmann *et al.*, 2009) and in the Hawf Mountains in

Yemen (Kürschner *et al.*, 2004) which may also be remnants of past agricultural practises.

The *Cadia purpurea-Olea europaea* forest variant is restricted to rugged slopes in the wettest areas of Jabal Qamar. *Croton confertus* and *C. purpurea* are generally restricted to such areas in Jabal Qamar, and are not highly constant species of the *Anogeissus* forest as described by Kürschner *et al.* (2004). *Olea europaea* subsp. *cuspidata* was traditionally of the greatest importance for its hard wood which was used as a building material, firewood, and fertiliser, and for making weapons and cooking utensils. Stands of this species indicated fertile soils and were subsequently cut and burnt for agricultural plots. This heavy demand has led to sharp declines in its population in Dhofar (Miller, Morris & Stuart-Smith, 1988), however substantial numbers were recorded in this variant, perhaps due to inaccessibility for harvesting the wood and low suitability of the terrain for agriculture. Traditionally it was an unfavoured browse species (Miller, Morris & Stuart-Smith, 1988) but we observed high browsing pressure on accessible foliage, particularly on suckers at the base of the trunk. *Croton confertus* was traditionally used in medicine, and as a fire and construction wood, and akin to our results, Miller *et al.* (1988) noted it was common at lower altitudes in the wet monsoon-affected zones. *Cadia purpurea* is a light-demanding shrub and is unpalatable for livestock. It is an indicator of disturbance in similar Afromontane *Olea-Juniper* forests in Ethiopia (Aynekulu, Denich & Tsegaye, 2009; Aynekulu *et al.*, 2016; Giday *et al.*, 2018), and thus we suspect it has become the dominant understory shrub as the forest has degraded. Interestingly, it was never included in the Plants of Dhofar book (Miller, Morris & Stuart-Smith, 1988).

Our results suggest that a steeper and more rugged terrain is the significant environmental factor which separates the *Euclea racemosa-Jasminum grandiflorum* shrubland variant from the *Cadia purpurea-Olea europaea* forest variant, which both persist at similar altitudes and fog densities. Soil physical conditions such as depth and water retention and soil chemical conditions may vary with the differing terrain, influencing community composition and vegetation structure. The *Euclea racemosa-Jasminum grandiflorum* shrubland had the highest species richness of all the variants, partly owing to locally high fog moisture levels but also to its past inaccessibility to livestock and human activity.

Kürschner *et al.* (2004) identified a sparse altitudinal variant of the *Anogeissus* forest with evergreen Afromontane species from Yemen. In Jabal Qamar, the *Euclea racemosa-Jasminum grandiflorum* shrubland variant and the *Cadia purpurea-Olea europaea* forest variant which both occur between 600 and 900 m a.s.l. could potentially be nested within this variant description, with common character species including *D. viscosa* subsp. *angustifolia*, *Jasminum grandiflorum*, *Pavetta longifolia*, *O. europaea* subsp. *cuspidata* and *R. somalensis*. Similar semi-evergreen species assemblages have been described in Dhofar above elevations of 500 m by Miller *et al.* (1988) and together may provide further evidence for the former existence of a continuous belt of semi-evergreen to evergreen woodland across the southern Arabian mountains (Kürschner *et al.*, 2004).

Our evidence points to the conclusion that the *Maytenus dhofarensis-Ficus sycomorus* sparse woodland variant is the result of long-term anthropogenic disturbance on forests in the monsoon influenced zone of Jabal Qamar. Examples of this variant showed signs of long-term degradation including deforested areas, tree limb management, isolated mature *A. dhofarica* trees, diagnostic unpalatable species, soil compaction, desiccation cracks, stunted phytomorphology and dead stumps (Appendix 16). Classified under this variant, was a very sparsely wooded site with scattered stunted *Commiphora spp*. and adults of *A. dhofarica*, *Ficus vasta* and *Ficus sycomorus*, which were some of the oldest trees in the study. Their size has enabled them to survive high stocking rates and they may have been protected by pastoralists to provide shelter and shade. Similarly, the sharp spines of *Maytenus dhofarensis* may have enabled it to persist under high stocking rates (Scholes & Archer, 1997) (Appendix 16). Kürschner *et al.* (2004) considered such areas in Yemen, and Patzelt (2011) in Dhofar, a result of forest clearance in favour of pastures for livestock. We add that clearance for timber for construction of houses and livestock shelters, and for firewood which was traded with communities living in arid areas of Dhofar may also have occurred (Miller, Morris & Stuart-Smith, 1988, H Al Hikmani 2018, personal communication, 10 September).

The *Jatropha dhofarica-Zygocarpum dhofarense* sparse woodland variant covers a vast area of the plateau in Jabal Qamar, yet is undescribed in the literature. *Zygocarpum dhofarense* has been described as comparatively rare and a species of steeper slopes (Miller, Morris & Stuart-Smith, 1988), yet it was abundant in this variant, although, it is certainly one of the most favoured browse species for camels (Miller, Morris & Stuart-Smith, 1988) and was usually heavily damaged and stunted unless inaccessible to livestock. We suspect this variant may be less clearly defined in Yemen and other parts of Dhofar as its existence in Jabal Qamar is based on the interplay between the periphery of the Khareef fog and the elevation of the mountain plateau. For example, this variant is predominantly absent between Dhalkut and Sarfait where the mountain plateau has its minimum elevation at 1000 m a.s.l., whereas in Rakhyut the plateau begins at 800 m a.s.l. and this variant occupies these hills until reaching its upper altitudinal range. At this point, there is a gradual transition from this sparse woodland variant, to a community dominated by stunted *Jatropha dhofarica, Commiphora* sp. *and A. obesum,* and then to the *Euphorbia balsamifera* cushion shrub community (Al-Zidjali, 1995; Kilian, Hein & Hubaishan, 2002; Kürschner *et al.*, 2004; Knees *et al.*, 2007; Patzelt, 2015). It is likely this *Jatropha dhofarica-Zygocarpum dhofarense* sparse woodland variant, which represents the most important and valuable rangeland area in Jabal Qamar, has lost populations of palatable tree and shrub species due to very high stocking rates throughout much of the year. Subsequently, populations of unpalatable *J. dhofarica* and *A. obesum* have remained stable or increased. Kürschner *et al.* (2004) described the typical *Anogeissus* forest variant as having a high cover-abundance of *Euphorbia smithii* and *J. dhofarica,* however our research indicates that these two species do not co-dominate in Jabal Qamar. While *J. dhofarica* dominates on flat plateau woodlands, *E. smithii* is associated with steeper slopes at lower altitudes with higher fog densities.

*Blepharispermum hirtum* is endemic to the southern mountains of Dhofar and Yemen, and apparently formed the dominant vegetation along the entire length of the foothills of the monsoon-affected mountains (Miller, Morris & Stuart-Smith, 1988). In Jabal Qamar, this variant is confined to low altitude, flat areas, with below-average fog densities and transitions to the *Premna resinosa-Hybanthus durus* forest variant on steeper slopes or rougher terrain. Kürschner *et al.* (2004) described an absence of *J. dhofarica* amongst the shrub layer of this variant but this is not the case in Jabal Qamar.

The *Premna resinosa-Hybanthus durus* forest variant occupies much of the steep escarpments of the Wilayat of Rakhyut, parts of which are quite remote (Appendix 17). This variant had the second lowest fog density (mean of 0.49). Moreover, we suspect net soil water availability is very low as this variant occupies sloped-south facing slopes with high solar insulation, steep fast-draining slopes, or slopes in a leeward position where the Khareef fog does not build up to high densities. Terrain roughness is also low, increasing surface runoff. Accordingly, the drought tolerant woody species, *Premna resinosa, H. durus, E. smithii* and *Acacia senegal* are diagnostic (Miller, Morris & Stuart-Smith, 1988). This variant is probably most similar to the *Acacia-Commiphora* woodland which is typical for most semi-arid escarpments of the southern mountains of Arabia (Miller & Cope, 1996; Ghazanfar & Fisher, 1998).

This research focused on the woodlands and forests within the monsoon-influenced zone of Jabal Qamar between 300 and 900 m above sea level. Outside of this range, different environmental conditions give rise to other distinct vegetation communities. Where wadis meet the ocean, the estuaries known as *khors* constitute ecologically important features along the Jabal Qamar coastline. Saline lagoons are usually present with species of *Typha*, *Phragmites* and *Juncus*, situated behind sandy beaches, and fed by freshwater springs and tidal influxes of seawater (Ball, 2014). Khors with low human disturbance and a freshwater source represent valuable refuges for biodiversity (Ball, Al Fazari & Borrell, 2015). Large trees such as *Tamarindus indica*, *Ficus vasta* and *Phoenix dactylifera* (data not shown) occupy the estuarine plains fed by shallow groundwater. Depending on the local topography, the khor may be bordered by gently-sloping dry coastal shrubland consisting of species such as *J. dhofarica*, *A. obesum*, *Cissus quadrangularis*, *C. habessinica*, *C. gileadensis*, *Delonix elata*, *Boscia arabica* and *Lannea triphylla* (data not shown). Numerous wadis of varying sizes intersect Jabal Qamar. They have formed over millennia through water erosion of the limestone, especially during historical periods of greater precipitation. The largest in Jabal Qamar are Wadi Sayq, Wadi Rakhyut, Wadi Sarfait and Wadi Hawta. Due to the relative inaccessibility of these wadis for livestock grazing and anthropogenic activities and their complex geological structures of ledges, overhangs and caves as well as boulder-strewn wadi beds, they offer valuable refuge for biodiversity (Ball, 2014; Ball, Al Fazari & Borrell, 2015; Ball & Borrell, 2016). The steep wadi sides connect with steep coastal cliffs often hundreds of meters high, which host relatively uncharted floral communities.

At altitudes greater than 1000 m a.s.l. the *Euphorbia balsamifera* cushion shrub community (Al-Zidjali, 1995; Kilian, Hein & Hubaishan, 2002; Kürschner *et al.*, 2004; Knees *et al.*, 2007; Patzelt, 2015) occupies the mountain plateau outside the zone influenced by continuous Khareef mists. This habitat can be easily observed neighbouring several stretches of the main road in Jabal Qamar. *Euphorbia balsamifera* may be accompanied by *Cissus quadrangularis* and stunted individuals of *C. habessinica*, *C. gileadensis*, *J. dhofarica*, *A. obesum* and *P. resinosa* (data not shown). At least 118 species have been recorded from this endemic plant community (Patzelt, 2015). Additional plant communities have been qualitatively described for Jabal Qamar by Patzelt (2015) including a drought-deciduous *Cocculus balfourii-Euphorbia cactus* cliff community (1200–1600 m a.s.l.), a *Seddera glomerata-Aloe dhufarensis* succulent community (1200–1500 m a.s.l.), a xeromorphic *Euphorbia schimperi-Dracaena serrulata* rock community (800–1200 m a.s.l.), a *Tetraena decumbens-Boswellia sacra* community (500–1200 m a.s.l.) and a *Launaea castanosperma-Heliotropium bacciferum* community (300–1000 m a.s.l.).

## 4.6 Conclusion

In this research we conducted the first detailed phytosociological study of the *Anogeissus* forest in the Dhofar Mountains of Oman. From a sample of 7,200 woody plants and 900 quadrat samples across thirty sites, we have identified seven variants of the *Hybantho durae-Anogeissetum dhofaricae* in Jabal Qamar. Six are new and one was pre-described by Kürschner *et al.* (2004). An analysis of associated abiotic and biotic variables and a review of key literature suggest that the *Dodonaea viscosa* subs. *angustifolia* shrubland variant is a result of historical agricultural practises whilst the *Maytenus dhofarensis-Ficus sycomorus* sparse woodland is the result of historical deforestation. The other variants are undergoing varying levels of degradation due to overstocking. The distinction between the variants is more acute in Jabal Qamar than in neighbouring mountain ranges, due to high local variability in available fog moisture as a result of the interplay between the complex topography and the Khareef fog. Future vegetation ecology studies in the region should strive to account for local variability in topoclimatic and disturbance factors, in addition to altitudinal gradient, in order to better understand vegetation responses in the region, especially given the ongoing impact of overstocking on local vegetation communities.

## 4.7 Literature cited

Al-Zidjali T. (1995) Oman: Country Report to the FAO International Technical Conference on Plant Genetic Resources. Muscat.

Aynekulu E., Aerts R., Denich M., Negussie A., Friis I., Demissew S. & Boehmer H. J. (2016) Plant diversity and regeneration in a disturbed isolated dry Afromontane forest in northern Ethiopia. Folia Geobotanica. Folia Geobotanica, 51, 115–127.

Aynekulu E., Denich M. & Tsegaye D. (2009) Regeneration Response of Juniperus procera and Olea europaea subsp cuspidata to Exclosure in a Dry Afromontane Forest in Northern Ethiopia. Mountain Research and Development, 29, 143–152.

Ball L. (2014) An investigation of odonate communities within Wadi Sayq, Dhofar Province, Oman (Insecta: Odonata). Check List, 10, 857–863.

Ball L. & Borrell J. (2016) An inventory of herpetofauna from Wadi Sayq, Dhofar, Oman. Journal of Threatened Taxa, 8, 9454–9460.

Ball L., Al Fazari W. & Borrell J. (2015) Birds of Wadi Sayq, Dhofar, Oman: British Exploring Society expeditions January–March 2012 and 2013. Sandgrouse, 37, 2–12.

Beasom S. L. & Haucke H. H. (1975) A Comparison of Four Distance Sampling Techniques in South Texas Live Oak Mottes. Management, 28, 142–144.

Belbin L. & McDonald C. (1993) Comparing 3 Classification Strategies for Use in Ecology. Journal of Vegetation Science, 4, 341–348.

Brinkmann K., Patzelt A., Dickhoefer U., Schlecht E. & Buerkert A. (2009) Vegetation patterns and diversity along an altitudinal and a grazing gradient in the Jabal al Akhdar mountain range of northern Oman. Journal of Arid Environments. Elsevier Ltd, 73, 1035–1045.

Briske D. D., Fuhlendorf S. D. & Smeins F. E. (2003) Vegetation dynamics on rangelands: A critique of the current paradigms. Journal of Applied Ecology, 40, 601–614.

De Cáceres M. & Jansen F. (2016) Package ‘indicspecies’ (Version 1.7.6). R Package Vignettes, 1–31.

De Cáceres M. & Legendre P. (2009) Associations between species and groups of sites: indices and statistical inference. Ecology, 90, 3566–3574.

Chao A., Chazdon R. L., Colwell R. K. & Shen T. J. (2006) Abundance-based similarity indices and their estimation when there are unseen species in samples. Biometrics, 62, 361–371.

Cottam G. & Curtis J. T. (1956) The Use of Distance Measures in Phytosociological Sampling. Ecology, 37, 451–460.

Dahdouh-Guebas F. & Koedam N. (2006) Empirical estimate of the reliability of the use of the Point-Centred Quarter Method (PCQM): Solutions to ambiguous field situations and description of the PCQM+ protocol. Forest Ecology and Management, 228, 1–18.

Dereje M. & Uden P. (2005) The browsing dromedary camel: I. Behaviour, plant preference and quality of forage selected. Animal Feed Science and Technology, 121, 297–308.

Dias A. S., dos Santos K., dos Santos F. A. M. & Martins F. R. (2017) How liana loads alter tree allometry in tropical forests. Plant Ecology, 218, 119–125.

Dickhoefer U., Buerkert A., Brinkmann K. & Schlecht E. (2010) The role of pasture management for sustainable livestock production in semi-arid subtropical mountain regions. Journal of Arid Environments. Elsevier Ltd, 74, 962–972.

Dufrene M. & Legendre P. (2013) Species Assemblages and Indicator Species : The Need for a Flexible Asymmetrical Approach. Ecological Monographs, 67, 345–366.

El-Mahi A. T. (2011) Old ways in a changing space: the issue of camel pastoralism in Dhofar. Sultan Qaboos University Journal - Agricultural and Marine Sciences, 16, 51–64.

Ellenberg H. & Mueller Dombois D. (1967) Tentative physiognomic-ecological classification of plant formations of the earth. Berichte des Geobotanischen Institutes der Eidg. Techn. Hochschule Stiftung Rübel, 37, 21–55.

Friesen J., Zink M., Bawain A. & Müller T. (2018) Hydrometeorology of the Dhofar cloud forest and its implications for groundwater recharge. Journal of Hydrology: Regional Studies. Elsevier, 16, 54–66.

Ghazanfar S. A. (1996) Analysis of the vegetation of saline, brackish and freshwater pools in Dhofar, Sultanate of Oman. Plant Life in Southwest and Central Asia, 501–508.

Ghazanfar S. A. (1998) Status of the flora and plant conservation in the sultanate of Oman. Biological Conservation, 85, 287–295.

Ghazanfar S. A. (1999) Coastal vegetation of Oman. Estuarine, Coastal and Shelf Science. Academic Press, 49, 21–27.

Ghazanfar S. A. & Fisher M. (1998) Vegetation of the Arabian Peninsula. Dordrecht: Kluwer Academic Publishers.

Giday K., Humnessa B., Muys B., Taheri F. & Azadi H. (2018) Effects of livestock grazing on key vegetation attributes of a remnant forest reserve: The case of Desa'a Forest in northern Ethiopia. Global Ecology and Conservation, 14, 1–14.

Greenacre M. & Primicerio R. (2013) Multivariate Analysis of Ecological Data. Bilbao: Fundación BBVA.

Hildebrandt A. & Eltahir E. A. B. (2006) Forest on the edge: Seasonal cloud forest in Oman creates its own ecological niche. Geophysical Research Letters, 33, 2–5.

Hildebrandt A. & Eltahir E. A. B. (2007) Ecohydrology of a seasonal cloud forest in Dhofar: 2. Role of clouds, soil type, and rooting depth in tree-grass competition. Water Resources Research, 43, 1–13.

Hildebrandt A. & Eltahir E. A. B. (2008) Using a horizontal precipitation model to investigate the role of turbulent cloud deposition in survival of a seasonal cloud forest in Dhofar. Journal of Geophysical Research, 113, 1–11.

Hill M. (1979) TWINSPAN - A Fortran Program for Arranging Multivariate Data in an Ordered Two-way Table by Classification of The Individuals and Attributes. New York: Cornell University.

Al Khulaidi A. (2006) Environmental and human determinates of vegetation distribution in the Hadhramaut region, Republic of Yemen. University of Edinburgh, UK.

Kilian N., Hein P. & Hubaishan M. A. (2002) New and noteworthy records for the flora of Yemen, chiefly of Hadhramaout and Al-Mahra. Willdenowia, 32, 239–269.

Knees S., Laser S., Miller A. & Patzelt A. (2007) A New Species of Barleria (Acanthaceae) from Oman. Edinburgh Journal of Botany, 64, 107–112.

Kürschner H., Hein P., Kilian N. & Hubaishan M. A. (2004) The Hybantho durae-Anogeissetum dhofaricae ass. nova - Phytosociology, structure and ecology of an endemic South Arabian forest community. Phytocoenologia, 34, 569–612.

Lamprey H. F. (1976) Preliminary Proposals for the Conservation and Management of the Jebel Qara Region. Nairobi: UNESCO Regional Office of Science and Technology for Africa.

Lawton R. M. (1978) A reconnaissance survey of the Jabal Qara grazing land ecosystem, with particular reference to the impact of development. London: Overseas Development Agency, Land Resources Development Centre.

Legendre P. & Legendre L. (1998) Numerical Ecology. 2nd edn. Amsterdam: Elsevier.

McCune B., Urban D. L. & Grace J. B. (2002) Analysis of Ecological Communities. Gleneden Beach: MjM Software Design.

Meister J., Hubaishan M. A., Kilian N. & Oberprieler C. (2005) Chloroplast DNA variation in the shrub Justicia areysiana (Acanthaceae) endemic to the monsoon affected coastal mountains of the southern Arabian Peninsula. Botanical Journal of the Linnean Society, 148, 437–444.

Miller A. G. & Cope T. A. (1996) Flora of the Arabian Peninsula and Socotra, Vol. 1. 1st edn. Edinburgh: Edinburgh University Press.

Miller A. G., Morris M. & Stuart-Smith S. (1988) Plants of Dhofar, The Southern Region of Oman: Traditional, Economic, and Medicinal Uses. Muscat: Office of the Adviser for Conservation of the Environment, Diwan of Royal Court.

Mosti S., Raffaelli M. & Tardelli M. (2012) Contribution to the Flora of Central-Southern Dhofar (Sultanate of Oman). Webbia, 67, 65–91.

Oksanen A. J., Blanchet F. G., Kindt R., Legendre P., Minchin P. R., Hara R. B. O., Simpson G. L., Solymos P., Stevens M. H. H. & Wagner H. (2018) Package 'vegan' (Version 2.6). R Package Vignettes, 1–292.

Oman National Centre for Statistics and Information (2017a) 2017 Statistical Year Book. Muscat: National Centre for Statistics and Information.

Oman National Centre for Statistics and Information (2017b) National Centre for Statistics and Information Data Portal. (Accessed: 22 August 2017).

Patzelt A. (2011) The Themeda Quadrivalvis Tall-Grass Savannah of Oman at the Crossroad Between Africa and Asia. Edinburgh Journal of Botany, 68, 301–319.

Patzelt A. (2015) Synopsis of the Flora and Vegetation of Oman, with Special Emphasis on Patterns of Plant Endemism. Abhandlungen der Braunschweigischen Wissenschaftlichen Gesellschaft, 282–317.

Peacock J. ., Ferguson M. ., Alhadrami G. A., McCann I. R., Al Hajoj A., Saleh A. & Karnik R. (2003) Conservation through utilization: a case study of the indigenous forage grasses of the Arabian Peninsula. Journal of Arid Environments, 54, 15–28.

Pereira I. M., Rios J. M., do Vale V. S., Gomes-Klein V. L., Coneglian A. & Rocha E. C. (2018) Forest Structure and the Species Composition of the Parque Estadual Mata Atlântica, Located in Goiás State, Brazil. International Journal of Ecology, Article ID 1219374.

R Core team (2013) R: A Language and Environment for Statistical Computing. Vienna.

Radcliffe-Smith A. (1980) The Vegetation of Dhofar. in Oman Office of the Government Adviser for Conservation of the Environment (ed.) The scientific results of the Oman flora and fauna survey 1977 (Dhofar), Journal of Oman Studies Special Report No. 2. Muscat: Ministry of Information and Culture, 59–86.

Raffaelli M. & Tardelli M. (2006) Phytogeographic zones of Dhofar (Southern Oman). Bocconea, 19, 103–108.

Scholes R. J. & Archer S. R. (1997) Tree-Grass Interactions in Savannas. Annual Review of Ecology and Systematics, 28, 517–544.

Tardelli M. & Raffaelli M. (2006) Some aspects of the vegetation of Dhofar (Southern Oman). Bocconea, 19, 109–112.

Tichý L. & Milan C. (2006) Statistical determination of diagnostic species for site groups of unequal size. Journal of Vegetation Science, 17, 809–818.

Whitehead E. E., Hutchinson C. F., Timmermann T. M. & Varady R. G. (1988) Arid Lands: Today and Tomorrow. Colorado: Westview Press.

Wilson A. & MacLeod N. (1991) Overgrazing: Present or absent? Journal of Range Management, 44, 475–482.

Yim O. & Ramdeen K. T. (2015) Hierarchical Cluster Analysis: Comparison of Three Linkage Measures and Application to Psychological Data. The Quantitative Methods for Psychology, 11, 8–21.

# 5 Stacking plant species distribution models and NDVI to map forest loss in Dhofar, Oman

## 5.1 Abstract

Here we developed a novel method to quantify historic deforestation which we applied to the Jabal Qamar mountain range in Dhofar. Conventional time series analysis of remotely sensed imagery has been found problematic in Dhofar for several reasons. Firstly, it is suspected that much deforestation occurred prior to satellite technology. Secondly, annual variation in the intensity of the monsoon fog results in annual variation of vegetation reflectance values. Finally, forest loss due to livestock browsing is gradual and patchy. However, quantifying the extent of forest loss in Dhofar is important as horizontal precipitation is critical to the water economy of the region. In our method, species distribution models of 18 characteristic large shrub and tree species were stacked to provide a historical baseline range of the *Anogeissus* forest which was then analysed in relation to unforested areas. Unforested areas with suitability for ten or more species were selected to give an estimate of 17.1% deforestation. The cartographic outputs provide a means to visualise the probability of historic anthropogenic deforestation across the landscape.

## 5.2 Introduction

There is increasing pressure on governments, businesses and organisations to not only protect ecosystems in their current states, but to restore them to their former natural conditions prior to large-scale human modification. For example, mining operations in Australia aim to restore exhausted mines to an older baseline condition than existed prior to mining activities (Bell, 2001; Doley, Audet & Mulligan, 2012) and in Scotland there is increasing interest in restoring the once continuous tracts of native forest (Newton, Stirling & Crowell, 2001; Mansourian, Vallauri & Dudley, 2005).

Current methods for determining an ecosystem's historical baseline condition use satellite imagery or aerial photography, local knowledge, written or fossil records, analysis of stable isotopes of soil carbon, dendrochronology or biogenic opals (Scholes & Archer, 1997), however these inherently have limited historical and/or spatial availability. In many instances, conservation objectives struggle to define a reference ecosystem condition (Newton, Stirling & Crowell, 2001) or aim for a seemingly natural baseline condition, which is in fact still in a human modified state due to loss or extinction of knowledge, which has been termed 'shifting baseline syndrome' (Papworth *et al.*, 2009). Species distribution models, which map potential species distributions based on occurrence records, offer a potential alternative to determining a baseline ecosystem condition.

Species distribution models (SDMs) are most commonly used to identify suitable habitat for endangered species to inform research objectives or prioritise areas for conservation (Wilting *et al.*, 2010; Adhikari, Barik & Upadhaya, 2012; Yang *et al.*, 2013), but are also used to map historical or future species or habitat distributions under various climatic or environmental scenarios (Werneck *et al.*, 2011; De-Souza & de Marco, 2014; Aguirre *et al.*, 2017; Manchego *et al.*, 2017; Silva *et al.*, 2017). Such models use current occurrence records and environmental parameters are adjusted according to the conditions prevailing during the past or future period of interest in order to observe species distribution change. This is known as 'hind-casting' and models are usually tested with archaeological evidence (Araújo & Guisan, 2006; Svenning *et al.*, 2011).

Alternatively, and carried out here for the first time, we can assume current environmental conditions but use old location records in order to map species distributions more recently, but prior to large-scale human modification. The baseline condition of interest will depend on the rate of degradation (length of time over which humans have changed the environment) and therefore the age of the location records must match accordingly. This is a valuable, yet seemingly underutilised tool that could be used to reliably inform conservation and restoration objectives, which we have applied to map the historic extent and subsequent deforestation of the *Anogeissus* forest in the Dhofar Mountains of southern Oman.

Quantifying deforestation is usually achieved using time series analysis of remotely sensed imagery (most commonly Landsat) however the baseline condition is limited to the historical availability of imagery suitable for vegetation analyses (1980- ). Thus, it is best suited to mapping and quantifying recent and rapid deforestation scenarios such as commercial logging of tropical forests (Lorena & Lambin, 2009; Verbesselt *et al.*, 2012). In Dhofar however substantial deforestation for cattle pastures occurred prior to satellite technology (Oman Office of the Government Adviser for Conservation of the Environment, 1980; Kürschner *et al.*, 2004; Patzelt, 2011).

There are a number of other problems with using conventional time series analysis of remotely sensed imagery in Dhofar (Galletti, Turner & Myint, 2016). Firstly, high interannual variability in the intensity of the Khareef and subsequently the vegetation reflectance values can give misleading results, especially when using image differencing of two dates (Lu *et al.*, 2004). Secondly, high rock cover in some areas falsifies vegetation reflectance values, especially at course resolutions (e.g. MODIS). Thirdly, the variety of habitat types and lack of ground truth data, as well as inconsistency in plant community distributions between the mountain ranges, can lead to misclassification of habitats in land-cover change detection, even at a course level (e.g. grasslands, shrublands and forests). Finally, if the complex process of cross-calibrating imagery from different satellites is carried out unsatisfactorily, results can be inaccurate and misleading (Vicente-Serrano, Pérez-Cabello & Lasanta, 2008).

In this article we aim to quantify deforestation in Jabal Qamar, a mountain range in western Dhofar, relative to a historical ecosystem baseline condition predicted by species distribution models (SDMs) of adults of 18 characteristic large shrub and tree species. The Dhofar Mountains in the south of Oman are suspected to have lost significant cover of the regionally endemic drought-deciduous *Anogeissus* cloud forest (Kürschner *et al.*, 2004) due to anthropogenic disturbances such as deforestation and livestock browsing. However, remnant isolated trees (many of which were protected to provide shade for people and livestock) or stands of trees, as well as more continuous tracts of forest in less accessible locations provide reliable evidence of the former distribution of the forest.

Quantifying a historical baseline of the coverage of the *Anogeissus* forest in Dhofar is of significant value as the forest is critical to the water economy of the region (Kürschner *et al.*, 2004). During the Khareef the shrub and tree canopies collect substantial quantities of fog moisture through horizontal precipitation capture, sustaining both the forest itself and groundwater supplies (Hildebrandt & Eltahir, 2006, 2007, 2008; Friesen *et al.*, 2018). Moreover, the *Anogeissus* forests are of global conservation importance. They are endemic to the southern coasts of Yemen and Oman and support threatened populations of endemic wildlife species such as the critically endangered Arabian leopard *panthera pardus nimr*, an umbrella and flagship species in Oman (Spalton & Al Hikmani, 2014).

## 5.3 Methods

Our methodological approach involved stacking species distribution models calculated from occurrence records of adults of 18 characteristic large shrub and tree species to produce a layer of species richness of the historical distribution of the *Anogeissus* forest. We then overlaid a layer of unforested land and interpreted unforested areas with high species richness or suitability for many species as having a high probability of deforestation.

### 5.3.1 Species occurrences

A total of 778 occurrence records of 18 large shrub and tree species (Table 5.1) were collected during fieldwork in September 2016 – April 2017. These species are characteristic of the *Anogeissus* forests and collectively they make up the *Anogeissus* forests as seen from satellite imagery, due to their relatively large size and high abundances. The records were collected both opportunistically (447 records) and from thirty systematically sampled sites (331 records). Their locations are shown in Figure 5.1. The opportunistic records were of mature adult individuals encountered throughout the study area of the monsoon-influenced southern escarpments and plateau (0–1200 m a.s.l.), which were mapped onto high resolution satellite imagery on ArcGIS Collector for iPad. At the sample sites (300–500 m and 700–900 m a.s.l.) 120 adult woody plants were sampled at each site using the point-centered quarter method (Cottam & Curtis, 1956). Only occurrences of adults were used in the models to project the oldest baseline condition. The occurrence data was spatially rarefied at 200m, 600m, and 1000m in areas of high, medium and low topographic heterogeneity, respectively, using the SDM toolbox for ArcGIS (Brown, 2014). This graduated filtering method maximises the number of spatially independent locations, while preserving the most occurrence data (Brown, 2014).

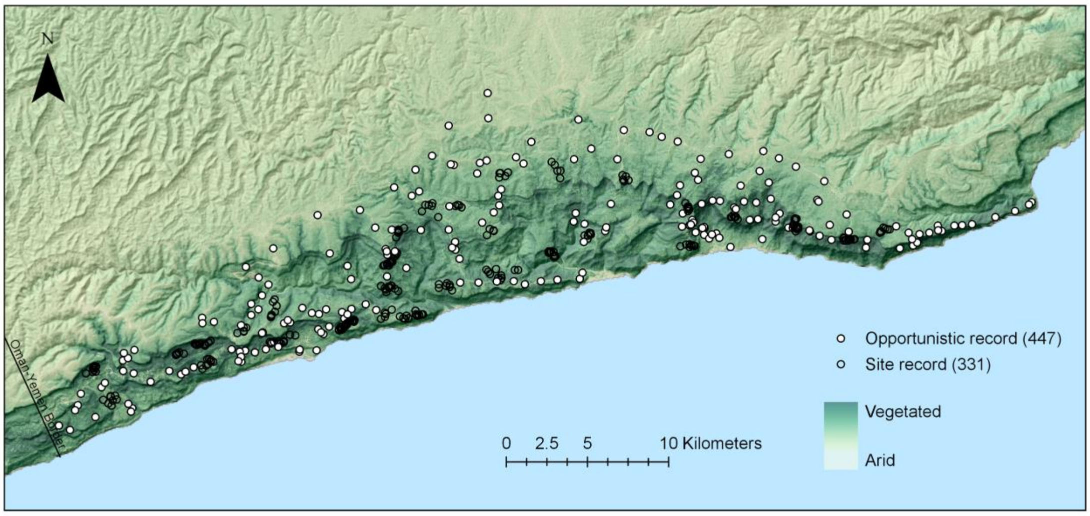



**Figure 5.1. Map showing locations of opportunistic and site-based occurrence records of the 18 species.**

### 5.3.2 Environmental variables

Elevation, slope, aspect and topographic position index (TPI), calculated from TanDEM-X 12m global DEM (Wessel, 2016), were selected for use in the models for the following reasons. Elevation has long been considered an important variable governing vegetation composition in Dhofar (Radcliffe-Smith, 1980; Raffaelli & Tardelli, 2006), specifically temperature and moisture availability vary with altitude. Slope affects radiant energy income, hydrology and soils with subsequent effects on vegetation (Holland & Steyn, 1975; Nearing, 1997). We transformed circular aspect to a topographic radiation aspect index (TRASP aspect) which assigns lowest values to cool, north-facing slopes and highest values to hotter, dryer south-facing slopes (Roberts & Cooper, 1989). In Dhofar, this also acts as a proxy for exposure to southerly Khareef fogs – at lower latitudes aspect can be more related to the direction of rain-bearing winds than solar insulation (Holland & Steyn, 1975). TPI has been shown to be an important variable influencing vegetation communities (Guisan, Weiss & Weiss, 1999), particularly in areas like Dhofar where complex topography affects drainage and moisture availability. Geology was not included as a variable as it acted as a proxy for topographic factors (Dubuis *et al.*, 2011) and is uniformly limestone (Ministry of Petroleum and Minerals, 1986), and soils were not included, as fine resolution data is unavailable, and course-resolution data shows the soil is consistent across the study area (Ministry of Agriculture and Fisheries & FAO, 1990).

A layer of spatial variability in fog density was also included in the model as horizontal precipitation during the monsoon season is critical for sustaining woody vegetation in

Dhofar (Hildebrandt & Eltahir, 2006, 2007, 2008; Kacimov, Hildebrandt & Obnosov, 2010; Manchego *et al.*, 2017; Friesen *et al.*, 2018), and previous research by the author has found fog density to heavily influence vegetation community composition (Chapter 3). Spatial variability in fog density was derived from the near-infrared (NIR) bands of thirteen Landsat 5 and four Landsat 7 products (Welch & Wielicki, 1986), and the ultra blue bands of twenty Landsat 8 products, acquired during Khareef seasons between 1990 and 2017. Through visual inspection a minimum threshold reflectance value was defined for each image to distinguish only the highly reflective fog layer, and the background values set to NULL. The images were then rescaled to a 0-1 range, stacked, and the mean calculated (Appendix 8). Areas with higher reflectance values were interpreted as denser and more moisture-laden fog, as the fogs upper altitude (cloud top) is limited to the altitude of the plateau due to warmer northerly winds from the desert (Kürschner *et al.*, 2004).

### 5.3.3 Species distribution modelling

We used ensemble species distribution models (E-SDMs) which use multiple statistical models to predict species distributions and provides more accurate predictions than just using a single model (Thuiller, 2004; Araújo *et al.*, 2005; Marmion *et al.*, 2009). Modelling was conducted in the SSDM package (Schmitt *et al.*, 2018) in R studio (R Core team, 2013). Four commonly used techniques, generalised linear models (GLM), generalised additive models (GAM), multivariate adaptive regression splines (MARS) and maximum entropy (MAXENT) models were used in the E-SDM with five repetitions.

It was inappropriate to input absence data to the models as current absences are not representative of past absences. We are testing whether species have been lost from areas due to anthropogenic activities and the evidence we have of historical distributions is the current presence of mature adult individuals. Thus, pseudo-absences were created using the recommendations of Barbet-Massin *et al.* (2012) and Wisz & Guisan (2009), which are integrated into the SSDM package. In these studies, models were built and tested using simulated data to identify how, where and how many pseudo-absences should be generated to build reliable species distribution models. For all four methods pseudo-absence locations were generated randomly within the analysis extent. For GLM, GAM and MAXENT 10,000 pseudo-absences were generated but for MARS, a machine-learning technique, 100 pseudo-absences were generated in each run, with equal weight given to presences and absences. By sampling throughout the monsoon-influenced zone and the plateau using both systematic and opportunistic methods, our occurrence data has low collectors' bias (provides a good representation of species relative abundances) and spatial bias, and thus it is robust as a presence-only dataset (Gomes *et al.*, 2018).

Cross-validation involves testing of predicted distributions using a proportion of the occurrence data. In our model, training and evaluation datasets were split using a holdout fraction of 0.75 (Schmitt *et al.*, 2017). To compute the binary map threshold the sensitivity-specificity equality (Cantor *et al.*, 1999) was used as recommended by Liu *et al.* (2005) and Liu, White and Newell (2013). To evaluate the relative importance of each environmental variable, a simple Pearson's correlation *r* was calculated between predictions of the full model and a model with each variable removed (Thuiller *et al.*, 2009). The area under the receiving operating characteristic (ROC) curve (AUC) statistic (DeLeo, 1993; Fielding & Bell, 1997) was used to select the best SDMs to be included in the ESDM, with an inclusion threshold of > 0.75. The SDM's were weighted according to their AUC statistic and the ESDMs evaluated using the AUC statistic.

The widely used AUC statistic is a good measure of model accuracy as it is both threshold independent yet evaluates both the false-positive error rate and the true positive rate (Fielding & Bell, 1997; Aguirre-Gutiérrez *et al.*, 2013). One expects to observe lower AUC values with increasing number of location records when using pseudo-absences, because the maximum attainable AUC value decreases with increasing number of records (maximum AUC = (1 - area occupied) / 2) (Phillips, Anderson & Schapire, 2006; Raes & ter Steege, 2007; Bean, Stafford & Brashares, 2012; Aguirre-Gutiérrez *et al.*, 2013).

### 5.3.4 Stacking species distribution models and NDVI

An NDVI layer derived from a Sentinel 2 dataset acquired in April 2017 was used to differentiate forested areas from unforested areas. The Sentinel 2 Level 1C product was topographically and atmospherically corrected using ESA Snap and the Sentinel 2 toolbox (Zuhlke *et al.*, 2015). Imagery from April was chosen because many trees and shrubs undergo a second generative growth phase producing leaves and flowers (which was particularly pronounced in 2017 due to a strong Khareef the previous summer), whilst herbaceous species have completely died off, making differentiating between forested and unforested areas straightforward. Through visual inspection of NDVI values in forested and unforested areas a threshold value of 0.2 was selected. Inspection of the resultant layer over 30 well-known study sites showed this threshold accurately distinguished areas with no canopy cover or only subshrub cover, from areas with large shrub or tree cover.

Stacked species distribution models (S-SDM) are simply the summation of SDMs or E-SDMs of multiple species to calculate layers of species richness (Ferrier & Guisan, 2006). In our study we produced two types of S-SDM; one by summing the E-SDM suitability probability values (pS-SDM), and another, by summing the binary E-SDM suitability values (bS-SDM) (Dubuis *et al.*, 2011; Calabrese *et al.*, 2014).

We used the pS-SDM only to test for a correlation between species richness and NDVI as it has a wide range of possible values, unlike the bS-SDM which is a count of species richness. A Spearman's rank correlation coefficient found a significant positive correlation ($r_s = 0.74$) from a sample of 5757 random points (Appendix 18). This suggests that consulting NDVI maps could provide a rapid means to identify areas of high species richness and ecological importance for botanical research or conservation in the *Anogeissus* forests.

For our main analysis, to map forest loss in the research area, the bS-SDM was extracted in unforested areas. The bS-SDM was favoured over the pS-SDM as it has been found to give more accurate predictions and reduce over-prediction of plant distributions (Mateo *et al.*, 2012). The species richness values of the bS-SDM were also more interpretable for our deforestation analysis. Cells with values less than one in the bS-SDM were ignored to exclude areas with no suitability for any species. The remaining summed binary suitability values were preserved as a proxy for the probability of historic deforestation, such that with increasing species richness we can be more certain that a species-rich continuous-canopy forest previously occurred, and that anthropogenic activities have caused unforested areas. Conversely, in areas with low species richness, we cannot assume that a continuous-canopy forest (or a dense monoculture for species richness of 1) previously occurred and that unforested areas have been deforested. For example, we cannot assume continuous-canopy forest historically existed in northern-draining wadis where we observe sparse populations of *Acacia gerrardii*, at coastal lagoons where *Tamarindus indica* and *Ficus vasta* may be present, in gently-sloping dry coastal shrublands where *C. habessinica*, *C. gileadensis, Delonix elata* and *Boscia arabica* can be common, or on the plateau where *Euphorbia balsamifera* may be accompanied by stunted individuals of *C. habessinica* and *C. gileadensis*.

Thus, to obtain an estimate of deforestation, we selected a threshold of areas suitable for 10 or more species, as previous research by the author (Chapter 4) has found an average of 10.3 (min 8, max 13, median 10.5, SD 1.5, SE 0.4) of the 18 characteristic large shrub and tree species coexisting in continuous-canopy forest. To refine this estimate we subtracted unforested areas suitable for 10 or more species from the total area suitable for 10 or more species to give an estimate of deforestation of continuous canopy forest.

Path distances to houses, camps, roads, tracks and waterpoints were sampled in 820 locations in both deforested (according to the threshold species richness value of 10 or more species) and forested areas (n=1640), within an altitudinal range of 200 to 700m above sea level, to test the hypothesis that deforested areas are closer to centres of human or livestock activity. Terrain roughness was sampled to test the hypothesis that deforestation occurs on even terrain. In addition, the variables used in the SDMs and species richness (bS-SDM) were sampled. Mann-Whitney U tests determined significant differences between forested and deforested areas, and box plots were used to visualise the results.

## 5.4 Results

The number of location records used in the E-SDMs for each species, three evaluative metrics (AUC, sensitivity and specificity) and the relative importance of the environmental variables are shown in Table 5.1. Fog density is the most important variable for most species, akin to our previous findings in an ordination study (Chapter 3), however for *Blepharispermum hirtum*, *Boscia arabica* and *Delonix elata*, elevation and other topographic variables are more important as these species have restricted altitudinal ranges.

**Table 5.1. Number of location records of eighteen large tree and shrub species used in the ESDMs with three metrics of ESDM evaluation and relative importance of the environmental variables. An AUC of < 0.5 shows the model is no better than random whereas an AUC of 1 indicates highly accurate predictions.**

| Species | Number of records | AUC | Sensitivity | Specificity | Elevation | Fog Density | Slope | Topographic Position Index | TRASP aspect |
|---|---|---|---|---|---|---|---|---|---|
| Acacia gerrardii | 36 | 0.775 | 0.778 | 0.772 | 9% | 52% | 17% | 9% | 14% |
| Acacia senegal | 70 | 0.782 | 0.792 | 0.773 | 14% | 54% | 13% | 9% | 10% |
| Allophylus rubifolius | 37 | 0.839 | 0.846 | 0.832 | 19% | 48% | 12% | 9% | 12% |
| Anogeissus dhofarica | 139 | 0.768 | 0.765 | 0.771 | 12% | 59% | 10% | 9% | 9% |
| Blepharispermu m hirtum | 31 | 0.822 | 0.817 | 0.827 | 37% | 17% | 17% | 23% | 6% |
| Boscia arabica | 11 | 0.922 | 0.972 | 0.871 | 57% | 14% | 10% | 12% | 8% |
| Commiphora gileadensis | 39 | 0.811 | 0.823 | 0.799 | 28% | 44% | 7% | 12% | 9% |
| Commiphora habessinica | 52 | 0.778 | 0.773 | 0.784 | 16% | 48% | 9% | 20% | 7% |
| Croton confertus | 25 | 0.840 | 0.844 | 0.836 | 23% | 46% | 8% | 11% | 12% |
| Delonix elata | 18 | 0.823 | 0.819 | 0.827 | 43% | 9% | 12% | 26% | 10% |
| Dodonaea viscosa subsp. angustifolia | 41 | 0.860 | 0.870 | 0.850 | 16% | 53% | 9% | 10% | 12% |
| Euclea racemosa | 38 | 0.837 | 0.839 | 0.836 | 20% | 56% | 11% | 6% | 7% |
| Euphorbia smithii | 46 | 0.844 | 0.838 | 0.851 | 27% | 49% | 9% | 8% | 7% |
| Ficus sycomorus | 13 | 0.928 | 1.000 | 0.855 | 9% | 45% | 20% | 19% | 8% |
| Ficus vasta | 61 | 0.820 | 0.822 | 0.819 | 15% | 61% | 7% | 9% | 8% |
| Maytenus dhofarensis | 41 | 0.828 | 0.825 | 0.832 | 13% | 56% | 15% | 9% | 7% |
| Olea europaea | 53 | 0.831 | 0.833 | 0.829 | 21% | 52% | 7% | 12% | 8% |
| Tamarindus indica | 27 | 0.831 | 0.823 | 0.838 | 22% | 42% | 10% | 12% | 14% |

Table 5.2 shows how much forest has been lost in areas of varying species richness according to the bS-SDM. Using our threshold of areas suitable for more than 10 or more species, we observe a total loss of 4,363 hectares of forest. In comparison to the total suitable area for any of the shrubs and trees (47,836 hectares) it is a loss of 9.1 percent and excluding the large area on the plateau only suitable for *A. gerrardii* (36,572 hectares) a loss of 11.9 percent. However, these percentages assume coverage of continuous canopy forest in species-poor areas. For a more accurate percentage loss of continuous-canopy forest we can subtract 4,363 hectares from the total area suitable for 10 or more species (25,473 hectares) which gives an estimated loss of 17.1 percent.

**Table 5.2. Total suitable area and unforested suitable area for each level of species richness (bS-SDM), including cumulative summations by decreasing species richness.**

| bS-SDM species richness | Total suitable area (hectares) | Cumulative total suitable area (hectares) | Unforested suitable area (hectares) | Cumulative unforested suitable area (hectares) |
|---|---|---|---|---|
| 18 | 366 | 366 | 140 | 140 |
| 17 | 3349 | 3715 | 605 | 745 |
| 16 | 5123 | 8838 | 713 | 1458 |
| 15 | 5591 | 14429 | 854 | 2312 |
| 14 | 5067 | 19495 | 844 | 3156 |
| 13 | 1777 | 21272 | 410 | 3566 |
| 12 | 1444 | 22716 | 273 | 3839 |
| 11 | 1029 | 23745 | 254 | 4093 |
| 10 | 955 | 24700 | 270 | 4363 |
| 9 | 912 | 25612 | 278 | 4641 |
| 8 | 1437 | 27048 | 617 | 5258 |
| 7 | 1377 | 28425 | 597 | 5855 |
| 6 | 1353 | 29778 | 577 | 6432 |
| 5 | 1327 | 31105 | 534 | 6966 |
| 4 | 1330 | 32435 | 625 | 7590 |
| 3 | 1187 | 33621 | 659 | 8249 |
| 2 | 1842 | 35463 | 1108 | 9358 |
| 1 | 10920 | 46383 | 7825 | 17182 |

Figure 5.2 shows a heat map of unforested suitable areas where the colour gradient represents bS-SDM species richness. The map clearly shows deforested areas suitable for many species at the centre of the monsoon-influenced southern escarpments. We see the Wilayat of Dhalkut, to the west of Wadi Sayq, appears to have lost substantial forest cover, particularly around Sarfait, and on the lowland plateau east of Hafof. In the Wilayat of Rakhyut forest loss is notable on lowland plateaus north of Rakhyut and south-east of Mathoop. A substantial unforested area on the plateau on the edge of the monsoon influenced zone (blue) is only suitable for *Acacia gerrardii*, which unlike the other species, has its optimum at high altitudes on the plateau and in northerly draining wadis and depressions. Similarly, on the low coastal slopes around the towns of Dhalkut and Rakhyut and the village of Al Hawta we see unforested areas suitable for a limited number of species.

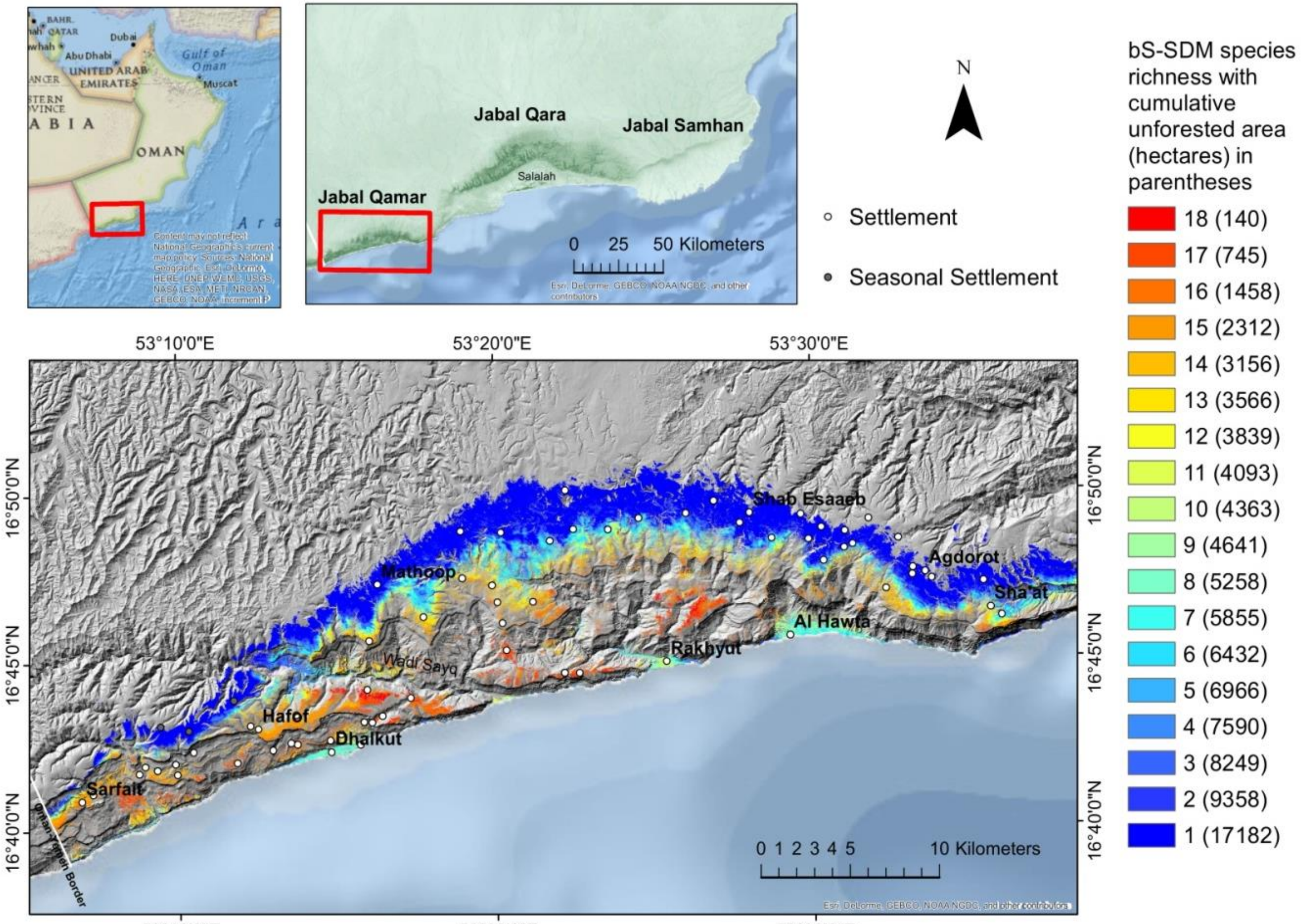



**Figure 5.2. Map of unforested suitable areas in Jabal Qamar. The heat shaded area is unforested land and the colour represents the species richness according to the bS-SDM. Higher values in red are unforested areas with suitability for many tree and shrub species, and lower values in blue are unforested areas with suitability for few species. The cumulative unforested suitable area (hectares) by decreasing species richness is shown (repeated from last column in Table 5.2).**

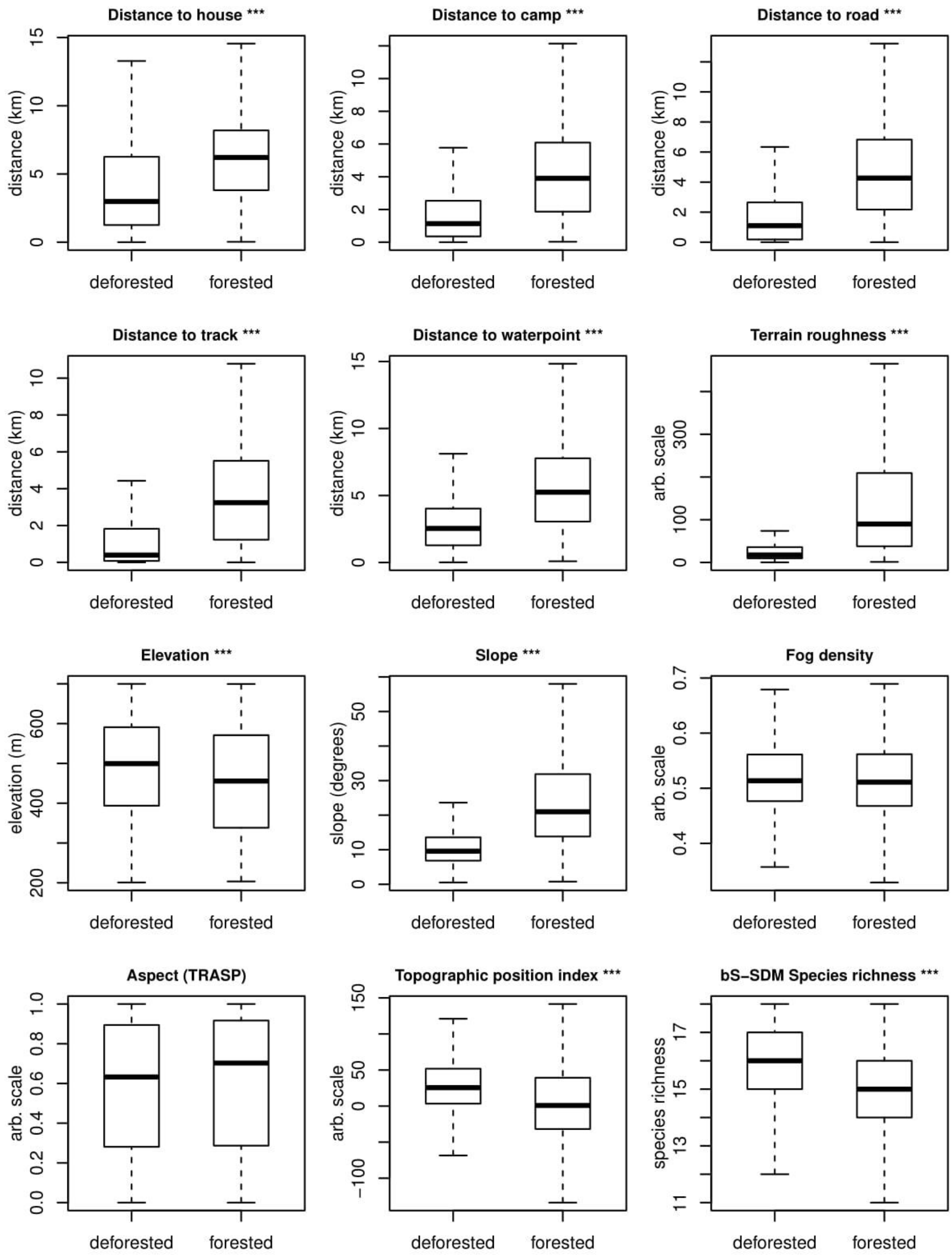



**Figure 5.3. Boxplots comparing a range of variables in forested and deforested areas. The significance of the difference between mean values according to Mann Whitney U tests are shown as significance stars where *** p < 0.001, ** p < 0.01 and * p < 0.05.**

The results in Figure 5.3 show that deforested areas (4,363 hectares) were found to be significantly closer to human settlements ($U = 191510$, $p < 0.001$), camps ($U = 160140$, $p < 0.001$), roads ($U = 158780$, $p < 0.001$), vehicle tracks ($U = 158670$, $p < 0.001$) and waterpoints ($U = 173800$, $p < 0.001$). The terrain was significantly smoother ($U = 111380$, $p < 0.001$), elevation significantly higher ($U = 380600$, $p <$

0.001) and slope gradients significantly shallower ($U = 120260$, $p < 0.001$) in deforested areas. Deforested areas had on average a significantly higher TPI ($U = 435070$, $p < 0.001$) which means they are situated on hilltops and plateaus rather than within wadis. Fog density ($U = 350930$, $p < 0.125$) and aspect (TRASP) ($U = 322180$, $p < 0.144$) were not significantly different between deforested and forested areas. Predicted species richness (bS-SDM) was significantly higher ($U = 436990$, $p < 0.001$) in deforested areas.

## 5.5 Discussion

Our study demonstrates a new method for quantifying forest loss over a longer time-scale than can be achieved using time series analysis of remotely sensed imagery. Forest loss is determined in relation to a baseline condition set by the age of the adult trees used as occurrence data in the SDMs, rather than by the earliest available satellite imagery. Nonetheless, this method has several limitations. Firstly, fieldwork is required to record location data of adult trees, although in some situations location data could be sourced from herbarium records which could provide an older baseline condition. Secondly, this method does not take in to account colonization of unpalatable shrub and tree species, or deforestation and recolonization/reforestation processes. Thirdly, the suitability models are based on extant individuals and thus the historical baseline of the distribution models is limited by the age of the species location records and the date of the baseline condition is undeterminable unless tree ring analysis is conducted. Thus, to achieve the oldest and presumably most natural historical baseline, one must aim to record the locations of the oldest trees, or source records from herbarium specimens or the accounts of local people.

A number of steps in the analysis warrant further discussion. It is important to only run the analysis on the dominant large shrub and tree species which differentiate the forested and unforested areas selected from the NDVI layer, which requires some familiarity with the plant communities of the study area. The methods used for the SDMs must be carefully considered. Predicted distributions can be sensitive to factors such as the distribution and sample size of occurrence data (Bean, Stafford & Brashares, 2012; Saupe *et al.*, 2012), the specific models chosen, and the parameters of the models, and several authors stress the need for in-depth analyses of the influence of such factors on predicted distributions (Araújo & Guisan, 2006; Wisz *et al.*, 2008; Marmion *et al.*, 2009).

Whilst the sensitivity to choice of species number can be inferred from Table 5.2, one must also carefully select the appropriate NDVI threshold value to distinguish forested and unforested areas as only a slight difference in this value may result in substantially different predictions of unforested areas. Figure X.X shows the results of a sensitivity analysis of the NDVI threshold for our study. We see that with increasing suitability for the forest (>1, >5, >10 or >15 species) there is increasing resilience of the resultant suitable area, to changes in the NDVI threshold. Indeed, we know that NDVI and species richness (pS-SDM) are positively correlated (Appendix 18) and therefore in species-rich areas the difference between NDVI values in unforested and forested areas is high. Thus, by refining our results based on a threshold of 10 or more species, we reduce the sensitivity of the results to the precise NDVI threshold selected.

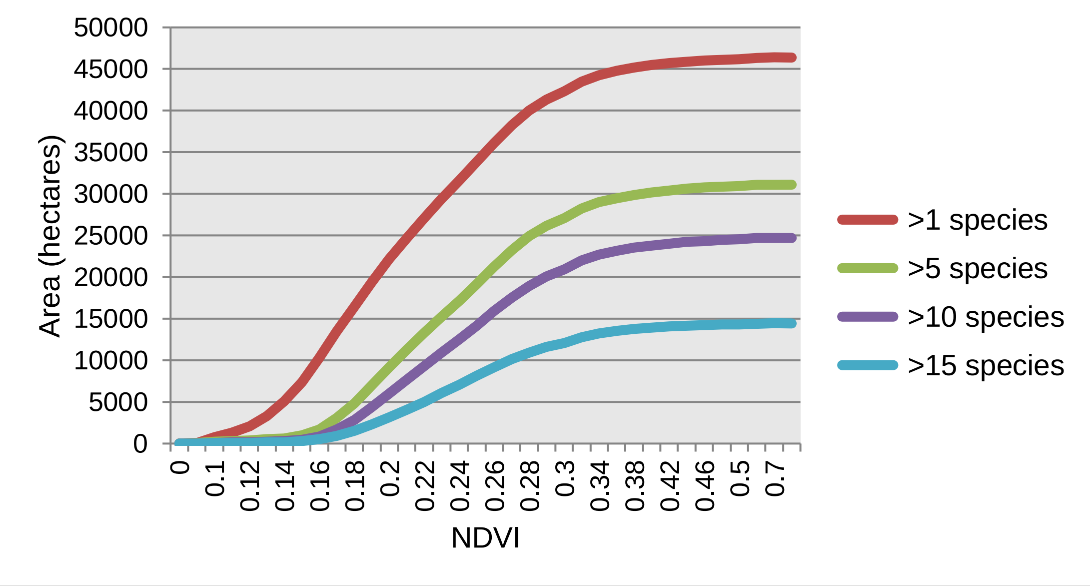



**Figure 5.4. Sensitivity analysis of the effect of change in the NDVI threshold on the total unforested suitable area across four species richness thresholds (>1, >5, >10 and >15 species). With increasing suitability for the forest (species richness) there is increasing resilience of the resultant suitable area to changes in the NDVI threshold.**

The threshold bS-SDM species richness values on which to base the final estimate of the area of deforestation can be informed by pre-existing data on species richness, density and canopy cover from published phytosociological studies (e.g. Chapter 4). Arguably of greater merit than the numeric estimate of deforestation however, are the maps of deforestation which enable quick and easy identification of deforested areas with an associated scale of probability (or likelihood or severity) of deforestation. In areas with suitability for many species we can be quite certain that deforestation has taken place; firstly, because these areas have been predicted to be suitable for species-rich forest and secondly, because these predictions are the harmonious results of multiple models. In areas with suitability for fewer species but little woody cover it is also reasonable to assume that the natural baseline condition should be a higher density of shrubs and trees. For example, our map shows a substantial unforested area on the fringe of the monsoon-influenced zone that is suitable for between five and ten species which we would expect to have greater shrub and tree cover. Indeed, the small wadis which intersect these hills are forested and the loss of woody cover may be due to the close proximity of villages.

By means of a comparison of variables in deforested and forested areas our results provide strong evidence that deforestation is human-induced, as deforested areas were significantly closer (> 2 km) to anthropogenic features such as houses, roads, vehicle tracks and camps. The significant and substantial differences in slope and terrain roughness also suggest that flat and accessible areas with an even terrain have been most susceptible to deforestation. Furthermore, despite fog density being the most powerful variable influencing vegetation communities in the study area (Chapter 3) it was not significantly different in forested and deforested areas.

Deforested areas in Dhofar are the result of a combination of resource-use activities including harvesting of wood for tools, construction and fuel (Miller, Morris & Stuart-Smith, 1988), conversion of forest to cattle pasture (Kürschner *et al.*, 2004) or agricultural plots (Miller, Morris & Stuart-Smith, 1988), and since the 1970s, due to browsing pressure from high numbers of camels, cattle and goats (Lamprey, 1976; Lawton, 1978; Oman Office of the Government Adviser for Conservation of the Environment, 1980; Wilson & MacLeod, 1991; Ghazanfar, 1998; Peacock *et al.*, 2003; Hedges & Lawson, 2006; Tardelli & Raffaelli, 2006; Directorate-General of Nature Conservation, 2010; El-Mahi, 2011b). Our results show substantial areas of deforestation on lowland plateaus in the monsoon-influenced zone. These warmer, wetter areas with deeper and more fertile soils would have been best suited for pasture or agriculture due to the optimum growing conditions. These areas today have high densities of permanent livestock encampments. At higher altitudes on the plateau hills, but still within the monsoon-influenced zone, loss of woody cover is more likely a result of livestock browsing, due to the close proximity of many villages. Indeed, severe browsing damage, stunting, and branch bending practises were noted in this area during fieldwork. On the edge of the monsoon-influenced zone the xeric conditions are unfavourable for most species so a continuous cover of woodland most likely never existed. However, due to these ecologically limiting conditions the vegetation in this area may have been particularly susceptible to human disturbance from the nearby development of roads and villages, and to browsing pressure from thousands of livestock which have occupied this area during, and often outside, the Khareef season for several decades. With this is mind and considering the existence of isolated mature *A. gerrardii* trees, we cannot dismiss the accounts of local pastoralists describing a loss of woody species in these plateau areas.

The maps of deforestation represent useful resources for future research or conservation practise (Appendix 19). They could inform replanting programmes by identifying areas which are likely to reforest most readily, or they can be used to inform land-use planning. Furthermore, they could be utilised for animal species conservation, for example, by enabling visualisation of the most transformed or unnatural habitats within a species' range or utilised in SDMs to predict historical species distributions.

## 5.6 Conclusion

In this research we stacked species distribution models derived from adult individuals of characteristic large shrub and tree species to estimate the coverage of the *Anogeissus* forest prior to large-scale human modification. We then used an NDVI layer to identify unforested areas, and extracted the stacked SDM. The resultant heat maps showed unforested areas, and the predicted species richness serves as a proxy for the probability of deforestation. This method is relatively simple yet provides useful cartographic outputs with multiple applications for conservation. Identifying deforested areas and monitoring future deforestation in Jabal Qamar, Dhofar, and the wider South Arabian cloud forest environment is of great importance as the forest provides a range of valuable ecosystem services such as groundwater recharge, forage resources and tourism interests. Furthermore the endemic ecosystem has high scientific value and harbours unique biological assemblages.

## 5.7 Literature cited


Adhikari D., Barik S. K. & Upadhaya K. (2012) Habitat distribution modelling for reintroduction of Ilex khasiana Purk., a critically endangered tree species of northeastern India. *Ecological Engineering*, *40*, 37–43.

Aguirre-Gutiérrez J., Carvalheiro L. G., Polce C., van Loon E. E., Raes N., Reemer M. & Biesmeijer J. C. (2013) Fit-for-Purpose: Species Distribution Model Performance Depends on Evaluation Criteria - Dutch Hoverflies as a Case Study. *PLoS ONE*, *8*, e63708.

Aguirre N., Eguiguren P., Maita J., Ojeda T., Sanamiego N., Furniss M. & Aguirre Z. (2017) Potential impacts to dry forest species distribution under two climate change scenarios in southern Ecuador. *Neotropical Biodiversity*, *3*, 18–29.

Araújo M. B., Whittaker R. J., Ladle R. J. & Erhard M. (2005) Reducing uncertainty in projections of extinction risk from climate change. *Global Ecology and Biogeography*, *14*, 529–538.

Araújo M. B. & Guisan A. (2006) Five (or so) challenges for species distribution modelling. *Journal of Biogeography*, *33*, 1677–1688.

Barbet-Massin M., Jiguet F., Albert C. H. & Thuiller W. (2012) Selecting pseudo-absences for species distribution models: How, where and how many? *Methods in Ecology and Evolution*, *3*, 327–338.

Bean W. T., Stafford R. & Brashares J. S. (2012) The effects of small sample size and sample bias on threshold selection and accuracy assessment of species distribution models. *Ecography*, *35*, 250–258.

Bell L. C. (2001) Establishment of native ecosystems after mining - Australian experience across diverse biogeographic zones. *Ecological Engineering*, *17*, 179–186.

Brown J. L. (2014) SDMtoolbox: a python-based GIS toolkit for landscape genetic, biogeographic and species distribution model analyses. *Methods in Ecology and Evolution*, *5*, 694–700.

Calabrese J. M., Certain G., Kraan C. & Dormann C. F. (2014) Stacking species distribution models and adjusting bias by linking them to macroecological models. *Global Ecology and Biogeography*, *23*, 99–112.

Cantor S. B., Sun C. C., Tortolero-Luna G., Richards-Kortum R. & Follen M. (1999) A Comparison of C/B Ratios from Studies Using Receiver Operating Characteristic Curve Analysis. *Journal of Clinical Epidemiology*, *52*, 885–892.

Cottam G. & Curtis J. T. (1956) The Use of Distance Measures in Phytosociological Sampling. *Ecology*, *37*, 451–460.

De-Souza R. A. & de Marco P. (2014) The use of species distribution models to predict the spatial distribution of deforestation in the western Brazilian Amazon. *Ecological Modelling*, *291*, 250–259.

DeLeo J. M. (1993) Receiver operating characteristic laboratory (ROCLAB): Software for developing decision strategies that account for uncertainty. *1993 (2nd) International Symposium on Uncertainty Modeling and Analysis*, 318–325.

Directorate-General of Nature Conservation (2010) *Fourth National Report To the Convention on Biological Diversity*. Muscat: Ministry of Environment and Climate Affairs.

Doley D., Audet P. & Mulligan D. R. (2012) Examining the Australian context for post-mined land rehabilitation: Reconciling a paradigm for the development of natural and novel ecosystems among post-disturbance landscapes. *Agriculture, Ecosystems and Environment*. Elsevier B.V., *163*, 85–93.

Dubuis A., Pottier J., Rion V., Pellissier L., Theurillat J. P. & Guisan A. (2011) Predicting spatial patterns of plant species richness: A comparison of direct macroecological and species stacking modelling approaches. *Diversity and Distributions*, *17*, 1122–1131.

El-Mahi A. T. (2011) Old ways in a changing space: the issue of camel pastoralism in Dhofar. *Sultan Qaboos University Journal - Agricultural and Marine Sciences*, *16*, 51–64.

Ferrier S. & Guisan A. (2006) Spatial modelling of biodiversity at the community level. *Journal of Applied Ecology*, *43*, 393–404.

Fielding A. H. & Bell J. F. (1997) A review of methods for the assessment of prediction errors in conservation presence / absence models. *Environmental Conservation*, *24*, 38–49.

Friesen J., Zink M., Bawain A. & Müller T. (2018) Hydrometeorology of the Dhofar cloud forest and its implications for groundwater recharge. *Journal of Hydrology: Regional Studies*. Elsevier, *16*, 54–66.

Galletti C. S., Turner B. L. & Myint S. W. (2016) Land changes and their drivers in the cloud forest and coastal zone of Dhofar, Oman, between 1988 and 2013. *Regional Environmental Change*. Springer Berlin Heidelberg, *16*, 2141–2153.

Ghazanfar S. A. (1998) Status of the flora and plant conservation in the sultanate of Oman. *Biological Conservation*, *85*, 287–295.

Gomes V. H. F., Ijff S. D., Raes N., Amaral I. L., Salomão R. P., Coelho L. D. S., Matos F. D. D. A., Castilho C. V., Filho D. D. A. L., López D. C., *et al.* (2018) Species Distribution Modelling: Contrasting presence-only models with plot abundance data. *Scientific Reports*, *8*, 1–12.

Guisan A., Weiss S. B. & Weiss A. D. (1999) GLM versus CCA spatial modeling of plant species distribution. *Plant Ecology*, *143*, 107–122.

Hedges S. & Lawson D. (2006) *Monitoring the Illegal Killing of Elephants: Dung Survey Standards for the Mike Programme*. Nairobi: CITES MIKE Programme.

Hildebrandt A. & Eltahir E. A. B. (2006) Forest on the edge: Seasonal cloud forest in Oman creates its own ecological niche. *Geophysical Research Letters*, *33*, 2–5.

Hildebrandt A. & Eltahir E. A. B. (2007) Ecohydrology of a seasonal cloud forest in Dhofar: 2. Role of clouds, soil type, and rooting depth in tree-grass competition. *Water Resources Research*, *43*, 1–13.

Hildebrandt A. & Eltahir E. A. B. (2008) Using a horizontal precipitation model to investigate the role of turbulent cloud deposition in survival of a seasonal cloud forest in Dhofar. *Journal of Geophysical Research*, *113*, 1–11.

Holland P. G. & Steyn D. G. (1975) Vegetational Responses to Latitudinal Variations in Slope Angle and Aspect. *Journal of Biogeography*, *2*, 179–183.

Kacimov A., Hildebrandt A. & Obnosov Y. (2010) Impact of the Dhofar cloud forest on subsurface hydrological fluxes: Modelling on the scale of a near-root zone soil continuum. *Proceedings of the Mary State Technical University: Forest, Ecology, Management of Natural Resources*, 21–29.

Kürschner H., Hein P., Kilian N. & Hubaishan M. A. (2004) The Hybantho durae-Anogeissetum dhofaricae ass. nova - Phytosociology, structure and ecology of an endemic South Arabian forest community. *Phytocoenologia*, *34*, 569–612.

Lamprey H. F. (1976) *Preliminary Proposals for the Conservation and Management of the Jebel Qara Region*. Nairobi: UNESCO Regional Office of Science and Technology for Africa.

Lawton R. M. (1978) *A reconnaissance survey of the Jabal Qara grazing land ecosystem, with particular reference to the impact of development*. London: Overseas Development Agency, Land Resources Development Centre.

Liu C., Berry P. M., Dawson T. P. & Pearson R. G. (2005) Selecting thresholds of occurrence in the prediction of species distributions. *Ecography*, *28*, 385–393.

Liu C., White M. & Newell G. (2013) Selecting thresholds for the prediction of species occurrence with presence-only data. *Journal of Biogeography*, *40*, 778–789.

Lorena R. B. & Lambin E. F. (2009) The spatial dynamics of deforestation and agent use in the Amazon. *Applied Geography*, *29*, 171–181.

Lu D., Mausel P., Brondízio E. & Moran E. (2004) Change detection techniques. *International Journal of Remote Sensing*, *25*, 2365–2401.

Manchego C. E., Hildebrandt P., Cueva J., Espinosa C. I., Stimm B. & Günter S. (2017) Climate change versus deforestation: Implications for tree species distribution in the dry forests of southern Ecuador. *PLoS ONE*, *12*, e0190092.

Mansourian S., Vallauri D. & Dudley N. (2005) *Forest Restoration in Landscapes: Beyond Planting Trees*. New York: Springer.

Marmion M., Parviainen M., Luoto M., Heikkinen R. K. & Thuiller W. (2009) Evaluation of consensus methods in predictive species distribution modelling. *Diversity and Distributions*, *15*, 59–69.

Mateo R. G., Felicísimo Á. M., Pottier J., Guisan A. & Muñoz J. (2012) Do stacked species distribution models reflect altitudinal diversity patterns? *PLoS ONE*, *7*, e32586.

Miller A. G., Morris M. & Stuart-Smith S. (1988) *Plants of Dhofar, The Southern Region of Oman: Traditional, Economic, and Medicinal Uses*. Muscat: Office of the Adviser for Conservation of the Environment, Diwan of Royal Court.

Ministry of Agriculture and Fisheries & FAO (1990) *Sultanate of Oman Soil Map (OMA/87/011)*. Rome: FAO.

Ministry of Petroleum and Minerals (1986) *Geological map, Oman 1:100,000*. London: Colourmap Scanning Ltd.

Nearing M. A. (1997) A Single, Continuous Function for Slope Steepness Influence on Soil Loss. *Soil Science Society of America Journal*, *61*, 917.

Newton A., Stirling M. & Crowell M. (2001) Current Approaches to Native Woodland Restoration in Scotland. *Botanical Journal of Scotland*, *53*, 169–195.

Oman Office of the Government Adviser for Conservation of the Environment (1980) *The scientific results of the Oman flora and fauna survey 1977 (Dhofar), Journal of Oman Studies Special Report No. 2*. Muscat: Ministry of Information and Culture.

Papworth S. K., Rist J., Coad L. & Milner-Gulland E. J. (2009) Evidence for shifting baseline syndrome in conservation. *Conservation Letters*, *2*, 93–100.

Patzelt A. (2011) The Themeda Quadrivalvis Tall-Grass Savannah of Oman at the Crossroad Between Africa and Asia. *Edinburgh Journal of Botany*, *68*, 301–319.

Peacock J. ., Ferguson M. ., Alhadrami G. A., McCann I. R., Al Hajoj A., Saleh A. & Karnik R. (2003) Conservation through utilization: a case study of the indigenous forage grasses of the Arabian Peninsula. *Journal of Arid Environments*, *54*, 15–28.

Phillips S. J., Anderson R. P. & Schapire R. E. (2006) Maximum entropy modeling of species geographic distributions. *Ecological Modelling*, *190*, 231–259.

R Core team (2013) R: A Language and Environment for Statistical Computing. Vienna.

Radcliffe-Smith A. (1980) The Vegetation of Dhofar. in Oman Office of the Government Adviser for Conservation of the Environment (ed.) *The scientific results of the Oman flora and fauna survey 1977 (Dhofar), Journal of Oman Studies Special Report No. 2*. Muscat: Ministry of Information and Culture, 59–86.

Raes N. & ter Steege H. (2007) A null-model for significance testing of presence-only species distribution models. *Ecography*, *30*, 727–736.

Raffaelli M. & Tardelli M. (2006) Phytogeographic zones of Dhofar (Southern Oman). *Bocconea*, *19*, 103–108.

Roberts D. W. & Cooper S. V (1989) Concepts and techniques of vegetation mapping. in Ferguson, D., Morgan, P., and Johnson, F. D. (eds) *Land classification based on vegetation: applications for resource management*. Washington: USDA Forest Service, 90–96.

Saupe E. E., Barve V., Myers C. E., Soberón J., Barve N., Hensz C. M., Peterson A. T., Owens H. L. & Lira-Noriega A. (2012) Variation in niche and distribution model performance: The need for a priori assessment of key causal factors. *Ecological Modelling*, *237–238*, 11–22.

Schmitt A. S., Pouteau R., Justeau D., Boissieu F. De & Birnbaum P. (2018) Package 'SSDM'. *R Package Vignettes*.

Schmitt S., Pouteau R., Justeau D., de Boissieu F. & Birnbaum P. (2017) SSDM: An R package to predict distribution of species richness and composition based on stacked species distribution models. *Methods in Ecology and Evolution*, *8*, 1795–1803.

Scholes R. J. & Archer S. R. (1997) Tree-Grass Interactions in Savannas. *Annual Review of Ecology and Systematics*, *28*, 517–544.

Silva T. R., Medeiros M. B., Noronha S. E. & Pinto J. R. R. (2017) Species distribution models of rare tree species as an evaluation tool for synergistic human impacts in the Amazon rainforest. *Revista Brasileira de Botanica*, *40*, 963–971.

Spalton A. & Al Hikmani H. (2014) *The Arabian Leopards of Oman*. 1st edn. London: Stacey International.

Svenning J. C., Fløjgaard C., Marske K. A., Nógues-Bravo D. & Normand S. (2011) Applications of species distribution modeling to paleobiology. *Quaternary Science Reviews*, *30*, 2930–2947.

Tardelli M. & Raffaelli M. (2006) Some aspects of the vegetation of Dhofar (Southern Oman). *Bocconea*, *19*, 109–112.

Thuiller W. (2004) Patterns and uncertainties of species' range shifts under climate change. *Global Change Biology*, *10*, 2020–2027.

Thuiller W., Lafourcade B., Engler R. & Araújo M. B. (2009) BIOMOD - A platform for ensemble forecasting of species distributions. *Ecography*, *32*, 369–373.

Verbesselt J., Kalomenopoulos M., Souza C. & Herold M. (2012) Near real-time deforestation monitoring in tropical ecosystems using satellite image time series. *2012 IEEE International Geoscience and Remote Sensing Symposium*, 2020–2023.

Vicente-Serrano S. M., Pérez-Cabello F. & Lasanta T. (2008) Assessment of radiometric correction techniques in analyzing vegetation variability and change using time series of Landsat images. *Remote Sensing of Environment*, *112*, 3916–3934.

Welch R. M. & Wielicki B. A. (1986) The stratocumulus nature of fog. *Journal of Climate and Applied Meteorology*, *25*, 101–111.

Werneck F. P., Costa G. C., Colli G. R., Prado D. E. & Sites Jr J. W. (2011) Revisiting the historical distribution of Seasonally Dry Tropical Forests: new insights based on palaeodistribution modelling and palynological evidence. *Global Ecology and Biogeography*, *20*, 272–288.

Wessel B. (2016) *TanDEM-X Ground Segment, DEM Products Specification Document*. Cologne: German Aerospace Center (DLR).

Wilson A. & MacLeod N. (1991) Overgrazing: Present or absent? *Journal of Range Management*, *44*, 475–482.

Wilting A., Cord A., Hearn A. J., Hesse D., Mohamed A., Traeholdt C., Cheyne S. M., Sunarto S., Jayasilan M. A., Ross J., *et al.* (2010) Modelling the species distribution of flat-headed cats (Prionailurus planiceps), an endangered South-East Asian small felid. *PLoS ONE*, *5*, e9612.

Wisz M. S., Hijmans R. J., Li J., Peterson A. T., Graham C. H., Guisan A., Elith J., Dudík M., Ferrier S., Huettmann F., *et al.* (2008) Effects of sample size on the performance of species distribution models. *Diversity and Distributions*, *14*, 763–773.

Wisz M. S. & Guisan A. (2009) Do pseudo-absence selection strategies influence species distribution models and their predictions? An information-theoretic approach based on simulated data. *BMC Ecology*, *9*, 8.

Yang X. Q., Kushwaha S. P. S., Saran S., Xu J. & Roy P. S. (2013) Maxent modeling for predicting the potential distribution of medicinal plant, Justicia adhatoda L. in Lesser Himalayan foothills. *Ecological Engineering*, *51*, 83–87.

Zuhlke M., Fomferra N., Brockmann C., Peters M., Veci L., Malik J. & Regner P. (2015) SNAP (Sentinel Application Platform) and the ESA Sentinel 3 Toolbox. in *Sentinel-3 for Science Workshop, Proceedings of a workshop held 2-5 June, 2015 in Venice, Italy*. Paris: ESA, 21.

# 6 Discussion

## 6.1 Summary of key findings

This research represents the first detailed analysis of the rangeland system in Dhofar. Using interdisciplinary methods we examined the socio-ecological system with a focus on understanding the social drivers and ecological impacts of overstocking to inform local decision making and provide additional insights for rangeland science.

In Chapter 2 we applied a socio-ecological systems framework to structure and analyse our findings in the context of sustainability. In addition to documenting the modern pastoral system in Dhofar, our results revealed a unique combination of social processes influencing overstocking. A number of these, such as reduced mobility, human and livestock population growth and a loss of traditional knowledge are well-documented processes in pastoral societies that can lead to overuse of rangeland resources, while others are lesser known.

For example in Dhofar, livestock ownership is principally motivated by pastoral values embedded in modern cultural norms and many livestock owners are passionate about livestock keeping, despite the financial costs involved. Pastoral systems which have expanded primarily due to cultural traditions in the face of economic losses for pastoralists are rare. Nevertheless, attitudes towards livestock keeping are not uniform and some wealthier or better-educated families are giving up pastoral activities, indicating a change in cultural norms.

Unlike in Africa, livestock in Dhofar are not relied upon for subsistence lifestyles or regularly sold for profit. Nor are they accumulated as a response to unpredictable forage resource availability (Sandford, 1983; McPeak, 2005). Rather, keepers are reluctant to sell surplus animals (Peacock *et al.*, 2003) due to strong pastoral values, and as they provide an insurance strategy against unpredictable socio-economic events such as medical costs or loss of government employment.

Pertinent to the status quo of overstocking is the availability of household wealth for daily provisioning of feedstuffs which deems the price of local livestock uncompetitive against imported livestock, and maintains livestock populations beyond the carrying capacity of the rangelands. Subsequently, dependence on the rangelands is minimal, leaving little incentive for collective action or conservation. We identified

a) too many resource users b) in an unproductive system c) with undervalued resources, as key variables preventing self-organization in Dhofar (Ostrom, 2009).

In Chapter 3 we analysed the impacts of livestock browsing on the woody plant layer of the *Anogeissus* forests. Observed impacts included increased frequencies of unpalatable species, decreased plant density, reduced advanced growth, altered population age structures, and altered plant phytomorphology from management practises, bark stripping and browsing. Thus, our findings provide evidence to support previous claims that livestock are degrading the Dhofar Mountain ecosystems (Lamprey, 1976; Lawton, 1978; Oman Office of the Government Adviser for Conservation of the Environment, 1980; Wilson & MacLeod, 1991; Ghazanfar, 1998; Peacock *et al.*, 2003; Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005; Tardelli & Raffaelli, 2006; Hedges & Lawson, 2006; Directorate-General of Nature Conservation, 2010; El-Mahi, 2011b; Patzelt, 2012). In addition, we found that elevation alone does not provide the best explanation for species distributions; rather species are distributed along a gradient of moisture availability, as a result of the interplay between the mountain topography and the Khareef fog. We also found aspect influences vegetation due to fog exposure rather than solar exposure (Holland & Steyn, 1975), high rock cover may inhibit livestock browsing, and stocking rates decrease with increasing distance from anthropogenic features.

In Chapter 4 we identified seven variants of the *Anogeissus* forest within Jabal Qamar. Six were new and a seventh was previously described by Kürschner *et al.* (2004). A review of the literature and associated topoclimatic and disturbance factors suggested that the *Dodonaea viscosa* subs. *angustifolia* shrubland variant was a result of historical agricultural practises whilst the *Maytenus dhofarensis-Ficus sycomorus* sparse woodland was the result of historical deforestation. Within all variants long-term stocking rates prevail as the primary driver of vegetation change.

In Chapter 5 we employed a novel method to quantify long-term anthropogenic deforestation in the study area. Our results gave an estimated loss of 4363 hectares (17.1%) of continuous-canopy *Anogeissus* forest. We suggest that unforested areas at the core of the monsoon-influenced zone are the result of anthropogenic deforestation for pasture, timber, firewood and agriculture.

## 6.2 Contributions to rangeland science

This research represents a rare case study of camel, cattle and goat pastoralism in a drought-deciduous cloud forest rangeland. These unique ecological conditions in addition to atypical social, cultural and economic settings provide a new angle from which to synthesise additional insights into rangeland dynamics (Lynam & Stafford Smith, 2004; Sayre *et al.*, 2012) and the complex socio-ecological systems that govern rangeland use (Ostrom, 1990; Sayre *et al.*, 2012).

Firstly we want to draw attention to feedstuff provision in rangeland systems, which is considered a global sustainability issue (Godfray *et al.*, 2010; Herrero *et al.*, 2013; Mottet *et al.*, 2017). We have demonstrated how it can sustain livestock populations beyond the carrying capacity of the rangelands, but it also has major implications for the relevance and applicability of numerous rangeland theories and concepts which have been developed to explain rangeland use by pastoralist communities. Given the current trend of global livestock sector growth, we may see an increased use of feedstuffs amongst smallholder pastoralist communities.

Firstly, feedstuff provision can inhibit density-dependence of livestock, which is a major factor in defining equilibrium and non-equilibrium rangeland dynamics (Ellis & Swift, 1988). Secondly, it undermines the requirement for self-organization (Ellis & Swift, 1988; Ostrom, 1999), mobility (Scoones, 1995; Fratkin, 1997; Niamir-Fuller, 1999) and territoriality (Dyson-Hudson & Smith, 1978; Moritz, Scholte, *et al.*, 2013; Moritz, 2016) amongst pastoralist communities, themes which have received a great deal of attention in recent decades (Sayre, 2017). Thirdly, it can increase the cost of livestock ownership and the price of livestock, and thus may impact local and national livestock market sectors, as well as socioeconomic and political processes at multiple scales. Fourthly, it presents a new explanation for degradation in open-access equilibrium rangelands, besides the ‘tragedy of the commons’ scenario (Herskovits, 1926; Hardin, 1968; Lamprey, 1983; Moritz, Scholte, *et al.*, 2013; Moritz, 2016). Finally, it can result in rangeland degradation to an extent which is acknowledged by most local stakeholders, including pastoralists themselves, and thus leaves little doubt as to whether the rangelands are overstocked – a question which has challenged rangeland scientists for almost a century (Perevolotsky & Seligman, 1998; Sayre, 2017).

We also want to draw attention to the socio-cultural factors driving overstocking in Dhofar, which have wider relevance to future studies on pastoralism in Arabia where wealth, prestige, heritage and sports may be more important motivators of livestock ownership than economic gain. Such processes may be overlooked in rangeland studies due to western researcher pre-disposition to find economically rationale explanations for pastoralist decision making.

The long-debated concept of equilibrium and non-equilibrium rangeland dynamics remains crucial for both natural and social science approaches to understanding rangeland systems. Accordingly, there is a need for empirical studies to examine the theory in site-specific contexts (Sullivan & Rohde, 2002). Our evidence suggests that the Dhofar rangelands tend towards an equilibrium environment, with livestock as the principle driver of vegetation change. This is to some extent unsurprising given the number of previous claims that livestock are causing ecosystem degradation in Dhofar (Lamprey, 1976; Lawton, 1978; Oman Office of the Government Adviser for Conservation of the Environment, 1980; Wilson & MacLeod, 1991; Ghazanfar, 1998; Peacock *et al.*, 2003; Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD, 2005; Tardelli & Raffaelli, 2006; Hedges & Lawson, 2006; Directorate-General of Nature Conservation, 2010; El-Mahi, 2011b; Patzelt, 2012).

A major assumption of non-equilibrium rangeland dynamics is that livestock populations are regulated in a density-dependent manner by limited forage availability (Wiens, 1984; Briske, Fuhlendorf & Smeins, 2003; Vetter, 2005; Gillson & Timm Hoffman, 2007). This does not occur in Dhofar due to the provision of feedstuff which supports a high year-round livestock pressure, which is the principle driver of vegetation change and rangeland degradation. Feedstuff provision has facilitated degradation in other MENA nations (Blench, 1995; Masri, 2001).

Ellis & Swift (1988) determined that rangelands exhibit non-equilibrium dynamics when the coefficient of variation (CV) of rainfall is above 33%. Values for Dhofar range from CV 37% in Salalah (mean annual precipitation of 85 mm) to CV 58% in Qairoon Hairiti (mean annual precipitation of 236 mm) (Ghazanfar & Fisher, 1998). However, these values do not account for the Khareef fog which reliably contributes three times as much moisture via horizontal precipitation capture (Hildebrandt &

Eltahir, 2006). Thus, although mean annual rainfall in the Dhofar Mountains is low (250mm: arid-semi arid) and the CV of rainfall is greater than 33%, horizontal precipitation lowers annual variability and substantially increases net precipitation which reaches the ground. Thus the climate favours equilibrium dynamics, especially in areas of sufficient height and mass of woody cover for reliable horizontal precipitation (Hildebrandt & Eltahir, 2008). Subsequently, in grasslands, moisture availability may be a limiting factor, and the CV of rainfall may remain above 33%. Thus, we may be seeing heterogeneous rangeland dynamics across the landscape (Stafford Smith, 1996; Oba, Stenseth & Lusigi, 2000; Vetter, 2005).

Evidence suggests that formerly-forested grasslands in Dhofar are unable to readily recover to forest due to a reduced capacity for horizontal precipitation (Kürschner *et al.*, 2004; Hildebrandt & Eltahir, 2006, 2007, 2008; Friesen *et al.*, 2018). We found some evidence of forest recovery in sparse woodlands, wherein juveniles of some palatable species were tolerant of higher stocking rates and drier conditions. However, an empirical exploration of forest-grassland interactions was outside the scope of this research, which focused on wooded environments. Figure 6.1 shows the processes known to facilitate or inhibit forest-grassland or grassland-forest transitions in African savannahs, which is the most comparable biome to Dhofar (Huntley & Walker, 1982; Scholes & Archer, 1997; Jeltsch, Weber & Grimm, 2000; Sankaran *et al.*, 2005). With regards to grassland-forest transitions (forest recovery), woody plant encroachment could occur in Dhofar given the right conditions of low browsing pressure, high grazing pressure and a continued absence of fires, although cattle may still target seedlings after grasses have senesced (Scheffer *et al.*, 2001). A similar process is likely facilitating unpalatable woody species dominance. Grassland-forest transitions may also be facilitated by increased bud burst and seed production as a stress response (Huntley & Walker, 1982), seedling protection within an unpalatable herbaceous layer (Smit *et al.*, 2007), niche separation by root depth, and increased soil infiltrability due to varying soil physical properties (Scholes & Archer, 1997).

Niche separation by phenology (Scholes & Archer, 1997) may facilitate increases of several dominant tree species in Dhofar (e.g. *A. dhofarica* and *A. rubifolius*). They bud prior to the Khareef (Miller, Morris & Stuart-Smith, 1988) affording them a competitive advantage over grasses (Rutherford & Panagos, 1982) which initiate leaves sequentially and at staggered intervals over the growing season (Archer &

Tieszen, 1980). Models and empirical observations predict that wherever seasonality is strong and predictable this type of phenological niche separation, usually results in dominance by trees, and is thus considered an equilibrium model (Scholes & Archer, 1997).

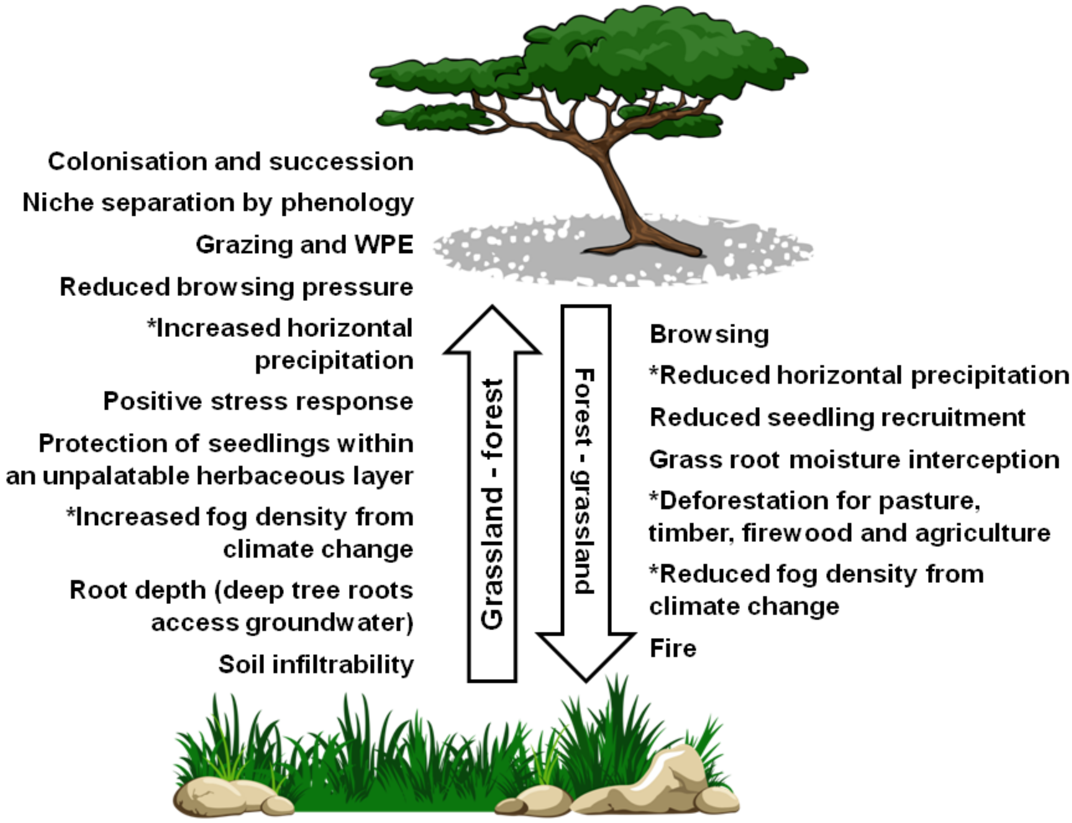



**Figure 6.1. Tree-grass interactions in savannahs, with additional processes for Dhofar marked with an asterisk.**

At present, there is insufficient evidence to challenge the findings of Hildebrandt & Eltahir (2006, 2007, 2008) and Friesen *et al.* (2018) which suggest that formerly-forested grasslands in Dhofar are unable to readily recover to forest due to a reduced capacity for horizontal precipitation. Indeed, the importance of the Khareef fog to vegetation communities was apparent in our results (Chapter 3). Forests and grasslands in Dhofar likely persist as alternative stable states (Holling, 1973) which violates a major assumption of equilibrium dynamics - that through internal regulation the vegetation will return to its pre-disturbance condition (Briske, Fuhlendorf & Smeins, 2003, 2005). Moreover, from a theoretical perspective, the ‘grassland state’ may be particularly stable and resistant to perturbations, as the forest-grassland transition results from both an internal (variable) alteration, in the form of deforestation or browsing, and an external (parameter) alteration, in the form of decreased horizontal precipitation (Beisner, Haydon & Cuddington, 2003).

Consequently, we suggest that grasslands in Dhofar are also at equilibrium. Indeed, the existence of alternative stable states provides poor justification for non-equilibrium dynamics. Not only were forest-grassland transitions mediated by anthropogenic rather than climatic processes, but livestock grazing pressure prevails as the principle driver of vegetation change in grasslands (Patzelt, 2011). Furthermore, grasslands, perhaps even more so than forests, exhibit equilibrium properties such as a greater capacity for internal regulation and reversibility of change, or recovery (Briske, Fuhlendorf & Smeins, 2003). Finally, it is likely that grasslands have persisted for prolonged periods in the monsoon-influenced zones of Dhofar (deforested areas suitable for many species in Chapter 5), certainly prior to the post-1970s boom in livestock numbers, which further supports their persistence as a stable ecosystem.

## 6.3 Implications for conservation

We echo the recommendations of previous authors that rangeland studies should be conducted on a case-by-case basis with an appreciation for the unique and distinct social and ecological processes that occur within and between rangelands. This research and that of other authors (Blench, 1995; Masri, 2001; Peacock *et al.*, 2003; Gallacher & Hill, 2006b, 2006a, 2008; Breulmann *et al.*, 2007; Gallacher, 2010; Louhaichi & Tastad, 2010) has illustrated the disparity between Arabian and African pastoral systems, the latter of which are more often the focus of rangeland studies. Thus, effort should be made to develop a region-specific understanding of Arabian rangelands where theories and concepts are developed from local case studies, in order to better inform management decisions. This research represents the first detailed analysis of pastoralism in Dhofar and thus holds substantial value to inform local decision making.

### 6.3.1 Social aspects

Our research has highlighted different attitudes, behaviours and socioeconomic circumstances amongst livestock keepers in Jabal Qamar (Section 2.5.3). Whilst many hold strong pastoral values and are passionate about livestock keeping, some wealthier households are losing interest whilst others feel peer-pressured into keeping livestock. Moreover, our evidence suggests cultural norms are changing. Therefore future management decisions must appreciate the current variation in attitudes and behaviours towards livestock keeping at the household level, and anticipate further change in parallel with the development and modernisation of the Dhofar region.

No informants were aware of the importance of the *Anogeissus* forest for the local water economy (Friesen *et al.*, 2018) or the uniqueness of the Dhofar ecosystem on a global scale. We suggest young Omanis should be better educated about the local environment and the services it provides for human wellbeing. It would be interesting to run a research study to evaluate the effectiveness of environmental workshops on changing the attitudes and behaviours of young people.

Most livestock keepers are aware that overstocking is degrading the vegetation. Moreover, many livestock keepers explained that they are awaiting a solution from the government (Section 2.5.3). This is important as it means livestock keepers are open to new ideas and may be willing to adjust their management techniques. New management techniques will be more successful if they either directly or indirectly address the problems faced by livestock keepers which were identified in this study (Section 2.5.1).

When asked about a solution to overstocking, many livestock keepers explained that the price of feedstuff should be reduced, reliance on it should increase, and livestock should be kept for longer periods in fenced pens (Section 2.5.3). Greater effort is required to assess the feasibility of using locally-adapted crop species for feedstuff production in order to increase the sustainability of production and reduce the cost to consumers. Current fodder crops, alfalfa *Medicago sativa* and Rhodes grass *Chloris gayana* do not occur naturally and are not adapted to the prevailing conditions of drought, temperature and salinity, and require vast quantities of water. Peacock *et al.*, (2003) suggests indigenous forage species such as Buffel grass *Cenchrus ciliaris*, which is used as a fodder crop in other parts of the world, could be utilized.

It is promising to hear of a new government-driven initiative spearheaded by the Oman Food Investment Holding Co (OFIC), which seeks to establish a market for rural camel and cattle dairy products in Dhofar. This represents a step in the right direction, however, without an informed regulatory framework involving guidelines for milk suppliers, there is the potential that such a project, which for the first time places a market value on rural livestock, could foster further growth in rural livestock populations, and place greater pressure on rangeland resources.

There has been recent discussion by policy makers on the option of moving rural livestock to the northern desert slopes (*Nejd*) in Dhofar. Although this would relieve pressure in the monsoon-influenced zone, it would shift pressure to an area with a lower carrying capacity and limited groundwater resources, require significant infrastructural development and abolish the local pastoral culture. There would no longer be a control of fast-growing weeds around settlements, and the mountains would lose tourism potential, both camel-based, and from the disappearance of footpaths and trails in remote areas.

### 6.3.2 Ecological aspects

We have provided empirical evidence to support previous claims that overstocking is detrimental to the Dhofar mountain ecosystems (Chapter 3). Moreover, a number of impacts such as altered population age structures, increased frequency of unpalatable species and low advanced growth are indicative of long-term change in forest composition and structure, and thus highlight the urgency of the situation.

A number of species had small populations or low advanced growth (Section 3.4) and thus their populations should be monitored. The impact of branch bending on tree survivability should be assessed as over half of *Anogeissus dhofarica* trees had been subject to branch bending management practises (Table 3.4), which could pose a serious threat for this regionally endemic dominant forest species. An assessment of the chemical composition of the most bark-stripped tree species in Dhofar (Table 3.4) should be carried out to understand why livestock supplement their diet with bark of these species. Simultaneously, a review of the nutrient composition of livestock feedstuffs should be carried out. Although bark stripping is not a major threat to the vegetation, it is perceived as a substantial problem by pastoralists who often remove the teeth from their camels.

Using our results we can identify potentially vulnerable areas in Jabal Qamar. The Broad-leaved *Blepharispermum hirtum* variant shows a restricted range in Jabal Qamar, and Miller, Morris and Stuart-Smith (1988) suggested it once formed the dominant vegetation along the entire length of the mountain foothills. Its largest expanse is below Sarfait where a vehicle track which runs North to South down the escarpment has led to numerous camps and very high stocking rates (Appendix 19), and subsequently the habitat is much degraded. This area of livestock activity could be acting as a barrier to Arabian leopard movement and gene flow into and out of Yemen as a long-running camera trap survey has failed to record leopards between this area and the Yemen border (H Al Hikmani, pers. comm.).

The *Cadia purpurea-Olea europaea* forest variant is likely to have a restricted range across all Dhofar as it is dependent on high densities of fog which build up at the base of cliffs, which are unique to Jabal Qamar. Concerningly, we have found evidence that the high frequencies of unpalatable *Cadia purpurea* which dominates the understory may be a response to disturbance, as reported from Afromontane *Olea-Juniper* forests in Ethiopia (Aynekulu, Denich & Tsegaye, 2009; Aynekulu *et al.*, 2016; Giday *et al.*, 2018).

The most species-rich habitat in Jabal Qamar occurs below Agdorot and Sha'at at the eastern end of Jabal Qamar (Appendix 19). This area is vulnerable to increasing numbers of livestock due to a new vehicle track which has been built through this area. We strongly suggest no additional access routes should be constructed into previously inaccessible areas.

The quality, quantity and seasonal availability of forage resources and subsequent rangeland use often differs across a rangeland (Vetter, 2005). The piosphere model is applicable to our study area with high stocking rates and highly degraded rangelands close by, and low stocking rates and less degraded rangelands, further from settlements, camps, roads and tracks (Andrew, 1988). The same trend was observed for long-term deforestation. Several of our sites had been subject to low long-term stocking rates due to inaccessibility (for example sites 22, 23, 25 and 26, see Appendix 6) and thus persist in a reasonably intact condition (little phytomorphological damage, high plant densities and few unpalatable species). Therefore, these areas are best suited to evaluate the effects of non-livestock disturbance regimes, such as climate change, cyclone damage and tree pests on the *Anogeissus* forests.

There has been a recent effort to decree the date on which livestock can return to the southern escarpment following the Khareef, with the aim of allowing a rest period for the vegetation to reproduce and set seed. This would certainly be an improvement on current regimes and may well have positive effects on vegetation productivity, although it is unclear how different habitats and their structural layers might respond. Owing to its more rapid life cycle, herbaceous vegetation may show the most substantial increase in biomass, which could inhibit forest recovery through competition, or facilitate it through increased horizontal precipitation capture and protection of juvenile woody plants. A number of other processes might also occur (Figure 6.1). Improving our understanding of forest-grassland interactions in Dhofar is crucial, and would be best studied using plot-based or exclosure methods. Therefore, future research should aim to establish fenced exclosures and monitor rangeland succession, interaction and colonization dynamics over time. Such research could inform management questions, such as whether livestock removal from grasslands is a worthwhile forest restoration strategy, and could utilise the deforested areas we have identified in this study (Appendix 19).

Our novel methods to observe spatial variability in fog density (Appendix 8) and map long-term deforestation in Dhofar (Chapter 5) stand out as particularly useful tools for future research and conservation in Dhofar. The former will enable a better understanding of plant species ecology and distributions and could support current efforts to identify important plant areas in southern Arabia (Al-Abbasi *et al.*, 2010). The latter enables visualisation of the historical distribution of the *Anogeissus* forest and areas of deforestation, and thus could inform forest regeneration projects.

It must be recognised that this study was limited to Jabal Qamar and thus it may not be representative of all Dhofar. It is likely that similar attitudes and behaviours are present amongst pastoralists in Jabal Qara and Jabal Samhan however triangulation of key themes with communities or key informants would be important prior to any interventions in these mountain ranges. In addition, it is suspected that forest loss has been greater in Jabal Qara, with grasslands and unpalatable species-dominated shrublands dominating large areas. Larger human and livestock populations as well as a flatter terrain have likely facilitated larger-scale deforestation. Lower topographic complexity, mean annual precipitation and spatial variability in fog density, means that not all variants of the *Anogeissus* forest we identified are necessarily present in Jabal Qara.

## 6.4 Concept for sustainable livestock production in Jabal Qamar

In this section we propose a new concept, based on our findings, to integrate economically and environmentally sustainable livestock production into the future development of Jabal Qamar. In the following paragraphs we present a chronology of the key steps and highlight the benefits for sustainability and the linkages to our findings. Figure 6.2 summarises the range of incentives based on the FAO classification system of Garrett and Neves (2016). A short concept video can be viewed here ([https://streamable.com/lqemx](https://streamable.com/lqemx)) which shows a livestock farming zone with feedstuff storage and a dairy collection centre, and an example of a branded meat product.

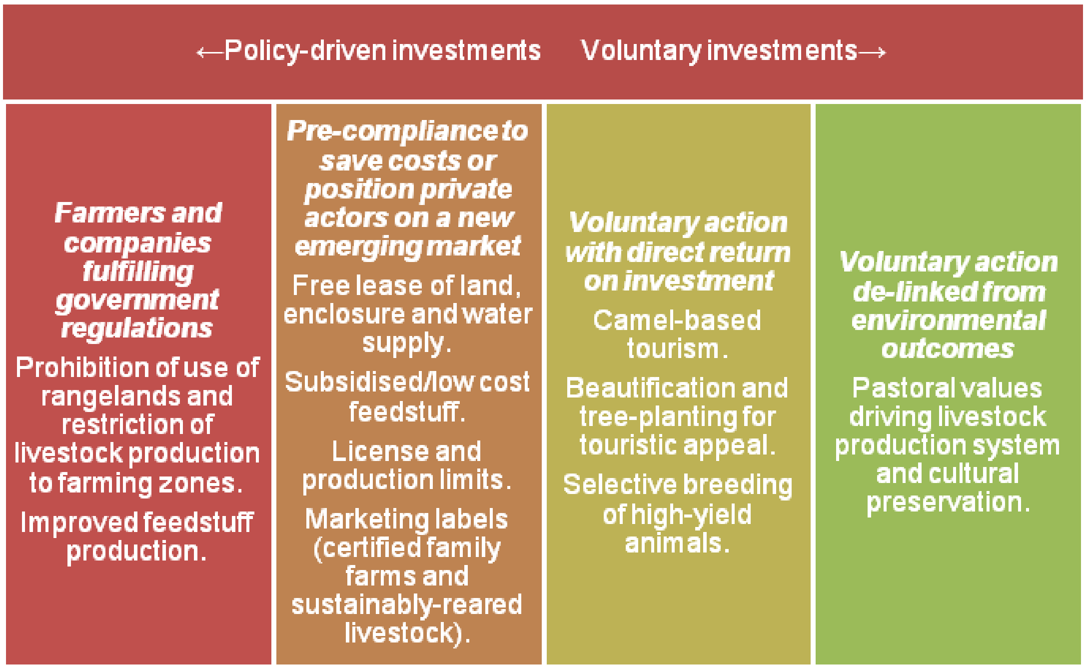



**Figure 6.2. Summary of concept incentives based on the classification system of Garrett and Neves (2016).**

The first step in our concept requires that livestock ownership becomes regulated through a license-based system. This would enable the number of livestock keepers and livestock to be monitored as there is currently no reliable system in place. Licensed livestock keepers would become part of a recognised, certified group of sustainable livestock producers, whom make revenue from livestock production but adhere to specific regulations. Individuals who are losing interest in pastoral activities, and who might choose not to become a licensed livestock producer, could sell their livestock without loss of face. Conversely, those who hold strong pastoral values and are passionate about livestock husbandry could continue to keep livestock, many of whom would benefit most from an income from livestock production.

Licensed livestock producers would freely lease a small plot of land (farm) in a livestock farming zone near to the main Salalah-Sarfait road. Each livestock farming zone would be local to each village. The farms should be leased with a fenced enclosure and a mains water supply (or water tank). The environment near the main road is suitable as it is already ecologically degraded, has low biodiversity value being on the fringe of the monsoon-influenced zone, and is close to pre-existing infrastructure.

Nowadays few livestock keepers herd their animals deep into the rangelands, and our results showed that many livestock keepers would be willing to keep their livestock in fenced enclosures if feedstuff was cheaper. Therefore, licensed livestock producers would be entitled to low-cost livestock feed which would be delivered weekly to the feed stores in the livestock farming zones. Livestock producers could also be entitled to low cost veterinary care.

To reduce the price of livestock feed for livestock keepers, feedstuff production in Dhofar should be made more efficient to reduce both its water requirements and its production costs. Current fodder crops, Alfalfa *Medicago sativa* and Rhodes grass *Chloris gayana*, are not adapted to the climate and require vast quantities of water. Indigenous forage species such as *Cenchrus ciliaris* could be utilized (Peacock *et al.*, 2003).

Licensed livestock producers should make revenue from sales of milk and livestock. They should be equipped to obtain milk from their livestock (e.g. milk churn) which they subsequently deposit at their local milk collection centre. The quantity and quality of the milk should be assessed and recorded by staff at the collection centre. Live animals should be regularly collected from each livestock farming zone. Production limits could be used to ensure equal revenue for producers. Livestock keepers are currently reluctant to sell livestock, so clear presentation of the benefits will be required to encourage sales.

This concept depends on investment in the rural livestock production system. Investment is required to; a) establish livestock farms; b) improve the efficiency of feedstuff production to reduce the consumer price; c) establish a system of meat and dairy collection, processing and packaging, and; d) develop a high-end market for Dhofari livestock products. Dhofar Cattle Feed Co. (SAOG) is the largest producer of feedstuffs and the largest producer of dairy products in Oman and is thus perfectly positioned to spearhead efforts. The Oman Food Investment Holding Co (OFIC) Milk Collection and Dairy Processing Project has yet to be launched and could be integrated. The meat and dairy products, branded as sustainably-reared Dhofari produce, should be sold as high-end products in Muscat and other Gulf nations. Consumers here perceive Dhofar as a rich and bountiful environment and over half a million Arab tourists visit Dhofar during the Khareef each year to escape high summer temperatures elsewhere in the Arabian Peninsula. Visitors often enjoy local meat at pop-up barbeques in mountain areas. Thus the Khareef, mountains and forests could be incorporated into an appealing brand. Rural livestock are currently a neglected resource but they have economic and nutritional value and should be sustainably integrated into Oman’s economy.

Grazing and browsing of natural vegetation should at first be strictly prohibited as many severely degraded areas require complete destocking. Livestock should then be gradually reintroduced according to appropriate sustainable stocking regimes within a broader strategic land use planning framework. Designing sustainable regimes which are accepted by local communities will be a challenge. The designs should incorporate conservation objectives such as wildlife corridors and protected areas as well as facilitate access, through livestock trails, to certain areas for recreation. Vegetation, soil and wildlife monitoring should take place throughout the process in different habitats to examine the conservation benefits and improve our local understanding of rangeland dynamics. Specific focus should be on rangeland recovery following removal of livestock and during prescribed stocking regimes. A research-implementation-research gap should be avoided.

The long term vision would be to beautify livestock farms and farming zones through tree-planting, with the additional ecological benefit of restoring native tree and shrub cover on the mountain plateau. The main road through Jabal Qamar is currently bordered by aesthetically unappealing gravel plains which have lost native tree and shrub cover.

The livestock farms would provide an excellent tourist attraction, especially as temperatures are cooler on the plateau than at lower elevations. There would be great potential for camel-based tourism in the form of petting, milking and camel rides which would provide additional revenue for licensed livestock producers, increase tourism in Jabal Qamar, and preserve and popularise Dhofari pastoral culture. Tourists to Dhofar are enchanted by camels and livestock producers would enjoy sharing their heritage with visitors. In addition, multi-day camel trekking trips, either north into the Nejd or south into the *Anogeissus* forests would likely prove popular. There is currently little infrastructure for trekking tourism in Dhofar.

## 6.5 Concluding remarks

This thesis provides the first detailed analysis of the socio-ecological system surrounding pastoralism in Dhofar. We found that available household wealth from non-livestock employment enables daily feedstuff provisioning which makes local livestock uncompetitive against imported livestock and maintains livestock populations beyond the carry capacity of the environment. Subsequently, the rangelands, which receive reliable precipitation, exhibit equilibrium properties, with overstocking impacting the composition and structure of the vegetation. Feedstuff provision is found to be a critical variable which deems many rangeland concepts inapplicable. Despite the expense, strong socio-cultural forces motivate livestock ownership, although some better-educated or wealthier individuals are losing interest. In addition, we identified seven variants of the *Anogeissus* forest which persist due to local variability in topoclimatic and anthropogenic factors, and we quantify long-term deforestation at 17.1 percent.

Our findings contribute valuable insights for rangeland science but demonstrate the need for an improved understanding of pastoralism and rangelands in Arabia, founded on case studies. In addition, by providing a robust overview of the local drivers and impacts of overstocking our research represents a useful resource to inform local decision making.

## 6.6 Literature cited


Al-Abbasi T. M., Al-Farhan A., Al-Khulaidi A. W., Hall M., Llewellyn O. A., Miller A. G. & Patzelt A. (2010) Important Plant Areas in the Arabian Peninsula. *Edinburgh Journal of Botany*, *67*, 25–35.

Andrew M. H. (1988) Grazing impact in relation to livestock watering points. *Trends in Ecology and Evolution*, *3*, 336–339.

Archer S. & Tieszen L. L. (1980) Growth and Physiological Responses of Tundra Plants to Defoliation. *Arctic and Alpine Research*, *12*, 531–552.

Aynekulu E., Aerts R., Denich M., Negussie A., Friis I., Demissew S. & Boehmer H. J. (2016) Plant diversity and regeneration in a disturbed isolated dry Afromontane forest in northern Ethiopia. *Folia Geobotanica*. Folia Geobotanica, *51*, 115–127.

Aynekulu E., Denich M. & Tsegaye D. (2009) Regeneration Response of Juniperus procera and Olea europaea subsp cuspidata to Exclosure in a Dry Afromontane Forest in Northern Ethiopia. *Mountain Research and Development*, *29*, 143–152.

Beisner B. E., Haydon D. T. & Cuddington K. (2003) Alternative stable states in ecology. *Frontiers in Ecology and the Environment*, *1*, 376–382.

Blench R. (1995) *Rangeland degradation and socio-economic changes among the Bedu of Jordan: results of the 1995 IFAD survey*. London: Pastoral Development Network & Overseas Development Institute.

Breulmann M., Böer B., Wernery U., Wernery R., Shaer H. El, Alhadrami G., Gallacher D., Peacock J., Chaudhary S. A., Brown G., *et al.* (2007) *The Camel from Tradition To Modern Times*. Doha: UNESCO.

Briske D. D., Fuhlendorf S. D. & Smeins F. E. (2003) Vegetation dynamics on rangelands: A critique of the current paradigms. *Journal of Applied Ecology*, *40*, 601–614.

Briske D. D., Fuhlendorf S. D. & Smeins F. E. (2005) State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. *Rangeland Ecology and Management*, *58*, 1–10.

Directorate-General of Nature Conservation (2010) *Fourth National Report To the Convention on Biological Diversity*. Muscat: Ministry of Environment and Climate Affairs.

Dyson-Hudson R. & Smith E. A. (1978) Human Territoriality: An Ecological Reassessment. *American Anthropologist*, *80*, 21–41.

El-Mahi A. T. (2011) Old ways in a changing space: the issue of camel pastoralism in Dhofar. *Sultan Qaboos University Journal - Agricultural and Marine Sciences*, *16*, 51–64.

Ellis J. E. & Swift D. M. (1988) Stability of African Pastoral Ecosystems: Alternate Paradigms and Implications for Development. *Journal of Range Management*, *41*, 450.

Fratkin E. (1997) Pastoralism: Governance and Development Issues. *Annual Review of Anthropology*, *26*, 235–261.

Friesen J., Zink M., Bawain A. & Müller T. (2018) Hydrometeorology of the Dhofar cloud forest and its implications for groundwater recharge. *Journal of Hydrology: Regional Studies*. Elsevier, *16*, 54–66.

Gallacher D. & Hill J. (2006) Effects of camel vs oryx and gazelle grazing on the plant ecology of the Dubai Desert Conservation Reserve. *Journal of arid environments*, *66*, 85–95.

Gallacher D. J. (2010) Arid rangeland degradation in an oil-rich Gulf state; inertia of perceived heritage and pro-agricultural policies. in Veress, B. and Szigethy, J. (eds) *Horizons in Earth Science Research*. New York: Nova Science Publishers, 335–350.

Gallacher D. J. & Hill J. P. (2006) Effects of camel grazing on the ecology of small perennial plants in the Dubai (UAE) inland desert. *Journal of Arid Environments*, *66*, 738–750.

Gallacher D. J. & Hill J. P. (2008) *Intensification of rangeland grazing in an oil-rich state; causes, consequences and possible solutions*. Dubai: Zayed University.

Garrett L. & Neves B. (2016) *Incentives for Ecosystem Services: Spectrum*. Rome: FAO.

Ghazanfar S. A. (1998) Status of the flora and plant conservation in the sultanate of Oman. *Biological Conservation*, *85*, 287–295.

Ghazanfar S. A. & Fisher M. (1998) *Vegetation of the Arabian Peninsula*. Dordrecht: Kluwer Academic Publishers.

Giday K., Humnessa B., Muys B., Taheri F. & Azadi H. (2018) Effects of livestock grazing on key vegetation attributes of a remnant forest reserve: The case of Desa'a Forest in northern Ethiopia. *Global Ecology and Conservation*, *14*, 1–14.

Gillson L. & Timm Hoffman M. (2007) Rangeland ecology in a changing world. *Science*, *315*, 53–54.

Godfray H. C. J., Beddington J. R., Crute I. R., Haddad L., Lawrence D., Muir J. F., Pretty J., Robinson S., Thomas S. M. & Toulmin C. (2010) Food Security: The Challenge of Feeding 9 Billion People. *Science*, *327*, 812–818.

Hardin G. (1968) The Tragedy of the Commons. *Science*, *162*, 1243–1248.

Hedges S. & Lawson D. (2006) *Monitoring the Illegal Killing of Elephants: Dung Survey Standards for the Mike Programme*. Nairobi: CITES MIKE Programme.

Herrero M., Havlik P., Valin H., Notenbaert A., Rufino M. C., Thornton P. K., Blummel M., Weiss F., Grace D. & Obersteiner M. (2013) Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences*, *110*, 20888–20893.

Herskovits M. J. (1926) The Cattle Complex in East Africa. *American Anthropologist*, *28*, 230–272.

Hildebrandt A. & Eltahir E. A. B. (2006) Forest on the edge: Seasonal cloud forest in Oman creates its own ecological niche. *Geophysical Research Letters*, *33*, 2–5.

Hildebrandt A. & Eltahir E. A. B. (2007) Ecohydrology of a seasonal cloud forest in Dhofar: 2. Role of clouds, soil type, and rooting depth in tree-grass competition. *Water Resources Research*, *43*, 1–13.

Hildebrandt A. & Eltahir E. A. B. (2008) Using a horizontal precipitation model to investigate the role of turbulent cloud deposition in survival of a seasonal cloud forest in Dhofar. *Journal of Geophysical Research*, *113*, 1–11.

Holland P. G. & Steyn D. G. (1975) Vegetational Responses to Latitudinal Variations in Slope Angle and Aspect. *Journal of Biogeography*, *2*, 179–183.

Holling C. S. (1973) Resilience and Stability of Ecological Systems. *Annual Review of Ecology and Systematics*, *4*, 1–23.

Huntley B. & Walker B. (1982) *Ecology of Tropical Savannas*. New York: Springer.

Jeltsch F., Weber G. E. & Grimm V. (2000) Ecological buffering mechanisms in savannas: A unifying theory of long-term tree-grass coexistence. *Plant Ecology*, *150*, 161–171.

Kürschner H., Hein P., Kilian N. & Hubaishan M. A. (2004) The Hybantho durae-Anogeissetum dhofaricae ass. nova - Phytosociology, structure and ecology of an endemic South Arabian forest community. *Phytocoenologia*, *34*, 569–612.

Lamprey H. F. (1976) *Preliminary Proposals for the Conservation and Management of the Jebel Qara Region*. Nairobi: UNESCO Regional Office of Science and Technology for Africa.

Lamprey H. F. (1983) Pastoralism yesterday and today: The overgrazing problem. in Bourlière, F. (ed.) *Tropical Savannas (Ecosystems of the World)*. Amsterdam: Elsevier, 643–666.

Lawton R. M. (1978) *A reconnaissance survey of the Jabal Qara grazing land ecosystem, with particular reference to the impact of development*. London: Overseas Development Agency, Land Resources Development Centre.

Louhaichi M. & Tastad A. (2010) The Syrian steppe: Past trends, current status, and future priorities. *Rangelands, 32*, 2–7.

Lynam T. J. P. & Stafford Smith M. (2004) Monitoring in a complex world - Seeking slow variables, a scaled focus, and speedier learning. *African Journal of Range and Forage Science*, *21*, 69–78.

Masri A. (2001) *Country pasture/forage resource profiles: Syria*. Rome: FAO.

McPeak J. (2005) Individual and collective rationality in pastoral production: Evidence from northern Kenya. *Human Ecology*, *33*, 171–197.

Miller A. G., Morris M. & Stuart-Smith S. (1988) *Plants of Dhofar, The Southern Region of Oman: Traditional, Economic, and Medicinal Uses*. Muscat: Office of the Adviser for Conservation of the Environment, Diwan of Royal Court.

Ministry of Regional Municipalities Environment and Water Resources & UNEP & UNCCD (2005) *National Action Programme to Combat Desertification in the Sultanate of Oman*. Muscat: Ministry of Regional Municipalities, Environment, and Water Resources.

Moritz M., Scholte P., Hamilton I. M. & Kari S. (2013) Open Access, Open Systems: Pastoral Management of Common-Pool Resources in the Chad Basin. *Human Ecology*, *41*, 351–365.

Moritz M. (2016) Open property regimes. *International Journal of the Commons*, *10*, 1–22.

Mottet A., de Haan C., Falcucci A., Tempio G., Opio C. & Gerber P. (2017) Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. *Global Food Security*, *14*, 1–8.

Niamir-Fuller M. (1999) *Managing Mobility in African Rangelands: The Legitimization of Transhumance*. Rugby: Intermediate Technology Publications.

Oba G., Stenseth N. C. & Lusigi W. J. (2000) New perspectives on sustainable grazing management in arid zones of sub-Saharan Africa. *BioScience*, *50*, 35–51.

Oman Office of the Government Adviser for Conservation of the Environment (1980) *The scientific results of the Oman flora and fauna survey 1977 (Dhofar), Journal of Oman Studies Special Report No. 2*. Muscat: Ministry of Information and Culture.

Ostrom E. (1990) *Governing the Commons: The Evolution of Institutions for Collective Action*. 14th edn. Cambridge: Cambridge University Press.

Ostrom E. (1999) Coping With Tragedies of the Commons. *Annual Review of Political Science*, *2*, 493–535.

Ostrom E. (2009) A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science*, *325*, 419–422.

Patzelt A. (2011) The Themeda Quadrivalvis Tall-Grass Savannah of Oman at the Crossroad Between Africa and Asia. *Edinburgh Journal of Botany*, *68*, 301–319.

Patzelt A. (2012) Notes on the Flora and Vegetation of Wadi Kharfut on Jabal Qamar, Oman. in Borrell, J. et al. (eds) *The British Exploring Society Empty Quarter Expedition 2012*. London: British Exploring Society, 94–100.

Peacock J. ., Ferguson M. ., Alhadrami G. A., McCann I. R., Al Hajoj A., Saleh A. & Karnik R. (2003) Conservation through utilization: a case study of the indigenous forage grasses of the Arabian Peninsula. *Journal of Arid Environments*, *54*, 15–28.

Perevolotsky A. & Seligman N. G. (1998) Role of Grazing in Mediterranean Rangeland Ecosystems. *BioScience*, *48*, 1007–1017.

Rutherford M. C. & Panagos M. D. (1982) Seasonal woody plant shoot growth in Burkea africana - Ochna pulchra savanna. *South African Journal of Botany*, *1*, 104–116.

Sandford S. (1983) *Management of pastoral development in the Third World*. Chichester: Wiley.

Sankaran M., Hanan N. P., Scholes R. J., Ratnam J., Augustine D. J., Cade B. S., Gignoux J., Higgins S. I., Roux X. Le, Ludwig F., *et al.* (2005) Determinants of woody encroachment and cover in African savannas. *Nature*, *438*, 846–849.

Sayre N. F., Debuys W., Bestelmeyer B. T. & Havstad K. M. (2012) ‘The range problem’ after a century of rangeland science: New research themes for altered landscapes. *Rangeland Ecology and Management*, *65*, 545–552.

Sayre N. F. (2017) *The Politics of Scale: A History of Rangeland Science*. Chicago: University of Chicago Press.

Scheffer M., Carpenter S., Foley J. A., Folke C. & Walker B. (2001) Catastrophic shifts in ecosystems. *Nature*, *413*, 591–596.

Scholes R. J. & Archer S. R. (1997) Tree-Grass Interactions in Savannas. *Annual Review of Ecology and Systematics*, *28*, 517–544.

Scoones I. (1995) *Living with Uncertainty: New Directions in Pastoral Development in Africa*. Rugby: Intermediate Technology Publications.

Smit C., Vandenberghe C., Den Ouden J. & Müller-Schärer H. (2007) Nurse plants, tree saplings and grazing pressure: Changes in facilitation along a biotic environmental gradient. *Oecologia*, *152*, 265–273.

Stafford Smith M. (1996) Management of Rangelands: Paradigms at their Limits. in Hodgson, J. and Illius, A. (eds) *The Ecology and Management of Grazing Systems*. Wallingford: CAB International, 325–357.

Sullivan S. & Rohde R. (2002) On non-eqilibrium in arid and semi-arid grazing systems. *Journal of Biogeography*, *29*, 1595–1618.

Tardelli M. & Raffaelli M. (2006) Some aspects of the vegetation of Dhofar (Southern Oman). *Bocconea*, *19*, 109–112.

Vetter S. (2005) Rangelands at equilibrium and non-equilibrium: Recent developments in the debate. *Journal of Arid Environments*, *62*, 321–341.

Wiens J. A. (1984) On Understanding a Non-Equilibrium World: Myth and Reality in Community Patterns and Processes. in Strong, D. R. et al. (eds) *Ecological Communities: Conceptual Issues and the Evidence*. Princeton: Princeton University Press, 439–457.

Wilson A. & MacLeod N. (1991) Overgrazing: Present or absent? *Journal of Range Management*, *44*, 475–482.

# 7 Appendices

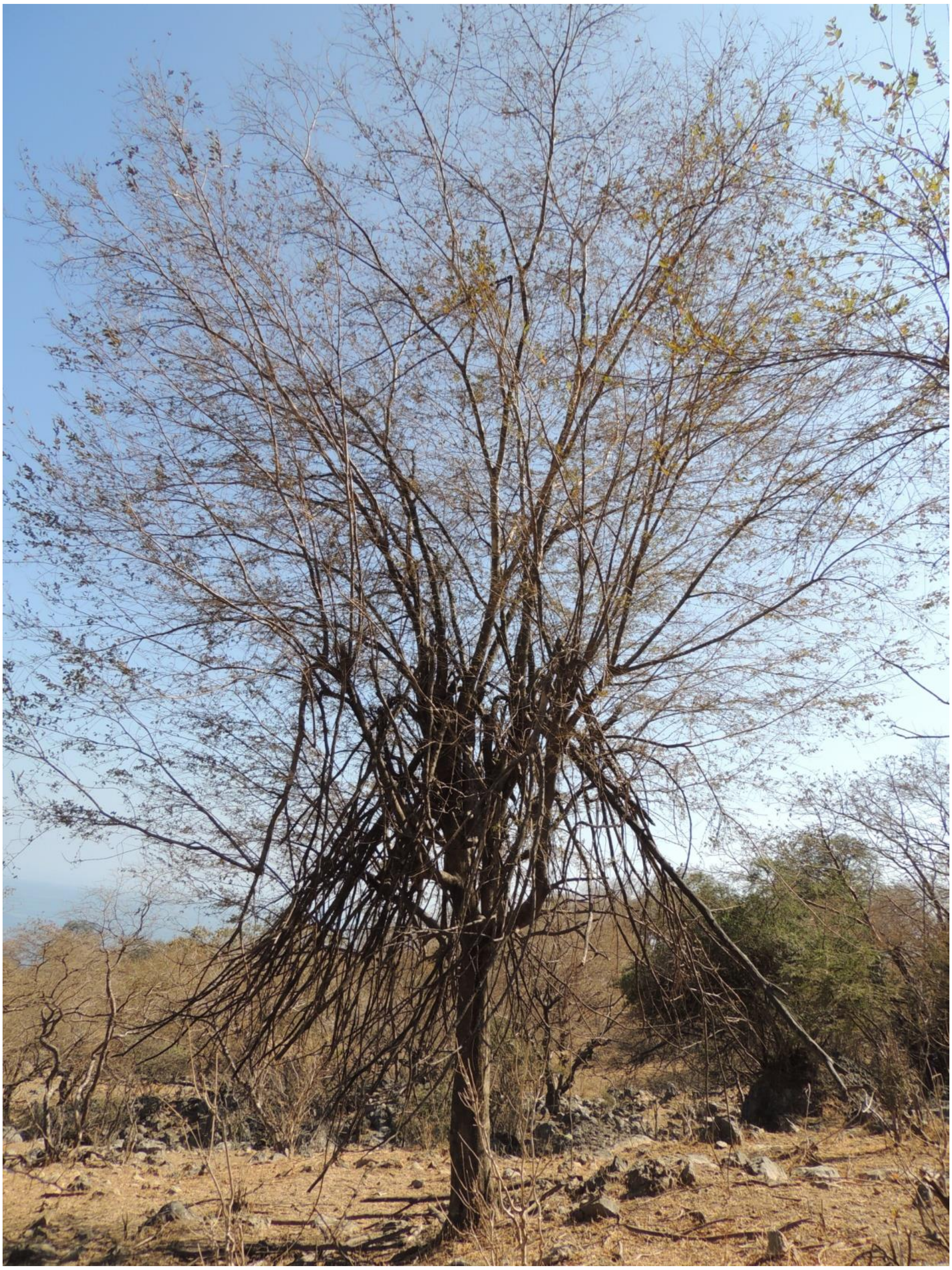
**Appendix 1. The results of the Likert scale showing proportions of responses for each level of agreement, where greener cells represent more responses.**

| | Statement | Strongly disagree | Disagree | Agree | Strongly agree | n= |
|---|---|---|---|---|---|---|
| 1 | I would like to have more livestock. | 8.1% | 19.5% | 52.8% | 19.5% | 123 |
| 2 | I would sell all of my livestock if I could get a good price. | 26.2% | 29.4% | 25.4% | 19.0% | 126 |
| 3 | No matter what, I will always keep livestock. | 8.9% | 19.4% | 45.2% | 26.6% | 124 |
| 4 | I have more livestock than I need. | 24.6% | 36.9% | 33.6% | 4.9% | 122 |
| 5 | My children want to keep livestock. | 16.7% | 19.2% | 45.0% | 19.2% | 120 |
| 6 | I spend less time with my animals compared to my father. | 20.3% | 30.1% | 34.1% | 15.4% | 123 |
| 7 | I would be happy to keep less livestock to protect the trees and grass. | 13.6% | 6.4% | 40.8% | 39.2% | 125 |
| 8 | We should protect the grass and trees for our children’s livestock animals to eat. | 5.0% | 12.5% | 35.0% | 47.5% | 120 |
| 9 | People cannot live in the mountains without grass and trees. | 7.3% | 13.7% | 39.5% | 39.5% | 124 |
| 10 | We should protect the grass and trees for the wildlife. | 3.4% | 1.7% | 39.5% | 55.5% | 119 |
| 11 | There should be a policy to limit how many livestock people can own. | 13.9% | 25.4% | 34.4% | 26.2% | 122 |
| 12 | I do not have enough time to look after my livestock. | 21.7% | 41.7% | 25.0% | 11.7% | 120 |
| 13 | If livestock feedstuff was cheaper I would keep my animals in one location. | 12.4% | 7.4% | 38.8% | 41.3% | 121 |
| 14 | The government should implement new laws to protect the environment from livestock grazing. | 9.8% | 14.6% | 48.8% | 26.8% | 123 |
| 15 | I know where my livestock go to graze. | 7.6% | 11.8% | 56.3% | 24.4% | 119 |
| 16 | I would enjoy breeding and selling livestock as a business. | 13.8% | 20.3% | 43.9% | 22.0% | 123 |
| 17 | In Dhofar people only eat Dhofari meat. | 6.7% | 9.2% | 34.5% | 49.6% | 119 |
| 18 | More roads should be built for livestock grazing. | 7.6% | 5.9% | 52.9% | 33.6% | 119 |
| 19 | Tourists will be less interested in visiting the Dhofar Mountains if there are less trees, grass and wildlife. | 8.9% | 11.4% | 30.9% | 48.8% | 123 |
| 20 | My animals regularly go away from my house for several days. | 15.7% | 28.1% | 43.8% | 12.4% | 121 |
| 21 | There are more livestock animals now than in 1999, before the government bought many animals. | 12.4% | 16.5% | 44.6% | 26.4% | 121 |
| 22 | I have noticed the area of wildlife has decreased in my lifetime. | 5.7% | 24.6% | 33.6% | 36.1% | 122 |
| 23 | I search for good places to graze my livestock. | 10.3% | 11.9% | 40.5% | 37.3% | 126 |

**Appendix 2. The dynamic conceptual framework (DCF) was constructed over the course of the fieldwork period to map themes and their interrelatedness.**

**Appendix 3. Bark stripping by camels on a large adult *Jatropha dhofarica* tree.**

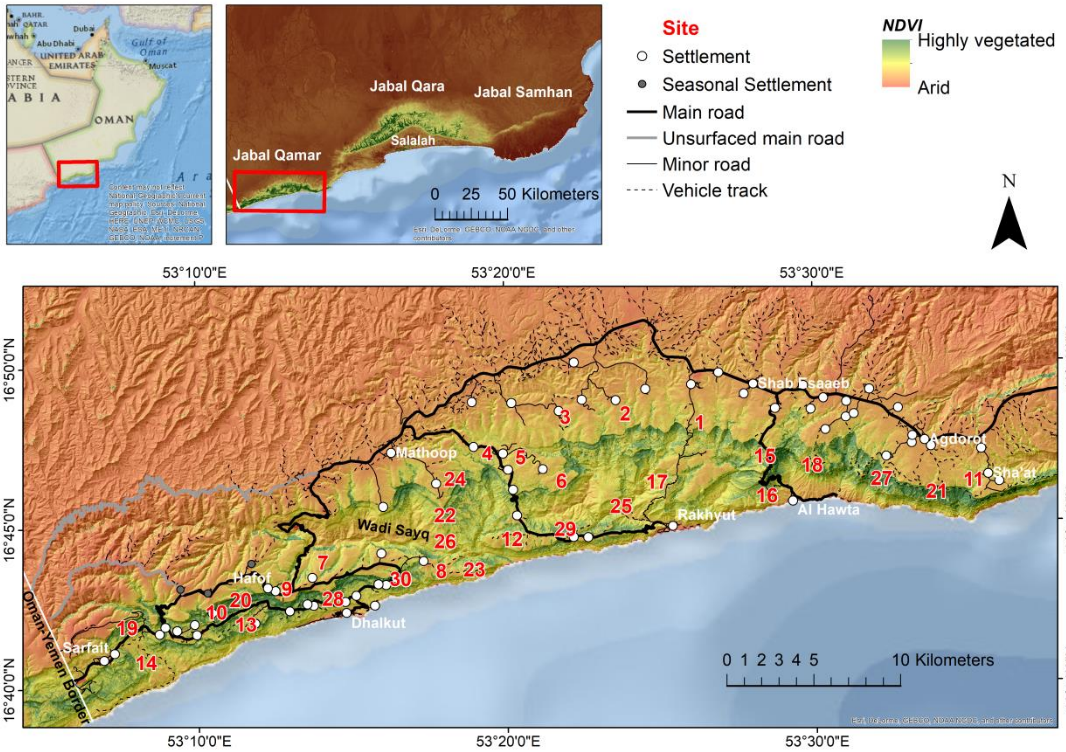
**Appendix 4. Branch bending practised on an *Anogeissus dhofarica* tree to enable livestock to reach the foliage. Fifty-seven percent of adult *A. dhofarica* trees (n=534) had been subject to branch bending.**



**Appendix 5. DRC thresholds for adults and juveniles, where diameters greater than or equal to the DRC threshold values are adults.**

| Woody species | DRC thresholds |
|---|---|
| Acacia gerrardii Benth. | >5 |
| Acacia senegal Willd. | >5 |
| Acridocarpus orientalis A. Juss | >4 or 2x >3 |
| Adenium obesum (Forssk.) Roem. & Schult. | 3x >5 or 2x >10 (at first fork) |
| Allophylus rubifolius (Hochst. ex A.Rich.) Engl. | >5 or 2x >4 |
| Anogeissus dhofarica A.J.Scott | >6 |
| Azima tetracantha Lam. | >4 or 2x >3 |
| Blepharis dhofarensis A.G.Mill. | >5 |
| Blepharispermum hirtum Oliv. | >5 or 2x >4 |
| Unidentified sp. | - |
| Boscia arabica Pestal. | >5 |
| Cadia purpurea (G.Piccioli) Aiton | >4 or 2x >3 |
| Calotropis procera (Aiton) Dryand. | >5 |
| Carissa spinarum L. | >5 |
| Caesalpinia erianthera Chiov. | >4 or 2x >3 |
| Commiphora gileadensis (L.) C.Chr. | >5 |
| Commiphora habessinica (O.Berg) Engl. | >5 |
| Cordia ovalis R. Br. | >3 |
| Cordia perrottetii Wight | >3 |
| Croton confertus Baker | >5 or 2x >4 or 3x >3 |
| Delonix elata (L.) Gamble | >5 |
| Dodonaea viscosa subsp. angustifolia (L.f.) J.G.West | >5 |
| Ehretia obtusifolia Hochst. ex A.DC. | >2 |
| Euclea racemosa subsp. schimperi (A.DC.) F.White | >5 or 2x >4 or 3x >3 |
| Euphorbia smithii S.Carter | >5 |
| Ficus sycomorus L. | >6 |
| Ficus vasta Forssk. | >6 |
| Flueggea virosa (Roxb. ex Willd.) Royle | >1 |
| Gomphocarpus fruticosus subsp. setosus (Forssk.) Goyder & Nicholas | >1 |
| Grewia bicolor Juss. | >3 |
| Grewia villosa Willd. | >3 |
| Hildebrandtia africana Vatke | >2 or 4x >1 |
| Hybanthus durus (Baker) O.Schwartz | >2 or 3x >1 |
| Jasminum grandiflorum L. | >3 |
| Jatropha dhofarica Radcl.-Sm. | >5 |
| Lawsonia inermis L. | >3 |
| Maytenus dhofarensis Sebsebe | >5 or 2x >4 or 3x >3 |
| Olea europaea subsp. cuspidata (Wall. & G.Don) Cif. | >5 |
| Pavetta longiflora Vahl | >5 |
| Premna resinosa (Hochst.) Schauer | >2 |
| Rhamnus staddo A.Rich. | >2 |
| Rhus somalensis Engl. | >5 or 2x >4 |
| Searsia pyroides (Burch.) Moffett | >2 or 3x >1 |
| Solanum incanum L. | >2 |
| Tamarindus indica L. | >6 |
| Woodfordia uniflora (A. Rich.) Koehne | >3 |
| Zygocarpum dhofarense (Hillc. & J.B.Gillett) Thulin & Lavin | >2 or 4x >1 |

**Appendix 6. Map of Jabal Qamar showing numbered vegetation sampling site locations. Two inset maps show the whole Dhofar Mountains and their location in Oman.**



**Appendix 7. Exhaustive list of variables with values for each site.**

| site | cluster | elevation (m a.s.l.) | elevation range (high/low) | landform class | geology (bedrock type) |
|---|---|---|---|---|---|
| 1 | E | 837.781 | high | high plateau | bioclastic limestone |
| 2 | E | 844.651 | high | high plateau | bioclastic limestone |
| 3 | E | 814.674 | high | high plateau | bioclastic limestone |
| 4 | E | 801.960 | high | high plateau | bioclastic limestone |
| 5 | E | 784.785 | high | high plateau | bioclastic limestone |
| 6 | E | 721.789 | high | high plateau | bioclastic limestone |
| 7 | E | 702.113 | high | high plateau | bioclastic limestone |
| 8 | D | 342.920 | low | flat lowland | bioclastic limestone |
| 9 | A | 761.111 | high | high plateau | bioclastic limestone |
| 10 | B | 734.072 | high | sub-plateau slopes | scree |
| 11 | E | 842.829 | high | high plateau | bioclastic limestone |
| 12 | A | 453.701 | low | flat lowland | bioclastic limestone |
| 13 | D | 431.604 | low | flat lowland | scree |
| 14 | F | 387.088 | low | flat lowland | scree |
| 15 | G | 700.303 | high | sub-plateau slopes | micritic limestone |
| 16 | G | 365.377 | low | lowland slopes | micritic limestone |
| 17 | G | 400.852 | low | lowland slopes | chalky dolomite |
| 18 | A | 704.294 | high | high plateau | micritic limestone |
| 19 | B | 822.693 | high | sub-plateau slopes | scree |
| 20 | B | 705.809 | high | sub-plateau slopes | scree |
| 21 | C | 749.035 | high | sub-plateau slopes | micritic limestone |
| 22 | G | 684.180 | high | sub-plateau slopes | micritic limestone |
| 23 | G | 367.462 | low | lowland slopes | bioclastic limestone |
| 24 | C | 811.437 | high | sub-plateau slopes | yellow-green marl |
| 25 | G | 315.380 | low | lowland slopes | bioclastic limestone |
| 26 | F | 335.469 | low | flat lowland | micritic limestone |
| 27 | C | 728.285 | high | sub-plateau slopes | micritic limestone |
| 28 | D | 340.906 | low | flat lowland | scree |
| 29 | G | 349.686 | low | lowland slopes | bioclastic limestone |
| 30 | G | 314.115 | low | lowland slopes | bioclastic limestone |

| site | fog exposure (arb. 0-1) | fog density (arb. 0-1) | heat load index (McCune and Keon, 2002) | solar radiation (3√kWH/m2) | slope (degree incline) |
|---|---|---|---|---|---|

| | | | | | |
|---|---|---|---|---|---|
| 1 | 0.530 | 0.412 | 1.062 | 122.239 | 7.388 |
| 2 | 0.940 | 0.401 | 1.063 | 122.412 | 10.327 |
| 3 | 0.490 | 0.344 | 1.074 | 122.123 | 9.522 |
| 4 | 0.870 | 0.493 | 1.070 | 122.498 | 7.606 |
| 5 | 0.080 | 0.387 | 1.057 | 121.363 | 7.769 |
| 6 | 0.555 | 0.489 | 1.059 | 121.271 | 8.748 |
| 7 | 0.310 | 0.422 | 1.073 | 121.056 | 9.824 |
| 8 | 0.250 | 0.491 | 1.063 | 119.743 | 11.491 |
| 9 | 0.000 | 0.532 | 1.055 | 120.654 | 7.904 |
| 10 | 0.725 | 0.571 | 1.065 | 121.330 | 7.655 |
| 11 | 0.375 | 0.452 | 1.070 | 122.016 | 12.427 |
| 12 | 0.840 | 0.545 | 1.082 | 120.732 | 11.724 |
| 13 | 0.885 | 0.633 | 1.073 | 120.771 | 9.645 |
| 14 | 0.865 | 0.554 | 1.063 | 119.992 | 11.437 |
| 15 | 0.274 | 0.504 | 1.108 | 120.290 | 22.534 |
| 16 | 0.495 | 0.504 | 1.100 | 119.664 | 17.464 |
| 17 | 0.047 | 0.441 | 1.099 | 117.775 | 18.044 |
| 18 | 0.935 | 0.572 | 1.074 | 121.900 | 10.544 |
| 19 | 0.860 | 0.516 | 1.076 | 121.396 | 17.795 |
| 20 | 0.960 | 0.625 | 1.104 | 121.531 | 14.372 |
| 21 | 0.919 | 0.532 | 1.121 | 120.956 | 24.303 |
| 22 | 0.990 | 0.551 | 1.104 | 120.182 | 21.070 |
| 23 | 0.000 | 0.484 | 1.067 | 115.683 | 17.605 |
| 24 | 0.925 | 0.519 | 1.076 | 120.916 | 18.975 |
| 25 | 0.740 | 0.478 | 1.087 | 118.446 | 22.194 |
| 26 | 0.515 | 0.487 | 1.081 | 120.213 | 12.921 |
| 27 | 0.555 | 0.643 | 1.106 | 119.620 | 23.499 |
| 28 | 0.875 | 0.592 | 1.076 | 119.443 | 15.972 |
| 29 | 0.000 | 0.473 | 1.067 | 117.525 | 18.471 |
| 30 | 0.000 | 0.518 | 1.075 | 111.971 | 26.110 |

| site | compound topographic index (Gessler et al. 1995) | topographic radiation aspect index | slope aspect (Stage, 1976) | linear aspect | terrain curvature (concavity/convexity) |
|---|---|---|---|---|---|
| 1 | 7.720 | 0.582 | 78.633 | 187.431 | 18229.538 |
| 2 | 7.722 | 0.743 | 88.967 | 152.368 | -11554.227 |
| 3 | 7.147 | 0.760 | 78.567 | 214.957 | -1490.585 |
| 4 | 6.937 | 0.805 | 95.000 | 170.130 | -8974.215 |
| 5 | 7.281 | 0.328 | 61.967 | 156.254 | 3999.790 |
| 6 | 7.081 | 0.363 | 65.567 | 132.523 | 11578.913 |
| 7 | 7.207 | 0.487 | 69.467 | 161.032 | 7375.861 |
| 8 | 6.800 | 0.480 | 76.433 | 131.344 | 8289.939 |
| 9 | 7.861 | 0.179 | 45.433 | 224.815 | 2391.643 |
| 10 | 6.510 | 0.490 | 80.567 | 139.340 | 9394.023 |
| 11 | 7.142 | 0.577 | 77.467 | 183.744 | 12364.252 |
| 12 | 6.593 | 0.826 | 96.067 | 193.696 | 20831.878 |
| 13 | 7.101 | 0.679 | 96.167 | 146.691 | 2993.756 |
| 14 | 6.870 | 0.485 | 88.300 | 119.231 | 17942.898 |
| 15 | 6.066 | 0.872 | 110.767 | 231.810 | -6708.696 |
| 16 | 5.989 | 0.838 | 107.333 | 224.524 | 16541.566 |
| 17 | 6.049 | 0.669 | 88.200 | 280.886 | 10858.121 |
| 18 | 6.976 | 0.729 | 99.767 | 168.047 | -6235.868 |
| 19 | 7.286 | 0.557 | 97.433 | 137.628 | 18367.994 |
| 20 | 6.995 | 0.775 | 121.600 | 154.311 | -3124.412 |
| 21 | 6.820 | 0.874 | 136.367 | 191.706 | 30162.662 |
| 22 | 6.153 | 0.649 | 127.700 | 144.707 | 13756.284 |
| 23 | 6.304 | 0.226 | 57.400 | 286.877 | 8274.574 |
| 24 | 6.475 | 0.512 | 106.533 | 114.592 | 11030.474 |
| 25 | 5.970 | 0.711 | 120.033 | 169.718 | 2541.163 |
| 26 | 6.448 | 0.762 | 107.167 | 179.072 | -657.636 |
| 27 | 6.603 | 0.832 | 105.733 | 254.602 | -9069.428 |

| | | | | | |
|---|---|---|---|---|---|
| 28 | 7.370 | 0.652 | 98.167 | 150.350 | -11919.049 |
| 29 | 6.743 | 0.391 | 81.567 | 183.450 | -8858.690 |
| 30 | 6.218 | 0.252 | 74.500 | 294.795 | 3528.259 |

| site | terrain roughness | terrain rugosity (arb. 0-1) | distance to road (km) | distance to road or vehicle track (km) | distance to house (km) |
|---|---|---|---|---|---|
| 1 | 17.949 | 0.499 | 0.763 | 0.613 | 2.318 |
| 2 | 20.436 | 0.499 | 1.234 | 1.234 | 1.219 |
| 3 | 18.452 | 0.498 | 0.589 | 0.434 | 0.750 |
| 4 | 11.173 | 0.495 | 0.479 | 0.479 | 0.559 |
| 5 | 11.837 | 0.505 | 0.406 | 0.330 | 0.863 |
| 6 | 14.378 | 0.507 | 1.016 | 1.016 | 1.134 |
| 7 | 20.376 | 0.504 | 0.909 | 0.909 | 0.902 |
| 8 | 28.432 | 0.498 | 1.451 | 0.158 | 1.612 |
| 9 | 11.879 | 0.510 | 0.286 | 0.283 | 0.577 |
| 10 | 14.314 | 0.493 | 1.808 | 0.442 | 0.708 |
| 11 | 31.997 | 0.503 | 0.428 | 0.428 | 0.958 |
| 12 | 27.359 | 0.505 | 1.852 | 0.516 | 2.932 |
| 13 | 18.816 | 0.496 | 0.460 | 0.194 | 0.777 |
| 14 | 24.234 | 0.504 | 2.718 | 0.574 | 3.960 |
| 15 | 105.483 | 0.501 | 0.791 | 0.791 | 4.559 |
| 16 | 58.609 | 0.504 | 1.120 | 0.642 | 5.112 |
| 17 | 64.127 | 0.500 | 2.262 | 2.262 | 8.192 |
| 18 | 20.848 | 0.499 | 3.448 | 0.168 | 3.384 |
| 19 | 72.746 | 0.501 | 2.011 | 2.011 | 2.654 |
| 20 | 44.312 | 0.503 | 2.280 | 2.188 | 3.031 |
| 21 | 138.635 | 0.496 | 4.036 | 0.534 | 4.136 |
| 22 | 91.942 | 0.503 | 2.645 | 2.642 | 2.753 |
| 23 | 64.359 | 0.500 | 4.489 | 1.058 | 4.651 |
| 24 | 89.067 | 0.499 | 1.720 | 1.718 | 1.723 |
| 25 | 104.009 | 0.500 | 3.993 | 2.533 | 5.386 |
| 26 | 33.710 | 0.499 | 6.213 | 5.486 | 6.312 |
| 27 | 160.926 | 0.500 | 2.893 | 1.140 | 4.275 |
| 28 | 59.308 | 0.487 | 1.287 | 1.225 | 2.152 |
| 29 | 68.612 | 0.491 | 2.752 | 2.330 | 5.745 |
| 30 | 152.914 | 0.499 | 1.140 | 1.126 | 1.796 |

| site | distance to camp (km) | distance to house or camp (km) | distance to waterpoint (km) | rock cover (%) | soil pH |
|---|---|---|---|---|---|
| 1 | 0.881 | 0.881 | 2.289 | 17.500 | 7.87 |
| 2 | 1.219 | 1.219 | 6.461 | 20.700 | 7.89 |
| 3 | 0.347 | 0.347 | 1.832 | 21.000 | 8.09 |
| 4 | 0.559 | 0.559 | 3.307 | 12.700 | 7.88 |
| 5 | 0.863 | 0.863 | 1.219 | 17.833 | 8.24 |
| 6 | 1.134 | 1.134 | 2.240 | 17.167 | 7.94 |
| 7 | 0.902 | 0.902 | 3.503 | 19.600 | 8.26 |
| 8 | 1.612 | 1.612 | 4.062 | 5.567 | 8.12 |
| 9 | 0.577 | 0.577 | 0.692 | 8.300 | 7.96 |
| 10 | 0.708 | 0.708 | 4.449 | 5.733 | 7.91 |
| 11 | 0.958 | 0.958 | 1.386 | 35.033 | 7.85 |
| 12 | 2.850 | 2.850 | 1.971 | 7.967 | 8.06 |
| 13 | 0.400 | 0.400 | 1.797 | 3.433 | 8.21 |
| 14 | 0.489 | 0.489 | 0.918 | 7.100 | 8.28 |
| 15 | 0.781 | 0.781 | 4.315 | 3.900 | 8.08 |
| 16 | 1.005 | 1.005 | 8.835 | 5.233 | 8.31 |
| 17 | 2.516 | 2.516 | 5.522 | 9.500 | 8.36 |
| 18 | 0.599 | 0.599 | 0.465 | 6.233 | 8.16 |
| 19 | 2.405 | 2.405 | 5.418 | 20.500 | 8.06 |
| 20 | 2.346 | 2.346 | 3.661 | 14.467 | 8 |
| 21 | 0.929 | 0.929 | 4.092 | 18.700 | 8.08 |
| 22 | 2.753 | 2.753 | 4.605 | 5.833 | 8.23 |

| | | | | | |
|---|---|---|---|---|---|
| 23 | 4.651 | 4.651 | 5.304 | 2.333 | 8.27 |
| 24 | 1.723 | 1.723 | 3.053 | 21.267 | 7.93 |
| 25 | 0.644 | 0.644 | 10.625 | 8.867 | 8.34 |
| 26 | 6.312 | 6.312 | 2.864 | 5.690 | 8.41 |
| 27 | 1.980 | 1.980 | 2.312 | 11.433 | 7.68 |
| 28 | 1.600 | 1.600 | 3.622 | 5.867 | 8.04 |
| 29 | 3.685 | 3.685 | 2.322 | 8.700 | 8.26 |
| 30 | 1.191 | 1.191 | 2.448 | 6.533 | 8.17 |

| site | canopy cover (%) | total adult basal area ($cm^2$) | average adult browsing damage (1-5 scale) | average juvenile browsing damage (1-5 scale) | average adult and juvenile browsing damage (1-10 scale) |
|---|---|---|---|---|---|
| 1 | 15.000 | 14848.0 | 4.877 | 4.674 | 9.551 |
| 2 | 14.500 | 17899.8 | 4.854 | 4.405 | 9.259 |
| 3 | 23.750 | 36212.9 | 4.927 | 4.891 | 9.818 |
| 4 | 34.250 | 34319.8 | 4.864 | 4.627 | 9.490 |
| 5 | 28.250 | 19982.2 | 4.823 | 4.767 | 9.589 |
| 6 | 31.250 | 35864.3 | 4.663 | 4.809 | 9.472 |
| 7 | 2.750 | 6991.3 | 4.771 | 4.678 | 9.449 |
| 8 | 6.000 | 84856.7 | 4.791 | 4.818 | 9.609 |
| 9 | 14.750 | 9323.7 | 5.000 | 4.576 | 9.576 |
| 10 | 61.500 | 50401.3 | 4.487 | 4.810 | 9.298 |
| 11 | 10.250 | 5118.9 | 4.495 | 4.382 | 8.877 |
| 12 | 41.750 | 18180.3 | 4.694 | 4.657 | 9.352 |
| 13 | 27.955 | 77977.9 | 4.941 | 4.804 | 9.745 |
| 14 | 35.682 | 37146.8 | 4.787 | 4.818 | 9.605 |
| 15 | 42.000 | 22406.7 | 4.484 | 4.330 | 8.814 |
| 16 | 65.000 | 27894.9 | 4.534 | 4.569 | 9.103 |
| 17 | 46.500 | 18448.0 | 4.376 | 4.487 | 8.863 |
| 18 | 19.750 | 10607.4 | 4.717 | 4.642 | 9.359 |
| 19 | 54.650 | 34780.1 | 4.729 | 4.739 | 9.468 |
| 20 | 72.400 | 48159.4 | 4.337 | 4.481 | 8.818 |
| 21 | 65.111 | 32816.4 | 3.778 | 4.102 | 7.880 |
| 22 | 67.600 | 12775.0 | 2.479 | 3.198 | 5.677 |
| 23 | 62.500 | 14338.9 | 3.581 | 4.127 | 7.708 |
| 24 | 46.667 | 13155.6 | 3.959 | 4.119 | 8.079 |
| 25 | 68.100 | 15305.3 | 2.820 | 3.103 | 5.923 |
| 26 | 42.000 | 16832.5 | 2.725 | 3.561 | 6.286 |
| 27 | 57.767 | 25262.0 | 2.764 | 3.157 | 5.922 |
| 28 | 45.034 | 38794.1 | 4.667 | 4.343 | 9.010 |
| 29 | 52.433 | 20372.0 | 4.039 | 4.167 | 8.206 |
| 30 | 62.000 | 30286.4 | 4.290 | 4.323 | 8.613 |

| site | average adult height | shannon adult diversity index | proportion of dead adults (%) | average adult point-plant distance | average adult DRC basal area ($cm^2$) |
|---|---|---|---|---|---|
| 1 | 242.617 | 1.890 | 10.830 | 4.631 | 123.733 |
| 2 | 290.658 | 1.491 | 10.000 | 4.215 | 149.165 |
| 3 | 402.400 | 1.576 | 10.000 | 5.728 | 301.774 |
| 4 | 403.800 | 1.819 | 13.330 | 5.132 | 285.998 |
| 5 | 332.058 | 1.553 | 13.330 | 5.388 | 166.518 |
| 6 | 433.225 | 1.602 | 14.170 | 6.284 | 298.869 |
| 7 | 196.408 | 1.763 | 3.330 | 2.919 | 58.261 |
| 8 | 309.525 | 2.159 | 6.670 | 26.841 | 707.139 |
| 9 | 223.508 | 1.176 | 13.330 | 3.935 | 77.697 |
| 10 | 490.294 | 2.076 | 15.830 | 3.491 | 420.011 |
| 11 | 136.433 | 1.839 | 6.670 | 2.333 | 42.761 |
| 12 | 267.558 | 1.895 | 15.830 | 3.619 | 151.503 |
| 13 | 360.517 | 2.135 | 10.830 | 9.366 | 649.816 |
| 14 | 364.092 | 1.759 | 8.330 | 4.489 | 309.557 |
| 15 | 369.125 | 2.500 | 5.830 | 2.986 | 187.937 |
| 16 | 441.883 | 2.024 | 10.000 | 3.134 | 232.458 |

| | | | | | |
|---|---|---|---|---|---|
| 17 | 428.958 | 2.297 | 12.500 | 2.761 | 153.733 |
| 18 | 216.150 | 1.708 | 8.330 | 3.523 | 88.395 |
| 19 | 398.748 | 1.858 | 16.670 | 3.104 | 289.834 |
| 20 | 505.242 | 2.541 | 18.330 | 3.474 | 401.329 |
| 21 | 431.317 | 2.931 | 5.830 | 3.771 | 273.646 |
| 22 | 366.475 | 2.310 | 0.830 | 2.063 | 106.459 |
| 23 | 402.542 | 2.282 | 8.330 | 2.629 | 125.966 |
| 24 | 295.958 | 2.547 | 15.830 | 2.857 | 109.630 |
| 25 | 465.417 | 2.088 | 17.500 | 2.564 | 127.544 |
| 26 | 354.458 | 1.702 | 2.500 | 3.219 | 140.270 |
| 27 | 395.567 | 2.710 | 5.830 | 3.028 | 210.516 |
| 28 | 394.350 | 1.965 | 13.330 | 4.707 | 323.285 |
| 29 | 390.883 | 2.484 | 8.330 | 2.852 | 169.766 |
| 30 | 447.167 | 2.518 | 3.330 | 3.594 | 252.387 |

| site | total adult bark stripped area ($cm^2$) | adult species richness | shannon juvenile diversity index | average juvenile height | average juvenile point-plant distance |
|---|---|---|---|---|---|
| 1 | 9900 | 12 | 2.098 | 60.508 | 2.481 |
| 2 | 43802 | 9 | 1.959 | 58.958 | 2.093 |
| 3 | 50176 | 8 | 1.826 | 50.083 | 2.249 |
| 4 | 26927 | 12 | 1.900 | 62.242 | 2.490 |
| 5 | 10039 | 8 | 1.958 | 51.183 | 2.482 |
| 6 | 6312 | 11 | 1.882 | 67.408 | 3.430 |
| 7 | 21 | 11 | 2.070 | 59.975 | 1.785 |
| 8 | 7860 | 12 | 2.020 | 43.242 | 11.290 |
| 9 | 1966 | 11 | 1.949 | 84.558 | 3.422 |
| 10 | 5992 | 11 | 2.061 | 77.667 | 1.773 |
| 11 | 6 | 12 | 2.016 | 45.325 | 1.048 |
| 12 | 3020 | 12 | 1.984 | 86.183 | 2.991 |
| 13 | 4344 | 15 | 2.203 | 43.100 | 5.233 |
| 14 | 23965 | 12 | 2.268 | 87.450 | 4.205 |
| 15 | 146 | 15 | 2.311 | 69.517 | 1.704 |
| 16 | 845 | 13 | 2.623 | 66.717 | 1.859 |
| 17 | 117 | 13 | 2.577 | 70.275 | 1.623 |
| 18 | 49 | 12 | 2.085 | 85.425 | 2.848 |
| 19 | 13673 | 14 | 2.286 | 86.125 | 2.068 |
| 20 | 1027 | 16 | 2.392 | 68.350 | 1.951 |
| 21 | 0 | 24 | 2.658 | 74.608 | 2.486 |
| 22 | 0 | 14 | 2.151 | 64.617 | 0.868 |
| 23 | 5400 | 16 | 2.149 | 92.642 | 1.302 |
| 24 | 320 | 17 | 2.912 | 65.408 | 1.748 |
| 25 | 340 | 15 | 2.590 | 70.517 | 1.370 |
| 26 | 100 | 14 | 2.260 | 73.175 | 1.822 |
| 27 | 122 | 22 | 2.517 | 80.342 | 1.833 |
| 28 | 6872 | 15 | 2.314 | 51.992 | 2.432 |
| 29 | 800 | 19 | 2.621 | 69.975 | 1.546 |
| 30 | 0 | 19 | 2.540 | 58.917 | 1.603 |

| site | average juvenile DRC basal area ($cm^2$) | juvenile species richness | adult density (individuals/hectare) | juvenile density (individuals/hectare) | Proportion adults with bent branches (%) |
|---|---|---|---|---|---|
| 1 | 5.874 | 14 | 326.919 | 1239.419 | 10.000 |
| 2 | 5.217 | 15 | 562.324 | 1965.466 | 31.579 |
| 3 | 2.908 | 15 | 276.632 | 1695.608 | 40.426 |
| 4 | 3.687 | 11 | 335.840 | 1378.587 | 37.705 |
| 5 | 4.909 | 12 | 294.298 | 1303.504 | 29.630 |
| 6 | 6.114 | 13 | 244.161 | 684.245 | 42.500 |
| 7 | 5.283 | 14 | 1104.149 | 2721.460 | 3.614 |
| 8 | 3.443 | 14 | 11.415 | 59.863 | 22.472 |
| 9 | 6.902 | 15 | 473.111 | 758.513 | 7.692 |
| 10 | 4.100 | 13 | 823.719 | 2775.252 | 15.278 |

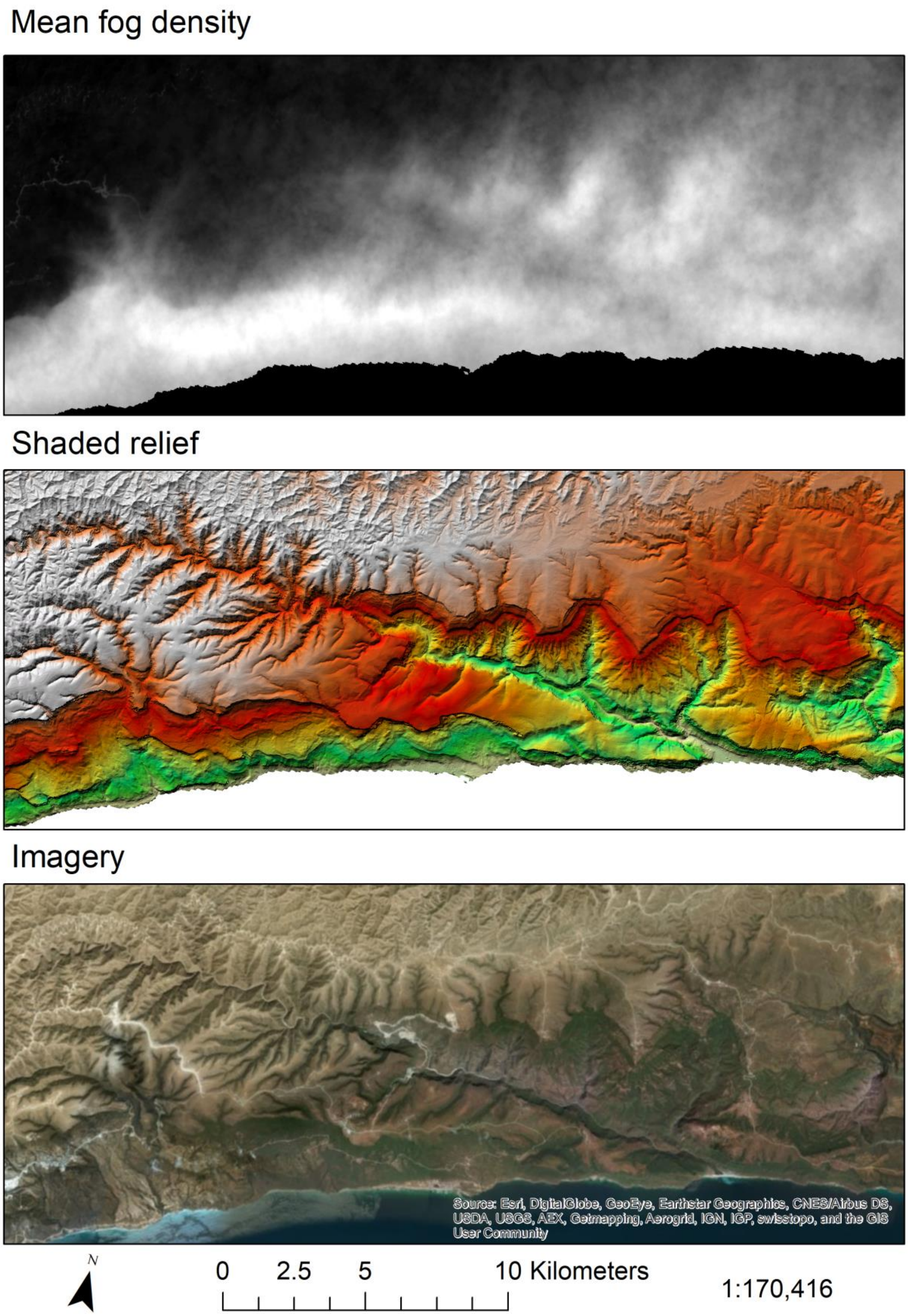
| | | | | | |
|---|---|---|---|---|---|
| 11 | 3.854 | 13 | 1908.775 | 8810.183 | 1.961 |
| 12 | 5.843 | 14 | 736.509 | 932.607 | 4.225 |
| 13 | 3.453 | 15 | 101.174 | 316.814 | 32.292 |
| 14 | 6.456 | 14 | 492.236 | 529.581 | 22.093 |
| 15 | 2.938 | 17 | 1150.606 | 3312.727 | 16.667 |
| 16 | 3.025 | 18 | 1063.396 | 2301.473 | 14.583 |
| 17 | 3.283 | 20 | 1345.155 | 3453.538 | 0.935 |
| 18 | 4.116 | 13 | 785.000 | 1121.718 | 5.769 |
| 19 | 4.480 | 15 | 1060.731 | 2371.170 | 32.258 |
| 20 | 2.094 | 18 | 844.676 | 2222.005 | 15.385 |
| 21 | 2.108 | 24 | 625.792 | 1366.062 | 5.714 |
| 22 | 1.595 | 13 | 2377.971 | 12774.960 | 1.739 |
| 23 | 1.695 | 14 | 1432.508 | 5376.387 | 1.000 |
| 24 | 2.446 | 23 | 1239.448 | 2673.281 | 2.985 |
| 25 | 1.706 | 22 | 1570.158 | 5098.399 | 1.786 |
| 26 | 3.388 | 14 | 974.763 | 2943.240 | 0.000 |
| 27 | 1.827 | 18 | 970.513 | 2495.539 | 7.339 |
| 28 | 2.843 | 17 | 402.345 | 1663.890 | 28.395 |
| 29 | 2.897 | 19 | 1254.006 | 3713.359 | 18.447 |
| 30 | 1.926 | 19 | 770.532 | 3494.741 | 14.151 |

| site | average adult broken branches (1-5 scale) | average adult bent branches (1-5 scale) | herbaceous species richness | bare ground cover (%) | grass cover (%) |
|---|---|---|---|---|---|
| 1 | 3.588 | 1.243 | 19 | 1.931 | 91.103 |
| 2 | 3.382 | 1.684 | 16 | 4.433 | 87.467 |
| 3 | 3.297 | 1.660 | 12 | 9.167 | 88.767 |
| 4 | 4.491 | 1.557 | 15 | 9.767 | 81.967 |
| 5 | 3.925 | 1.444 | 14 | 15.000 | 94.433 |
| 6 | 3.529 | 1.838 | 15 | 6.400 | 88.600 |
| 7 | 3.419 | 1.084 | 15 | 11.733 | 89.867 |
| 8 | 3.400 | 1.494 | 18 | 23.733 | 88.033 |
| 9 | 4.273 | 1.308 | 12 | 17.200 | 91.333 |
| 10 | 2.891 | 1.292 | 16 | 21.067 | 72.167 |
| 11 | 2.788 | 1.020 | 30 | 25.967 | 72.167 |
| 12 | 2.762 | 1.042 | 14 | 22.733 | 76.700 |
| 13 | 3.138 | 1.708 | 20 | 44.567 | 83.200 |
| 14 | 3.289 | 1.442 | 12 | 56.862 | 56.267 |
| 15 | 2.557 | 1.267 | 16 | 17.733 | 70.767 |
| 16 | 2.718 | 1.198 | 15 | 54.167 | 41.333 |
| 17 | 2.852 | 1.009 | 19 | 41.933 | 60.267 |
| 18 | 2.408 | 1.154 | 17 | 8.900 | 88.310 |
| 19 | 2.820 | 1.742 | 13 | 20.500 | 68.667 |
| 20 | 2.684 | 1.352 | 20 | 23.167 | 60.833 |
| 21 | 2.034 | 1.114 | 20 | 30.367 | 80.833 |
| 22 | 1.697 | 1.026 | 16 | 51.600 | 35.300 |
| 23 | 2.325 | 1.010 | 17 | 15.267 | 62.767 |
| 24 | 2.054 | 1.045 | 18 | 18.633 | 80.667 |
| 25 | 2.270 | 1.036 | 18 | 30.167 | 41.833 |
| 26 | 2.671 | 1.000 | 17 | 22.167 | 69.138 |
| 27 | 2.144 | 1.147 | 23 | 24.833 | 79.000 |
| 28 | 2.624 | 1.753 | 17 | 61.267 | 78.500 |
| 29 | 2.546 | 1.317 | 21 | 41.300 | 57.500 |
| 30 | 2.404 | 1.236 | 26 | 58.000 | 59.333 |

| site | herb cover (%) | current camel stocking rates (dung transect) (camels/hectar e) | current cattle stocking rates (dung transect) (cattle/hectare) | Long-term stocking rate (rank 0-30) | Long-term stocking rate (five-class scale) |
|---|---|---|---|---|---|
| 1 | 12.345 | 45.6 | 35.5 | 19 | high |
| 2 | 20.100 | 38 | 58.2 | 17 | medium |

| | | | | | |
|---|---|---|---|---|---|
| 3 | 15.233 | 90.8 | 20.5 | 25 | very high |
| 4 | 24.567 | 53.5 | 39.5 | 28 | very high |
| 5 | 9.433 | 94.8 | 51.3 | 26 | very high |
| 6 | 20.733 | 151.9 | 29.8 | 22 | high |
| 7 | 14.567 | 125.3 | 92.6 | 21 | high |
| 8 | 18.433 | 18.7 | 49.1 | 29 | very high |
| 9 | 14.933 | 44.5 | 75.9 | 27 | very high |
| 10 | 35.333 | 122.5 | 94.5 | 24 | high |
| 11 | 30.067 | 41.4 | 21.3 | 15 | medium |
| 12 | 26.900 | 59.3 | 49.1 | 13 | medium |
| 13 | 24.333 | 179.6 | 79.1 | 30 | very high |
| 14 | 46.567 | 376.1 | 97.3 | 23 | high |
| 15 | 36.100 | 53.3 | 8.7 | 9 | low |
| 16 | 61.833 | 103.9 | 14.1 | 12 | low |
| 17 | 43.333 | 42.5 | 1.4 | 5 | very low |
| 18 | 22.345 | 30.5 | 62.8 | 18 | medium |
| 19 | 34.000 | 144.8 | 50.5 | 20 | high |
| 20 | 42.667 | 101.8 | 59.4 | 14 | medium |
| 21 | 24.433 | 111.4 | 199.7 | 7 | low |
| 22 | 65.867 | 17 | 3.9 | 3 | very low |
| 23 | 44.667 | 23.5 | 14 | 4 | very low |
| 24 | 21.067 | 72.8 | 89.4 | 11 | low |
| 25 | 64.500 | 25 | 0 | 2 | very low |
| 26 | 38.966 | 2 | 0 | 1 | very low |
| 27 | 24.633 | 37.5 | 123.7 | 6 | very low |
| 28 | 22.500 | 102.7 | 29.2 | 16 | medium |
| 29 | 48.167 | 30.3 | 20.9 | 8 | low |
| 30 | 44.138 | 47.4 | 58.6 | 10 | low |

**Appendix 8. Map showing the layer of mean fog density used in the multivariate analysis, accompanied by a shaded relief and an aerial imagery map of the same area. One can see how the topography interacts with the fog.**



**Appendix 9. List of all recorded woody species with endemism status (N = not endemic, RE = regional endemic (south Arabian mountains), E = endemic to Dhofar) and IUCN Red List status (CR = critically endangered, EN = endangered, VU = vulnerable, NT = near threatened, LC = least concern, DD = data deficient, NE = Not evaluated). Total counts for the study area and advanced growth is shown.**

| Woody species | Endemism | Red List | Adult | Juvenile | Advanced growth | Total |
|---|---|---|---|---|---|---|
| Commiphora habessinica (O.Berg) Engl. | N | NE | 474 | 403 | 85% | 877 |
| Jatropha dhofarica Radcl.-Sm. | RE | NE | 371 | 363 | 98% | 734 |
| Anogeissus dhofarica A.J.Scott | RE | VU | 534 | 125 | 23% | 659 |
| Zygocarpum dhofarense (Hillc. & J.B.Gillett) Thulin & Lavin | RE | VU | 179 | 426 | 238% | 605 |
| Dodonaea viscosa subsp. angustifolia (L.f.) J.G.West | N | NE | 274 | 151 | 55% | 425 |
| Solanum incanum L. | N | NE | 119 | 302 | 254% | 421 |
| Maytenus dhofarensis Sebsebe | RE | NT | 153 | 210 | 137% | 363 |
| Commiphora gileadensis (L.) C.Chr. | N | NE | 194 | 146 | 75% | 340 |
| Blepharispermum hirtum Oliv. | RE | VU | 214 | 117 | 55% | 331 |
| Hybanthus durus (Baker) O.Schwartz | N | NE | 90 | 232 | 258% | 322 |
| Euphorbia smithii S.Carter | RE | NT | 223 | 95 | 43% | 318 |
| Allophylus rubifolius (Hochst. ex A.Rich.) Engl. | N | NE | 155 | 153 | 99% | 308 |
| Acacia senegal Willd. | N | NE | 44 | 220 | 500% | 264 |
| Cadia purpurea (G.Piccioli) Aiton | N | NE | 108 | 154 | 143% | 262 |
| Croton confertus Baker | N | NE | 89 | 39 | 44% | 128 |
| Euclea racemosa subsp. schimperi (A.DC.) F.White | N | NE | 60 | 47 | 78% | 107 |
| Blepharis dhofarensis A.G.Mill. | RE | VU | 9 | 89 | 989% | 98 |
| Jasminum grandiflorum L. | N | NE | 48 | 46 | 96% | 94 |
| Olea europaea subsp. cuspidata (Wall. & G.Don) Cif. | N | NE | 86 | 6 | 7% | 92 |
| Premna resinosa (Hochst.) Schauer | N | NE | 27 | 55 | 204% | 82 |
| Adenium obesum (Forssk.) Roem. & Schult. | N | NE | 37 | 16 | 43% | 53 |
| Acacia gerrardii Benth. | N | NE | 18 | 28 | 156% | 46 |
| Flueggea virosa (Roxb. ex Willd.) Royle | N | NE | 5 | 40 | 800% | 45 |
| Rhus somalensis Engl. | N | NE | 10 | 16 | 160% | 26 |
| Carissa spinarum L. | N | NE | 3 | 22 | 733% | 25 |
| Ficus vasta Forssk. | N | NE | 19 | 3 | 16% | 22 |
| Pavetta longiflora Vahl | RE | NE | 3 | 17 | 567% | 20 |
| Woodfordia uniflora (A. Rich.) Koehne | N | NE | 4 | 15 | 375% | 19 |
| Boscia arabica Pestal. | RE | VU | 2 | 14 | 700% | 16 |
| Grewia bicolor Juss. | N | NE | 2 | 13 | 650% | 15 |
| Delonix elata (L.) Gamble | N | LC | 13 | 2 | 15% | 15 |
| Tamarindus indica L. | N | LC | 11 | 3 | 27% | 14 |
| Gomphocarpus fruticosus subsp. setosus (Forssk.) Goyder & Nicholas | N | NE | 6 | 6 | 100% | 12 |
| Cordia ovalis R. Br. | N | NE | 1 | 8 | 800% | 9 |
| Ehretia obtusifolia Hochst. ex A.DC. | N | NE | 1 | 4 | 400% | 5 |
| Ficus sycomorus L. | N | NE | 5 | - | NA | 5 |
| Searsia pyroides (Burch.) Moffett | N | NE | 2 | 2 | 100% | 4 |
| Calotropis procera (Aiton) Dryand. | N | NE | 3 | - | NA | 3 |
| Caesalpinia erianthera Chiov. | N | NE | - | 3 | NA | 3 |
| Lawsonia inermis L. | N | NE | - | 3 | NA | 3 |
| Azima tetracantha Lam. | N | NE | 1 | 1 | 100% | 2 |
| Cordia perrottetii Wight | N | NE | 1 | 1 | 100% | 2 |
| Hildebrandtia africana Vatke | N | NE | - | 2 | NA | 2 |
| Acridocarpus orientalis A. Juss | N | NE | 1 | - | NA | 1 |
| Rhamnus staddo A.Rich. | N | NE | 1 | - | NA | 1 |
| Unidentified sp. | - | - | - | 1 | NA | 1 |
| Grewia villosa Willd. | N | NE | - | 1 | NA | 1 |
| | | Total | 3600 | 3600 | | 7200 |

**Appendix 10. Woody species count data for each site.**

| **Site number** | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
|---|---|---|---|---|---|---|---|---|---|---|
| | Count of individuals (A = Adult, J = Juvenille) | | | | | | | | | |
| **Woody species** | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J |
| Commiphora habessinica (O.Berg) Engl. | 23, 14 | 4, 4 | 7, 3 | 7, 20 | 21, 18 | 16, 13 | 44, 33 | 29, 16 | 3, 7 | -, 7 |
| Jatropha dhofarica Radcl.-Sm. | 30, 23 | 61, 35 | 48, 43 | 36, 27 | 45, 41 | 15, 25 | 34, 26 | 3, 2 | 8, 11 | 4, 3 |
| Anogeissus dhofarica A.J.Scott | 26, 2 | 24, - | 36, 2 | 42, - | 35, - | 60, 3 | 6, 4 | 21, 18 | 3, 4 | 7, 1 |
| Zygocarpum dhofarense (Hillc. & J.B.Gillett) Thulin & Lavin | 21, 36 | 14, 40 | -, 28 | 7, 32 | 1, 16 | 1, 43 | 4, 20 | -, - | 1, 3 | -, 13 |
| Dodonaea viscosa subsp. angustifolia (L.f.) J.G.West | -, - | -, - | -, - | 1, - | -, - | -, - | -, - | 10, 6 | 87, 51 | -, - |
| Solanum incanum L. | 8, 5 | 5, 7 | 4, 22 | 7, 16 | -, 17 | 3, 3 | 2, 4 | 15, 35 | 5, 22 | 5, 13 |
| Maytenus dhofarensis Sebsebe | 2, 14 | 1, 10 | -, 1 | -, 12 | -, 2 | -, 7 | -, 2 | 14, 6 | -, 5 | 3, 8 |
| Commiphora gileadensis (L.) C.Chr. | -, 3 | -, 1 | 13, - | 8, - | 6, - | 13, 1 | 10, 9 | 11, 24 | 4, 1 | 2, 5 |
| Blepharispermum hirtum Oliv. | -, - | -, - | 2, 1 | -, - | -, - | -, - | -, - | -, 1 | -, - | -, - |
| Hybanthus durus (Baker) O.Schwartz | -, 2 | -, 4 | 1, 10 | -, 4 | -, 9 | 1, 1 | 5, 11 | -, - | -, - | 4, 11 |
| Euphorbia smithii S.Carter | -, 1 | -, 5 | -, 1 | -, 1 | -, - | 1, 2 | 11, 3 | 1, - | 3, 2 | 21, 2 |
| Allophylus rubifolius (Hochst. ex A.Rich.) Engl. | 2, 7 | -, 3 | -, 1 | 2, 4 | -, 5 | 2, 17 | -, - | 8, 6 | -, - | 1, 2 |
| Acacia senegal Willd. | -, - | -, 2 | -, 1 | -, - | 2, 2 | -, - | 1, 2 | -, 1 | -, - | -, 4 |
| Cadia purpurea (G.Piccioli) Aiton | -, - | -, - | -, - | -, 1 | -, - | -, - | -, - | -, - | 2, 2 | 27, 46 |
| Croton confertus Baker | -, - | -, 1 | -, 1 | -, - | -, 1 | -, - | -, - | -, - | -, - | 10, 5 |
| Euclea racemosa subsp. schimperi (A.DC.) F.White | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Blepharis dhofarensis A.G.Mill. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Jasminum grandiflorum L. | 1, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, 4 | 4, - |
| Olea europaea subsp. cuspidata (Wall. & G.Don) Cif. | 4, - | 2, - | -, - | 5, - | -, - | -, - | -, - | -, - | 3, - | 31, - |
| Premna resinosa (Hochst.) Schauer | -, - | -, - | -, 1 | -, - | -, - | -, 1 | -, 2 | 2, - | -, - | -, - |
| Adenium obesum (Forssk.) Roem. & Schult. | 1, 3 | 8, 4 | 8, - | 3, 1 | 6, 2 | 7, 3 | 1, - | -, - | -, - | -, - |
| Acacia gerrardii Benth. | -, - | 1, 1 | 1, 3 | -, - | 4, 6 | -, - | 2, 2 | -, - | -, - | -, - |
| Flueggea virosa (Roxb. ex Willd.) Royle | -, - | -, - | -, - | -, - | -, - | -, - | -, 1 | -, 1 | -, 1 | -, - |
| Rhus somalensis Engl. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, 4 | -, - |
| Carissa spinarum L. | -, 8 | -, - | -, - | 1, 2 | -, - | -, - | -, - | -, - | -, 2 | -, - |
| Ficus vasta Forssk. | 1, - | -, - | -, - | 1, - | -, - | 1, - | -, - | 1, 1 | -, - | 1, - |
| Pavetta longiflora Vahl | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Woodfordia uniflora (A. Rich.) Koehne | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, 1 | -, - | -, - |
| Boscia arabica Pestal. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Grewia bicolor Juss. | -, - | -, 2 | -, 2 | -, - | -, 1 | -, - | -, - | -, - | -, - | -, - |
| Delonix elata (L.) Gamble | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Tamarindus indica L. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Gomphocarpus fruticosus subsp. setosus (Forssk.) Goyder & Nicholas | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | 1, 1 | -, - |
| Cordia ovalis R. Br. | -, - | -, 1 | -, - | -, - | -, - | -, 1 | -, - | -, 2 | -, - | -, - |
| Ehretia obtusifolia Hochst. ex A.DC. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Ficus sycomorus L. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | 4, - | -, - | -, - |
| Searsia pyroides (Burch.) Moffett | 1, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Calotropis procera (Aiton) Dryand. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | 1, - | -, - | -, - |
| Caesalpinia erianthera Chiov. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Lawsonia inermis L. | -, 1 | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Azima tetracantha Lam. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Cordia perrottetii Wight | -, 1 | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Hildebrandtia africana Vatke | -, - | -, - | -, - | -, - | -, - | -, - | -, 1 | -, - | -, - | -, - |
| Acridocarpus orientalis A. Juss | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Rhamnus staddo A.Rich. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Unidentified sp. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Grewia villosa Willd. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| **Site number** | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 |

| | Count of individuals (A = Adult, J = Juvenille) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Woody species** | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J |
| Commiphora habessinica (O.Berg) Engl. | 59, 22 | 38, 37 | 8, 4 | 1, 3 | 12, 2 | 17, 17 | 29, 19 | 10, 5 | 1, 3 | 7, 7 |
| Jatropha dhofarica Radcl.-Sm. | 9, 37 | 5, 3 | 5, 5 | 14, 9 | 10, 16 | 4, 7 | 6, 3 | -, - | 1, 3 | -, 1 |
| Anogeissus dhofarica A.J.Scott | 7, - | 6, 1 | 20, 26 | 25, 10 | 16, 1 | 44, 7 | 16, 6 | 4, - | 12, 2 | 10, - |
| Zygocarpum dhofarense (Hillc. & J.B.Gillett) Thulin & Lavin | 12, 16 | 1, 1 | 2, 8 | -, 2 | 12, 21 | 2, 15 | 6, 12 | 3, 9 | 2, 13 | 7, 14 |
| Dodonaea viscosa subsp. angustifolia (L.f.) J.G.West | -, - | 35, 33 | 7, - | -, - | 2, 1 | -, - | -, - | 61, 44 | -, - | -, - |
| Solanum incanum L. | 6, 6 | 6, 14 | 5, 21 | 5, 10 | 10, 12 | 3, 4 | -, - | 5, 9 | 1, 15 | 1, 6 |
| Maytenus dhofarensis Sebsebe | 1, 9 | 1, 9 | 39, 24 | -, 4 | 1, 7 | 3, 5 | -, 1 | 3, 4 | 4, 10 | 11, 10 |
| Commiphora gileadensis (L.) C.Chr. | 5, 6 | 7, 5 | -, 3 | 1, 2 | 6, 2 | 5, 10 | 14, 9 | 2, 6 | -, - | 1, 2 |
| Blepharispermum hirtum Oliv. | -, - | 3, - | -, - | 53, 32 | 8, 9 | 3, 3 | 10, 14 | -, - | -, - | -, - |
| Hybanthus durus (Baker) O.Schwartz | 6, 15 | -, - | -, - | -, 1 | 5, 12 | 6, 15 | 1, 6 | -, - | 3, 9 | 4, 9 |
| Euphorbia smithii S.Carter | 5, 1 | 13, 4 | 4, 1 | 3, - | 20, 2 | 16, 8 | 9, 2 | 18, 2 | 1, - | 7, 3 |
| Allophylus rubifolius (Hochst. ex A.Rich.) Engl. | -, - | 3, 4 | 17, 13 | 4, 17 | 10, 4 | 14, 11 | 8, 10 | -, 1 | 4, 6 | 6, 7 |
| Acacia senegal Willd. | -, - | 2, 2 | 3, 7 | 3, 10 | 3, 26 | -, 4 | -, 7 | -, - | 1, 10 | 3, 9 |
| Cadia purpurea (G.Piccioli) Aiton | -, - | -, - | -, 2 | 3, 13 | -, - | -, - | -, - | -, - | 44, 33 | 20, 36 |
| Croton confertus Baker | 4, - | -, - | 6, 1 | 6, 5 | -, - | -, - | -, 1 | -, - | 35, 10 | 7, 2 |
| Euclea racemosa subsp. schimperi (A.DC.) F.White | -, - | -, - | -, - | -, - | 1, - | -, - | 1, - | 4, 13 | 1, - | 7, 1 |
| Blepharis dhofarensis A.G.Mill. | -, - | -, - | -, - | -, - | -, 2 | -, - | -, 1 | -, - | -, - | -, 6 |
| Jasminum grandiflorum L. | -, 3 | -, - | -, 1 | -, - | 1, 1 | -, - | -, - | 8, 13 | 2, 1 | 10, 2 |
| Olea europaea subsp. cuspidata (Wall. & G.Don) Cif. | 4, - | -, - | 1, - | -, - | 2, - | -, - | -, - | -, 2 | 7, 1 | 17, - |
| Premna resinosa (Hochst.) Schauer | -, 2 | -, 1 | -, - | -, - | -, 1 | -, 3 | 9, 14 | -, - | -, - | -, - |
| Adenium obesum (Forssk.) Roem. & Schult. | 2, 1 | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Acacia gerrardii Benth. | -, - | -, - | -, - | -, - | -, - | -, - | 4, 10 | -, - | -, - | -, - |
| Flueggea virosa (Roxb. ex Willd.) Royle | -, - | -, - | -, 3 | -, - | -, - | 1, 5 | -, - | -, - | -, 1 | -, 2 |
| Rhus somalensis Engl. | -, 1 | -, - | -, - | -, - | 1, - | -, - | -, - | 1, 10 | -, - | -, - |
| Carissa spinarum L. | -, - | -, - | -, 1 | -, - | -, - | -, - | -, - | -, 2 | -, - | -, - |
| Ficus vasta Forssk. | -, - | -, - | 1, - | -, - | -, - | -, - | -, - | -, - | 1, - | 2, - |
| Pavetta longiflora Vahl | -, 1 | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, 3 | -, 2 |
| Woodfordia uniflora (A. Rich.) Koehne | -, - | -, 4 | -, - | -, 2 | -, - | -, - | -, - | -, - | -, - | -, - |
| Boscia arabica Pestal. | -, - | -, - | -, - | -, - | -, - | -, - | -, 1 | -, - | -, - | -, - |
| Grewia bicolor Juss. | -, - | -, 2 | -, - | -, - | -, 1 | -, - | -, 1 | -, - | -, - | -, - |
| Delonix elata (L.) Gamble | -, - | -, - | -, - | -, - | -, - | 1, 1 | 7, 1 | -, - | -, - | -, - |
| Tamarindus indica L. | -, - | -, - | 1, - | 2, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Gomphocarpus fruticosus subsp. setosus (Forssk.) Goyder & Nicholas | -, - | -, - | -, - | -, - | -, - | -, - | -, - | 1, - | -, - | -, - |
| Cordia ovalis R. Br. | -, - | -, - | -, - | -, - | -, - | 1, 1 | -, 1 | -, - | -, - | -, - |
| Ehretia obtusifolia Hochst. ex A.DC. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Ficus sycomorus L. | -, - | -, - | 1, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Searsia pyroides (Burch.) Moffett | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Calotropis procera (Aiton) Dryand. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Caesalpinia erianthera Chiov. | -, - | -, - | -, - | -, - | -, - | -, 3 | -, - | -, - | -, - | -, - |
| Lawsonia inermis L. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, 1 |
| Azima tetracantha Lam. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Cordia perrottetii Wight | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Hildebrandtia africana Vatke | -, - | -, - | -, - | -, - | -, - | -, - | -, 1 | -, - | -, - | -, - |
| Acridocarpus orientalis A. Juss | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Rhamnus staddo A.Rich. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Unidentified sp. | -, - | -, - | -, - | -, - | -, - | -, 1 | -, - | -, - | -, - | -, - |
| Grewia villosa Willd. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| **Site number** | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 |

| | Count of individuals (A = Adult, J = Juvenille) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Woody species** | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J | A, J |
| Commiphora habessinica (O.Berg) Engl. | 16, 9 | 14, 8 | 15, 23 | 11, 5 | 23, 13 | 12, 31 | 17, 21 | 6, 9 | 15, 12 | 9, 18 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Jatropha dhofarica Radcl.-Sm. | -, - | -, 2 | 4, 6 | 4, 3 | -, 4 | 12, 9 | -, - | 2, 6 | 10, 9 | 1, 4 |
| Anogeissus dhofarica A.J.Scott | 12, 2 | 8, 1 | 15, - | 8, 3 | 13, 3 | 4, 5 | 4, - | 20, 14 | 14, 4 | 16, 6 |
| Zygocarpum dhofarense (Hillc. & J.B.Gillett) Thulin & Lavin | 10, 6 | 25, 11 | 2, 5 | 5, 9 | 14, 13 | -, 3 | 8, 15 | -, - | 12, 18 | 7, 4 |
| Dodonaea viscosa subsp. angustifolia (L.f.) J.G.West | 1, - | -, - | -, - | 28, 10 | -, - | -, - | 7, 1 | 28, 3 | 1, - | 6, 2 |
| Solanum incanum L. | 4, 11 | 2, 2 | 2, 3 | 4, 10 | -, - | 4, 4 | 2, 11 | 3, 6 | 1, 9 | 1, 5 |
| Maytenus dhofarensis Sebsebe | 7, 3 | 2, - | 3, 1 | 7, 14 | -, - | -, 2 | 5, 1 | 39, 35 | 1, 1 | 6, 3 |
| Commiphora gileadensis (L.) C.Chr. | 9, 9 | 16, 10 | 3, 5 | 8, 5 | 30, 6 | 2, 4 | 8, 8 | 1, 2 | 3, 6 | 6, 2 |
| Blepharispermum hirtum Oliv. | -, - | 12, 9 | 10, 2 | -, - | 21, 8 | 66, 15 | 1, - | 1, 2 | 22, 18 | 2, 3 |
| Hybanthus durus (Baker) O.Schwartz | 5, - | 16, 20 | 15, 33 | -, 5 | 1, 17 | 3, 11 | 4, 12 | -, - | 7, 5 | 3, 10 |
| Euphorbia smithii S.Carter | 5, 7 | 10, 6 | 32, 11 | 4, 4 | 2, 3 | 5, 3 | 10, 9 | 1, 2 | 5, 7 | 16, 3 |
| Allophylus rubifolius (Hochst. ex A.Rich.) Engl. | 4, 1 | 2, 2 | 10, 3 | 5, 4 | 2, 1 | -, - | 1, 3 | 9, 9 | 14, 5 | 27, 7 |
| Acacia senegal Willd. | 9, 11 | 5, 8 | 1, 17 | -, - | -, 18 | 4, 22 | 2, 9 | 2, 9 | -, 12 | 3, 27 |
| Cadia purpurea (G.Piccioli) Aiton | -, - | -, - | 4, 1 | 5, 7 | -, - | -, - | -, - | -, - | -, - | 3, 13 |
| Croton confertus Baker | 1, - | -, - | 1, - | 3, 2 | 1, - | 1, - | -, - | 3, 2 | 5, 3 | 6, 5 |
| Euclea racemosa subsp. schimperi (A.DC.) F.White | 7, 6 | 2, - | -, - | 14, 13 | -, - | -, - | 22, 11 | -, 2 | -, - | 1, 1 |
| Blepharis dhofarensis A.G.Mill. | 4, 29 | 5, 37 | -, - | -, 4 | -, 8 | -, - | -, 2 | -, - | -, - | -, - |
| Jasminum grandiflorum L. | 4, 7 | -, - | -, - | 8, 4 | -, - | -, - | 7, 9 | 2, 1 | 1, - | -, - |
| Olea europaea subsp. cuspidata (Wall. & G.Don) Cif. | 2, - | -, - | -, - | 3, - | -, - | -, - | 4, 1 | 1, 1 | -, 1 | -, - |
| Premna resinosa (Hochst.) Schauer | 2, 2 | 1, 4 | 2, 9 | -, - | 5, 10 | -, 2 | -, - | -, - | 5, 3 | 1, - |
| Adenium obesum (Forssk.) Roem. & Schult. | -, - | -, - | -, - | 1, 2 | -, - | -, - | -, - | -, - | -, - | -, - |
| Acacia gerrardii Benth. | -, - | -, - | -, - | -, - | 3, 2 | 3, 4 | -, - | -, - | -, - | -, - |
| Flueggea virosa (Roxb. ex Willd.) Royle | -, - | -, - | -, 1 | -, 1 | -, 1 | -, - | -, - | -, 16 | -, 2 | 4, 5 |
| Rhus somalensis Engl. | 2, 1 | -, - | -, - | -, - | -, - | -, - | 6, - | -, - | -, - | -, - |
| Carissa spinarum L. | 1, 1 | -, - | -, - | -, 3 | -, - | -, - | -, 1 | -, - | 1, 2 | -, - |
| Ficus vasta Forssk. | 4, 1 | -, - | -, - | -, - | -, - | -, - | 5, 1 | -, - | 1, - | -, - |
| Pavetta longiflora Vahl | 2, 2 | -, - | -, - | -, 6 | -, - | -, - | 1, 3 | -, - | -, - | -, - |
| Woodfordia uniflora (A. Rich.) Koehne | 1, 5 | -, - | -, - | -, - | -, - | -, - | 3, 2 | -, 1 | -, - | -, - |
| Boscia arabica Pestal. | -, - | -, - | -, - | -, - | -, 6 | 1, 5 | -, - | -, - | 1, 2 | -, - |
| Grewia bicolor Juss. | 1, 1 | -, - | -, - | -, - | -, 1 | -, - | 1, - | -, - | -, 1 | -, 1 |
| Delonix elata (L.) Gamble | -, - | -, - | 1, - | -, - | 2, - | 2, - | -, - | -, - | -, - | -, - |
| Tamarindus indica L. | 5, 2 | -, - | -, - | -, - | -, - | 1, - | -, - | -, - | -, - | 2, 1 |
| Gomphocarpus fruticosus subsp. setosus (Forssk.) Goyder & Nicholas | 2, 1 | -, - | -, - | 2, 4 | -, - | -, - | -, - | -, - | -, - | -, - |
| Cordia ovalis R. Br. | -, 1 | -, - | -, - | -, - | -, 1 | -, - | -, - | -, - | -, - | -, - |
| Ehretia obtusifolia Hochst. ex A.DC. | -, - | -, - | -, - | -, - | 1, 4 | -, - | -, - | -, - | -, - | -, - |
| Ficus sycomorus L. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Searsia pyroides (Burch.) Moffett | -, - | -, - | -, - | -, 1 | 1, 1 | -, - | -, - | -, - | -, - | -, - |
| Calotropis procera (Aiton) Dryand. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | 2, - | -, - | -, - |
| Caesalpinia erianthera Chiov. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Lawsonia inermis L. | -, 1 | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Azima tetracantha Lam. | -, 1 | -, - | -, - | -, - | -, - | -, - | 1, - | -, - | -, - | -, - |
| Cordia perrottetii Wight | -, - | -, - | -, - | -, - | 1, - | -, - | -, - | -, - | -, - | -, - |
| Hildebrandtia africana Vatke | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Acridocarpus orientalis A. Juss | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | 1, - | -, - |
| Rhamnus staddo A.Rich. | -, - | -, - | -, - | -, - | -, - | -, - | 1, - | -, - | -, - | -, - |
| Unidentified sp. | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - | -, - |
| Grewia villosa Willd. | -, - | -, - | -, - | -, 1 | -, - | -, - | -, - | -, - | -, - | -, - |

**Appendix 11. A sapling *Anogeissus dhofarica* growing under the protection of a rock.**

**Appendix 12.** ***Acacia senegal*** **and** ***Maytenus dhofarensis*** **with stunted morphology due to camel browsing.**

**Appendix 13. A small fenced area shows the difference in sward height between grazed and ungrazed land in January (4 months after Khareef).**

**Appendix 14. List of all recorded herbaceous species with endemism status (N = not endemic, RE = regional endemic (south arabian mountains), E = endemic to Dhofar) and IUCN Red List status (CR = critically endangered, EN = endangered, VU = vulnerable, NT = near threatened, LC = least concern, DD = data deficient, NE = Not evaluated). Average total percentage site cover is shown.**

| Herbaceous species | Endemism | Red List | Avg. % Cover |
|---|---|---|---|
| Arthraxon junnarensis S.K.Jain & Hemadri | - | - | 32.45 |
| Apluda mutica L. | N | NE | 20.35 |
| Oplismenus burmanni (Retz.) P.Beauv. | N | NE | 8.53 |
| Rungia pectinata (L.) Nees | N | NE | 7.95 |
| Aristida sp. | - | - | 5.83 |
| Themeda quadrivalvis (L.) Kuntze | N | NE | 4.80 |
| Arundinella pumila (Hochst. ex A.Rich.) Steud. | N | NE | 4.24 |
| Impatiens balsamina L. | N | NE | 3.85 |
| Capillipedium parviflorum (R.Br.) Stapf | N | NE | 3.83 |
| Heteropogon contortus (L.) P.Beauv. ex Roem. & Schult. | N | NE | 3.79 |
| Mitreola petiolata (J.F. Gmel.) Torr. & A. Gray | N | NE | 3.53 |
| Setaria sp. | - | - | 3.23 |
| Launaea crassifolia (Balf. fil.) C. Jeffr. | N | NE | 3.15 |
| Justicia areysiana Deflers | RE | NE | 3.10 |
| Ruellia grandiflora (Forssk.) Pers. | N | NE | 3.06 |
| Oplismenus sp. (purple) | - | - | 2.20 |
| Senna obtusifolia (L.) H.S.Irwin & Barneby | N | NE | 2.09 |
| Dichanthium annulatum (Forssk.) Stapf | N | NE | 2.04 |
| Anagallis pumila Sw. | N | NE | 2.03 |
| Lepidagathis calycina Hochst. ex DC. | N | NE | 1.97 |
| Eragrostis sp. | - | - | 1.83 |
| Panicum trichoides Sw. | N | NE | 1.55 |
| Blumea axillaris (Lam.) DC. | N | NE | 1.52 |
| Viola stocksii Boiss. | N | NE | 1.50 |
| Megalochlamys violacea (Vahl) Vollesen | N | NE | 1.46 |
| Cleistachne sorghoides Benth. | N | NE | 1.42 |
| Cyperus alulatus J.Kern | LC | NE | 1.41 |
| Ocimum dhofarense (Sebald) A.J.Paton | E | NE | 1.35 |
| Cyperus sp. (white seed) | - | - | 1.33 |
| Digitaria tomentosa (J.Koenig ex Rottler) Henrard | N | NE | 1.33 |
| Eragrostis viscosa (Retz.) Trin. | N | NE | 1.33 |
| Bidens biternata (Lour.) Merr. & Sherff | N | NE | 1.32 |
| Enteropogon dolichostachyus (Lag.) Keng | N | NE | 1.31 |
| Hypoestes forskaolii (Vahl) Sol. ex Roem. & Schult. | N | NE | 1.30 |
| Cynodon dactylon (L.) Pers. | N | NE | 1.27 |
| Dactyloctenium aegyptium (L.) Willd. | N | NE | 1.26 |
| Rottboellia cochinchinensis (Lour.) Clayton | N | NE | 1.24 |
| Selaginella imbricata (Forsk.) Spring ex Decaisne | N | NE | 1.22 |
| Gladiolus candidus (Rendle) Goldblatt | N | NE | 1.21 |
| Triumfetta pentandra A. Rich. ex Guill. & Perr. | N | NE | 1.16 |
| Cucumis sativus L. | N | NE | 1.10 |
| Oldenlandia corymbosa L. | N | NE | 1.02 |

| | | | |
|---|---|---|---|
| Ipomoea nil (L.) Roth | N | NE | 1.00 |
| Digitaria sp.1 | - | - | 1.00 |
| Cyperus sp. | - | - | 0.98 |
| Luffa acutangula (L.) Roxb. | N | NE | 0.93 |
| Kohautia retrorsa (Boiss.) Bremek. | N | NE | 0.91 |
| Adiantum lunulatum Burm. f. | N | NE | 0.83 |
| Canscora concanensis C. B. Clark | N | NE | 0.83 |
| Cyperus longus L. | LC | NE | 0.83 |
| Fimbristylis bisumbellata (Forssk.) Bubani | LC | NE | 0.83 |
| Leucas dhofarensis Hedge & Sebald | E | NE | 0.83 |
| Achyranthes aspera L. | N | NE | 0.81 |
| Sclerocarpus africanus Jacq. ex Murray | N | NE | 0.79 |
| Pimpinella schweinfurthii Aschers. | N | NE | 0.78 |
| Barleria hochstetteri Nees | N | NE | 0.77 |
| Brachiaria eruciformis (Sm.) Griseb. | N | LC | 0.75 |
| Digitaria velutina (Forssk.) P.Beauv. | N | NE | 0.70 |
| Dyschoriste dalyi | N | NE | 0.67 |
| Chrysopogon macleishii Cope | E | NE | 0.67 |
| Remusatia vivipara (Roxb.) Schott | N | NE | 0.67 |
| Ruttya fruticosa Lindau | N | NE | 0.64 |
| Chloris sp. | - | - | 0.63 |
| Ammi majus L. | N | NE | 0.62 |
| Eustachys paspaloides (Vahl) Lanza & Mattei | N | NE | 0.59 |
| Asparagus racemosus Willd. | N | NE | 0.58 |
| Justicia heterocarpa T. Anderson | N | NE | 0.58 |
| Adiantum philippense L. | N | NE | 0.53 |
| Aloe praetermissa T.A.McCoy & Lavranos | E | NE | 0.51 |
| Dichanthium micranthum Cope | N | NE | 0.50 |
| Digitaria sp. | - | - | 0.50 |
| Ipomoea biflora (L.) Pers. | N | NE | 0.50 |
| Pimpinella sp. | - | - | 0.50 |
| Adiantum capillus-veneris L. | N | LC | 0.40 |
| Cissus quadrangularis L. | N | NE | 0.40 |
| Plectranthus barbatus Andrews | N | NE | 0.38 |
| Polygala senensis Kl. | N | NE | 0.37 |
| Aneilema forsskalii Kunth | N | NE | 0.37 |
| Plumbago zeylanica L. | N | NE | 0.37 |
| Ruellia patula Jacq. | N | NE | 0.33 |
| Alysicarpus glumaceus (Vahl) DC. | N | NE | 0.33 |
| Indigofera oblongifolia Forssk. | LC | NE | 0.33 |
| Chloris virgata Sw. | N | NE | 0.33 |
| Dorstenia foetida (Forssk.) Schweinf. | N | NE | 0.33 |
| Eustachys sp. | - | - | 0.33 |
| Ipomoea sp. | - | - | 0.33 |
| Meineckia sp. | - | - | 0.33 |
| Sida ovata Forssk. | N | NE | 0.27 |
| Urochloa panicoides P.Beauv. | LC | NE | 0.25 |
| Blumea lacera (Burm. fil.) DC. | N | NE | 0.23 |
| Vigna radiata (L.) R.Wilczek | N | NE | 0.23 |

| | | | |
|---|---|---|---|
| Endostemon tenuiflorus (Benth.) M.R.Ashby | N | NE | 0.22 |
| Orobanche dhofarensis M.J.Y. Foley | E | NE | 0.21 |
| Arthraxon sp.1 | - | - | 0.20 |
| Convolvulus prostratus Forsk. | N | NE | 0.20 |
| Digitaria ciliaris (Retz.) Koeler | N | NE | 0.20 |
| Parthenium hysterophorus L. | N | NE | 0.20 |
| Abelmoschus manihot (L.) Medik. | N | NE | 0.18 |
| Cyperus esculentus L. | LC | NE | 0.18 |
| Corchorus aestuans L. | N | NE | 0.17 |
| Corchorus trilocularis L. | N | NE | 0.17 |
| Tephrosia humilis Guill. & Perr. | N | NE | 0.17 |
| Tephrosia subtriflora Baker | N | NE | 0.13 |
| Justicia bentii V.A.W. Grah. | RE | NE | 0.13 |
| Eragrostis amabilis (L.) Wight & Arn. | N | NE | 0.12 |
| Commelina forskaolii Vahl | N | NE | 0.10 |
| Aleuritopteris scioana (Chiov.) Fraser-Jenk. | N | NE | 0.07 |
| Tephrosia sp. | - | - | 0.07 |
| Rhynchosia minima (L.) DC. | LC | NE | 0.03 |

**Appendix 15. Herbaceous species percentage covers for each site.**

| Site number | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
|---|---|---|---|---|---|---|---|---|---|---|
| **Herbaceous species** | % cover of site | | | | | | | | | |
| Arthraxon junnarensis S.K.Jain & Hemadri | 67.80 | 61.33 | 48.00 | 57.33 | 39.00 | 57.27 | 50.67 | 54.87 | 55.27 | 26.83 |
| Apluda mutica L. | 19.97 | 23.80 | 33.10 | 20.87 | 41.27 | 42.33 | 31.50 | 4.33 | 17.73 | 4.13 |
| Oplismenus burmanni (Retz.) P.Beauv. | - | - | - | - | - | - | - | - | 1.33 | 24.90 |
| Rungia pectinata (L.) Nees | 1.00 | - | - | 1.97 | - | 2.17 | - | 0.73 | - | 12.17 |
| Aristida sp. | - | - | - | - | - | - | - | - | - | - |
| Themeda quadrivalvis (L.) Kuntze | - | - | - | - | - | 1.77 | 4.43 | 16.37 | 3.27 | - |
| Arundinella pumila (Hochst. ex A.Rich.) Steud. | 0.50 | 1.67 | 0.83 | 4.17 | 3.33 | 0.50 | - | - | - | 8.40 |
| Impatiens balsamina L. | 4.90 | 3.17 | 3.33 | 10.27 | 0.27 | 9.57 | 2.33 | 0.17 | 9.03 | 3.00 |
| Capillipedium parviflorum (R.Br.) Stapf | - | - | - | - | - | - | - | - | - | - |
| Heteropogon contortus (L.) P.Beauv. ex Roem. & Schult. | - | - | - | - | - | 4.33 | - | - | - | - |
| Mitreola petiolata (J.F. Gmel.) Torr. & A. Gray | - | - | - | - | - | 1.70 | - | - | - | - |
| Setaria sp. | 5.40 | 4.27 | 12.40 | 0.60 | 13.63 | 0.17 | 4.60 | 5.00 | 5.10 | - |
| Launaea crassifolia (Balf. fil.) C. Jeffr. | - | - | - | - | - | - | - | - | - | - |
| Justicia areysiana Deflers | - | - | - | - | - | - | - | - | - | - |
| Ruellia grandiflora (Forssk.) Pers. | - | - | - | - | - | - | - | - | - | 0.17 |
| Oplismenus sp. (purple) | - | - | - | - | - | - | - | - | - | - |
| Senna obtusifolia (L.) H.S.Irwin & Barneby | 0.83 | 6.03 | 3.67 | 1.23 | 1.13 | 0.87 | - | 8.23 | 0.60 | - |
| Dichanthium annulatum (Forssk.) Stapf | 0.67 | - | - | - | 0.07 | - | - | 0.30 | 6.33 | - |
| Anagallis pumila Sw. | - | - | - | - | - | - | - | - | - | - |
| Lepidagathis calycina Hochst. ex DC. | - | - | - | - | - | - | - | - | - | - |
| Eragrostis sp. | - | - | - | - | - | - | - | - | - | - |
| Panicum trichoides Sw. | - | - | - | - | - | - | - | - | - | - |
| Blumea axillaris (Lam.) DC. | - | - | - | - | - | - | - | 1.40 | - | - |
| Viola stocksii Boiss. | - | - | - | - | - | - | - | - | - | 1.00 |
| Megalochlamys violacea (Vahl) Vollesen | - | - | - | - | - | - | - | - | - | - |
| Cleistachne sorghoides Benth. | - | - | - | - | 0.10 | - | - | - | - | - |
| Cyperus alulatus J.Kern | 0.03 | - | - | - | - | - | - | - | - | - |
| Ocimum dhofarense (Sebald) A.J.Paton | - | 2.83 | - | 0.93 | - | - | - | - | - | - |
| Cyperus sp. (white seed) | - | - | - | - | - | - | - | - | - | - |
| Digitaria tomentosa (J.Koenig ex Rottler) Henrard | - | - | - | - | - | - | - | - | - | - |
| Eragrostis viscosa (Retz.) Trin. | - | - | - | - | - | - | - | 1.33 | - | - |
| Bidens biternata (Lour.) Merr. & Sherff | - | - | 1.17 | 0.43 | - | 0.17 | - | - | - | 1.37 |
| Enteropogon dolichostachyus (Lag.) Keng | - | - | - | - | - | - | - | - | - | 0.07 |
| Hypoestes forskaolii (Vahl) Sol. ex Roem. & Schult. | - | - | - | 0.07 | - | - | - | - | - | 0.20 |
| Cynodon dactylon (L.) Pers. | - | - | - | - | - | - | - | - | - | - |
| Dactyloctenium aegyptium (L.) Willd. | 0.10 | 2.67 | - | - | 0.10 | - | 2.17 | - | - | - |
| Rottboellia cochinchinensis (Lour.) Clayton | 0.50 | 0.67 | - | - | - | 0.67 | - | - | - | - |
| Selaginella imbricata (Forsk.) Spring ex Decaisne | - | - | - | - | - | - | 2.07 | - | - | - |
| Gladiolus candidus (Rendle) Goldblatt | - | - | - | 0.67 | - | - | - | - | - | - |
| Triumfetta pentandra A. Rich. ex Guill. & Perr. | 0.20 | 0.83 | 0.17 | 0.67 | 1.17 | 0.67 | 0.43 | - | - | 1.37 |
| Cucumis sativus L. | - | 0.37 | - | - | 0.50 | - | - | - | - | 0.83 |
| Oldenlandia corymbosa L. | 0.70 | - | - | - | - | - | - | 1.17 | - | - |
| Ipomoea nil (L.) Roth | - | - | - | - | - | - | - | - | - | - |
| Digitaria sp.1 | - | - | - | - | - | - | - | - | - | - |
| Cyperus sp. | - | - | - | - | - | - | - | - | - | - |
| Luffa acutangula (L.) Roxb. | - | - | - | - | - | - | - | - | - | - |
| Kohautia retrorsa (Boiss.) Bremek. | - | - | - | - | - | - | - | - | - | - |
| Adiantum lunulatum Burm. f. | - | - | - | - | - | - | - | - | - | - |
| Canscora concanensis C. B. Clark | - | - | - | - | - | - | - | - | - | - |
| Cyperus longus L. | - | - | - | - | - | - | - | - | - | - |
| Fimbristylis bisumbellata (Forssk.) Bubani | - | - | - | - | - | - | - | - | - | - |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Leucas dhofarensis Hedge & Sebald | - | - | - | - | - | - | - | - | - | - |
| Achyranthes aspera L. | - | 0.50 | 0.50 | 0.33 | 0.83 | - | 0.83 | - | 0.27 | 0.77 |
| Sclerocarpus africanus Jacq. ex Murray | 0.10 | - | - | 2.00 | - | 0.17 | - | - | - | 1.33 |
| Pimpinella schweinfurthii Aschers. | - | - | - | - | - | - | - | - | - | - |
| Barleria hochstetteri Nees | - | - | - | - | - | - | - | - | - | - |
| Brachiaria eruciformis (Sm.) Griseb. | 0.27 | 1.00 | 0.50 | - | - | - | - | - | - | - |
| Digitaria velutina (Forssk.) P.Beauv. | - | - | - | - | - | - | - | 1.00 | - | - |
| Dyschoriste dalyi | - | 0.50 | - | - | - | - | 0.33 | - | - | - |
| Chrysopogon macleishii Cope | - | - | - | - | - | - | 0.67 | - | - | - |
| Remusatia vivipara (Roxb.) Schott | - | - | - | - | - | - | - | - | 1.00 | - |
| Ruttya fruticosa Lindau | - | - | - | - | - | - | - | - | - | - |
| Chloris sp. | - | - | - | - | - | - | - | - | - | - |
| Ammi majus L. | 0.23 | - | - | - | - | - | - | - | - | - |
| Eustachys paspaloides (Vahl) Lanza & Mattei | - | - | - | - | - | - | - | - | 0.30 | - |
| Asparagus racemosus Willd. | - | - | - | - | - | - | - | - | - | - |
| Justicia heterocarpa T. Anderson | - | - | - | - | - | - | - | - | - | - |
| Adiantum philippense L. | - | - | - | - | - | - | - | - | - | - |
| Aloe praetermissa T.A.McCoy & Lavranos | - | - | - | - | 0.67 | - | - | - | - | - |
| Dichanthium micranthum Cope | - | - | - | - | - | - | - | - | 0.50 | - |
| Digitaria sp. | - | - | - | - | - | - | - | - | - | - |
| Ipomoea biflora (L.) Pers. | - | - | - | - | - | - | - | - | - | - |
| Pimpinella sp. | - | - | - | - | - | - | - | - | - | - |
| Adiantum capillus-veneris L. | - | - | - | - | - | - | - | - | - | - |
| Cissus quadrangularis L. | - | - | - | - | 0.50 | - | - | - | - | - |
| Plectranthus barbatus Andrews | - | - | - | - | - | - | - | - | - | - |
| Polygala senensis Kl. | - | 0.17 | - | - | - | - | - | - | - | - |
| Aneilema forsskalii Kunth | - | - | - | - | - | - | - | - | - | - |
| Plumbago zeylanica L. | - | - | - | - | - | - | - | - | - | - |
| Ruellia patula Jacq. | - | - | - | - | - | - | - | - | - | - |
| Alysicarpus glumaceus (Vahl) DC. | - | - | - | - | - | - | - | 0.17 | - | - |
| Indigofera oblongifolia Forssk. | - | - | - | - | - | - | - | 0.33 | - | - |
| Chloris virgata Sw. | - | - | - | - | - | - | - | - | - | - |
| Dorstenia foetida (Forssk.) Schweinf. | - | - | - | - | - | - | - | - | - | 0.33 |
| Eustachys sp. | - | - | - | - | - | - | - | - | - | - |
| Ipomoea sp. | - | - | - | - | - | - | - | - | - | - |
| Meineckia sp. | - | - | - | - | - | - | - | - | - | - |
| Sida ovata Forssk. | - | - | - | - | - | - | - | - | - | - |
| Urochloa panicoides P.Beauv. | - | - | - | - | - | - | 0.33 | - | - | - |
| Blumea lacera (Burm. fil.) DC. | - | - | - | - | - | - | - | 0.23 | - | - |
| Vigna radiata (L.) R.Wilczek | - | - | - | - | - | - | 0.23 | - | - | - |
| Endostemon tenuiflorus (Benth.) M.R.Ashby | - | - | - | - | - | - | - | - | - | - |
| Orobanche dhofarensis M.J.Y. Foley | - | 0.33 | 0.27 | 0.33 | - | - | - | - | - | - |
| Arthraxon sp.1 | - | - | - | - | - | - | - | 0.20 | - | - |
| Convolvulus prostratus Forsk. | 0.23 | - | - | - | - | - | - | - | - | - |
| Digitaria ciliaris (Retz.) Koeler | - | - | - | - | - | - | - | 0.20 | - | - |
| Parthenium hysterophorus L. | - | - | - | - | - | - | - | - | - | - |
| Abelmoschus manihot (L.) Medik. | - | - | - | - | - | - | 0.20 | - | - | - |
| Cyperus esculentus L. | 0.03 | - | - | - | - | - | - | - | - | - |
| Corchorus aestuans L. | - | - | - | - | - | - | - | 0.17 | - | - |
| Corchorus trilocularis L. | - | - | - | - | - | - | - | - | - | - |
| Tephrosia humilis Guill. & Perr. | - | - | 0.17 | - | - | - | - | - | - | - |
| Tephrosia subtriflora Baker | 0.07 | - | - | - | - | - | - | - | - | - |
| Justicia bentii V.A.W. Grah. | - | - | - | - | - | - | - | - | - | - |
| Eragrostis amabilis (L.) Wight & Arn. | - | - | - | - | - | - | 0.10 | - | - | - |
| Commelina forskaolii Vahl | - | - | - | - | - | - | - | - | - | - |
| Aleuritopteris scioana (Chiov.) Fraser-Jenk. | - | - | - | - | - | - | - | - | - | - |
| Tephrosia sp. | - | - | - | - | - | - | - | - | - | - |
| Rhynchosia minima (L.) DC. | - | - | - | - | - | - | - | - | - | - |

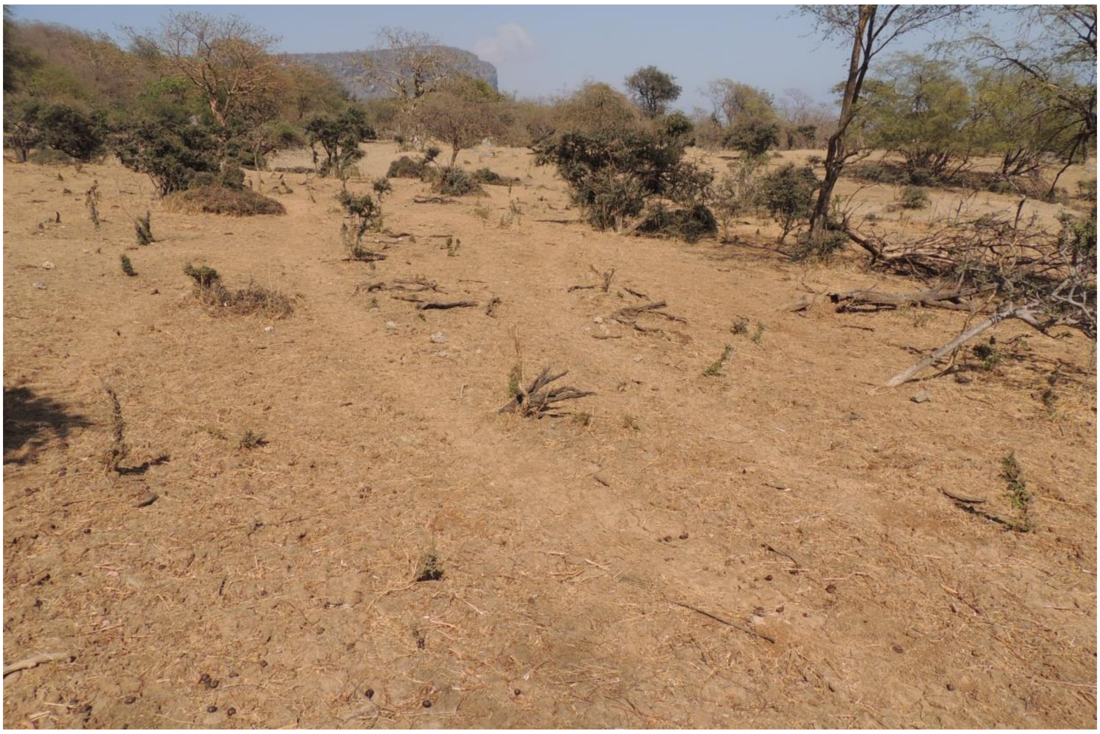
| Site number | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 |
|---|---|---|---|---|---|---|---|---|---|---|
| **Herbaceous species** | % cover of site | | | | | | | | | |
| Arthraxon junnarensis S.K.Jain & Hemadri | 27.60 | 52.63 | 42.53 | 6.23 | 35.27 | 10.60 | 15.67 | 48.83 | 42.93 | 10.00 |
| Apluda mutica L. | 0.77 | 9.47 | 6.00 | 24.27 | 32.50 | 10.30 | 34.37 | 62.50 | 13.67 | 2.00 |
| Oplismenus burmanni (Retz.) P.Beauv. | - | - | 0.50 | - | - | - | - | - | 3.00 | 30.50 |
| Rungia pectinata (L.) Nees | - | 7.53 | 0.93 | 3.47 | 22.57 | 23.80 | - | 5.33 | 12.57 | 11.10 |
| Aristida sp. | - | - | - | - | - | - | - | - | - | - |
| Themeda quadrivalvis (L.) Kuntze | - | 0.43 | - | - | - | - | - | 11.33 | - | - |
| Arundinella pumila (Hochst. ex A.Rich.) Steud. | 1.10 | - | 1.17 | 1.67 | 7.33 | 7.60 | 6.33 | 3.43 | 1.67 | 5.13 |
| Impatiens balsamina L. | 0.17 | 1.83 | 7.13 | 7.13 | 6.33 | 10.23 | 0.93 | 4.53 | 0.53 | 4.50 |
| Capillipedium parviflorum (R.Br.) Stapf | - | - | - | - | - | - | - | - | - | - |
| Heteropogon contortus (L.) P.Beauv. ex Roem. & Schult. | - | - | - | - | 8.00 | - | - | 0.33 | - | 1.33 |
| Mitreola petiolata (J.F. Gmel.) Torr. & A. Gray | - | 1.03 | - | - | - | 0.13 | 3.23 | - | - | - |
| Setaria sp. | 0.53 | 2.60 | 0.47 | 0.13 | 0.50 | - | - | 0.03 | - | - |
| Launaea crassifolia (Balf. fil.) C. Jeffr. | - | 0.20 | - | - | - | 0.67 | 0.33 | - | 0.27 | 0.33 |
| Justicia areysiana Deflers | - | - | - | - | - | - | - | - | - | - |
| Ruellia grandiflora (Forssk.) Pers. | - | - | - | - | - | - | - | - | - | 0.33 |
| Oplismenus sp. (purple) | - | - | - | - | - | - | - | - | - | - |
| Senna obtusifolia (L.) H.S.Irwin & Barneby | 1.00 | 1.27 | - | - | 0.17 | - | 0.50 | - | - | - |
| Dichanthium annulatum (Forssk.) Stapf | 0.37 | - | 0.70 | - | - | - | - | 0.77 | - | - |
| Anagallis pumila Sw. | 2.03 | - | - | - | - | - | - | - | - | - |
| Lepidagathis calycina Hochst. ex DC. | - | - | - | - | 0.27 | 0.10 | - | - | - | - |
| Eragrostis sp. | 1.83 | - | - | - | - | - | - | - | - | - |
| Panicum trichoides Sw. | - | - | - | - | - | - | - | - | - | - |
| Blumea axillaris (Lam.) DC. | - | 3.33 | 4.13 | - | - | - | - | - | 0.17 | 0.50 |
| Viola stocksii Boiss. | - | - | - | - | - | - | - | - | - | 0.50 |
| Megalochlamys violacea (Vahl) Vollesen | - | - | - | - | - | - | 0.10 | - | - | 0.47 |
| Cleistachne sorghoides Benth. | - | - | - | - | - | - | - | - | - | - |
| Cyperus alulatus J.Kern | 0.20 | - | - | - | - | - | - | - | - | - |
| Ocimum dhofarense (Sebald) A.J.Paton | 0.83 | - | - | - | 0.33 | - | - | - | 0.33 | - |
| Cyperus sp. (white seed) | - | - | - | - | - | - | - | 1.33 | - | - |
| Digitaria tomentosa (J.Koenig ex Rottler) Henrard | - | - | - | - | - | - | - | - | - | - |
| Eragrostis viscosa (Retz.) Trin. | - | - | - | - | - | - | - | - | - | - |
| Bidens biternata (Lour.) Merr. & Sherff | - | - | - | - | 1.17 | - | 7.73 | 0.17 | 1.50 | - |
| Enteropogon dolichostachyus (Lag.) Keng | - | - | - | - | - | - | - | - | - | 2.07 |
| Hypoestes forskaolii (Vahl) Sol. ex Roem. & Schult. | 1.23 | - | 0.10 | - | 0.50 | - | - | - | 1.93 | 3.73 |
| Cynodon dactylon (L.) Pers. | - | - | - | - | - | - | - | - | - | - |
| Dactyloctenium aegyptium (L.) Willd. | - | - | - | - | - | - | - | - | - | - |
| Rottboellia cochinchinensis (Lour.) Clayton | - | - | - | - | 2.67 | - | - | 2.17 | - | - |
| Selaginella imbricata (Forsk.) Spring ex Decaisne | 0.83 | - | - | - | - | - | 0.77 | - | - | - |
| Gladiolus candidus (Rendle) Goldblatt | - | - | - | - | 1.07 | 0.10 | - | - | - | - |
| Triumfetta pentandra A. Rich. ex Guill. & Perr. | 0.50 | 2.53 | - | - | - | 0.20 | 5.17 | - | 0.33 | - |
| Cucumis sativus L. | - | - | - | 5.00 | - | 0.17 | - | - | - | 0.83 |
| Oldenlandia corymbosa L. | 1.20 | - | - | - | - | - | - | - | - | - |
| Ipomoea nil (L.) Roth | - | - | - | 1.67 | - | - | 0.33 | - | - | - |
| Digitaria sp.1 | 1.00 | - | - | - | - | - | - | - | - | - |
| Cyperus sp. | - | - | 0.50 | - | - | - | - | - | - | - |
| Luffa acutangula (L.) Roxb. | - | - | 0.03 | 1.83 | - | - | - | - | - | - |
| Kohautia retrorsa (Boiss.) Bremek. | - | 0.07 | - | - | - | - | - | - | - | - |
| Adiantum lunulatum Burm. f. | - | - | - | - | - | - | - | - | - | 0.83 |
| Canscora concanensis C. B. Clark | - | - | - | - | - | - | - | 0.83 | - | - |
| Cyperus longus L. | - | - | 0.83 | - | - | - | - | - | - | - |
| Fimbristylis bisumbellata (Forssk.) Bubani | - | - | - | - | - | - | - | 0.83 | - | - |
| Leucas dhofarensis Hedge & Sebald | - | - | - | - | - | - | - | - | - | - |
| Achyranthes aspera L. | - | - | 1.33 | 4.13 | - | 0.07 | - | - | 0.50 | - |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Sclerocarpus africanus Jacq. ex Murray | 0.40 | - | 0.77 | - | - | - | - | - | - | - |
| Pimpinella schweinfurthii Aschers. | 0.90 | - | - | - | - | - | - | - | - | - |
| Barleria hochstetteri Nees | - | - | - | - | - | - | 0.30 | - | - | - |
| Brachiaria eruciformis (Sm.) Griseb. | 0.27 | - | 1.73 | - | - | - | - | - | - | - |
| Digitaria velutina (Forssk.) P.Beauv. | 0.40 | - | - | - | - | - | - | - | - | - |
| Dyschoriste dalyi | 1.17 | - | - | - | - | - | - | - | - | - |
| Chrysopogon macleishii Cope | - | - | - | - | - | - | - | - | - | - |
| Remusatia vivipara (Roxb.) Schott | - | - | - | - | - | - | - | - | - | - |
| Ruttya fruticosa Lindau | 0.13 | - | - | - | 0.83 | - | - | - | - | 0.67 |
| Chloris sp. | - | - | - | - | - | - | - | - | - | - |
| Ammi majus L. | 1.00 | - | - | - | - | - | - | - | - | - |
| Eustachys paspaloides (Vahl) Lanza & Mattei | 0.17 | - | - | - | - | - | 1.10 | - | - | - |
| Asparagus racemosus Willd. | - | - | - | - | - | 0.50 | - | - | - | - |
| Justicia heterocarpa T. Anderson | - | - | - | - | - | - | 0.20 | - | - | - |
| Adiantum philippense L. | - | - | - | - | - | - | - | - | - | - |
| Aloe praetermissa T.A.McCoy & Lavranos | 0.27 | - | - | - | - | 0.60 | - | - | - | - |
| Dichanthium micranthum Cope | - | - | - | - | - | - | - | - | - | - |
| Digitaria sp. | 0.50 | - | - | - | - | - | - | - | - | - |
| Ipomoea biflora (L.) Pers. | - | - | - | - | - | - | 0.50 | - | - | - |
| Pimpinella sp. | - | - | - | - | - | - | - | - | - | - |
| Adiantum capillus-veneris L. | - | - | - | - | - | - | - | - | - | 0.40 |
| Cissus quadrangularis L. | - | - | - | - | - | - | - | - | - | - |
| Plectranthus barbatus Andrews | - | - | - | - | - | - | - | - | - | - |
| Polygala senensis Kl. | - | - | - | - | - | 0.17 | 0.97 | - | - | - |
| Aneilema forsskalii Kunth | - | - | - | - | - | - | - | - | - | 0.50 |
| Plumbago zeylanica L. | - | - | - | - | - | - | - | - | - | 0.33 |
| Ruellia patula Jacq. | - | - | - | 0.17 | - | - | - | - | - | - |
| Alysicarpus glumaceus (Vahl) DC. | - | - | - | - | - | - | 0.50 | - | - | - |
| Indigofera oblongifolia Forssk. | - | - | - | - | - | - | - | 0.27 | - | - |
| Chloris virgata Sw. | - | - | - | - | 0.33 | - | - | - | - | - |
| Dorstenia foetida (Forssk.) Schweinf. | - | - | - | - | - | - | - | - | - | - |
| Eustachys sp. | - | - | - | - | - | - | - | - | - | - |
| Ipomoea sp. | - | - | - | - | - | - | - | - | - | - |
| Meineckia sp. | - | - | - | - | - | - | - | - | - | - |
| Sida ovata Forssk. | - | - | - | - | - | - | - | - | - | - |
| Urochloa panicoides P.Beauv. | 0.17 | - | - | - | - | - | - | - | - | - |
| Blumea lacera (Burm. fil.) DC. | - | - | - | - | - | - | - | - | - | - |
| Vigna radiata (L.) R.Wilczek | - | - | - | - | - | - | - | - | - | - |
| Endostemon tenuiflorus (Benth.) M.R.Ashby | - | 0.17 | - | - | - | - | 0.27 | - | - | - |
| Orobanche dhofarensis M.J.Y. Foley | 0.07 | - | - | - | - | - | - | 0.03 | - | - |
| Arthraxon sp.1 | - | - | - | - | - | - | - | - | - | - |
| Convolvulus prostratus Forsk. | - | - | 0.17 | - | - | - | - | - | - | - |
| Digitaria ciliaris (Retz.) Koeler | - | - | - | - | - | - | - | - | - | - |
| Parthenium hysterophorus L. | 0.20 | - | - | - | - | - | - | - | - | - |
| Abelmoschus manihot (L.) Medik. | - | - | 0.17 | - | - | - | - | - | - | - |
| Cyperus esculentus L. | - | - | 0.33 | - | - | - | - | - | - | - |
| Corchorus aestuans L. | - | - | - | - | - | - | - | - | - | - |
| Corchorus trilocularis L. | 0.17 | - | - | - | - | - | - | - | - | - |
| Tephrosia humilis Guill. & Perr. | - | - | - | - | - | - | - | - | - | - |
| Tephrosia subtriflora Baker | - | - | - | - | - | - | - | 0.20 | - | - |
| Justicia bentii V.A.W. Grah. | - | - | - | - | - | - | - | - | - | - |
| Eragrostis amabilis (L.) Wight & Arn. | - | - | - | - | - | - | - | - | - | - |
| Commelina forskaolii Vahl | - | - | 0.10 | - | - | - | - | - | - | - |
| Aleuritopteris scioana (Chiov.) Fraser-Jenk. | - | - | - | - | - | - | - | - | - | - |
| Tephrosia sp. | - | - | - | - | - | - | - | - | - | - |
| Rhynchosia minima (L.) DC. | - | 0.03 | - | - | - | - | - | - | - | - |
| **Site number** | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 |
| **Herbaceous species** | % cover of site | | | | | | | | | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Arthraxon junnarensis S.K.Jain & Hemadri | 2.90 | 11.50 | 17.50 | 38.00 | 18.93 | 22.67 | 12.50 | 17.73 | 16.90 | 4.10 |
| Apluda mutica L. | 18.37 | 3.00 | 41.00 | 14.73 | 14.33 | 47.27 | 12.10 | 9.40 | 14.03 | 1.43 |
| Oplismenus burmanni (Retz.) P.Beauv. | 15.13 | 2.60 | 0.50 | 3.60 | - | - | 1.67 | 8.97 | 2.03 | 16.10 |
| Rungia pectinata (L.) Nees | 10.20 | 6.07 | 23.73 | 3.43 | - | - | 3.03 | 1.33 | 10.67 | 3.23 |
| Aristida sp. | - | - | - | - | - | 5.83 | - | - | - | - |
| Themeda quadrivalvis (L.) Kuntze | - | - | 0.67 | - | - | - | - | - | - | 0.17 |
| Arundinella pumila (Hochst. ex A.Rich.) Steud. | 4.40 | 7.47 | 1.93 | 10.53 | 6.83 | 1.27 | 11.37 | 2.13 | - | 5.33 |
| Impatiens balsamina L. | 2.00 | - | - | 1.83 | - | - | - | 0.13 | 2.67 | 0.17 |
| Capillipedium parviflorum (R.Br.) Stapf | - | - | - | - | - | - | 3.83 | - | - | - |
| Heteropogon contortus (L.) P.Beauv. ex Roem. & Schult. | 2.53 | - | 6.50 | 1.83 | - | 2.67 | 12.40 | 0.33 | - | 1.43 |
| Mitreola petiolata (J.F. Gmel.) Torr. & A. Gray | - | - | 0.83 | - | 4.20 | 22.27 | - | 0.33 | 1.43 | 0.10 |
| Setaria sp. | - | 0.67 | 1.33 | - | 0.67 | - | - | - | - | - |
| Launaea crassifolia (Balf. fil.) C. Jeffr. | - | 3.23 | 2.83 | 0.23 | 27.00 | 1.60 | - | - | 0.17 | 0.93 |
| Justicia areysiana Deflers | - | 3.10 | - | - | - | - | - | - | - | - |
| Ruellia grandiflora (Forssk.) Pers. | - | 12.57 | - | - | 3.17 | 0.17 | - | - | - | 1.93 |
| Oplismenus sp. (purple) | - | - | - | - | - | - | 2.20 | - | - | - |
| Senna obtusifolia (L.) H.S.Irwin & Barneby | - | - | - | - | 0.33 | 4.70 | - | - | 2.77 | 0.03 |
| Dichanthium annulatum (Forssk.) Stapf | 3.83 | - | - | 0.13 | - | - | 7.27 | - | - | - |
| Anagallis pumila Sw. | - | - | - | - | - | - | - | - | - | - |
| Lepidagathis calycina Hochst. ex DC. | - | 6.90 | - | - | 3.73 | 0.50 | - | - | 0.33 | - |
| Eragrostis sp. | - | - | - | - | - | - | - | - | - | - |
| Panicum trichoides Sw. | - | - | 0.50 | - | - | - | - | - | 2.60 | - |
| Blumea axillaris (Lam.) DC. | - | - | - | 0.30 | - | - | - | 0.80 | - | - |
| Viola stocksii Boiss. | - | - | - | - | 3.00 | - | - | - | - | - |
| Megalochlamys violacea (Vahl) Vollesen | - | 4.63 | 0.27 | - | 3.10 | - | - | - | 0.17 | 1.50 |
| Cleistachne sorghoides Benth. | - | - | - | - | - | - | 2.73 | - | - | - |
| Cyperus alulatus J.Kern | 4.00 | - | - | - | - | - | - | - | - | - |
| Ocimum dhofarense (Sebald) A.J.Paton | - | 2.17 | 3.17 | 0.23 | - | - | - | - | - | - |
| Cyperus sp. (white seed) | - | - | - | - | - | - | - | - | - | - |
| Digitaria tomentosa (J.Koenig ex Rottler) Henrard | - | - | - | 1.33 | - | - | - | - | - | - |
| Eragrostis viscosa (Retz.) Trin. | - | - | - | - | - | - | - | - | - | - |
| Bidens biternata (Lour.) Merr. & Sherff | 0.10 | - | - | - | 0.83 | 0.90 | - | - | 0.33 | - |
| Enteropogon dolichostachyus (Lag.) Keng | 0.17 | - | - | - | - | - | 2.93 | - | - | - |
| Hypoestes forskaolii (Vahl) Sol. ex Roem. & Schult. | 0.67 | 0.10 | 0.77 | 2.73 | - | - | 1.73 | 1.57 | 1.87 | 2.37 |
| Cynodon dactylon (L.) Pers. | - | - | - | 1.27 | - | - | - | - | - | - |
| Dactyloctenium aegyptium (L.) Willd. | - | - | - | - | - | - | - | - | - | - |
| Rottboellia cochinchinensis (Lour.) Clayton | 2.50 | - | - | 0.47 | - | - | 0.27 | - | - | - |
| Selaginella imbricata (Forsk.) Spring ex Decaisne | - | - | - | - | - | - | - | - | - | - |
| Gladiolus candidus (Rendle) Goldblatt | - | 0.77 | - | - | - | - | 3.43 | - | - | - |
| Triumfetta pentandra A. Rich. ex Guill. & Perr. | 0.33 | - | 5.60 | 0.07 | - | 1.43 | - | 0.30 | 0.83 | 0.40 |
| Cucumis sativus L. | - | - | - | 0.03 | - | - | - | - | - | - |
| Oldenlandia corymbosa L. | - | - | - | - | - | - | - | - | - | - |
| Ipomoea nil (L.) Roth | - | - | - | - | - | - | - | - | - | - |
| Digitaria sp.1 | - | - | - | - | - | - | - | - | - | - |
| Cyperus sp. | 2.57 | - | - | - | - | - | 0.50 | 0.37 | - | - |
| Luffa acutangula (L.) Roxb. | - | - | - | - | - | - | - | - | - | - |
| Kohautia retrorsa (Boiss.) Bremek. | - | - | - | - | 1.00 | 1.67 | - | - | - | - |
| Adiantum lunulatum Burm. f. | - | - | - | - | - | - | - | - | - | - |
| Canscora concanensis C. B. Clark | - | - | - | - | - | - | - | - | - | - |
| Cyperus longus L. | - | - | - | - | - | - | - | - | - | - |
| Fimbristylis bisumbellata (Forssk.) Bubani | - | - | - | - | - | - | - | - | - | - |
| Leucas dhofarensis Hedge & Sebald | - | - | - | - | 0.83 | - | - | - | - | - |
| Achyranthes aspera L. | 1.40 | - | - | - | - | - | 0.40 | 0.20 | - | 0.07 |
| Sclerocarpus africanus Jacq. ex Murray | - | - | - | - | - | - | - | - | - | - |
| Pimpinella schweinfurthii Aschers. | - | - | - | - | 0.67 | - | - | - | - | - |

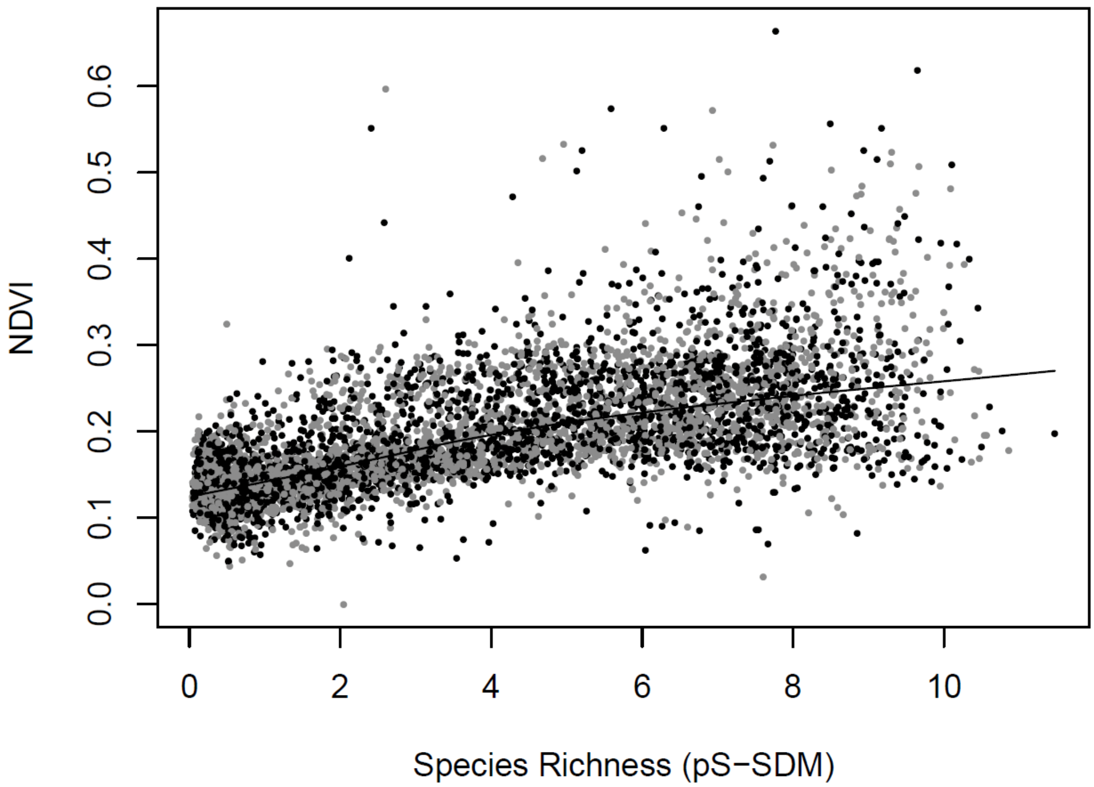
| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Barleria hochstetteri Nees | - | - | - | - | 1.83 | - | - | - | 0.37 | 0.60 |
| Brachiaria eruciformis (Sm.) Griseb. | - | - | - | - | - | - | - | 1.07 | - | 0.43 |
| Digitaria velutina (Forssk.) P.Beauv. | - | - | - | - | - | - | - | - | - | - |
| Dyschoriste dalyi | - | - | - | - | - | - | - | - | - | - |
| Chrysopogon macleishii Cope | - | - | - | - | - | - | - | - | - | - |
| Remusatia vivipara (Roxb.) Schott | - | - | - | - | - | - | 0.33 | - | - | - |
| Ruttya fruticosa Lindau | - | - | - | - | 0.83 | - | 0.33 | - | 0.20 | 1.50 |
| Chloris sp. | - | - | - | - | - | - | - | - | - | 0.63 |
| Ammi majus L. | - | - | - | - | - | - | - | - | - | - |
| Eustachys paspaloides (Vahl) Lanza & Mattei | - | 0.47 | 0.33 | - | 0.17 | - | 1.93 | - | 0.27 | - |
| Asparagus racemosus Willd. | - | 0.67 | - | - | - | - | - | - | - | - |
| Justicia heterocarpa T. Anderson | - | - | - | - | - | 1.50 | - | 0.03 | - | - |
| Adiantum philippense L. | - | - | - | - | - | - | 0.53 | - | - | - |
| Aloe praetermissa T.A.McCoy & Lavranos | - | - | - | - | - | - | - | - | - | - |
| Dichanthium micranthum Cope | - | - | - | - | - | - | - | - | - | - |
| Digitaria sp. | - | - | - | - | - | - | - | - | - | - |
| Ipomoea biflora (L.) Pers. | - | - | - | - | - | - | - | - | - | - |
| Pimpinella sp. | - | - | 0.50 | - | - | - | - | - | - | - |
| Adiantum capillus-veneris L. | - | - | - | - | - | - | - | - | - | - |
| Cissus quadrangularis L. | - | - | - | - | - | - | - | - | - | 0.30 |
| Plectranthus barbatus Andrews | - | - | - | - | - | - | 0.67 | - | - | 0.10 |
| Polygala senensis Kl. | - | - | - | 0.17 | - | - | - | - | - | - |
| Aneilema forsskalii Kunth | - | - | - | - | - | - | 0.23 | - | - | - |
| Plumbago zeylanica L. | - | - | - | - | - | - | - | - | - | 0.40 |
| Ruellia patula Jacq. | - | - | - | - | - | - | - | - | 0.67 | 0.17 |
| Alysicarpus glumaceus (Vahl) DC. | - | - | - | - | - | - | - | - | - | - |
| Indigofera oblongifolia Forssk. | - | - | - | - | - | - | - | - | 0.40 | - |
| Chloris virgata Sw. | - | - | - | - | - | - | - | - | - | - |
| Dorstenia foetida (Forssk.) Schweinf. | - | - | - | - | - | - | - | - | - | - |
| Eustachys sp. | - | - | - | - | - | - | - | - | 0.33 | - |
| Ipomoea sp. | - | - | - | - | - | 0.33 | - | - | - | - |
| Meineckia sp. | - | - | - | - | - | 0.33 | - | - | - | - |
| Sida ovata Forssk. | - | - | - | - | - | 0.27 | - | - | - | - |
| Urochloa panicoides P.Beauv. | - | - | - | - | - | - | - | - | - | - |
| Blumea lacera (Burm. fil.) DC. | - | - | - | - | - | - | - | 0.23 | - | - |
| Vigna radiata (L.) R.Wilczek | - | - | - | - | - | - | - | - | - | - |
| Endostemon tenuiflorus (Benth.) M.R.Ashby | - | - | - | - | - | - | - | - | - | - |
| Orobanche dhofarensis M.J.Y. Foley | - | - | - | - | - | - | - | - | - | - |
| Arthraxon sp.1 | - | - | - | - | - | - | - | - | - | - |
| Convolvulus prostratus Forsk. | - | - | - | - | - | - | - | - | - | - |
| Digitaria ciliaris (Retz.) Koeler | - | - | - | - | - | - | - | - | - | - |
| Parthenium hysterophorus L. | - | - | - | - | - | - | - | - | - | - |
| Abelmoschus manihot (L.) Medik. | - | - | - | - | - | - | - | - | - | - |
| Cyperus esculentus L. | 0.17 | - | - | - | - | - | - | - | - | - |
| Corchorus aestuans L. | - | - | - | - | - | - | - | - | - | - |
| Corchorus trilocularis L. | - | - | - | - | - | - | - | - | - | - |
| Tephrosia humilis Guill. & Perr. | - | - | - | - | - | - | - | - | - | - |
| Tephrosia subtriflora Baker | - | - | - | - | - | - | - | - | - | - |
| Justicia bentii V.A.W. Grah. | - | - | - | - | - | - | 0.13 | - | - | - |
| Eragrostis amabilis (L.) Wight & Arn. | - | - | - | - | - | - | - | - | 0.13 | - |
| Commelina forskaolii Vahl | - | - | - | - | - | - | - | - | - | - |
| Aleuritopteris scioana (Chiov.) Fraser-Jenk. | 0.07 | - | - | - | - | - | - | - | - | - |
| Tephrosia sp. | - | - | - | - | - | - | - | - | - | 0.07 |
| Rhynchosia minima (L.) DC. | - | - | - | - | - | - | - | - | - | - |

**Appendix 16. Photographs of heavily degraded *Anogeissus* forest (*Maytenus dhofarensis-Ficus sycomorus* sparse woodland). Top photo shows soil compaction, desiccation cracks, a vehicular trail, stunted phytomorphology and dead stumps. Bottom photo shows branch bending management practised on a large mature *Anogeissus dhofarica* tree (back right) and unpalatable *Cissus quadrangularis* and *Calotropis procera*. In both photos *Maytenus dhofarensis* appears somewhat resilient, possibly due to its hard wood and sharp spines.**

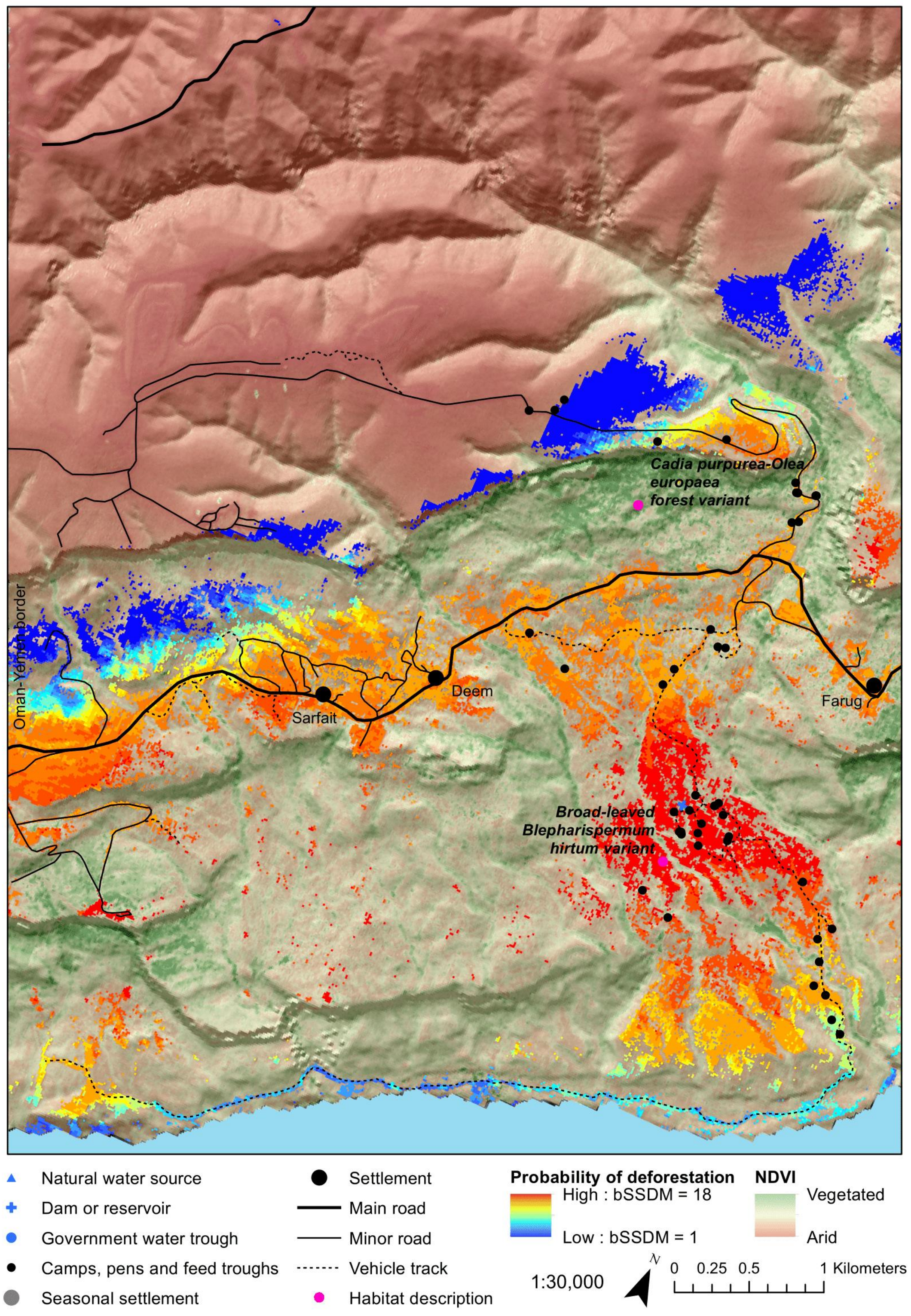


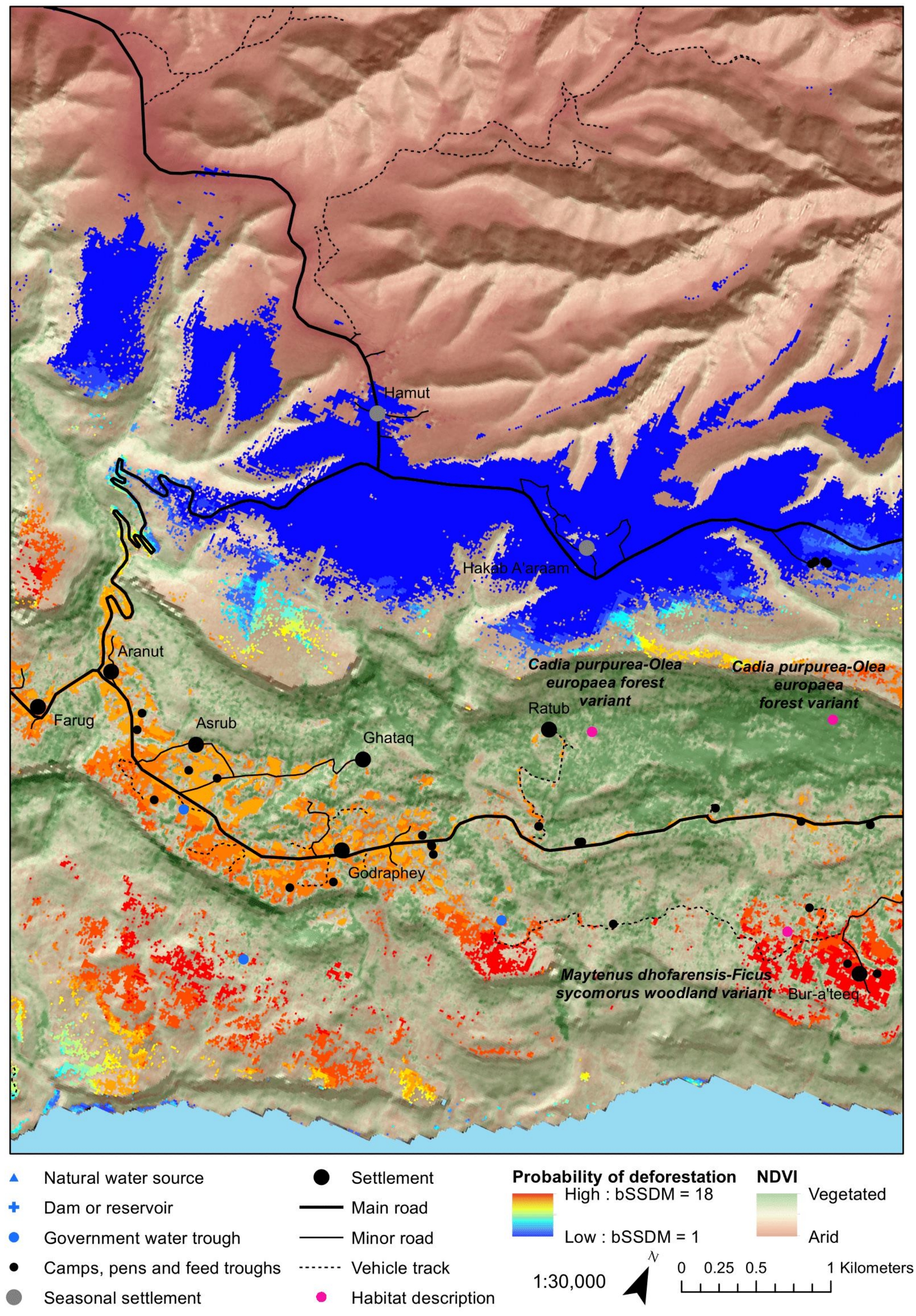
**Appendix 17. A remote area of *Premna resinosa-Hybanthus durus* forest to the northwest of Rakhyut with numerous livestock trails.**

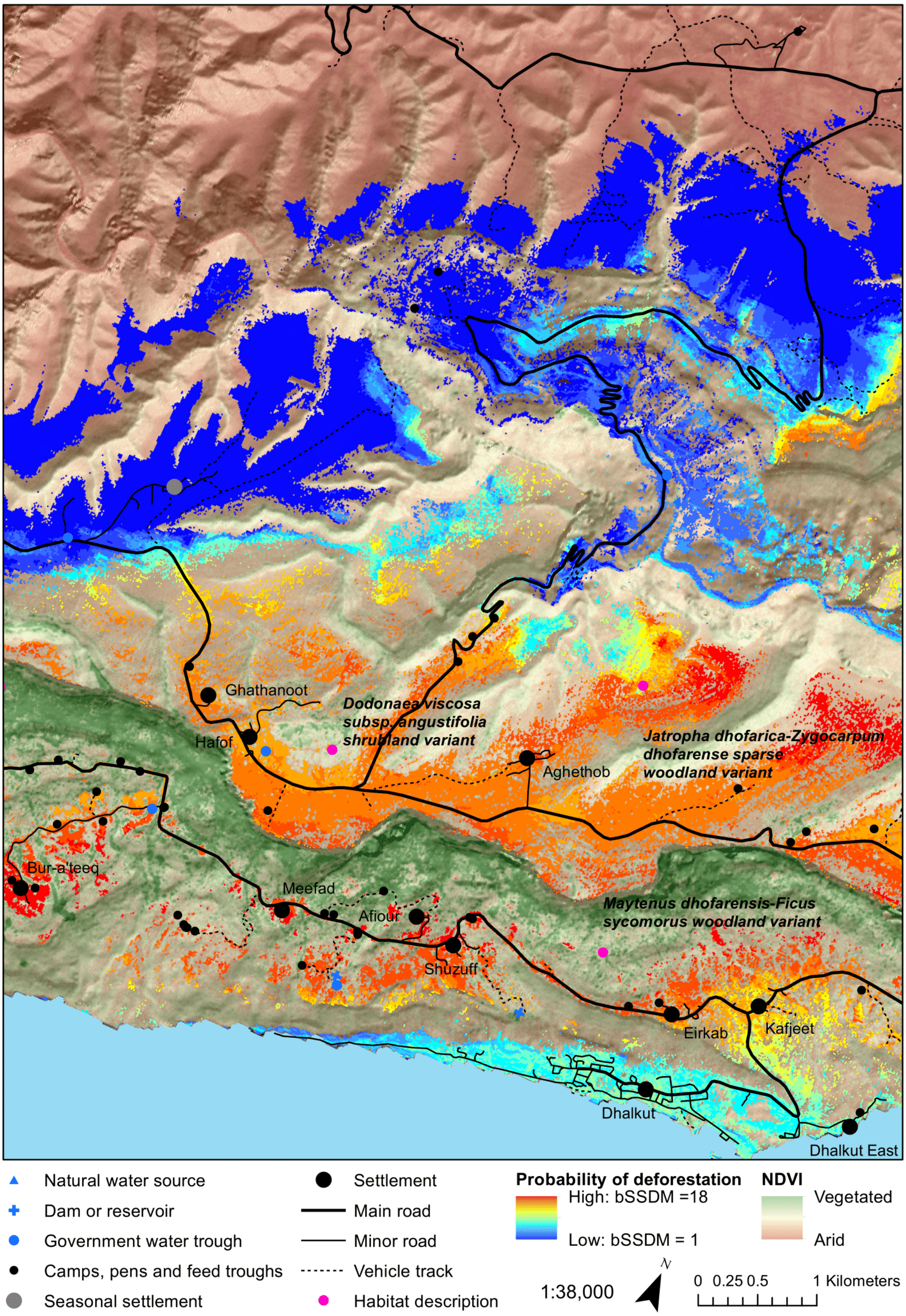
**Appendix 18. Scatter plot of NDVI against species richness (pS-SDM) from a sample of 5757 random points.**



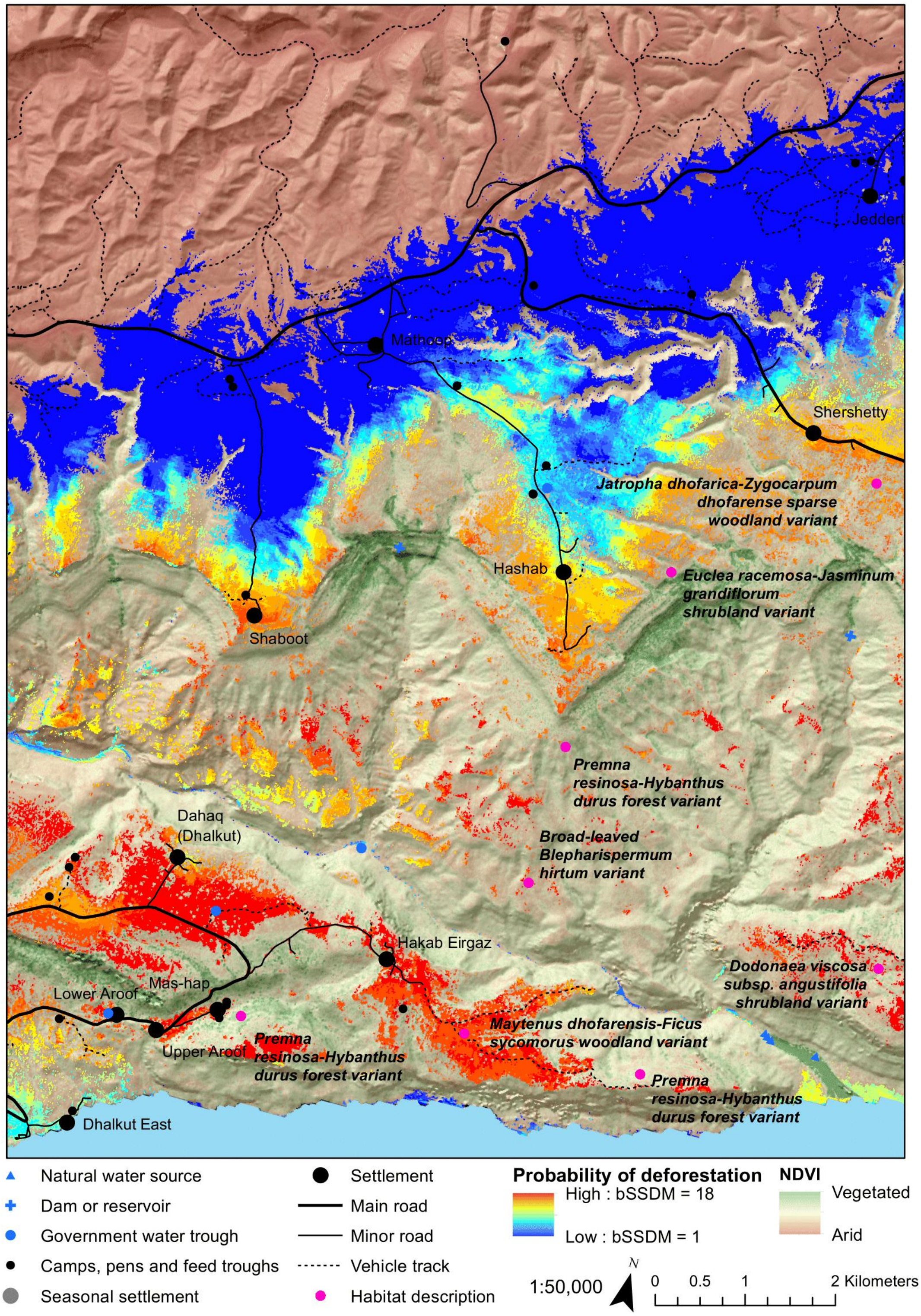
**Appendix 19. Detailed map series of Jabal Qamar, from Sarfait in the West to Sha'at in the East. Water sources, camps, settlements and roads are marked. The sampling sites are marked with their respective variant names (Chapter 4). A layer of probability of deforestation (Chapter 5) is displayed over a base map of NDVI with hillshade.**



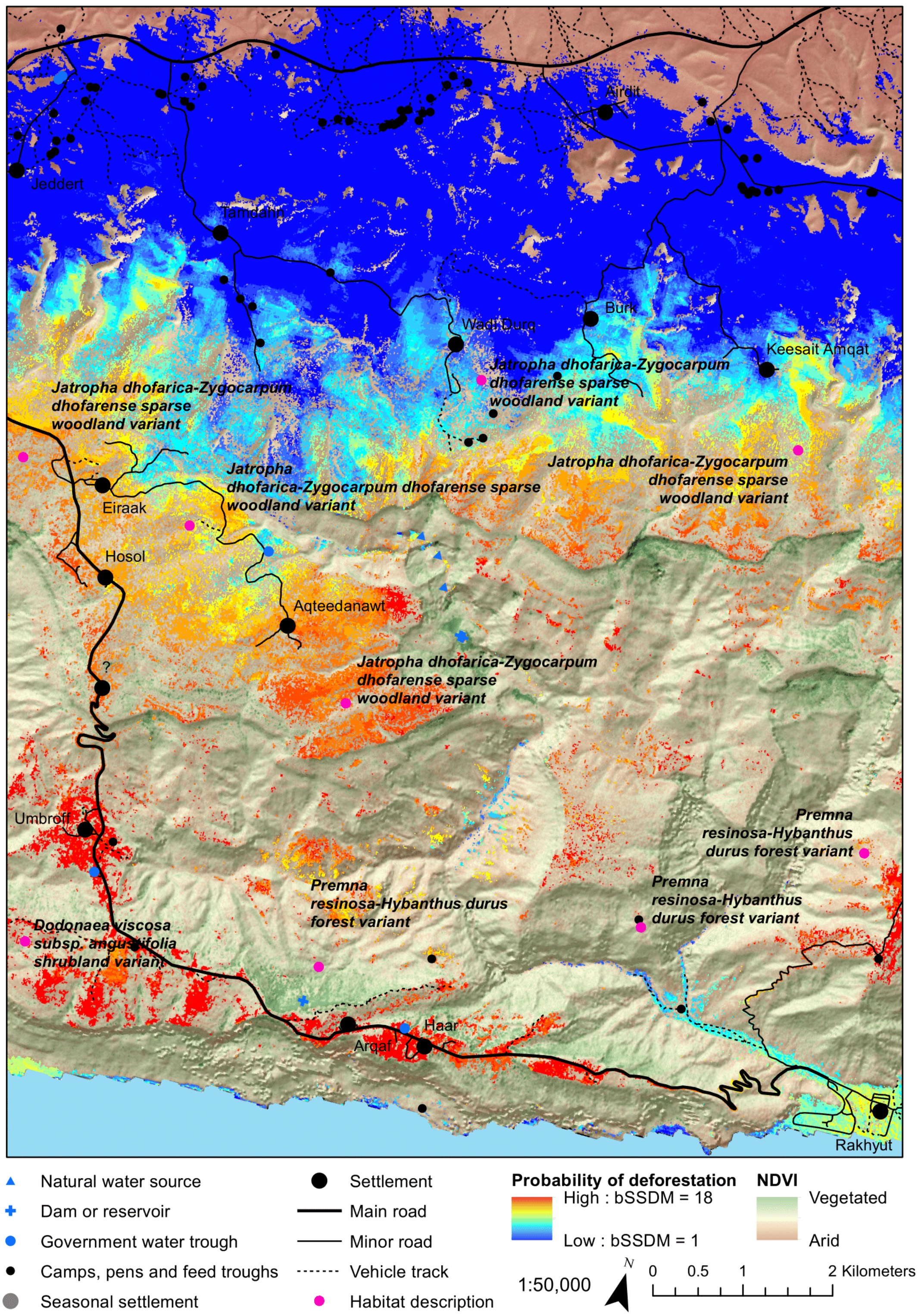

Hamut
Hakab A'araam
Aranut
Farug
Asrub
Ghataq
Ratub
Godraphey
Bur-a'teeq
Cadia purpurea-Olea europaea forest variant
Cadia purpurea-Olea europaea forest variant
Maytenus dhofarensis-Ficus sycomorus woodland variant
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High : bSSDM = 18
Low : bSSDM = 1
NDVI
Vegetated
Arid
1:30,000
N
0 0.25 0.5 1 Kilometers



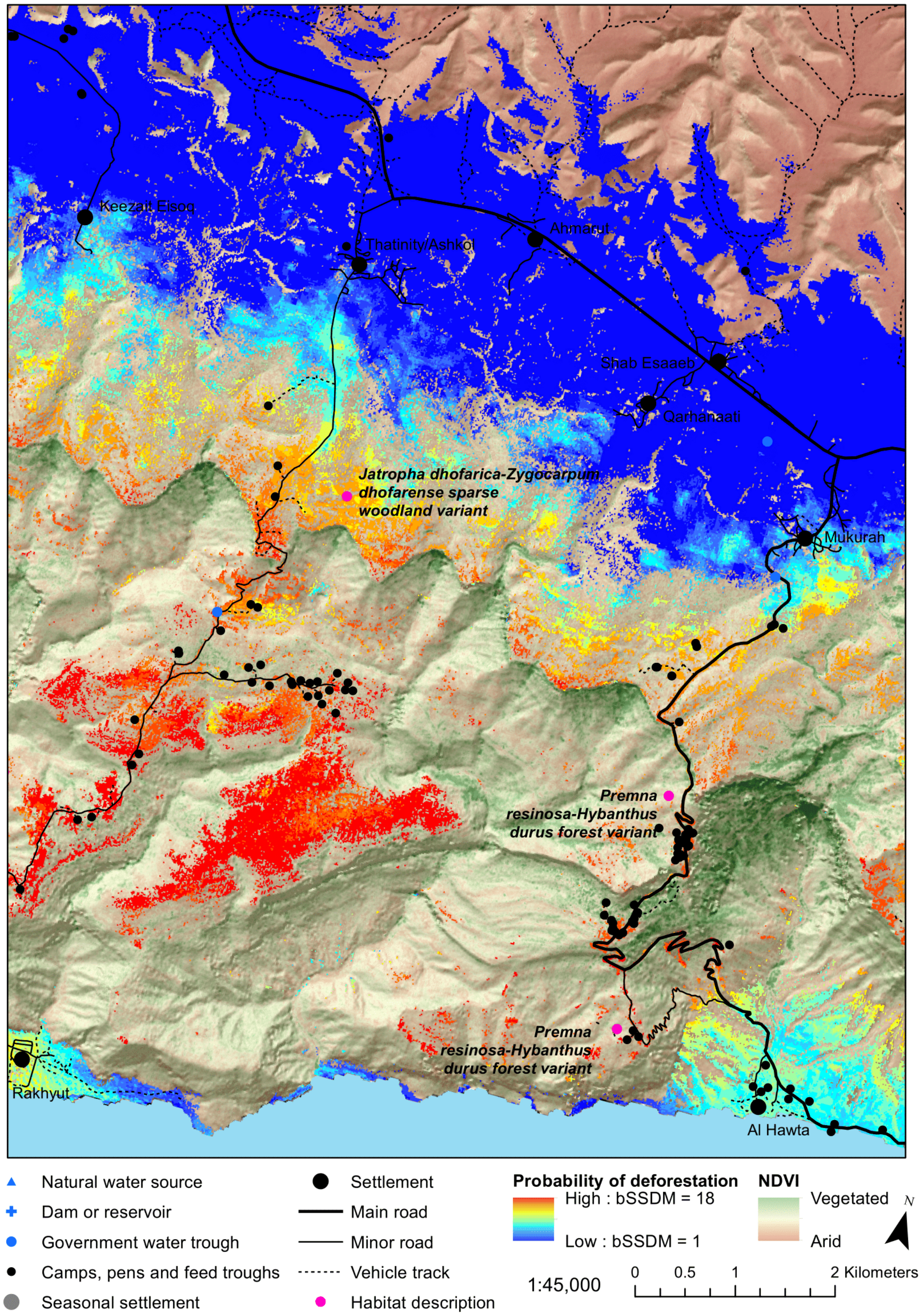
Ghathanoot
Hafof
Dodonaea viscosa subsp. angustifolia shrubland variant
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Aghethob
Bur-a'teeq
Meefad
Afiour
Shuzuff
Maytenus dhofarensis-Ficus sycomorus woodland variant
Eirkab
Kafjeet
Dhalkut
Dhalkut East
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High: bSSDM =18
Low: bSSDM = 1
NDVI
Vegetated
Arid
1:38,000
N
0 0.25 0.5 1 Kilometers



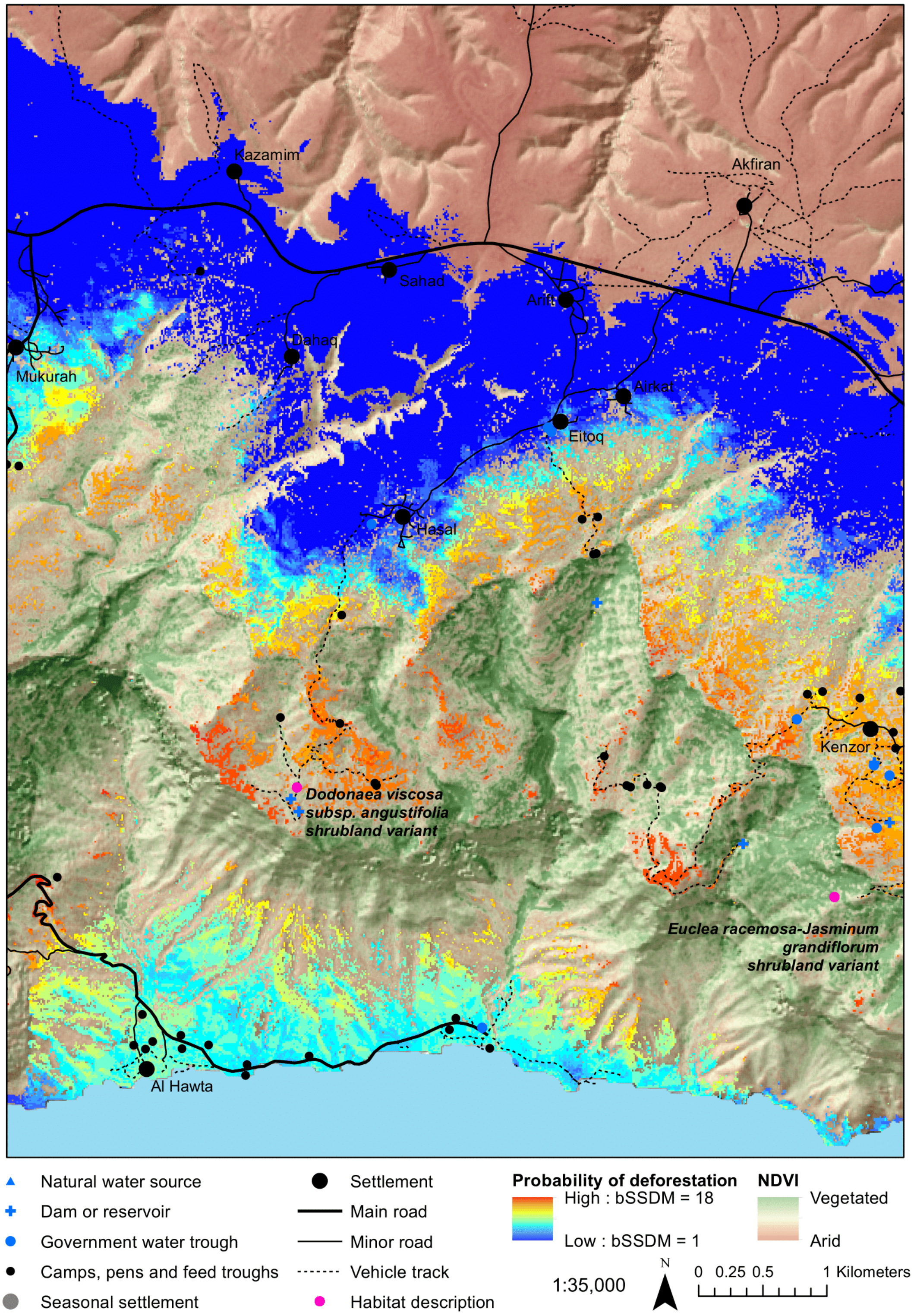
Jedderr
Mathoop
Shershetty
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Hashab
Euclea racemosa-Jasminum grandiflorum shrubland variant
Shaboot
Premna resinosa-Hybanthus durus forest variant
Broad-leaved Blepharispermum hirtum variant
Dahaq (Dhalkut)
Hakab Eirgaz
Dodonaea viscosa subsp. angustifolia shrubland variant
Mas-hap
Lower Aroof
Upper Aroof
Premna resinosa-Hybanthus durus forest variant
Maytenus dhofarensis-Ficus sycomorus woodland variant
Premna resinosa-Hybanthus durus forest variant
Dhalkut East
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High : bSSDM = 18
Low : bSSDM = 1
NDVI
Vegetated
Arid
1:50,000
N
0 0.5 1 2 Kilometers



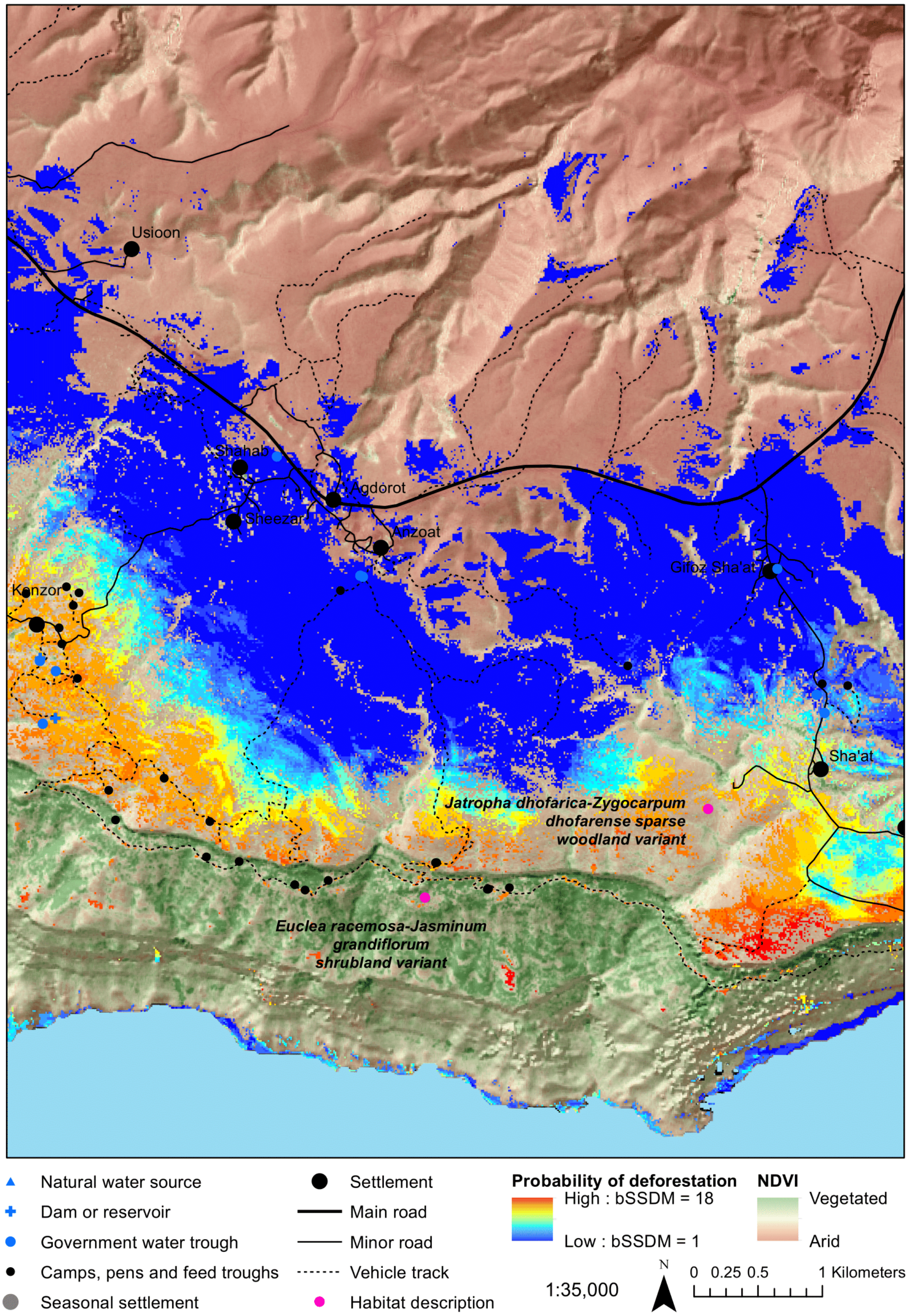
Jeddert
Tamdahn
Airdit
Burk
Wadi Durq
Keesait Amqat
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Eiraak
Hosol
Aqteedanawt
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
?
Umbroff
Premna resinosa-Hybanthus durus forest variant
Premna resinosa-Hybanthus durus forest variant
Premna resinosa-Hybanthus durus forest variant
Dodonaea viscosa subsp. angustifolia shrubland variant
Haar
Arqaf
Rakhyut
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High : bSSDM = 18
Low : bSSDM = 1
NDVI
Vegetated
Arid
1:50,000
N
0 0.5 1 2 Kilometers



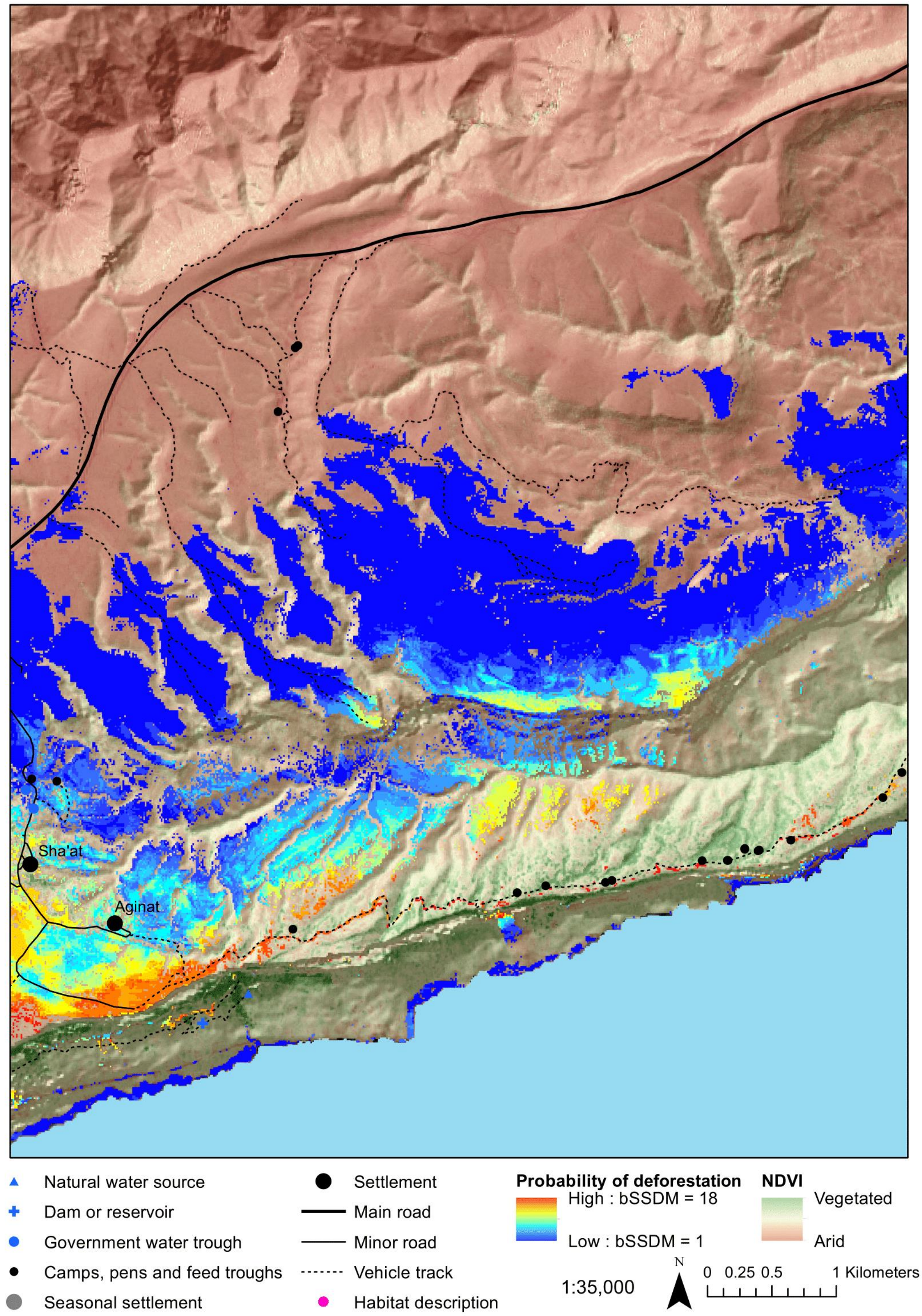
Keezait Eisoq
Thatinity/Ashkol
Ahmarut
Shab Esaaeb
Qarhanaati
Mukurah
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Premna resinosa-Hybanthus durus forest variant
Premna resinosa-Hybanthus durus forest variant
Rakhyut
Al Hawta
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High : bSSDM = 18
Low : bSSDM = 1
NDVI
Vegetated
Arid
N
1:45,000
0 0.5 1 2 Kilometers



Kazamim
Akfiran
Sahad
Arif
Mukurah
Dahaq
Airkat
Eitoq
Hasal
Kenzor
Dodonaea viscosa
subsp. angustifolia
shrubland variant
Euclea racemosa-Jasminum
grandiflorum
shrubland variant
Al Hawta
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High : bSSDM = 18
Low : bSSDM = 1
NDVI
Vegetated
Arid
1:35,000
N
0 0.25 0.5 1 Kilometers



Usioon
Shahab
Agdorot
Sheezar
Anzoat
Gifoz Sha'at
Kenzor
Sha'at
Jatropha dhofarica-Zygocarpum dhofarense sparse woodland variant
Euclea racemosa-Jasminum grandiflorum shrubland variant
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High : bSSDM = 18
Low : bSSDM = 1
NDVI
Vegetated
Arid
1:35,000
N
0 0.25 0.5 1 Kilometers


Sha'at
Aginat
Natural water source
Dam or reservoir
Government water trough
Camps, pens and feed troughs
Seasonal settlement
Settlement
Main road
Minor road
Vehicle track
Habitat description
Probability of deforestation
High : bSSDM = 18
Low : bSSDM = 1
NDVI
Vegetated
Arid
1:35,000
N
0 0.25 0.5 1 Kilometers