Surface footprint of mining in South African Grasslands: proximity and threat to agriculture and biodiversity

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By

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Summary

Mining, prevalent in the South African grasslands, can have a wide range of impacts on surrounding regions, over long distances. This includes impacts that can decrease local biodiversity and agricultural productivity. Therefore, it is important to understand the potential impacts mining in the grasslands can have on biodiversity and agriculture. In this dissertation, I investigate if mining poses a disproportionate risk to the high-value agricultural land and biodiversity priority zones within the South African grasslands using mapping approaches. To determine whether the mining area within the high-value agricultural area, and near biodiversity priority zones, was disproportionate, the location of mines was randomised across the grasslands 1000 times, creating a null model of mine distribution in the grasslands. This allowed for the comparison of the observed, and the expected (based on the 1000 randomisations) mine cover within, and surrounding high value agricultural and biodiverse areas. Additionally, the high-value agricultural water sources and river courses, at risk of pollution from mining, were identified using buffers surrounding mines, and information from hydrologically condition digital elevation models. There was disproportionate mining area within sorghum and sunflower high-value agricultural land, but not maize or soybean high-value agricultural land. The water pollution risk mining posed varied per crop, ranging between 25% and 74%, with the highest pollution risk seen in water sources and rivers surrounding sunflower high-value agricultural land water. In biodiversity priority zones, I found evidence of disproportionate mine cover within 1-10 km of protected area boundaries, with 44.84% of all mining area occurring within ten kilometres of protected areas. Additionally, mines occurred within 99 legally protected areas. I found disproportionate mine cover within one kilometre of wetland boundaries; additionally 96.84% of all mining activities occur within ten kilometres of wetlands. In contrast, there was no evidence for disproportionate mine cover within, or surrounding Important Bird Area, with only 15.58% of mine area within 10 km of important bird areas. Mining seems to especially threaten the high-value agricultural land of sorghum and sunflower Furthermore, protected areas appear to be at a disproportionate risk from the long-distance impacts of mining, rather than the short-distance impacts. This is supported by the Protected Areas Act (2003) which only references managing the impacts of land-uses directly bordering protected areas. Additionally, mining appears to pose a large risk to bordering wetlands. In contrast, the high-value agricultural land of maize and soybean, and Important bird areas, appear to be less threatened by mining, than expected from the null models. The long-distance effects of mining need to be incorporated into land-use, and agricultural production and conservation planning. This study justifies the growing concern in literature, and in policy, regarding the long-distance, and long-term impacts of mining on agricultural productivity, and biodiversity.

LIST OF ABBREVIATIONS

BPZ	Biodiversity priority zones
HVAL	High-value agricultural land
PAs	Protected areas
IBAs	Important bird areas
ESR	Endemic species richness
PADDD	Protected area downgrading, downsizing, and degazettement
EIA	Environmental Impact Assessment
GIS	Geographic information system
DEM	Digital elevation map
NLCM	National land cover map
CSIR	Council for Scientific and Industrial Research
NRF	National Research Foundation
SANORD	Southern African-Nordic Centre
IFC	International Finance Corporation
IUCN	International Union for Conservation of Nature
TGCI	Temperate Grasslands Conservation Initiative
SDGs	Sustainable development goals
SA	South Africa
GDP	Gross domestic profit

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Introduction

Mining is an important part of the global economy, with numerous positive impacts, as it stimulates economic growth, promotes foreign investment, and assists in poverty alleviation through job creation (Broadman & Isik, 2007). Mining can also result in infrastructural improvements, allowing for improved transport networks and infrastructural access (Fitzherbert et al., 2008; Sayer, Ghazoul, Nelson, & Klintuni Boedhihartono, 2012). For example, in agricultural communities, improved infrastructure and transport networks can results in increased net agricultural output due to faster transport of agricultural products, reducing spoilage (Gajigo & Lukoma, 2011). The positive effects of mining, such as increased foreign investment or improved transport networks, are especially noticeable in remote and developing regions (Broadman & Isik, 2007). In all, mining can benefit local communities due to increased access to resources and economic connectivity (Faye, McArthur, Sachs, & Snow, 2004), hence its important economic role in many countries.

The suite of positive impacts stemming from mining activities are, however, accompanied by various negative consequences, contributing to the current global biodiversity crisis (S. L. Maxwell, Fuller, Brooks, & Watson, 2016), and potentially impacting long-term food security. Mining affects both biodiversity and agricultural productivity through effects such as air, water, and soil pollution, and transformation of land (Sandker, Ruiz-Perez, & Campbell, 2012).

For example, mining can lead to sterilization and loss of topsoil, soil erosion, soil compaction and subsidence, decreased soil water retention, an absence of soil-forming fine materials, and leaching of soil nutrients (Bradshaw, 1997; Chamber of Mines of South Africa & Coalteck Research Association, 2007; Dudka & Adriano, 1997; J. Harris, Birch, & Short, 1989; J. A. Harris, Birch, & Palmer, 1996; Limpitlaw, Aken, Lodewijks, & Viljoen, 2005; Rethman, 2006; Tongway & Ludwig, 2011). Mining can also lead to the introduction and promotion of weed and invasive species, which can compete with native, and crop plant species (Li et al., 2010). Mine structures, especially the erection of fencing can also result in habitat fragmentation (Department of Environmental Affairs, Department of Mineral Resources, Chamber of Mines, South African Mining and Biodiversity Forum, & South African National Biodiverity Institute, 2013; Edwards et al., 2014; Sonter, Ali, & Watson, 2018), which, along with mining activities such as vegetation clearing, leads to the loss of accessible habitat for local species (Department of Environmental Affairs et al., 2013;

Edwards et al., 2014; Sonter et al., 2018). Mining has also been shown to lead to increased poaching in surrounding areas, both as a source of food, and profit (Alamgir et al., 2017). Furthermore, mining activities can also have extensive effects on local hydrology (Bebbington & Williams, 2008; Tiwary, 2001), altering water availability and hydrological cycles (Tiwary, 2001), leading to water pollution (Adler, Claassen, Godfrey, & Turton, 2007; Department of Environmental Affairs et al., 2013; Dudka & Adriano, 1997), and water acidification from e.g. acid mine drainage (Roback & Richardson, 1969; Winterbourn, McDiffett, & Eppley, 2000). Additionally, whilst rare, extreme events, such as the failure of two tailings dams in Brazil, are recent examples of large-scale water pollution, resulting in total biological degradation of downstream ecological communities, with long-term effects (Asner, Llactayo, Tupayachi, & Luna, 2013; Department of Environmental Affairs et al., 2013; Li et al., 2010). Many of the impacts of mining are long-lasting. Some radioactive or trace elements can persist in soil and water for decades, and possibly centuries after mine closure (Dudka & Adriano, 1997; Kabata-Pendias, 1993). Similarly, the effects of mining on soil fertility and site soil profile are often irreversible (Chamber of Mines of South Africa & Coalteck Research Association, 2007). All of these impacts can greatly reduce the agricultural value of mined land and surrounding regions, rendering it unviable, or less productive, for agriculture (see Chapter 1) (Chamber of Mines of South Africa & Coalteck Research Association, 2007; Limpitlaw et al., 2005), and can lead to a drastic decrease in local biodiversity (see Chapter 2) (Sonter, Barrett, & Soares-Filho, 2014; Wickham et al., 2013).

A particular difficulty of many biodiversity-mining management systems is that many of the impacts of mining vary greatly between mine types. For example, surface mining results in considerable overburden (Kesler, Simon, & Simon, 2015), whereas underground mining, which produces less overburden, can cause large-scale water pollution (Akcil & Koldas, 2006). Additionally, even within the same type of mining process, some mined minerals can result in more extensive environmental damage. For example, surface copper mining produces the largest amount of waste material (i.e. overburden and tailings) of any metal mining activity, and involves some of the largest earth-moving operations in the world, causing extensive mechanical soil damage (Dudka & Adriano, 1997; Jolly, 1985). The geographic reach of mining impacts is also highly variable. For example, in remote regions, mine development might require the additional construction of considerable support infrastructure, such as transport networks (i.e. roads, railways, shipping ports), facilities for mining staff (Alamgir et al., 2017; Lechner, Chan, & Campos-Arceiz, 2018), water and electricity access, and, in some cases, tailings dams (Salomons & Förstner, 1988). These ancillary infrastructures greatly increase the footprint of mining activities (Edwards et al., 2014; Sonter et al., 2018). Additionally, accidental escape of pollutants from mine sites, such as trace metals (Pb, As, Zn) and petroleum, can pollute both downstream and underground water sources far beyond the boundaries of the mining project (Razo, Carrizales, Castro, Díaz-Barriga, & Monroy, 2004). Even then, the distance that mines can impact surrounding areas depends strongly on local conditions. For example, in high-rainfall regions highly mobile metals (e.g. Pb, As, and Zn), can travel hundreds of kilometres downstream, whereas in low precipitation environments they might not escape past the mine boundary (Durán, Rauch, & Gaston, 2013). Lastly, the impact of mining on biodiversity can vary drastically between different regions, due to differences in mineral and biodiversity spatial overlap, as well as changes and differences in mining practices, legislature and conservation practices (Jenkins & Yakovleva, 2006). As a consequence, it is difficult to predict the collective impacts of mining activities, and often impacts might be underestimated (Edwards et al., 2014; Sonter et al., 2018; Sonter et al., 2017).

Unfortunately, the underestimation of the impacts of mining is often an issue seen in mine planning and regulation. For example, in international-standard Environmental Impact Assessment (EIA) methods, often only the direct, local consequences of mining on biodiversity (e.g. vegetation clearing, soil disturbance) are considered (Sonter et al., 2018), and the long-distance and long-term impacts (e.g. increased poaching, introduction of alien species) are largely ignored (Department of Environmental Affairs et al., 2013; Laurance, 2008; Weng et al., 2013). Further, strict access control to mine sites, and lack of transparency within the mining sector, prevents efforts to assess the potential impacts of mining activities, and the distances at which they can affect surrounding areas.

Even though methods for the remediation of mine impacts have been developed, they are often either not correctly applied, not applied at all, or ineffective at restoring the previous biodiversity or agricultural value of the land (Department of Environmental Affairs et al., 2013; J. A. Harris et al., 1996; Ruiz-Jaen & Aide, 2005). For example, there are more than 6 000 and 50 000 abandoned mines in South Africa and Australia respectively, most of which have seen very little, or no remediation efforts (Auditor-General South Africa, 2009; Department of Mineral Resources, 2009; Unger, Lechner, Glenn, Edraki, & Mulligan, 2012). Even if an attempt was made to rehabilitate these abandoned mines, the cost would be prohibitive (Auditor-General South Africa, 2009; Weyer, Truter, Lechner, & Unger, 2017). Hence, even though mining accounts for only a fraction of global land-use, relative to other land-uses (Sonter et al., 2018), and rehabilitation plans are often in place, the disturbance potential of mines on both natural and agricultural systems is second only to urban development (Tongway & Ludwig, 2011).

The potential impacts of mining on agriculture are especially worrying, as the world is facing growing food security instability, particularly in Sub-Saharan Africa (Hall, Dawson, Macdiarmid, Matthews, & Smith, 2017). Sub-Saharan African agriculture has always faced numerous challenges with conventional crop production, and hence, food security. The leading causes of food insecurity in Africa include low precipitation, water availability, and poor soil quality. In addition, roughly 30% of Africa's population (> 218 million people) experiences daily food insecurity (Funk & Brown, 2009). This situation will likely worsen as the African population is expected to increase to 2.5 billion by 2050 (United Nations Department of Economic and Social Affairs, 2019). Furthermore, much of sub-Sahara African agricultural land is at high risk of degradation (Oldeman, Hakkeling, & Sombroek, 1991), whilst climate change will decrease crop yields (Department of Environmental Affairs, 2011; Thornton, 2003). Hence, as food insecurity in Africa is expected to increase by at least 43% by 2030 should agricultural yields not double over the period (Funk & Brown, 2009; Stockholm Environmental Institute, 2005), it is important to understand all the potential threats faced by agriculture. Africa contains between 30-40% of the earth's terrestrial mineral resources, many of which are as of yet largely unexploited. Hence, Africa is likely still on the verge of a rapid large scale mining boom (Taylor, Schulz, & Doebrich, 2009). South Africa has a long history of mining, with much of its early economic structure built on the production of gold and diamonds. As a consequence, mining led to the establishment of much of the country's early mass infrastructure, due to the industry need for roads, railways, harbours, water, housing, food, and labour (Van Rhjin, 1959).

In the 1950s, mining activities accounted for 18% of the country's gross domestic profit (GDP), compared to the 24% of the manufacturing sector (Van Rhjin, 1959). However, as stated by the governor of the South African Reserve Bank at the time, "the importance of the mining industry cannot be judged solely by its direct net contribution to the national income": at the time, mineral resources accounted for over 50% of South Africa's exports (Van Rhjin, 1959). Though the economic contribution of the mining sector has receded in recent years, contributing to only roughly 8% of the country's GDP, or just over R 352 billion (Minerals Council South Africa, 2019; South African Market Insight, 2019), the sector remains an important contributor to the country's GPD, and foreign investment, employing over 455 000 people in 2018 (Minerals Council South Africa, 2019).

In this study, I assess the risk mines pose to biodiversity and agriculture in the South African grassland biome. The South African grassland biome accounts for roughly 30% of the country's terrestrial land area (South African National Biodiversity Institute, 2018), and provides a wide range of important ecosystem services, such as contributing to the global water supply, carbon sequestration, and pollination (Bengtsson et al., 2019; Carlier, Rotar, Vlahova, & Vidican, 2009; Lemaire, Hodgson, & Chabbi, 2011; O'Mara, 2012; Sala & Paruelo, 1997). The South African grasslands are also critically important for South African biodiversity. For example, the grasslands contain five centres of plant endemism, namely the Drakensberg Alpine, Barberton, Wolkberg, Sekhukhune and Soutpansberg centres (Van Wyk & Smith, 2001). Additionally, 52 of the 122 important bird areas within South Africa are located within the grasslands (BirdLife South Africa, 2015; SANBI, 2013).

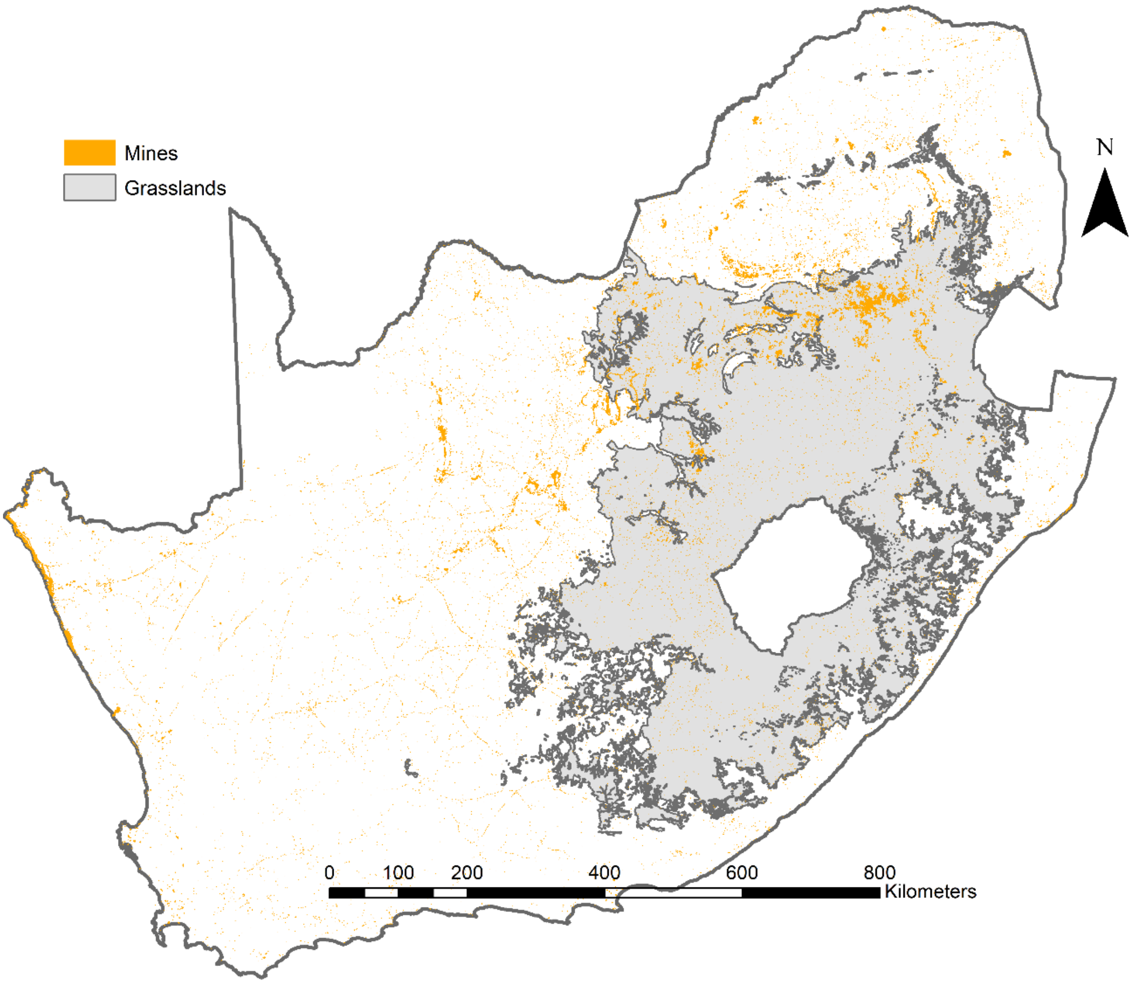

Figure 1. Boundaries of the surface footprint of mines in the South African grasslands (GeoTerraImage, 2019; South African National Biodiversity Institute, 2018).

However, South Africa's grassland biome is one of the country's most threatened biomes, with more than 35% of the biome irreversibly transformed to non-natural land-use

(Reyers, Nel, Egoh, Jonas, & Rouget, 2005). One of the largest causes of transformation in the grassland is agriculture, which also poses one of the largest future potential threats to grassland habitats in South Africa (Neke & Du Plessis, 2004). However, mining also poses a particular threat in the grasslands, as the grasslands are the heavies mined biome in South Africa (Fig. 1) (GeoTerraImage, 2019). Mining is also still predicted to expand rapidly in South Africa. For example, 63% of the South African Mpumalanga province's area, which accounts for a large section of the South African grasslands, has received applications for mineral prospecting rights (Fig. 2). Furthermore, less than 3% of the grassland biome of South Africa is under formal protection (SANBI, 2013). Based on the total habitat loss, habitat fragmentation, and the additional anticipated future threats, the South African grasslands have been classified as critically endangered (Olson & Dinerstein, 1998; Reyers, Fairbanks, Van Jaarsveld, & Thompson, 2001), and the South African biome most urgently in need of conservation (South African National Biodiversity Institute, 2018).

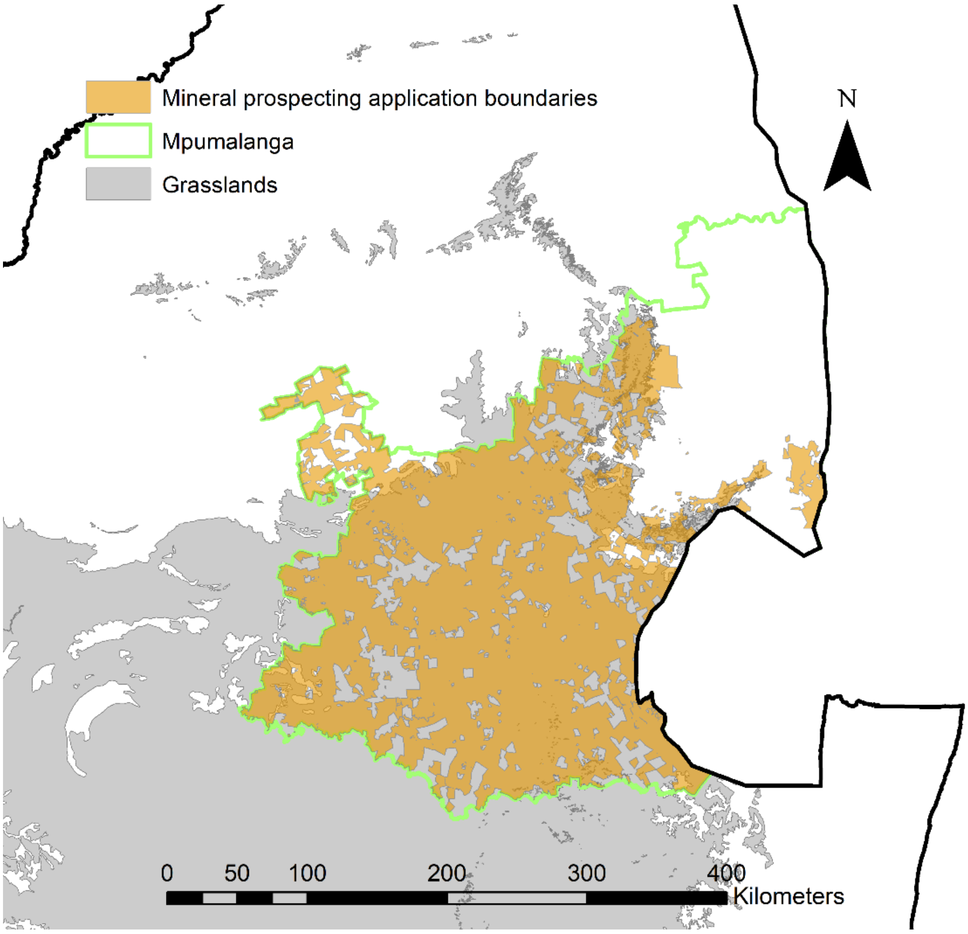

Figure 2. Boundaries of mineral extraction prospecting applications within the Mpumalanga province, South Africa (Prospecting data obtained from Mervyn Lötter, Mpumalanga Parks and Tourism, in December 2019).

The grasslands of South Africa are also critically important for agriculture, accounting for over 55% of the country's developed agricultural area (GeoTerraImage, 2019; South African National Biodiversity Institute, 2018), and accounts for 80% of maize, 73% of soybean, 72% of sorghum, and 76% of sunflower (all economically important crops) high-value agricultural land in South Africa (ISCW, 2006). High-value agricultural land is scarce in South Africa, with only 12% of South African land being suitable for the production of rain-fed crops (Goldblatt, 2015). Additionally, agriculture in the South African grasslands faces strong pressure from mining. For example, approximately half of the South African high-value agricultural land is within Mpumalanga, where approximately 12% of its high-value agricultural land is projected to be lost at current mining rates, and a further 13.6% already set aside under prospecting rights (Bureau for Food and Agricultural Policy, 2012). Furthermore, though mining activities cover less than 1% of the South African land area, mining activities contribute to 8% of South Africa's GDP (South African Market Insight, 2019). In contrast, agriculture contributes 2.6% to GDP but covers 11% of the land area (South African Market Insight, 2019). This economic disparity can lead to mining in high-value agricultural land, despite the scarcity, and importance to food security, of high-value agricultural land.

Mining corporations are increasingly investing in conservation offsets, better ecological management and restoration of mining sites (Reed & Miranda, 2007). In some cases, mines even lead to improved conservation, relative to many other land use types (e.g. agriculture) due to large exclusion zones, intended as security measures, around such sites (Loock, Williams, Emslie, Matthews, & Swanepoel, 2018). Mines can also bring in financial resources that can potentially contribute to the creation and maintenance of conservation zones, and stimulate local agriculture. In some countries, mining sites also follow much stricter environmental regulation than other land-use types (McKinney, 2002; Sax & Gaines, 2003).

However, for the ecological and agricultural remediation to match ecological damage of mines, better knowledge of the risk mines pose is required. Currently, many mining-biodiversity management strategies are centred around the use of legal mandates and economic incentives (International Finance Corporation, 2016). Many of these management styles can be effective, at least in theory. For example, requiring EIAs as a requisite for stock exchange listings and financial support from the International Finance Corporation (IFC) (International Finance Corporation, 2016) has the potential to increase the implementation of EIAs by multinational mining corporations (Edwards et al., 2014). However, these strategies are of little consequence if these methods are not based on research specific to that area, allowing for optimal planning of biodiversity integration and conservation during mining practices, and restoration of agricultural value following mine closure (Bridge, 2004; Department of Environmental Affairs et al., 2013; Edwards et al., 2014). Further, there are many small scale national, local or 'artisanal' mining operations that do not confer to any environmental protection norms (Hentschel, Hruschka, & Priester, 2002; Hinton, Veiga, & Veiga, 2003). Though these mining operations are usually smaller than to those of international mining companies, there can be many more of this type of mining operation (Hentschel et al., 2002). Therefore, to effectively mitigate the loss of biodiversity and agricultural value during/after mining operations will require both incentives to mitigate biodiversity and agricultural value loss, as well as practical guidelines for the protection of biodiversity and agricultural value during, and restoration following mining activities. However, the first steps in developing such mitigation and restoration procedures are to better understand the impact mining has already had on biodiversity and agriculture.

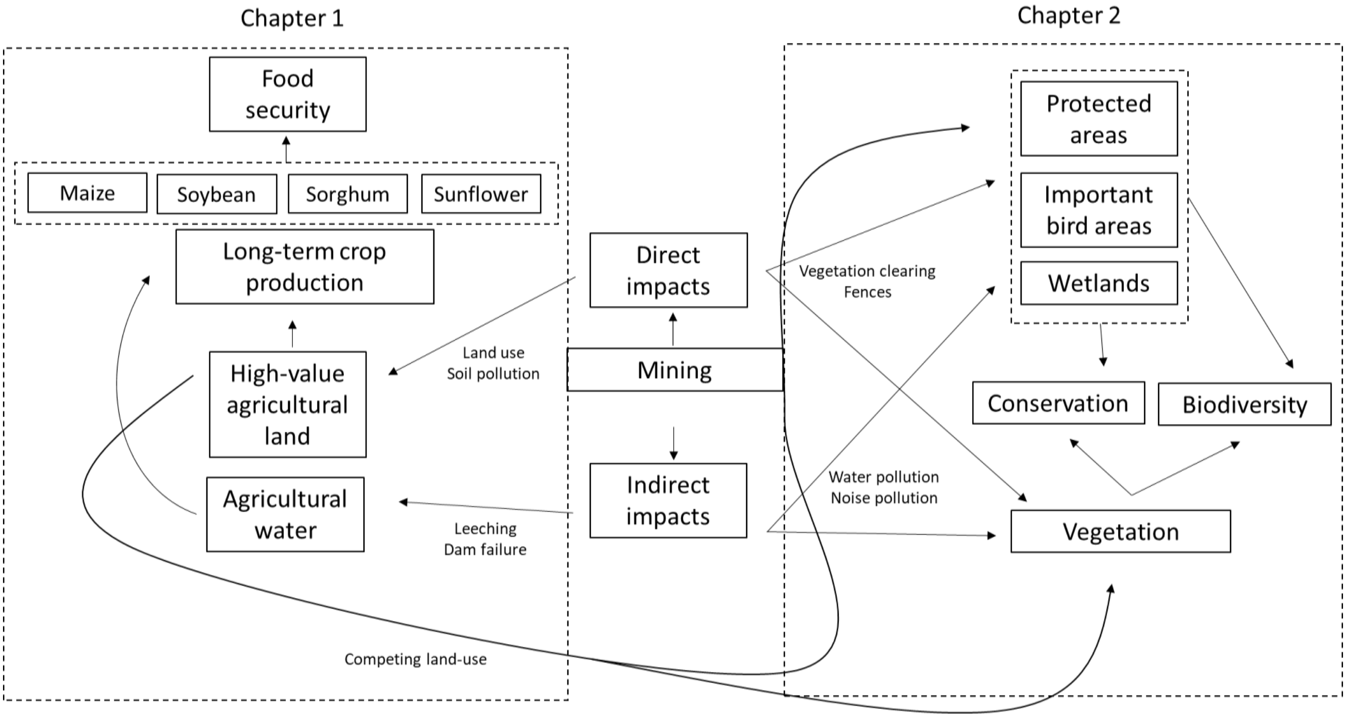

Figure 3. Some of the effects of the direct, and indirect impacts of mining on agriculture and biodiversity in South Africa.

Therefore, the goals of this study were to assess the potential impacts mining has already had, and the risk mining poses, to agriculture and biodiversity in South Africa's grasslands. Hence, in this dissertation, I tested whether mining in the South African grasslands poses a disproportionate risk to areas important for agriculture and biodiversity (Fig. 3). To achieve this, I assessed the pattern of mine establishment within high-value agricultural land, and identified the high-value agricultural water sources and river courses at risk of pollution from mining (Chapter 1). Additionally, to determine if mining poses a disproportionate risk to important biodiversity area, I assessed the proximity of mines in the South African grasslands to biodiversity priority zones (i.e. protected areas, important bird areas, and wetlands), and the relationship between the conservation value of a vegetation type and mining within its borders (Chapter 2). In this study, I used a macroecological approach, using Geographic Information System (GIS) to assess mining impacts using large-scale publicly available datasets. This allowed me to assess the potential risk mining poses for the entire grassland biome of South Africa; this work thus provides a first quantitative assessment of mining impacts across a large geographic area in South Africa. A limitation of this approach is that multiple assumptions and generalisations, discussed in the respective chapters, had to be made.

Though mining is prevalent in the South African grasslands, a region important for South African agriculture and biodiversity, little information is available on the risks mining pose to agriculture and biodiversity, at a large scale. This dissertation contributes to the understanding of the risks mining pose to important agricultural and biodiversity areas in South Africa. The dissertation shows, that currently, the high value agricultural land of sorghum, and sunflower are at a disproportionate risk from the impacts of mining. Additionally, areas surrounding protected areas and wetlands, both of high conservation concern, are also at a disproportionate risk from the impacts of mining. This work highlights the threats from mining to agricultural land important for future food security as well as biodiversity conservation. Additionally, this work can be used to inform future land use planning by placing greater consideration on the potential long-distance impacts of mining on South African biodiversity and conservation.

Chapter 1

Surface footprint of mining activities in South African grasslands: potential impacts on agriculture

1. Introduction

The world is facing growing food security instability (Hall et al., 2017). Though the area of both arable and permanent crops (9.6%), and permanent meadows and pastures (8.7%) has increased over the last 50 years, such increases mainly took place between 1961 and 1991 (Food and Agriculture Organisation of the United Nations, 2008; O'Mara, 2012). The lack of agricultural area increases since 1991 is partly attributed to a loss of agricultural land due to increased urbanisation and industrialisation, especially on high-value agricultural land. As a result, new agricultural land, often of lower agricultural value, must be developed in order to maintain the current total agricultural area, and meet food demands (O'Mara, 2012). This lower value agricultural land also requires increased investment, i.e. fertiliser and/or irrigation to meet the production values of lost higher value agricultural land (O'Mara, 2012), which many farmers cannot afford (Food and Agriculture Organisation of the United Nations, 2008). Hence, investigating and mitigating the impact of industrial land-uses on the existing high-value agricultural regions of the world has become increasingly important.

Mining is one land use that can have severe impacts on the agricultural productivity of land, and hence food security. For example, mining can lead to sterilization and loss of topsoil, soil erosion, soil compaction and subsidence, decreased soil water retention, an absence of soil-forming fine materials, leaching of soil nutrients, and promotion of weed species (Bradshaw, 1997; Chamber of Mines of South Africa & Coalteck Research Association, 2007; J. Harris et al., 1989; J. A. Harris et al., 1996; Limpitlaw et al., 2005; Rethman, 2006; Tongway & Ludwig, 2011). Mining activities can also have extensive effects on local hydrology (Bebbington & Williams, 2008; Tiwary, 2001), altering water availability and hydrological cycles (Tiwary, 2001), leading to water pollution (Adler et al., 2007; Department of Environmental Affairs et al., 2013; Dudka & Adriano, 1997), and water acidification from e.g. acid mine drainage (Roback & Richardson, 1969; Winterbourn et al., 2000). Many of the impacts of mining are long-lasting. For example, some radioactive or trace elements can persist in soil and water for decades, and possibly centuries after mine closure (Dudka & Adriano, 1997; Kabata-Pendias, 1993). Similarly, the effects of mining on soil fertility and site soil profile are often irreversible (Chamber of Mines of South Africa & Coalteck Research Association, 2007). All of these impacts can greatly reduce the agricultural value of land that is mined and surrounding regions, rendering it unviable, or less productive, for agriculture (Chamber of Mines of South Africa & Coalteck Research Association, 2007; Limpitlaw et al., 2005).

Even though methods for the remediation of mine impacts have been developed, they are often either not correctly applied, not applied at all, or ineffective at restoring the previous agricultural value of the land (Department of Environmental Affairs et al., 2013; J. A. Harris et al., 1996; Ruiz-Jaen & Aide, 2005). For example, there are more than an estimated 6 000, and 50 000 abandoned mines in South Africa and Australia respectively (Auditor-General South Africa, 2009; Department of Mineral Resources, 2009; Unger et al., 2012), most of which have seen very little, or no remediation efforts. Even if an attempt was made to rehabilitate these abandoned mines, the cost would be prohibitive (Auditor-General South Africa, 2009; Weyer et al., 2017). Hence, even though mining accounts for only a fraction of global land-use, relative to other land-uses (Sonter et al., 2018), and rehabilitation plans are often in place, the disturbance potential of mines is second only to urban development (Tongway & Ludwig, 2011), and can lead to an extensive, long-term decrease in agricultural production.

Sub-Saharan African agriculture has always faced numerous challenges with conventional crop production, and hence, food security. Though African crop output is increasing, this increase is largely attributed to agricultural expansion, rather than increased agricultural productivity, with the average annual crop yield per hectare remaining essentially consistent (Food and Agriculture Organisation of the United Nations, 2008). The leading causes of food insecurity include low precipitation, water availability, and poor soil quality. For example, the average grain crop yield in sub-Saharan Arica since the 1960s has largely been below 1 t.ha^{-1}, compared to yields of 2.5 t.ha^{-1} and 4.5 t.ha^{-1} for South Asia and East Asia respectively. In addition, roughly 30% of Africa's population (> 218 million people) experiences daily food insecurity (Funk & Brown, 2009). This situation will likely worsen as the African population is expected to increase to 2.5 billion by 2050 (United Nations Department of Economic and Social Affairs, 2019). Much of sub-Sahara African agricultural land is at high risk of degradation (Oldeman et al., 1991), and climate change will decrease crop yields (Department of Environmental Affairs, 2011; Thornton, 2003). Hence, food insecurity in Africa is expected to increase by at least 43% by 2030 (Funk & Brown, 2009) should agricultural yields not double over the period (Stockholm Environmental Institute, 2005).

In this study, I assess the risk mines pose to agriculture in the South African Grassland biome. Grasslands account for roughly 40% of the earth's terrestrial surface and (Lemaire et al., 2011; Suttie, Reynolds, & Batello, 2005; White, Murray, Rohweder, Prince, & Thompson, 2000), accounting for an estimated 80% of global agricultural land (Boval & Dixon, 2012; I. A. Wright, Jones, Davies, Davidson, & Vale, 2006). The South African Grassland biome accounts for roughly 30% of the country's terrestrial land area (South African National Biodiversity Institute, 2018). Only 12% of land is suitable for the production of rain-fed crops (Goldblatt, 2015); of this the grasslands comprise an important centre of agricultural activities, accounting for over 55% of the country's agricultural area (GeoTerraImage, 2019; South African National Biodiversity Institute, 2018).

Though mining activities cover less than 1% of South African land area, mining activities contribute to 8% of South Africa's GDP, this is in contrast agriculture which contributes 2.6% to GDP but covers 11% of land area (South African Market Insight, 2019). Therefore, mining offers high short-term economic returns, as opposed to the higher long-term returns of agriculture. This economic power disparity, combined with the low availability of high-value agricultural land, has the potential to lead to disproportionate loss, and a lack of conservation, of high-value agricultural land, in lieu of short-term high economic return, known as the 'Resource Curse' (Auty, 2002).

Therefore, in this study, I specifically determined whether HVAL is disproportionately targeted for mining in South Africa's grasslands. As there are no *a priori* indicators that mine establishment should be target HVAL, I do not expect to see disproportionate mining in HVAL, both currently, or as early as 1990. However, if there is evidence of disproportionate mining in HVAL, it would indicate that food security should be given more consideration during mine land-use planning. Additionally, I do not expect to see difference in mine cover within different crop HVAL. Hence, I tested whether there is disproportionate mining area within the HVAL of four economically important crops, i.e., maize, soybean, sorghum, and sunflower, for two time periods, namely 1990 and 2018, and for new mines established between 1990 and 2018. Additionally, I determined the area and location of water sources and river courses in HVAL of the four crops that are at potential risk of water pollution from mining.

2. Methods

First, I tested whether there was disproportionate mine cover within the HVAL (as opposed to lower value agricultural lands) in 1990 and 2018, and whether there has been a shift towards mine establishment in HVAL between 1990 and 2018, for four economically important crops (i.e. maize, soybean, sorghum, and sunflower). The surface footprint of mines (hereafter referred to collectively as mines) were extracted from the 1990 and 2018 South African National Land Cover Maps (GeoTerraImage, 2014, 2019) (Table 1, Table S1.1.1). Additionally, the area of all mines established between 1990 and 2018 was extracted by excluding the mines identified in the 1990 national land cover map from those identified in the 2018 national land cover map, henceforth referred to as ‘new mines (1990-2018)’. In these layers, mines are mapped as any areas that include the surface footprint of mines (i.e. extraction sites, dumps, tailings, overburden), but excludes underground mines (Table S1.1). The 1990 land cover map was created using semi-automated mapping models and multi-seasonal Landsat 5 satellite imagery from between 1989 and 1991, and the 2018 land cover map using automated mapping models and multi-seasonal Sentinel 2 satellite imagery from 1 January to 31 December 2018 (GeoTerraImage, 2014, 2019).

The Institute for Soil, Climate, and Water of the South African Agricultural Research Council created maps of the agricultural potential (unsuitable, suitable, moderate, marginal, high) for maize, soybean, sorghum, and sunflower (ISCW, 2006). Agricultural potential was determined by matching the best available data on the crop requirements (i.e. climate and soil), and the respective environmental attribute as defined in the National Asset natural resource information systems (ISCW, 2006), for South Africa. In the creation of these maps, the agricultural potential was determined based solely on the crop requirements and environmental attributes, and did not take existing land-uses into consideration (ISCW, 2006). From these maps I extracted the regions classified as high-value for each of the four respective crops (Table 1). It is also important to note that there can be overlap between the HVAL for the four crops, due to overlapping abiotic tolerances.

Table 1. Data extracted for this study, and the names and citations of the source datasets from which these data were extracted. The resolution of the original data are also shown for raster data.

Data used	Source Dataset	Spatial resolution	Citation
Boundaries of mining activities 2018	**2018 National Land Cover Map**	**30m**	**GeoTerraImage (2019)**
Boundaries of mining activities 1990	**1990 National Land Cover Map**	**30m**	**GeoTerraImage (2014)**
Potential maize agricultural value	**Land Suitability for Biofuel Crops**	**Vector**	**ISCW (2006)**
Potential soybean agricultural value	**Land Suitability for Biofuel Crops**	**Vector**	**ISCW (2006)**
Potential sorghum agricultural value	**Land Suitability for Biofuel Crops**	**Vector**	**ISCW (2006)**
Potential sunflower agricultural value	**Land Suitability for Biofuel Crops**	**Vector**	**ISCW (2006)**
Hydrologically conditioned digital elevation model	**WWF HydroSHEDS Hydrologically Conditioned DEM**	**15 arc seconds**	**Lehner, Verdin, and Jarvis (2008)**
Water flow accumulation	**WWF HydroSHEDS Flow Accumulation**	**15 arc seconds**	**Lehner et al. (2008)**
Grassland permanent and seasonal rivers	**South Africa Rivers**	**Vector**	**Department of Water and Sanitation (2018)**

The extracted mine polygons were used to determine the total mine area within HVAL in 1990 and 2018, as well as the mine area within HVAL of new mines established between 1990 and 2018, for all four crop species. Thereafter, I used a randomisation procedure to test whether the observed mine area within the HVAL was more, less, or not significantly different to if the mines were randomly distributed within the grasslands of South Africa. To do this, I randomised the placement, without overlap, of mining polygons with the same number, size and shape of current mines across the study area (Durán et al., 2013). The randomisation was conducted in ArcPy by creating the same number of random points throughout the grasslands as there are individual mining polygons in the shapefiles. Next, using the function ‘UpdateCursor’, I reassigned the centroid of each polygon to the location of one of the randomly dispersed points. The randomisation was repeated 1000 times to create a null model of mine cover. For each randomisation, the area of the randomly placed ‘mines’ within the HVAL was determined.

I then tested whether the observed mining area within HVAL differed from the expected mining area in HVAL based on the null model of mine cover, i.e. the area of the randomly distributed mines (eq. 1.1). For this I used a one-sample Z-test (Sprinthall, 2014):

$$Z = \frac{\bar{y} - m_o}{\frac{\hat{\sigma}}{\sqrt{n}}} \qquad (1.1)$$

Where n is the sample size, $\hat{\sigma}$ is the standard deviation, and $\bar{y}$ is the mean for the expected mining area within the HVAL, and m_o is the observed mining area within the HVAL.

This procedure was repeated for all four crop species for the total area covered by mines in 1990 and in 2018 mines, as well as for the new mines established between 1990 and 2018.

Further, I wished to determine the potential risk mines pose to the water sources and river courses of HVAL. I assumed that pollutants originating and flowing downstream from mines would pollute water sources and rivers with negative impacts on agriculture (Arao et al., 2010; Department of Water Affairs and Forestry, 1996a; Gola, Malik, Shaikh, & Sreekrishnan, 2016). This analysis was conducted only for mines from the 2018 National Land Cover Map (GeoTerraImage, 2019). First, the elevation of the highest point of each mine was identified by the overlap of the mining polygons and a hydrologically conditioned digital elevation map (DEM) (Lehner et al., 2008). The distance at which pollutants can spread from mines varies greatly, depending in the local hydrology, mine type, and pollutant type, ranging from tens of metres to tens of kilometres (Axtmann & Luoma, 1991; Csiki & Martin, 2008; Durán et al., 2013). However, for the purpose of this study, I used ten kilometres to represent the maximum potential distance pollutants can disperser from mining sites (Durán et al., 2013; Edwards et al., 2014) (Table 2). Hence, a buffer of ten kilometres was created around the centre points of each of the identified 2018 mines (Durán et al., 2013). Many mines have mine buildings (e.g. site offices) further away from water pollution sources (i.e. mine dumps and tailings). Hence, the buffers were created around the centre of the mine, so as not to overestimate the pollution risk based on mine buildings further away from the actual pollution sources.

Table 2. Maximum detected water pollution distances for various metals. Modified from (Durán et al., 2013).

Authors	Metal	Maximum distance of water pollution from mining source (km)
Razo et al. (2004)	**Copper, gold, lead, zinc, and silver**	**5**
Telmer et al. (2006)	**Copper**	**50**
Kodirov and Shukurov (2009)	**Copper and zinc**	**4**
Lafabrie et al. (2009)	**Cobalt**	**5**
Taylor et al. (2009)	**Copper, zinc, and lead**	**30**
Huang et al. (2010)	**Copper and zinc**	**10**
Lefcort et al. (2010)	**Copper and zinc**	**2.5**
Axtman and Luoma (1991)	**Zinc**	**400**
Taylor et al. (2009)	**Zinc**	**16**
Csiki and Martin (2013)	**Arsenic, barium, chromium, copper, nickel, and lead**	**60**

Next, the HVAL within the ten-kilometre buffers around mines was extracted. The assumption was made that HVAL within the buffer area, with lower elevations would be polluted, but that higher elevation areas would not be polluted by lower-lying mines. Hence, the HVAL areas within the ten-kilometre buffer of the mines with lower elevations than the maximum elevation of the mine were extracted. Rivers and other regions of high-waterflow, have high metal mobility, and can therefore disperse pollutants over a greater distance, posing a greater risk to surrounding agriculture. Additionally, high waterflow regions are also more likely to be drawn from for crop irrigation (Oelofse, 2009; Razo et al., 2004). Therefore, the waterflow per pixel was extracted from the WWF HydroSHEDS Flow Accumulation raster (Lehner et al., 2008), for all HVAL within the ten kilometre buffer of mines. The output map therefore represents the HVAL water sources potentially at risk from pollution by mines, indicating the waterflow of the regions, thereby highlighting regions were water pollution would have the greatest potential impact.

Lastly, the output map of HVAL water sources at risk of pollution from mining was used to identify and isolate the established rivers (but not other water sources such as rivers, dams, underground water) within HVAL at risk of pollution from mining. First, all South African rivers, both permanent and seasonal, as identified by the Department of Water and Sanitation

(2018), within HVAL were extracted. Then, I identified all rivers overlapping with the above output map of all HVAL water sources at risk of pollution from mining, representing the HVAL river courses at risk of pollution from mining.

All data layers used in this study (Table 1) were either in, or reprojected to WGS 84/UTM Zone 35S prior to analysis. If not already in the correct resolution, raster datasets were resampled to a 30m resolution using nearest neighbour assignment. Additionally, all data layers were cut to the extent of the grassland biome of South Africa, as defined in the 2018 version of the Vegetation map of South Africa, Lesotho, and Swaziland (Eswatini) (Ladislav Mucina & Rutherford, 2006; South African National Biodiversity Institute, 2018). All methods described were repeated for the HVAL of all four the selected crops, i.e. maize, soybean, sorghum, and sunflower.

3. Results

Mining accounted for 0.25% of the South Africa grassland biome area. Only the ‘suitable’ (0.88%) and ‘moderate’ (1.22%) agricultural value land classes for maize and soybean, and the ‘high’ agricultural value land classes of sorghum (0.46%) and sunflower (0.6%) showed a higher proportion of their total area transformed to mining (Table 3). Mining accounted for 0.15% and 0.006% of maize and soybean HVAL, compared to the total grassland area of 0.28% transformed to mining area (Table 3). In contrast mining covered 0.46% and 0.60% of sorghum and sunflower HVAL (Table 3).

The observed mine cover within HVAL of the four crops was significantly different from the mine cover expected from the null model. Mine area was lower than expected from the null model in the HVAL of maize and soybean, in both 1990 and 2018 (Figs. 1a,-b, Table S1.2), and for new mines (1990 – 2018) (Fig. 1c, Table S1.2). For soybean HVAL, the mine cover was lower than expected from the null model, with the observed mine area only equivalent to 1.78 %, and 2.22 % of its mean expected mine area in 1990, and 2018 respectively (Table S1.2). The 1990 mine cover expected from the null model in HVAL of sorghum showed no significant difference to the observed mine cover (Fig. 1a, Table S1.2). However, in 2018, and for new mines, the observed mine cover in sorghum HVAL exceeded the cover expected from the null model (Figs. 1b-c, Table S1.2). There was significantly higher observed mine cover, than expected from the null model, in the HVAL of sunflower, in both 1990 and 2018, and for new mines (Figs. 1a-c, Table S1.2).

Table 3. Total area, area covered by mining, and percentage of the total area covered by mining, of the high, moderate, marginal, and suitable agricultural land of four crops (maize, soybean, sorghum, and sunflower), in the South African grassland biome.

Crop	**Agricultural value category**	**Total area (km2)**	**Area mined (km2)**	**Area mined (%)**
Maize	**High**	**35431.71**	**53.31**	**0.15**
	Moderate	**77689.94**	**96.27**	**0.12**
	Marginal	**77817.14**	**74.13**	**0.1**
	Suitable	**52615**	**462.59**	**0.88**
	Total	**243553.8**	**686.3**	**0.28**
Soybean	**High**	**14308.26**	**0.91**	**0.006**
	Moderate	**35878.19**	**438.2**	**1.22**
	Marginal	**61523.36**	**63.65**	**0.1**
	Suitable	**47259.54**	**69.38**	**0.15**
	Total	**158969.36**	**572.14**	**0.36**
Sorghum	**High**	**113562.06**	**525.57**	**0.46**
	Moderate	**58555.82**	**84.66**	**0.14**
	Marginal	**31812.85**	**29.04**	**0.09**
	Suitable	**38972.04**	**45.66**	**0.12**
	Total	**242902.77**	**684.93**	**0.28**
Sunflower	**High**	**86127.53**	**520**	**0.6**
	Moderate	**52461.1**	**83.58**	**0.16**
	Marginal	**30992.88**	**32.58**	**0.11**
	Suitable	**34388.96**	**41.33**	**0.12**
	Total	**203970.48**	**677.47**	**0.33**

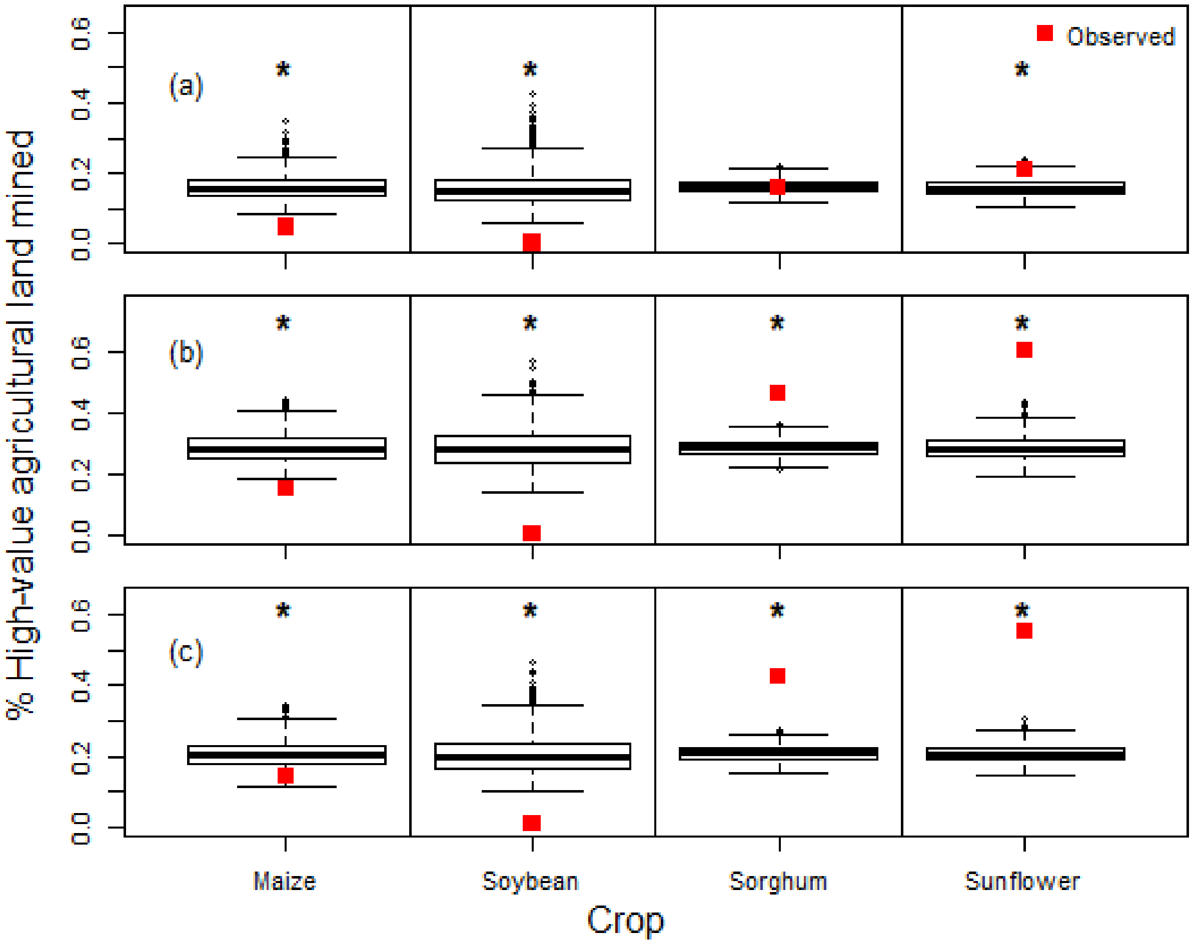

Figure 1. The percentage of the total high-value agricultural land (HVAL) area transformed to mine sites, for four economically important crops (maize, soybean, sorghum, and sunflower) within South African grasslands. The boxplots represent the expected mine cover based on 1000 randomisations of mine locations, and the red squares the observed mine cover in the HVAL of each crop for (a) 1990, (b) 2018, and (c) new mines established between 1990 and 2018. Significant difference between observed and expected mine cover are indicated by an asterisk. Note: statistical comparisons are made between each boxplot (expected mine area) and its corresponding red box (observed mine area), not between boxplots.

The observed annual mine cover increase between 1990 and 2018 in maize HVAL was 20% lower than the expected annual increase from the null models, and 97.8% lower than expected from the null model in soybean HVAL (Table S1.3). Between 1990 and 2018, sorghum and sunflower HVAL both show higher observed annual mine cover increase than expected from the null models, with their observed annual mine cover increase 240%, and 553% greater than their respective annual mine cover increases expected from the null models (Table S1.3).

The generated maps showed the proportion of HVAL water sources and river courses of each crop at risk of pollution from mines, and in the case of water sources, the flow accumulation of the respective area (Figs. 2-3). Large portions of the water sources and river courses of the HVAL for all four crops were at risk of pollution from mining (Fig. 4, Table S1.4, Figs. S1.1-1.8). Mining posed a threat to the water sources and river courses of the HVAL of all four crops in all regions of the grasslands, though generally, the more of a crop's HVAL was towards the north of the grasslands, the greater the risk mining posed. This is due to the high mine presence in many of the northern grassland regions, e.g. eMalahleni, Ogies, Johannesburg etc. Hence, mining posed the greatest threat to the water sources and river courses of sunflower and sorghum (Fig. 4, Figs. S1.5-1.8), followed closely by maize (Fig. 4, Fig. S1.1-1.2), and the least to those of soybean (Figs. 3-4, Table S1.4, Fig. S1.3-1.4).

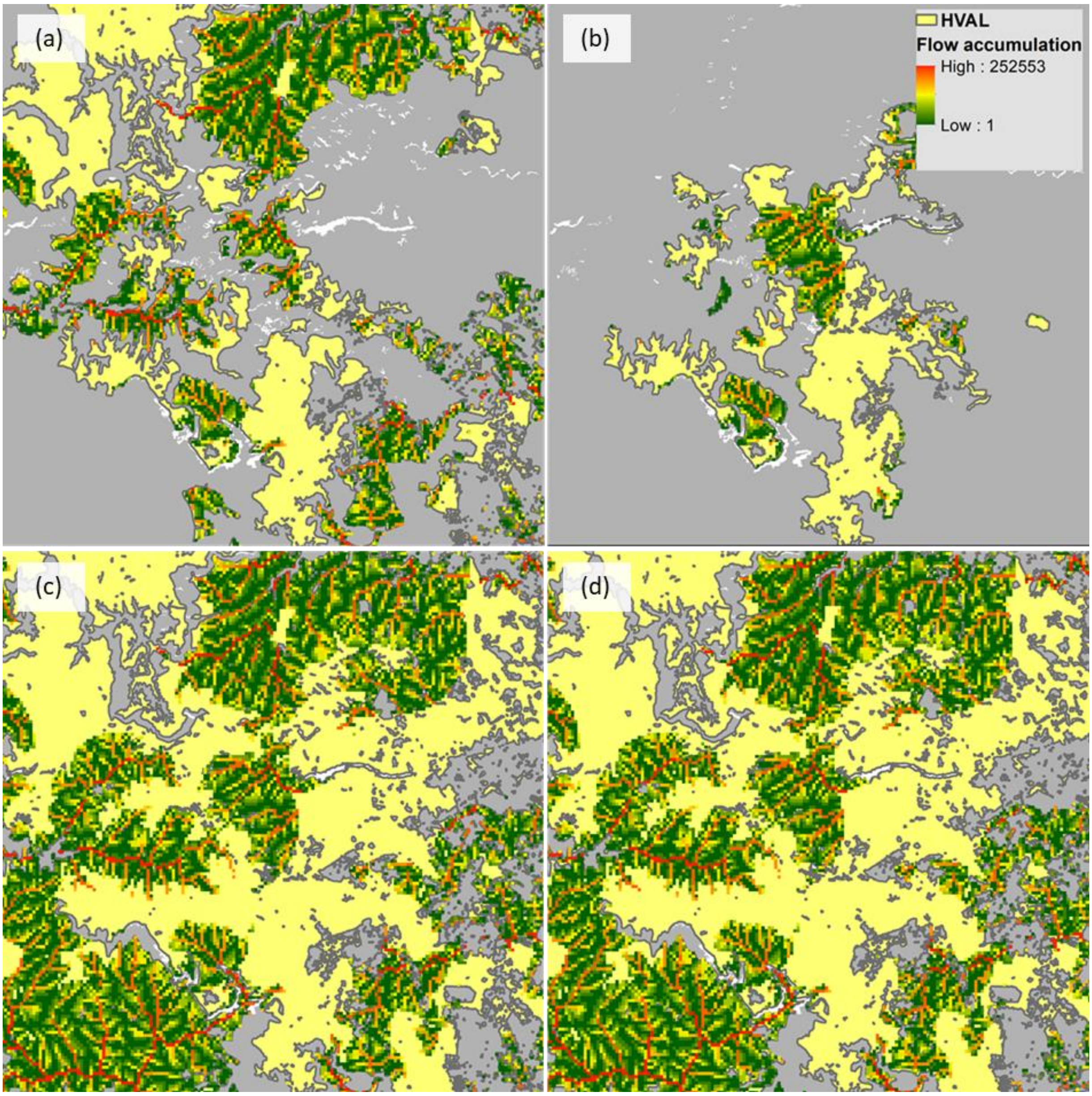

Figure 2. Sample of high-value agricultural land (HVAL) water source pollution risk by mines, for (a) maize, (b) soybean, (c) sorghum, and (d) sunflower, for a watershed near Wakkerstroom, South Africa. Areas with higher flow accumulation would increase pollutant mobility, and are more likely to be directly drawn from for crop irrigation, increasing the risk pollution of such an area poses.

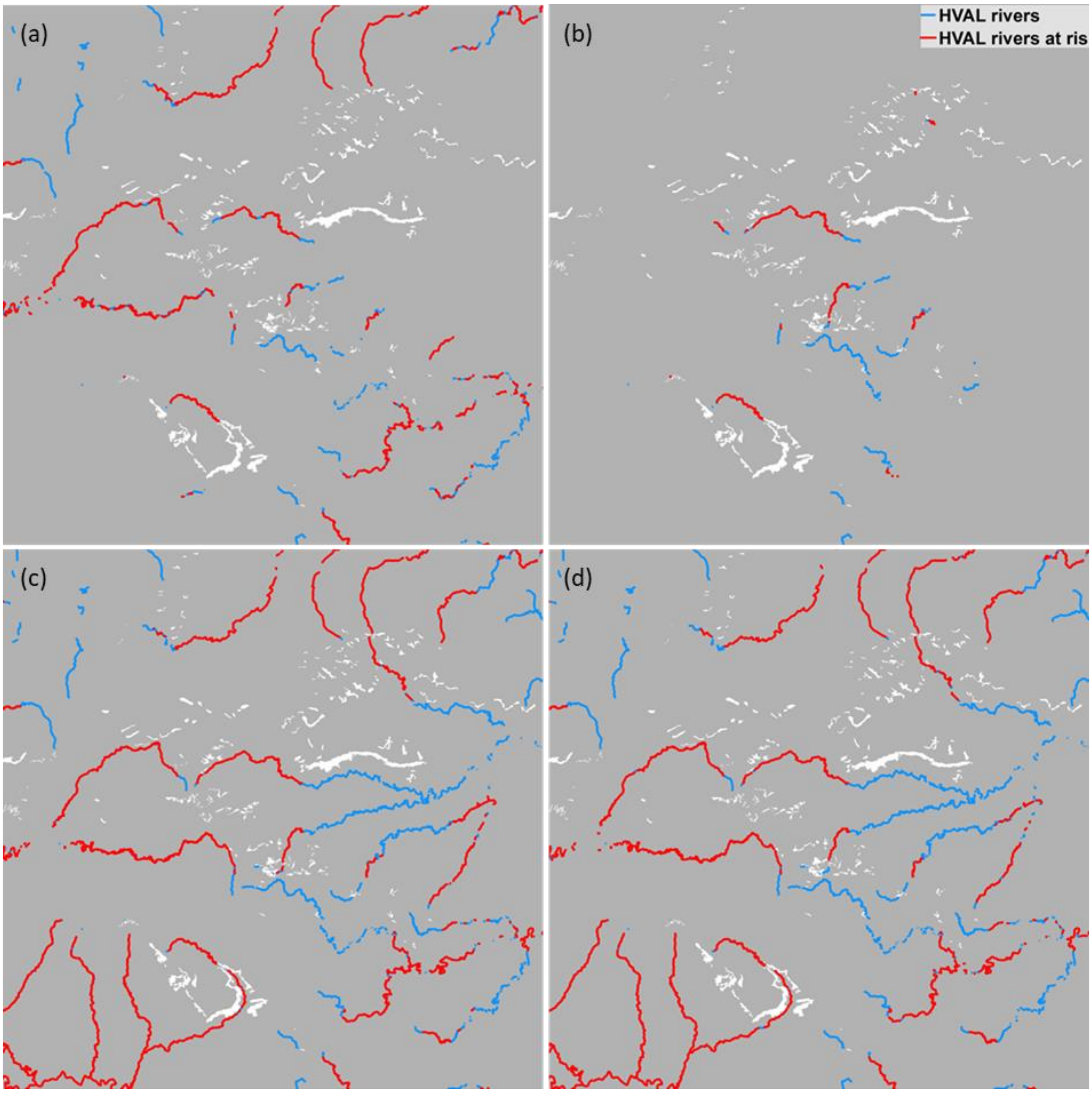

Figure 3. Sample of pollution risk by mines of high-value agricultural land (HVAL) river course, for (a) maize, (b) soybean, (c) sorghum, and (d) sunflower, for a watershed near Wakkerstroom, South Africa. Blue lines indicate all river courses within the HVAL, whilst the red lines indicate the HVAL river courses potentially at risk of water pollution from mining.

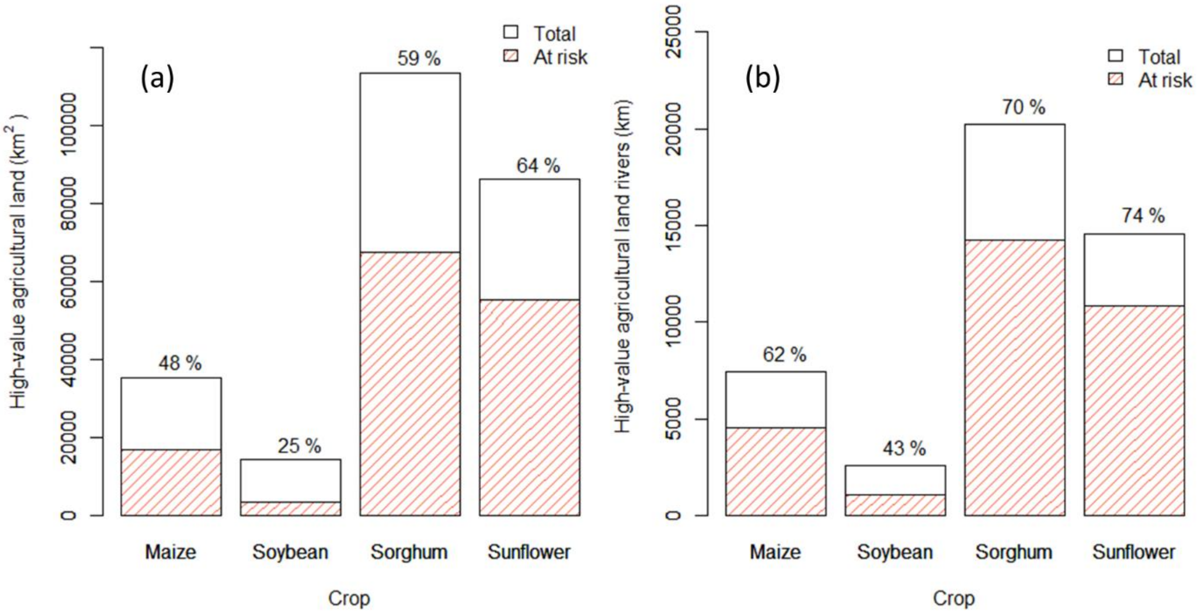

Figure 4 The (a) water source area, and (b) river course length, in high-value agricultural land (HVAL), both not at risk, and at risk of pollution (also indicated as a percentage of the area/length not at risk), from mining, in the South African grasslands, for four economically important crops (i.e. maize, soybean, sorghum, and sunflower).

4. Discussion

Though previous studies have raised concerns about shifts of mines towards high-value agricultural land (Tomlinson, 1980), this study shows mixed results. The presence of mines within the HVAL of the four crops and the change in establishment intensity from 1990 to 2018 vary, depending on the crop, and time period.

The percentage of land covered by mining in the HVAL of sorghum and sunflower, and the 'suitable' and 'moderate' value agricultural land of maize and soybean, suggest that these are the agricultural classes, for each of the crops, that are most likely to be experiencing disproportionate mining. Mining in HVAL of sorghum especially seems to be intensifying as mined HVAL of the crop has shifted from no different from expected in 1990, to larger than expected from random in 2018 (Fig. 1a-b, Table S1.2). The reason for this may be that mineral deposits in lower-value regions are becoming exhausted (Edwards et al., 2014). The disproportionate mine establishment in sunflower HVAL in both 1990 and 2018 suggests that disproportionate mineral exploitation within sunflower HVAL predated 1990, potentially starting in the 1980s, or earlier, as reported elsewhere (Tomlinson, 1980). Even though most mining took place before 1990, the observed annual mine cover increase in sunflower HVAL is more than five times greater than the expected between 1990 and 2018. Hence, these results suggest that mining in the grasslands have led to the disproportionate loss of HVAL of sorghum and sunflower. In contrast, not only was the mine area within the HVAL of maize and soybean lower than expected from the null model in both 1990 and 2018, but the annual mine cover increase between 1990 and 2018 in both maize and soybean HVAL is lower than expected.

Differences between patterns of HVAL mine placement observed between the four crops could be explained by the geographic distribution of the different HVAL croplands. Maize and soybean HVALs occur towards the eastern regions of the grasslands, where there is less mine area than in the central and northern regions, which in turn overlap largely with the HVAL distributions of sorghum and sunflower. Another potential explanation for this is that higher value is placed on conserving the HVAL of maize and soybean, than of sorghum and sunflower, due to the higher historical importance and production of maize and soybean (Bureau for Food and Agricultural Policy, Protein Research Foundation, Oil & Protein Seeds Development Trust/Oilseeds Advisory Committee, & Grain South Africa, 2020; C. K. Wright & Wimberly, 2013). If this is the case, the contribution and importance of these two crops for future food security is being considered in land-use planning. It is unlikely that the reason for the disproportionate mine cover in the high-value sorghum and sunflower land is due to lower real estate value and/or production value per hectare than that of maize or soybean (Bureau for Food and Agricultural Policy et al., 2020). It is also possible that mines have been established surrounding developed HVAL, due to the extensive support infrastructure required by mines, offered, in part, by existing agricultural communities (Alamgir et al., 2017; Lechner et al., 2018). This includes developed roads, access to water and electricity, and accommodation and access to resources for mine staff (Alamgir et al., 2017; Lechner, Baumgartl, Matthew, & Glenn, 2016; Salomons & Förstner, 1988).

Only 1.5-3% of South African land is considered suitable for high-value agricultural production (Bureau for Food and Agricultural Policy, 2012; Goldblatt, 2015). Half of this high-value agricultural land is within Mpumalanga, and much of it in the grassland biome, where approximately 12% of its HVAL is projected to be lost at current mining rates, and a further 13.6% set aside under prospecting rights (Bureau for Food and Agricultural Policy, 2012). As a result, if no remediation attempts are made following mine closure (Auditor-General South Africa, 2009; Department of Mineral Resources, 2009), high-value sorghum, and sunflower land could potentially be lost forever due to e.g. sterilization of topsoil, loss of topsoil, increased soil erosion etc. (Bradshaw, 1997; Chamber of Mines of South Africa & Coalteck Research Association, 2007; J. Harris et al., 1989; J. A. Harris et al., 1996; Limpitlaw et al., 2005; Rethman, 2006). Additionally, once transformed, this high-value land could become invaded, and serve as a source of weeds species, affecting surrounding agricultural land as well (J. Harris et al., 1989). With the predicted African population increase between 2005 and 2050 (United Nations, 2004), and the predicted drought frequency increase (Oldeman et al., 1991), the disproportionate loss of high-value sorghum and sunflower land, both moderately drought-tolerant crops (Hadebe, Modi, & Mabhaudhi, 2017; Hussain et al., 2018), could potentially contribute to a future increase in South African food insecurity (Funk & Brown, 2009).

The maps of high-value agricultural land water sources and river courses surrounding mines can help to identify the high-value agricultural water sources at the greatest risk of water pollution from mines. The higher the water flow accumulation within the high-value agricultural water sources, for the four respective crops, the higher the mobility of potential pollutants, contaminating a greater area of high-value agricultural land (Oelofse, 2009; Razo et al., 2004). Additionally, pollution of higher water flow accumulation water sources also increases the likelihood of a water source being used for crop irrigation, greatly increasing the potential impact of pollutants. Hence, the high proportion of high-value agricultural land water sources and river courses at risk of pollution from mining is a particular cause for concern. Given the potential impacts of heavy metals on crop and soil health (Department of Water Affairs and Forestry, 1996a), minimizing HVAL loss due to water pollution could become important in the near future. Maps such as these illustrate the potential risks associated with mining activities that need to be considered by land-use planners and policymakers.

Though the impacts of mines on agriculture are significant, it is important to note that mining is only one of multiple land-uses affecting high-value agricultural land. As such, the cumulative impact of mining on high-value agricultural land, when considered alongside other land-uses, such as urbanisation, are likely much more severe (Northey, Mudd, Saarivuori, Wessman-Jääskeläinen, & Haque, 2016; Weyer et al., 2017). Many assumptions and generalisations about the dispersal distances of metals from mine sites had to be made to come to estimates of areas potentially affected by pollution emanating from mines. More precise estimates would require more information on elements such as the local topography, hydrology, precipitation events and metal mobility. For example, there is high metal mobility variation between different metal types, with some dispersing over far greater distances, even for the same metal, depending on the local conditions such as water flow and topography (Durán et al., 2013). Additionally, the underground structures of mines (i.e. underground shafts, drifts etc.), which significantly contribute to water pollution (Côte, Moran, Hedemann, & Koch, 2010; Lechner et al., 2016), could not be included in these analyses. An accurate assessment of water pollution from mines within the grasslands would be a labour-intensive exercise, requiring thousands of samples (Ogola, Mitullah, & Omulo, 2002; Roback & Richardson, 1969; M. Taylor, Mackay, Kuypers, & Hudson-Edwards, 2009). Hence, broad assumptions were made in this study, to create a rough guide of potential water pollution of HVAL water sources and river courses by mines, to be refined upon in future studies.

Future land-use planning and policymaking in South Africa will need to consider the possible effects of mining on high-value agricultural land, as well as the importance of HVAL for future food security. Ideally, land-use planners should, as far as possible, limit mining within HVAL, and attempt to reduce the risk of water pollution of HVAL water sources and river courses by mining. With many countries running out of new agricultural area to expand into, emphasis will soon be placed on maximising agricultural productivity of the agricultural land currently available, and its associated water sources. In the case of four economically important crops in South Africa’s grasslands, I found mixed evidence. However, if the future food needs of South Africa are to be met, land-use planning will have to balance long-term food security with short-term financial gains.

Chapter 2

Surface footprint of mining in the South African Grassland biome: proximity and threat to biodiversity

1. Introduction

The biodiversity impacts of mining have often been underestimated, as they cover a relatively small land surface compared to other land-uses such as agriculture (Sonter et al., 2018). This marginalisation of the impact of mining on biodiversity is evident in the scientific literature, were fewer than 1% of articles in leading conservation journals (i.e. Conservation Biology and Conservation Letters) relating to the biodiversity impacts of mining (Sonter et al., 2017). However, mining activities already account for a relatively large portion of land in some regions. For example, 0.25% of South Africa's grasslands have been transformed by mining. Hence, as the depletion of higher-grade ores leads to a shift in mineral supply sources towards more biodiverse regions (Sonter et al., 2018), the threat of mining to biodiversity is likely to reach new levels.

Mining can have a range of impacts on biodiversity. The site-related impacts of mining activities can result in a rapid loss or alteration in local biodiversity. These impacts can include total loss of biodiversity due to vegetation clearing (Sonter et al., 2014; Wickham et al., 2013), habitat fragmentation (Department of Environmental Affairs et al., 2013; Edwards et al., 2014; Sonter et al., 2018), mechanical soil damage (Chamber of Mines of South Africa & Coalteck Research Association, 2007; Donker, 1992; J. Harris et al., 1989; King, 1988; C. D. Maxwell, 1991), soil erosion (Department of Environmental Affairs et al., 2013; Dudka & Adriano, 1997), soil pollution (Kabata-Pendias, 1993), and introduction and promotion of alien species (Li et al., 2010). Often, even in international-standard Environmental Impact Assessment (EIA) methods (Sonter et al., 2018), only these short-distance consequences of mining on biodiversity are considered. However, mining can also have impacts on that extend beyond the borders of the mine site. Examples of the long-distance impacts of mining activities include air pollution (Department of Environmental Affairs et al., 2013; Ghose & Majee, 2001), water pollution (Adler et al., 2007; Department of Environmental Affairs et al., 2013; Dudka & Adriano, 1997; Oelofse, 2009), noise pollution (Department of Environmental Affairs et al., 2013; Hockings, Anderson, & Matsuzawa, 2006; Hoskin & Goosem, 2010; Kerley et al., 2002; Kociolek, Clevenger, Clair, & Proppe, 2011), light pollution (Hölker, Wolter, Perkin, & Tockner, 2010), eutrophication (Ekholm & Lehtoranta, 2012), water availability (Tiwary, 2001), and acid mine drainage (Akcil & Koldas, 2006; Dudka & Adriano, 1997; Johnson & Hallberg, 2005; C. D. Maxwell, 1991).

Additionally, mining activities require extensive support infrastructure, such as transport networks (i.e. roads, railways, shipping ports), facilities for mining staff (Alamgir et al., 2017; Lechner et al., 2018), water and electricity access, and, in some cases, tailings dams (Salomons & Förstner, 1988). These ancillary infrastructures greatly increase the footprint of mining activities (Edwards et al., 2014; Sonter et al., 2018). Additionally, cumulative impacts often do not result from the impacts of a single mine, but rather the combination of multiple land use impacts in the region (Department of Environmental Affairs et al., 2013; Sonter et al., 2018), potentially resulting in a higher combined environmental stress than the sum of the environmental stress of all the individual mines. Together, these mining-related impacts can result in extensive and unpredicted impacts on biodiversity, potentially far greater than the direct impacts of mining activities (Edwards et al., 2014; Reed & Miranda, 2007; Sonter et al., 2018).

The consequences of both the short- and long-distance impacts of mines can last for long periods of time. For example, some trace metals can persist in soil for hundreds or thousands of years (Kabata-Pendias, 1993), whilst the local impacts of mining activities on soil can be near irreversible (Department of Environmental Affairs et al., 2013). Additionally, rare, extreme events, such as the failure of two tailings dams in Brazil, are recent examples of large-scale water pollution, resulting in total biological degradation of downstream ecological communities, with long-term effects (Asner et al., 2013; Department of Environmental Affairs et al., 2013; Li et al., 2010).

Even when all the potential effects of mining are considered, the presence of rich mineral resources can result in conservation and environmental protection legal frameworks being undermined or ignored, due to potential monetary gains (Laurance, 2004). This has been termed the “Resource Curse” (Auty, 2002). As a consequence of the Resource Curse, protected areas can experience changes in the status of their legal protection, often resulting in the loss of total protected area, i.e. downgrading, downsizing, and degazettement of protected areas (PADDD), to accommodate mining activities in that area (Mascia & Pailler, 2011). PADDD for mining activities has already been seen in at least five African countries (Edwards et al., 2014; Mascia & Pailler, 2011). Unfortunately, the regions experiencing the strongest "Resource Curse", and potentially PADDD, are also regions that consistently rank poorly on the Transparency International's corruption perceptions index (Transparency International, 2018).

In many regions, there is high overlap between mineral resources and biodiverse regions, exacerbating the biodiversity impacts of mining activities. For example, in Africa 44% (Edwards et al., 2014), and in Asia and South America, 25% of all active or abandoned major metal mines are located within ten kilometres of protected area boundaries (Durán et al., 2013). Further, the exploitation of all mineral reserves in Central Africa would directly impact an estimated 30-40% of ecologically important, high conservation value locations within this region (Edwards et al., 2014). As Africa contains roughly 30% of the earth's terrestrial mineral resources, many of which are as of yet largely unexploited, the continent is also on the verge of a rapid large scale mining boom (Taylor et al., 2009). Further, roughly 30% of all currently active major metal mines in Central Africa are located in important, stressed watersheds (Reed & Miranda, 2007); regions that are of special conservation concern given the current, and looming future water scarcity in Africa (Postel, 2000; Rijsberman, 2006).

However, mining corporations are increasingly investing in conservation offsets, better ecological management and restoration of mining sites (Reed & Miranda, 2007). In some cases, mines even lead to improved conservation, relative to many other land use types (e.g. agriculture) due to large exclusion zones, intended as security measures, around such sites (Loock et al., 2018). Mines can also bring in financial resources that can potentially contribute to the creation and maintenance of conservation zones. In some countries, mining sites also follow much stricter environmental regulation than other land-use types such as agriculture, and, due to a wider range of disturbances, possess larger overall site heterogeneity (McKinney, 2002; Sax & Gaines, 2003).

In this study, I investigate the pattern of mine establishment within the grasslands of South Africa. Grasslands are a large global biome, accounting for between 10-40% of the earth's surface, depending on the classification system utilised, and are located on every continent except Antarctica (Lemaire et al., 2011; Suttie et al., 2005; White et al., 2000).

Many grassland ecosystems harbour rich biodiversity (Habel et al., 2013), containing many economically and culturally important threatened, and endemic species. Globally grasslands contain 15%, 11%, and 29% of the world's Centres of Plant Endemism, Endemic Bird Areas, and ecoregions of outstanding biological distinctiveness, respectively (White et al., 2000). Additionally, grasslands provide a wide range of important ecosystem services, such as contributing to the global water supply, carbon sequestration, and pollination (Bengtsson et al., 2019; Carlier et al., 2009; Lemaire et al., 2011; O'Mara, 2012; Sala & Paruelo, 1997).

However, there are few ecosystems where the impacts of anthropogenic activities are more evident than the grassland regions of the world (Carbutt, Henwood, & Gilfedder, 2017). Globally, grasslands, and their associated biodiversity priority zones (e.g. protected area, important bird areas, and wetlands) face various threats. Despite their biodiversity and prevalence of endangered and endemic species, grasslands are globally both one of the most poorly protected habitats, with only 4.5% of temperate grasslands under formal protection (IUCN, 2020b), and the most extensively transformed terrestrial biome (Henwood, 2010; Hoekstra, Boucher, Ricketts, & Roberts, 2005), and have thus been described as a 'biome in crisis' (Barnosky et al., 2012; Hoekstra et al., 2005).

Habitat transformation has been identified as the primary cause of biodiversity loss (Asquith, 2001; Soule, 1991), with grasslands experiencing habitat loss due to intensive agriculture, overgrazing, and mineral extraction (Mark & McLennan, 2005; Sonter et al., 2018; C. K. Wright & Wimberly, 2013), resulting in grasslands being described as the ecological 'beast of burden' (Carbutt et al., 2017). Additionally, some evidence suggests that grasslands can rarely be rehabilitated to their natural state once severely disturbed, due to, for example, irreversible changes to limiting soil nutrients, such as phosphorus and potassium (Kahmen, Poschlod, & Thompson, 2004; Mentis, 2006).

In South Africa, the grassland biome accounts for roughly 30% of South Africa's land area (Ladislav Mucina & Rutherford, 2006; South African National Biodiversity Institute, 2018), and contains five centres of plant endemism, namely the Drakensberg Alpine, Barberton, Wolkberg, Sekhukhune and Soutpansberg centres (Van Wyk & Smith, 2001). Accordingly, the grasslands contain several rare, threatened, and endemic plant species, mostly forbs and dwarf shrubs, with a high concentration of endemic plant species in the eastern escarpment edge of the grassland biome (L. Mucina et al., 2006). Additionally, the South African grasslands harbour multiple threatened and/or endemic fish, bird, and mammal species, such as the Secretarybird (*Sagittarius serpentarius*), Black Wildebeest (*Connochaetes gnou*), and Drakensberg Rockjumper (*Chaetops aurantius*) (Lombard, 1995; Siebert, 2011; Skelton, Cambray, Lombard, & Benn, 1995; Van Wyk & Smith, 2001).

However, South Africa’s grassland biome is one of the country’s most threatened biomes, with more than 35% of the biome irreversibly transformed to non-natural land-use (Reyers et al., 2005). Based on the total habitat loss, habitat fragmentation, and the additional anticipated future threats, the South African grasslands have been classified as critically endangered (Olson & Dinerstein, 1998; Reyers et al., 2001), and the South African biome most urgently in need of conservation (South African National Biodiversity Institute, 2018).

Additionally, South African grasslands are especially at risk of transformation by mining activities. Instances of PADDD, though less prevalent in South Africa than many other southern African countries, have been reported (Mascia & Pailler, 2011), with 6.2% of private nature reserves, and 2.2% of state-owned nature reserves being degazetted between 1926 and 2018 (De Vos, Clements, Biggs, & Cumming, 2019). Additionally, South Africa is especially susceptible to corruption, with South Africa ranking 73rd in the 2018 Transparency International corruption perceptions index (Transparency International, 2018).

Less than 3% of the grassland biome of South Africa is under formal protection (SANBI, 2013). The majority of South Africa’s protected areas, including those in the grasslands, were proclaimed *ad hoc* (Pringle, Bond, & Clark, 1982). Though the reasons varied, proclamation was often based on their high scenic tourism potential, the prevalence of endemic diseases, or their lower value to alternative land-uses such as agriculture or urban development, rather than due to their biological conservation importance (Freitag, Nicholls, & van Jaarsveld, 1996; Pressey, Humphries, Margules, Vane-Wright, & Williams, 1993; Pringle et al., 1982). Hence, many of the few proclaimed grassland protected areas are even likely to have below-average conservation value.

The threats faced by the grasslands of the world have resulted in urgent calls for action (Millennium Ecosystem Assessment, 2005; World Resources Institute, 2000), with groups such as the International Union for Conservation of Nature (IUCN) launching the Temperate Grasslands Conservation Initiative (TGCI) in 2008 (Carbutt et al., 2017), and grasslands being included in the biodiversity goals set by both the United Nation's Sustainable Goals (SDGs) and Convention for Biological Diversity's 2020 Strategic Plan (CBD, 2011; International Council for Science, 2015). If these goals are to be met in the sensitive and biodiversity-rich grasslands of South Africa (Dean, Hoffinan, Meadows, & Milton, 1995; Neke & Du Plessis, 2004; O'Connor & Kuyler, 2009; Rutherford & Powrie, 2010), careful land-use decision will have to be made, considering all potential impacts of mining activities on surrounding biodiversity priority zones (Sonter et al., 2018; Sonter et al., 2017).

Hence, in this study, I investigate the impact of mining activities on the biodiversity of South African grasslands. I aimed to determine if mines have a disproportionate proximity to biodiversity priority zones (i.e. protected areas, important bird areas, and wetlands) within South Africa's grasslands. I also aimed to guide both future mine planning and conservation by identifying if grassland vegetation types with higher conservation value have more mining area within the borders.

2. Methods

The distance at which mining activities can affect biodiversity vary greatly, depending on multiple factors, such as the type of impact (e.g. water pollution, noise pollution, vegetation clearing), the local hydrology, and implemented preventative measures (de Castro Pena et al., 2017; Sonter et al., 2018) (Table 4). However, previous studies investigating the impacts of mining activities on biodiversity have often generalised the distances of the impacts in a buffer zone surrounding the mine, usually between five and ten kilometres (de Castro Pena et al., 2017; Durán et al., 2013; Edwards et al., 2014; Sonter et al., 2018). Likewise, I used the same buffers surrounding mine sites to generalise all the potential biodiversity impacts of mining.

Table 4. The maximum distance at which metals from mining activities have ecological/environmental impacts. Modified from Durán et al. (2013).

Authors	Mine type	Ecological/environmental effect	Maximum distance impact from mining source (km)
Hernández et al. (1999)	**Pyrite**	**Bird mortality**	**25**
Vásquez et al. (1999)	**Copper**	**Macroalgae abundance**	**3**
Razo et al. (2004)	**Copper–gold, lead–zinc–silver**	**Heavy metal concentration**	**5**
Telmer et al. (2006)	**Copper**	**Heavy metal concentration in lake sediments**	**50**
Yakovlev et al. (2008)	**Nickel**	**Soil quality**	**25**
Kodirov and Shukurov (2009)	**Copper and zinc**	**Heavy metal concentration in soil**	**4**
Kuznetsova (2009)	**Copper**	**Collembola communities in coniferous forests**	**7**
Lafabrie et al. (2009)	**Cobalt**	**Heavy metal concentration in seagrass**	**5**
Taylor et al. (2009)	**Copper, zinc and lead**	**Downstream water quality**	**30**
Bonifait and Villard (2010)	**Peat**	**Odonate abundance**	**1**
Chauhan (2010)	**Zinc**	**Deforestation**	**11**
Huang et al. (2010)	**Copper and zinc**	**Water acidity and heavy metal concentration**	**10**
Katpatal and Patil (2010)	**Coal**	**Flooding**	**15**
Lefcort et al. (2010)	**Copper and zinc**	**Stream insect diversity and abundance**	**2.5**
Vodyanitskii et al. (2011)	**Copper**	**Decrease of soil quality**	**30**

The surface footprint of mining activities were extracted from the 1990 and 2018 South African National Land Cover Maps (GeoTerraImage, 2014, 2019) (Table S2.1), and are henceforth collectively referred to as mines. I used protected areas (Department of Environmental Affairs, 2018), important bird areas (BirdLife South Africa, 2015), and wetlands (Council of Scientific and Industrial Research, 2018) to represent the biodiversity priority zones within the grasslands of South Africa. Due to the low presence of protected areas in South Africa (SANBI, 2013), it is crucial to understand the potential impact of mines on the existing protected areas.

The Important Bird Areas of South Africa, as identified by BirdLife South Africa (2015), were selected as they are good indicators of bird biodiversity, especially of endemic bird species, and have shown potential as indicators of local species richness (Bonn, Rodrigues, & Gaston, 2002; Schulze et al., 2004). Important bird area were designated based on their importance to the conservation of both global and regional bird species, the natural habitats of range-restricted and biome-restricted bird species, and the concentrations of congregatory bird species (BirdLife South Africa, 2015). Important bird areas are also likely to be among the areas most sensitive to the long-distance impacts of mining activities, such as noise pollution, light pollution, barriers to movement, edge effects, increased poaching and increased vehicular traffic (Hockings et al., 2006; Kociolek et al., 2011). For example, bird populations are very reliant on acoustic communication, using it to communicate with their offspring, attract mates, communicate threats and defend their territories, all of which can be disrupted due to noise pollution from mines (Kociolek et al., 2011; Lohr, Wright, & Dooling, 2003; Pohl, Slabbekoorn, Klump, & Langemann, 2009; Slabbekoorn & Ripmeester, 2008). Therefore, Important Bird Areas were important to consider in this study.

Wetlands were selected due to their high biodiversity, low formal protection, high number of critically endangered wetland ecosystems, and particular sensitivity to water pollution, one of the most devastating, long-distance impacts of mining (Berndt & Bavin, 2012; Lee & Bukaveckas, 2002; Pascoe, Blanchet, & Linder, 1994; Van Niekerk & Turpie, 2012). Additionally, wetlands are expected to be heavily impacted by mining in South Africa due to the high overlap between the current wetlands of South Africa, and underlying coal reserves (Exxaro; Kotze, 1997; Macfarlane et al., 2016; Stoop, 2010). Additionally, mining activities are highly water-dependent, with the mining industry, together with bulk industry, and power generation, account for 8% of South African water expenditure (Boccaletti, Stuchtey, & Van Olst, 2010). Overlap between the protected areas database and the wetlands map are unlikely to have confounded analyses, as only 4.25% of the wetlands in grasslands occur in protected area, with these making up only 2.87% of total protected area cover.

All data layers used in this study (Table 2) were cut to the extent of the grassland biome of South Africa, as defined in the 2018 version of the Vegetation map of South Africa, Lesotho, and Swaziland (Eswatini) (Ladislav Mucina & Rutherford, 2006; South African National Biodiversity Institute, 2018)

Table 5. Data extracted or recovered for this study. The title of the original datasets and their citations are provided.

Data used	Dataset title	Citation
Boundaries of mining activities 2018	**2018 National Land Cover Map**	**GeoTerraImage (2019)**
Boundaries of mining activities 1990	**1990 National Land Cover Map**	**GeoTerraImage (2014)**
Protected areas 2018	**South Africa protected areas database**	**Department of Environmental Affairs (2020)**
Important bird areas of South Africa	**The important bird areas of Southern Africa**	**BirdLife South Africa (2015)**
Wetlands of South Africa 2018	**National wetland map**	**Council of Scientific and Industrial Research (2018)**
Vegetation types of South Africa 2018	**2018 Vegetation Map of South Africa, Lesotho and Swaziland**	**South African National Biodiversity Institute (2018)**

First, I tested whether there was disproportionate mine area within protected areas, or within the 0-1 km, 1-5 km, and 5-10 km buffers surrounding protected areas, when compared to a null model of mine establishment within South Africa's grasslands. Polygons of all protected areas of South Africa were retrieved from the Protected Areas Database (Department of Environmental Affairs, 2018). Based on the mines extracted from the 2018 national land cover map (GeoTerraImage, 2019), I determined the mining area within protected areas, or within a 0-1 km, 1-5 km, or 5-10 km buffer around the boundary of the protected areas (Alamgir et al., 2017; de Castro Pena et al., 2017; Durán et al., 2013; Edwards et al., 2014). Thereafter, I used a randomisation procedure to test whether the actual mines area found within the protected areas or their buffers was more, less, or the same, compared to if mines were randomly distributed throughout the grasslands of South Africa. To do this, polygons of the same number, size, and shape as the polygons of current mining in South Africa's grasslands were randomly dispersed throughout the grasslands. This was done by creating the same number of random points throughout the grasslands as there are individual mining polygons in the shapefile, in ArcPy. Then, using 'UpdateCursor', I reassigned the centroid of each polygon to the location of one of the randomly dispersed points, without overlap. The randomisation was repeated 1000 times to create a null model of mine establishment. For each randomisation, the area of the randomly placed 'mines' within the protected areas, and its buffers, was quantified.

This process was repeated to assess whether current mines occur in and around the buffer areas of important bird areas (BirdLife South Africa, 2015), and wetlands (Council of Scientific and Industrial Research, 2018), in the grasslands more or less often than expected from the null model. Again, mining area within IBAs and wetlands, and within the same buffer distances surrounding their boundaries (i.e. within the area boundary, or within 0-1 km, 1-5 km, and 5-10 km of the boundary) was assessed and compared for the observed and expected (from the null models).

To determine if the observed mining area within biodiversity priority zones (i.e. protected areas, important bird areas, and wetlands), or within the surrounding buffers, differed from the expected mining area (eq. 2.1), I used a two-tailed one-sample Z-test (Sprinthall, 2014):

$$Z = \frac{\bar{y} - m_o}{\frac{\hat{\sigma}}{\sqrt{n}}} \tag{2.1}$$

Where, for the null model (i.e. the 1000 mine location randomisations), n is the sample size, $\hat{\sigma}$ is the standard deviation, and $\bar{y}$ is the mean of the expected mine area, within the protected areas, important bird areas, wetlands, or their respective buffer zones. Lastly, m_o is the observed mining area within each of the respective areas and buffers.

Next, I tested whether areas of higher conservation value contain more mining area. Obtaining information on the conservation values of different areas of the grasslands has many challenges, such as under-sampling and sampling biases (Botts, Erasmus, & Alexander, 2011; Hugo & Altwegg, 2017; Monsarrat, Boshoff, & Kerley, 2019). Therefore, conservation value was quantified at the level of grassland vegetation types. L. Mucina et al. (2006) identified the vegetation types (based on floristic characteristics) across the grasslands of South Africa, and provided a list of species endemic to each of the vegetation types. Endemic species richness has been used as an indicator of conservation value in the past, when data on all taxa was lacking (Bonn et al., 2002; Borges, Serrano, & Quartau, 2000; Loyola, Kubota, & Lewinsohn, 2007; Schulze et al., 2004). Therefore, the species endemic to each of the grassland vegetation types were extracted. Scores ranging from one to seven were assigned to the endemic species based on their IUCN red list status (Table 6) (IUCN, 2020a; SANBI, 2018). Next, the remaining extent of untransformed vegetation (i.e.

remaining natural area) for each vegetation type was calculated as the regions that were classified as untransformed in both the 1990 and 2018 National Land Cover Maps (GeoTerraImage, 2014, 2019) (Table S2.1). The conservation value per vegetation type was determined as the endemic species richness per 100 km^2 remaining natural vegetation, weighted by the red list status of the individual species (eq. 2.2). The conservation value (CV) for a vegetation type was thus calculated as:

$$CV_j = \frac{\sum_{ij}^{n} SpRS_{ij}}{RNA_j} \times 100 \tag{2.2}$$

where $SpRS_{ij}$ is the red list score of species i in vegetation type j, and RNA_j is the remaining natural area (km^2) of vegetation type j. Hence, for two vegetation types with the same number of endemic species, the vegetation type with more endangered species, and/or less remaining natural vegetation extent, would receive a higher conservation value. I then assessed whether a correlation exists between the mining area within a vegetation type and the conservation value,

All spatial analyses were conducted using ArcGIS Desktop (Version 10.6.1), and R Statistical software (R Core Team, 2018).

Table 6. Scores assigned to the IUCN red list categories of the endemic species, used to weigh the endemic species richness of the vegetation types.

Red list designation	Red list score
Least Concern	**1**
Data Deficient	**2**
Not Evaluated	**2**
Near Threatened	**3**
Vulnerable	**4**
Endangered	**5**
Critically Endangered	**6**
Extinct in the wild	**7**

3. Results

There was 19.88 km^2 of overlap between protected areas and mining areas in the grasslands (Figure 1, Figure S2.1, Table S2.2), spread across grassland protected areas (Table 7). In 2018, all 99 of these were listed as legally designated, and protected under law (Department of Environmental Affairs, 2018) (Figure S2.2). Of the total mine cover within the grassland biome of South Africa, 44.82% was within 10 km of protected areas. In contrast, only 33.17% of the total grassland area falls within 10 km of protected areas. Mine cover was lower than expected within protected areas (PAs), and within one kilometre of the boundary of PAs (Figure 1, Table S2.2). In contrast, mine cover was higher than expected within five, and ten kilometres of PA boundaries (Figure 1, Table S2.2).

Mine cover was lower than expected within, and within a 0-1, 1-5, and 5-10 km buffer of IBA boundaries (Figure 1, Table S2.2), with only 15.58% of South African mining area within 10 km of IBAs, compared to 25.73% of total grassland area occurring within 10 kms of IBAs. Mine cover was slightly lower than expected within wetlands, and within a 1-5 km buffer of wetland boundaries (Figure 1, Table S2.2). Additionally, mine cover was much lower than expected within a 5-10 km buffer of wetland boundaries (Figure 1, Table S2.2). In contrast, mine cover was much higher than expected within one kilometre of WA boundaries (Figure 1, Table S2.2). However, mines had a very high spatial relation to wetlands, with 96.84% of all mining area occurring within 10 km of wetland. This is in contrast to only 80.22% of total grassland area occurring within 10 km of wetlands.

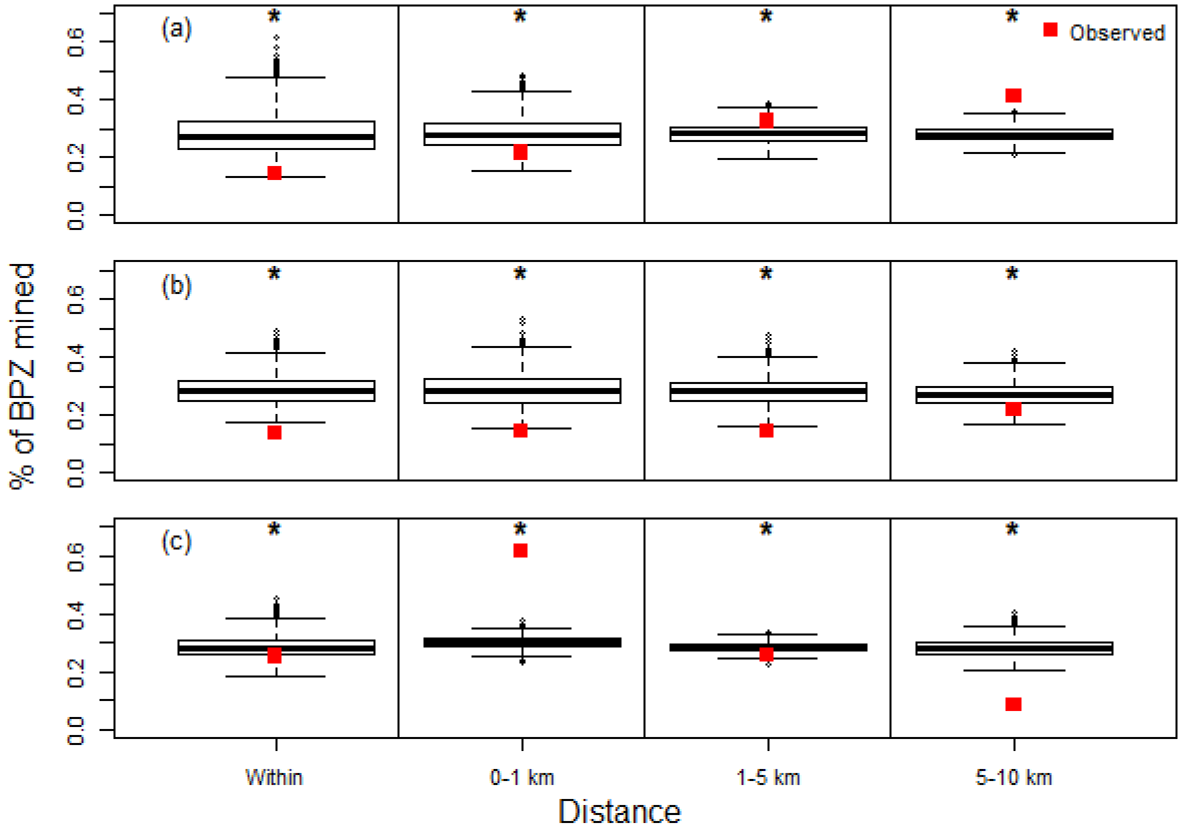

Figure 1. Comparison of the observed and expected mine area, represented by the red box and the boxplot respectively, within, and within a 0-1, 1-5, and 5-10 kilometre buffer, surrounding the boundaries of biodiversity priority zones (BPZ), i.e. (a) protected areas, (b) important bird areas, and (c) wetlands. The asterisk indicates a significant difference between the observed and expected mine area within a given area or buffer. Note: statistical comparisons are made between each boxplot (expected mine area) and its corresponding red box (observed mine area), not between boxplots. See Table S2.2 for absolute observed and expected mining area within each buffer distance.

Table 7. The designation of the 99 protected areas in South Africa's grasslands that have mines within their boundaries. Shown are the number of protected areas per designation type, as well as their relevant legislation (Department of Environmental Affairs, 2018; GeoTerraImage, 2019).

Designation Type	Number of PAs	Relevant Legislation
Nature Reserve	**85**	**National Environmental Management: Protected Areas Act, (Act No. 57 of 2003)**
Protected Environment	**8**	**National Environmental Management: Protected Areas Act, (Act No. 57 of 2003)**
Forest Wilderness Area	**1**	**National Forests Act. 1998 (Act No. 84 of 1998)**
National Park	**1**	**National Environmental Management: Protected Areas Act, (Act No. 57 of 2003)**
World Heritage Site	**2**	**World Heritage Convention Act, (Act No. 49 of 1999)**
Mountain Catchment Area	**1**	**Mountain Catchment Area Act, (Act No. 63 of 1970)**
Forest Nature Reserve	**1**	**National Forests Act. 1998 (Act No. 84 of 1998)**

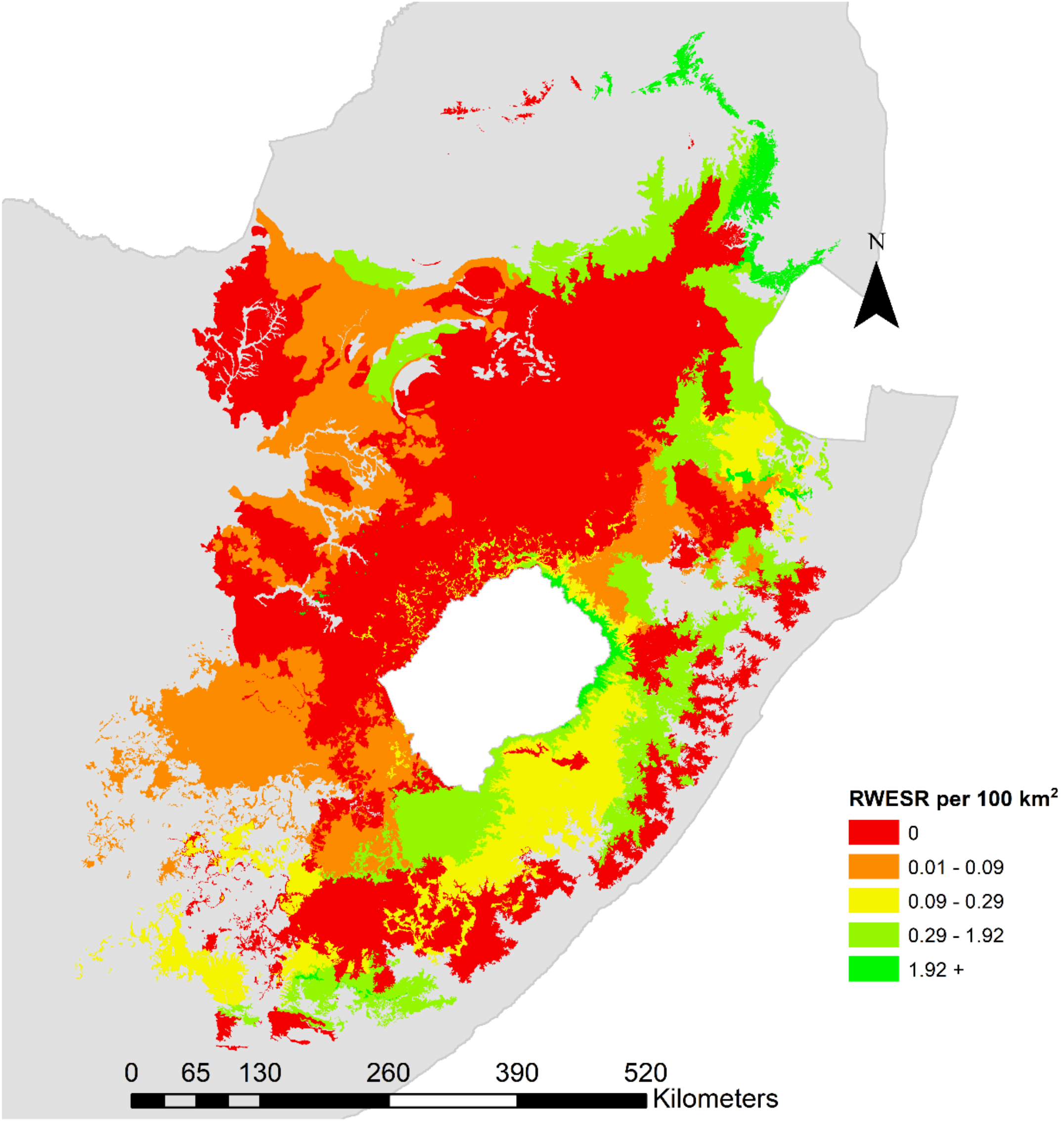

Figure 1. The conservation value per South African grassland vegetation type, based on the red-list weighted endemic species richness (RWESR) per 100 km^2 of the vegetation type's remaining natural extent, with species weighed by their IUCN red list status.

The vegetation types with higher conservation values were largely located in the eastern, and north-eastern regions of the grasslands (Figure 1). However, there was no correlation between the mining area within the vegetation types and the conservation value of the vegetation types ($p = 0.57$, $df = 71$, $R^2 \approx -0.07$). The ten vegetation types with the highest conservation value scores had less than 1 km^2 mining area within their borders (Table 8). In contrast, the ten vegetation types containing the highest mining area (km^2) largely had no, or very few reported endemic species, with an average conservation value of 0.065 (Table S2.4).

Table 8. The ten South African grassland vegetation types with the highest conservation values, based on the endemic species richness per 100 km^2 of the vegetation type's remaining natural extent, with species weighed by their IUCN red list status. Also indicated is the total mining area (km^2), and unmodified endemic species richness (ESR), per vegetation type.

Grassland Vegetation Type	Mining Area (km2)	ESR	Conservation value
Drakensberg-Amathole Afromontane Fynbos	**0**	**14**	**109.25**
Northern Escarpment Afromontane Fynbos	**0**	**3**	**32.38**
Drakensberg Afroalpine Heathland	**0**	**16**	**22.9**
Northern Escarpment Quartzite Sourveld	**0.56**	**38**	**11.91**
Woodbush Granite Grassland	**0**	**4**	**11.49**
Long Tom Pass Montane Grassland	**0.08**	**16**	**8.69**
Northern Zululand Mistbelt Grassland	**0.3**	**6**	**8.44**
uKhahlamba Basalt Grassland	**0**	**83**	**7.22**
Amathole Mistbelt Grassland	**0.01**	**9**	**7.14**
Soutpansberg Summit Sourveld	**0**	**5**	**6.59**

4. Discussion

In this study, I aimed to determine if there is disproportionate mine cover near biodiversity priority zones, as well as if there is disproportionate mining in vegetation types of higher conservation concern. I found evidence of disproportionate mine cover between 1-5 km, and 5-10 km of protected area boundaries, with 44.84% of all mining area within ten kilometres of protected areas, as well as mining area within 99 legally protected areas. Additionally, I found disproportionate mine cover within one kilometre of wetland boundaries, with 96.84% of all mining activities are within ten kilometres of wetlands. In contrast, there was no evidence for disproportionate mine cover within, or surrounding Important Bird Area, with only 15.58% of mine area within 10 km of important bird areas.

The disproportionate mine cover in close proximity to protected areas and wetlands increases the risk of long-term direct, site-related impacts , as well as long-distance impacts on biodiversity, through e.g. habitat destruction, emission of mobile pollutants, noise pollution, and mining-related support infrastructures (Alamgir et al., 2017; Durán et al., 2013; Kerley et al., 2002).

The lower mine establishment within protected areas, and within a one-kilometre buffer surrounding protected area boundaries suggest that the potential boundary effect of surrounding land uses, e.g. the direct impacts of mining activities, are considered when planning protected areas. The South African Protected Areas Act (2003) does not expressly make any provision for the creation of buffer zones surrounding protected areas (Paterson, 2009). However, there are multiple sections within this act, and the zoning schemes of the provincial planning legislation that can be used to create buffers surrounding protected areas (Paterson, 2009). Examples of these include stringent land-use restrictions within larger bioregions, which contain protected areas, as well as co-management and regulation of the development of economic opportunities, such as mining, within, and adjacent to protected areas (Paterson, 2009). However, in this study I found that 44.84% of all mining activities were located within ten kilometres of protected areas and mines were disproportionally established within a 1-5, and 5-10 km of protected area boundaries. This supports the findings of previous studies, with Edwards et al. (2014) reporting that 44% of all major metal mines are inside or within ten kilometres of protected areas, Durán et al. (2013) also finding disproportionate mines establishment within 1-5 km and 5-10 km of protected areas in Africa. Hence, these findings suggest that the long-distance impacts of mining activities should be closely monitored, as the biodiversity impact of long-distance effects are often largely ignored and underestimated in land-use and conservation planning (Laurance, 2008; Weng et al., 2013).

Protected areas are therefore more likely to suffer from, and be undermined by the indirect, induced, cumulative, and long-distance impacts of mining activities (Department of Environmental Affairs et al., 2013), such as air, water (Malm, Pfeiffer, Souza, & Reuther, 1990; Razo et al., 2004), soil (de Castro Pena et al., 2017; Edwards et al., 2014; Sonter et al., 2018), and noise pollution (Department of Environmental Affairs et al., 2013; Hockings et al., 2006; Kerley et al., 2002). With disproportionate mining within 1-5 and 5-10 km of protected areas, much of the biological impacts of mining will also likely be due to the extensive support infrastructure required by mining activities (Alamgir et al., 2017; Edwards et al., 2014; Lechner et al., 2018; Salomons & Förstner, 1988; Sonter et al., 2018). These impacts, though less obvious than the direct impacts of mining activities, can have a significant impact on the biodiversity of the surrounding natural area, i.e. protected areas (Alamgir et al., 2017; Edwards et al., 2014; Lechner et al., 2018; Sonter et al., 2018). Given the fact that the impacts of these activities on biodiversity are often underestimated, special attention should be given to these potential impacts when considering land-uses surrounding protected areas in the future.

Though significantly lower than expected from random placement, the mining area within protected areas, is cause for concern, as, according to the Protected Areas Act (2003), no mining activities should be initiated withing protected areas given the legal designation bestowed upon these areas. However, the act also states that, in the case where a mine was established within a protected area before the implementation of the Protected Areas Act (2003), the minister of national environmental management, following consultation with the cabinet member responsible for mineral and energy affairs, may approve the continuation of mining within the protected areas, potentially explaining some of the mining area seen within protected areas. Additionally, such mining activities might also be as a result of protected area downgrading, downsizing, and degazettement (PADDD) (Mascia & Pailler, 2011), or lax enforcement of protected area regulations, associated with the “Resource Curse” (Auty, 2002; Laurance, 2004). PADDD has been observed in South Africa in previous studies (Mascia & Pailler, 2011), with 6.2% of all private nature reserves, and 2.2% of state-owned nature reserves being officially and legally degazetted between 1926 and 2018 (De Vos et al., 2019). If the legal designation of the protected areas within the South African National Protected Area Database are accurate, then almost 100 protected areas, that have not been officially and legally degazetted, have been subjected to mining, suggesting that they are either no longer functioning as conservation regions and that the rate of PADDD is markedly higher than reported in previous studies (De Vos et al., 2019; Mascia & Pailler, 2011), or that they experienced mining activities prior to being gazetted. Hence, the protected area estimations for the grassland biome, already one of the most under-protected biomes in South Africa (SANBI, 2013), might be overestimated by South African conservation organisations and the IUCN.

Alternatively, many of the mines within protected area boundaries are more likely due to mines that were authorised before the establishment of nature reserves. Incorrect classification of mining activities within protected areas in the 2018 NLCM (GeoTerraImage, 2019), was eliminated as a possible explanation, as inspection of these overlaps largely indicated correct mine classification, confirming the presence of mine cover within the 99 protected areas. However the most plausible explanations are inaccuracies in the protected areas of South Africa database (Department of Environmental Affairs, 2018), or in some cases, areas that were initially designation in the 20th century, of which the documentation has only recently been rediscovered, after mines had already been approved and established (Mervyn Lötter, pers. comm.).

Important bird areas were selected based on their importance to the conservation of both global and regional bird species, the natural habitats of range-restricted and biome-restricted bird species, and the concentrations of congregatory bird species (BirdLife South Africa, 2015). As many South African mines were established by 1998, when the important birds areas of South Africa were first designated, the low overlap between mining areas and important bird areas might be due to low biodiversity near mining sites at the time of IBA designation. Alternatively, this could indicate that the site characteristics chosen for IBAs are given high value during local land-use planning. However, it is possible that IBA designation might decrease mining within biodiverse regions. As a result, important bird areas are at a reduced risk of being heavily affected by either the direct or indirect impacts of mining activities. This is fortunate, as bird populations are both among the most sensitive to some of the indirect impacts of mining activities, i.e. noise, artificial light, barriers to movement, and edge effects (Kociolek et al., 2011) and are good indicators of local species richness (Bonn et al., 2002; Schulze et al., 2004).

As 96.84% of all mining activities are within ten kilometres of wetlands areas, the critical ecosystem services provided by wetland areas are at risk of devastation from mines (Berndt & Bavin, 2012; Lee & Bukaveckas, 2002; Pascoe et al., 1994; Van Niekerk & Turpie, 2012). Additionally, the disproportionate mine cover within a one kilometre buffer of wetlands means wetland biodiversity is at an increased risk of being affected by both the direct, and long-distance impacts of mining. The high overlap between mining sites and wetland areas in the South African grasslands might be explained by the underlying geology of the wetlands, responsible for the formation of many of the coal seams in South Africa during the Permian age (Exxaro; Kotze, 1997; Macfarlane et al., 2016; Stoop, 2010). Another potential contributing factor, explaining the disproportionately high mine establishment within a one-kilometre buffer of wetland area boundaries, might be the high water dependency of many mining activities (Boccaletti et al., 2010; Ranchod, Sheridan, Pint, Slatter, & Harding, 2015). In mining, the majority of the water use and loss is involved in the processing plants, the tailings storage facilities and tailings discharge (Ranchod et al., 2015). Inversely, both the underlying geology, and the water dependency of mining activities could explain the disproportionately low mine establishment further away from wetlands, i.e. within a 1-5, and 5-10 km buffer of wetland boundaries. Though likely only a minor contributing factor, the extensive support infrastructure, such as staff accommodation and transport hubs required by mining activities (Alamgir et al., 2017; Lechner et al., 2018), could also contribute to the disproportionate mine presence within a one-kilometre buffer of wetland areas, as the necessary town are often established along waterways.

Though slightly lower than expected if mines were randomly dispersed throughout the grasslands, the mine presence within wetland areas is still a cause for concern. Metal discharge from mine sites vary in their mobility and toxicity to aquatic and terrestrial organisms (Durán et al., 2013). Some highly mobile metals, such as zinc and lead can cause a noticeable decrease in aquatic biodiversity along the watercourse in excess of ten kilometres from the source (Durán et al., 2013). There are many laws in place in South Africa to prevent and mitigate the threat mining activities pose to wetlands, such as the National Environmental Management Act (1998), National Water Act (1998), environmental provisions in the Mineral and Petroleum Resources Development Act (2002), and the South African Water Quality Guidelines for Aquatic Ecosystems (Department of Water Affairs and Forestry, 1996b) However, the majority of the relevant sections in these acts and guidelines concern the prevention or mitigation of the impacts of mining on wetlands, rather than restricting mining area within wetlands. The disproportionate mine cover within one kilometre of wetland boundaries, and mine area within wetlands, is a major cause for concern as, despite their high biodiversity, conservation value, and important ecosystem services, over 70% of South Africa’s wetland ecosystems have no form of protection, with 48% of these ecosystems already classified as critically endangered (Van Niekerk & Turpie, 2012).

As there is no correlation between the conservation value and the mining area per vegetation type, this might be evidence that mining has been regulated in high conservation value vegetation types. Alternatively, this might indicate that there is a lower overlap than expected between high-value resources reserves and biodiverse regions in South African grasslands (Edwards et al., 2014). Regardless, as endemic species have previously been shown to be suitable indicators of species richness (Bonn et al., 2002; Schulze et al., 2004), this might suggest that, at the vegetation type scale, there is not disproportionate mining in more biodiverse regions.

It is important to note that, despite the potentially severe biological impacts of mining, their relatively small surface area results in a smaller impact on biome level biodiversity, compared to other, much larger land-uses, such as agriculture and forestry (Sonter et al., 2018). Additionally, mines often have better ecological management and restoration (Reed & Miranda, 2007), and stricter environmental regulation (McKinney, 2002; Sax & Gaines, 2003), relative to other land-uses. Additionally, the large exclusion zones surrounding mine sites have been shown to protect habitat fragments of some species able to bypass the fencing, potentially improving conservation surrounding mines, relative to other land-uses such as agriculture (Loock et al., 2018). Indeed, mining corporations are increasingly investing in conservation offsets, something less common in larger land-use industries (Reed & Miranda, 2007).

Due to data availability, underground mine structures could not be considered in this study, despite their significant impact on water availability and pollution (Dudka & Adriano, 1997; Johnson & Hallberg, 2005; C. D. Maxwell, 1991). However, the surface footprint of underground mines, such as tailings and dumps were included as part of the identified mining surface footprint. Further, there is very limited information in South African, or indeed global literature concerning the specific distances at which mining effects biodiversity (Alamgir et al., 2017; de Castro Pena et al., 2017; Durán et al., 2013; Edwards et al., 2014). Some of these impacts might extend far beyond the buffer distances used in this study, such as water pollution by highly mobile metals (Durán et al., 2013; Kossoff et al., 2014; Razo et al., 2004). Additionally, the endemic species per vegetation type, extracted from the South African National Vegetation Maps (L. Mucina et al., 2006; South African National Biodiversity Institute, 2018), is by no means an extensive inventory of all endemic species of the grassland biome vegetation types. However, it represented the best available data source concerning the conservation value, based on endemic species, of the grassland vegetation types. Lastly, though weighing the conservation values of each vegetation types according to its proportion remaining natural area rightfully gives higher priority to more transformed vegetation types, this could potentially also give disproportionate priority to smaller vegetation types. Hence, additional conservation value information, presented in the supplementary material (e.g. endemic species richness not weighted by remaining natural area), should also be consulted.

Concerns for the conservation of biodiversity priority zones have largely focused on the direct, site-related impacts of land-use changes, such as commercial forestation, agriculture, urbanisation, and to a more limited extent, the direct, site-related impacts of legal mining activities (Sonter et al., 2018). However, as shown in this study, protected areas are at disproportionate risk from the long-distance impacts of mining, whilst wetlands are at higher risk of the direct, and short-distance impacts of mining (e.g. water pollution). Therefore, this study highlights both the need for a better understanding as to the potential impacts of these long-distance mining impacts on biodiversity, as well as the need to consider the land-uses surrounding mines and biodiversity priority zones. This study justifies the growing concern in the literature, and in policy concerning the long-distance impacts of mining activities (Laurance, 2008; Weng et al., 2013), and the need for further investigation.

General Conclusion

Here I have investigated whether mining in the South African grasslands poses a disproportionate risk to the agricultural productivity and biodiversity in the region. In Chapter 1 I assessed whether the observed mining area within the high-value agricultural land of four economically important crops (i.e. maize, soybean, sorghum, and sunflower), exceeded the expected mining area, if mines were randomly dispersed throughout the grasslands. Additionally, I determined what percentage of the high-value agricultural land water sources and river courses of the four crops are at risk of water pollution from mining. The African population is expected to increase to 2.5 billion by 2050 (United Nations Department of Economic and Social Affairs, 2019), and roughly a third of the population (> 218 million people) experiences daily food insecurity (Funk & Brown, 2009). Mining poses multiple risks to agriculture, including long-distance, and long-lasting impacts (Asner et al., 2013; Bradshaw, 1997; Chamber of Mines of South Africa & Coalteck Research Association, 2007; Department of Environmental Affairs et al., 2013; Dudka & Adriano, 1997; J. Harris et al., 1989; J. A. Harris et al., 1996; Li et al., 2010; Limpitlaw et al., 2005; Rethman, 2006; Tongway & Ludwig, 2011). As Africa contains between 30-40% of the earth’s terrestrial mineral resources, many of which are as of yet largely unexploited, Africa is likely still on the verge of a rapid large scale mining boom (Taylor et al., 2009). Hence, it is critically important to investigate all risks faced by agricultural productivity, and by extension, food security, in the region. Sub-Saharan Africa will be particularly sensitive to the impacts of mining on agriculture, due to the numerous challenges to agricultural productivity (and food security), in the region, such as loss of productive land (Oldeman et al., 1991), and reduced crop yields from climate change (Department of Environmental Affairs, 2011; Thornton, 2003). In South Africa, the extensive mining in the grassland biome poses a clear risk to South African agriculture. The reasons for this are three-fold: (1) the South African grasslands account for the majority of the developed agricultural land, and remaining high-value potential agricultural land in the country (GeoTerraImage, 2019; ISCW, 2006; South African National Biodiversity Institute, 2018); (2) high-value agricultural land is scarce in South Africa (Goldblatt, 2015); (3) mining is likely to expand rapidly in the region (Bureau for Food and Agricultural Policy, 2012). In this study, I illustrate that mining already poses a disproportionate risk to the high-value agricultural land of two of the four economically important crops examined here. Despite the small surface area of mining activities, compared to other land-uses (Sonter et al., 2018), the often irreversible damaged mining causes to the site soil (Chamber of Mines of South Africa & Coalteck Research Association, 2007; Department of Environmental Affairs et al., 2013), means that mining can have a long-term impact on agricultural productivity.

African agriculture is also often reliant on irrigation to meet crop production requirements. Irrigation is especially important in South Africa, were only 12% of land is suitable for rain-fed crops (Goldblatt, 2015). However, roughly a third of all metal mines in the Central Africa region are located in stressed watersheds, critical for agricultural water provision (Reed & Miranda, 2007), with figures in South Africa likely being similar. This, despite the fact that mining activities can affect water far beyond the boundary of the mine. Examples of mining impacts are high water extraction (Boccaletti et al., 2010), long-distance water pollution (Roback & Richardson, 1969; Winterbourn et al., 2000), and disruption of hydrological cycles (Tiwary, 2001). Mining could impact these water sources and river courses due to the slow, and accidental release of pollutants from mine sites (Razo et al., 2004), or through rare, but extreme, events such as the failing of tailings dams (Asner et al., 2013; Li et al., 2010). As shown in Chapter 1, in the case of sunflower HVAL river courses, this would mean that up to 74% of river courses flowing through this HVAL could be affected. I showed similar results for the water sources and river courses within the HVAL of maize, soybean, and sorghum as well. Pollution of such large tracts of irrigation water sources of scarce, high-value agricultural land, could pose a risk to South African food security, especially if the effects are long-lasting or irreversible (Kabata-Pendias, 1993).

In Chapter 2, I assessed whether there was disproportionate mine cover near biodiversity priority zones. Additionally, I determined if grassland vegetation types with higher conservation value had higher mining area. In many regions, there is high overlap between mineral resources and biodiverse regions, leading to competition between the two natural characteristics, with mining often given priority. This is a particular concern in Africa due to the high overlap of major metal mines with biodiverse regions, such as protected areas (Durán et al., 2013; Edwards et al., 2014), and important watersheds (Reed & Miranda, 2007). Additionally, a potential mining boom, leading to the exploitation of all mineral reserves in regions such as Central Africa, could directly impact 30-40% of ecologically important, high conservation value locations in such region (Edwards et al., 2014; Taylor et al., 2009). The grasslands of South Africa harbour rich biodiversity (Van Wyk & Smith, 2001), with the region supporting many important endemic and threatened species, across multiple taxa. However, this rich biodiversity is under threat, due to extensive transformation by non-natural land-uses, such as urban development, and mining (Reyers et al., 2005), hence the call for urgent conservation of the under-protected South African grassland biome (South African National Biodiversity Institute, 2018). Mining is a contributing factor to the threatened status of the South African biome, as is it is the South African biome with the most prevalent mining.

Some of the regions most important for biodiversity in South Africa, including the protected areas, Important Bird Areas, and wetlands, face many short-, and long-distance threats from mining (Department of Environmental Affairs et al., 2013; Hockings et al., 2006; Hoskin & Goosem, 2010; Kerley et al., 2002; Kociolek et al., 2011; Li et al., 2010; Sonter et al., 2018; Sonter et al., 2014; Wickham et al., 2013). However, only a small portion of the South African grasslands are under formal protection (SANBI, 2013), species characteristic of Important Bird Areas are often the most sensitive to the impacts of mining (Bonn et al., 2002; Kociolek et al., 2011; Schulze et al., 2004), and wetlands are one of the ecosystems most easily devastates by mining (Berndt & Bavin, 2012; Lee & Bukaveckas, 2002; Pascoe et al., 1994; Van Niekerk & Turpie, 2012). Unfortunately, as I have shown here, mining does pose a disproportionate risk, potentially through its long-distance impacts, to protected areas. Additionally, mining also poses a disproportionate risk, potentially through short-distance impacts and water pollution, to wetlands. In both cases, very large portions of the total mine area within the grasslands fall within a ten-kilometre buffer of their boundaries, consistent with previous studies (Durán et al., 2013; Edwards et al., 2014). Whilst some of the impacts of mining activities might cause slight harm to the biodiversity in these areas, such as noise pollution potentially, other impacts, such as water pollution, could completely decimate sensitive ecosystems.

Though the mining area within protected areas was not higher than expected, if mines were randomly dispersed throughout the grasslands, this is still a major cause for concern. The grassland protected areas with mine cover within their borders were all legally designated protected areas, i.e. regions were not mining should theoretically take place. Though there are many possible explanations for the observed mining area within the protected areas (see Chapter 2), the outcome is the same. Many studies have shown that mines near protected areas and other sensitive biodiversity areas can lead to serious damage of the area’s biodiversity (Department of Environmental Affairs et al., 2013; Edwards et al., 2014; Hockings et al., 2006; Hoskin & Goosem, 2010; Kerley et al., 2002; Kociolek et al., 2011; Sonter et al., 2018; Sonter et al., 2014; Wickham et al., 2013). As the grasslands are already poorly protected (SANBI, 2013), reduced biodiversity, and consequently conservation value, due to surrounding land-uses, should be avoided.

I found no link between the total mining area within a vegetation type, and the conservation value of a vegetation type. One potential explanation for this is fortunately, that there simply is not more mining in more biodiverse areas. However, as large-scale mining has taken place in South Africa since the 1880s, this might also indicate that extensive mining has already had an effect on biodiversity in that region.

As very little quantitative information is available on what risk mining poses to agriculture and biodiversity within South Africa, on a large scale, this study has the potential to guide policymakers and land-use planners. Additionally, this project hopes to guide future efforts to assess and monitor the risks that mining can pose, to allow better planning and management.

High-value agricultural land, and the associated water sources and river courses are important to the current, and future food security of South Africa. However, mining offers short-term financial gains, potentially to the detriment of long-term food security (Auty, 2002). Similarly, biodiversity priority zones within the South African grasslands harbour rich, but often sensitive biodiversity, adjacent to some of the most ecologically damaging and largest earth-moving operations in the world. Mining poses a disproportionate risk to high-value agricultural land, water sources and river courses, and biodiversity priority zones such as protected areas and wetlands, as shown in this study. Therefore, I urge policymakers and land-use planners so weight the short-term financial gains against the long-term food security and biodiversity of South Africa, and take actions to preserve the little high-value agricultural land, and the rich biodiversity within South Africa’s grasslands.

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Supplementary material

1. Chapter 1 supplementary material

Table S1.1. Mine classes, here collectively referred to as ‘mines’, as defined in the 1990 and 2018 National Land Cover Maps (GeoTerraImage, 2014, 2019).

1990 National Land Cover Map	2018 National Land Cover Map
Mine Buildings	**Mines: Surface Infrastructure**
Mine (1) Bare and Mine (2) Semi-bare	**Mines: Extraction Sites: Open Cast & Quarries combined**
	Mines: Extraction Sites: Salt Mines
	Mines: Waste (Tailings) & Resource Dumps

Table S1.2. The observed and mean expected (based on 1000 randomisations of mine polygons) mine area, within the high-value agricultural land (HVAL) of four economically important crops (maize, soybean, sorghum, and sunflower), in South African grasslands. Shown are the results comparing the expected versus observed mine cover in 1990, 2018, and for new mines established between 1990 and 2018 (i.e. excluding 1990 and earlier mines). Significant differences between the observed and expected mine areas are indicated in bold. CIs = confidence intervals, generated from 1000 randomisations.

Year	Crop	Observed mining area (km2)	Random mining area confidence intervals (km2)
1990	**Maize**	**16.84**	**56.89 ± 1.55**
	Soybean	**0.41**	**23.15 ± 0.93**
	Sorghum	**184.16**	**183.19 ± 2.64**
	Sunflower	**183.4**	**138.53 ± 2.44**
2018	**Maize**	**53.31**	**102.44 ± 2.05**
	Soybean	**0.91**	**40.96 ± 1.17**
	Sorghum	**525.57**	**325.23 ± 3.39**
	Sunflower	**520**	**199.43 ± 8.84**
New mines	**Maize**	**49.99**	**73.67 ± 1.69**
	Soybean	**0.84**	**29.74 ± 0.99**
	Sorghum	**479.65**	**235.54 ± 2.81**
	Sunflower	**474.56**	**178.88 ± 2.58**

Table S1.3. Observed and expected (based on 1000 randomisations of mine polygons) annual mine cover increase (km^2/year) between 1990 and 2018, in the HVAL of four economically important crops (i.e. maize, soybean, sorghum, and sunflower) in South African grasslands.

Crop	Observed annual mine cover increase (km2/year)	Expected annual mine cover increase (km2/year)
Maize	**1.3**	**1.63**
Soybean	**0.02**	**0.64**
Sorghum	**12.19**	**5.07**
Sunflower	**12.02**	**2.18**

Table S1.4. Water sources (km^2), and river courses (km), at risk of pollution from mining, for maize, soybean, sorghum, and sunflower high-value agricultural land.

Crop	Maize	Soybean	Sorghum	Sunflower
HVAL water sources				
At risk (km^2)	**16950.16**	**3534.714**	**67459.55**	**55291.81**
Total area (km^2)	**35431.71**	**14308.26**	**113562.1**	**86127.53**
% Of total at risk	**47.83895**	**24.70401**	**59.40324**	**64.1976**
HVAL rivers				
At risk (km)	**4559.125**	**1105.206**	**14240.02**	**10852.31**
Total length (km)	**7409.478**	**2588.934**	**20228.07**	**14573.68**
% Of total at risk	**61.53098**	**42.68961**	**70.39731**	**74.46513**

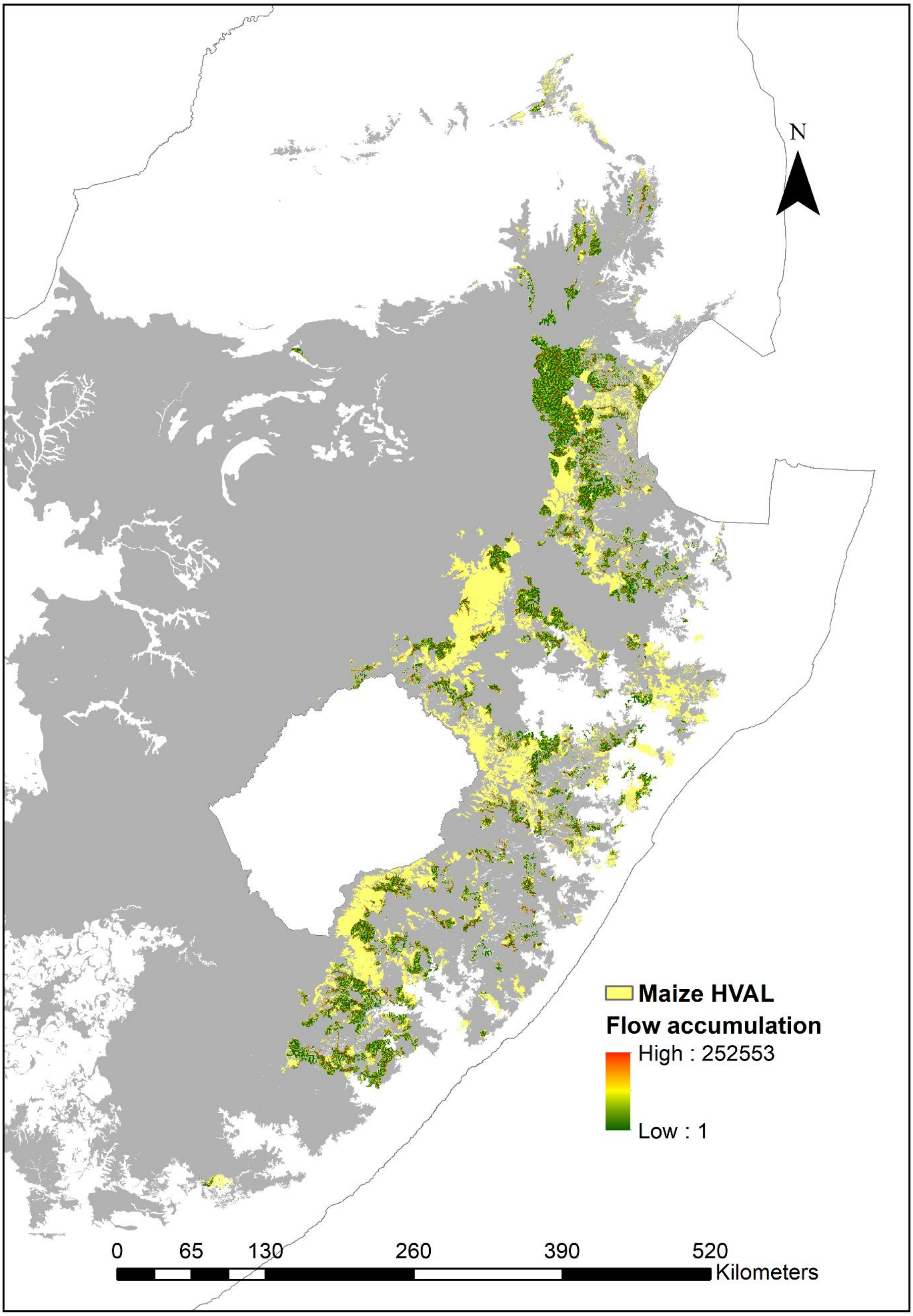

Figure S1.1. Water source pollution risk by mines, for maize high-value agricultural land (HVAL) water sources, in South Africa’s grasslands.

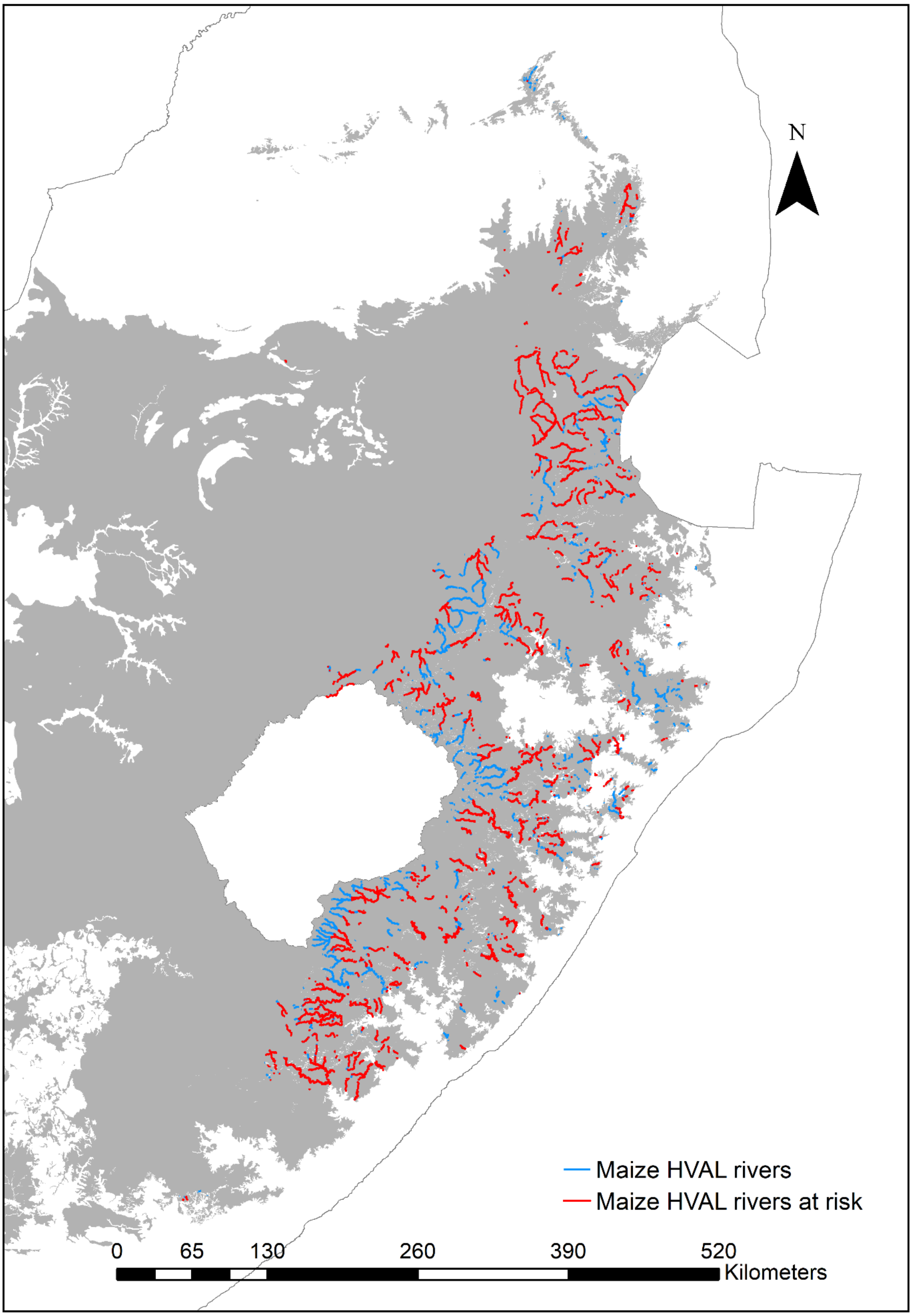

Figure S1.2. River courses at risk of pollution by mines, for maize high-value agricultural land (HVAL) river courses, in South Africa’s grasslands. Blue lines indicate all river courses within the HVAL of maize, whilst the red lines indicate the maize HVAL river courses potentially at risk of water pollution from mining.

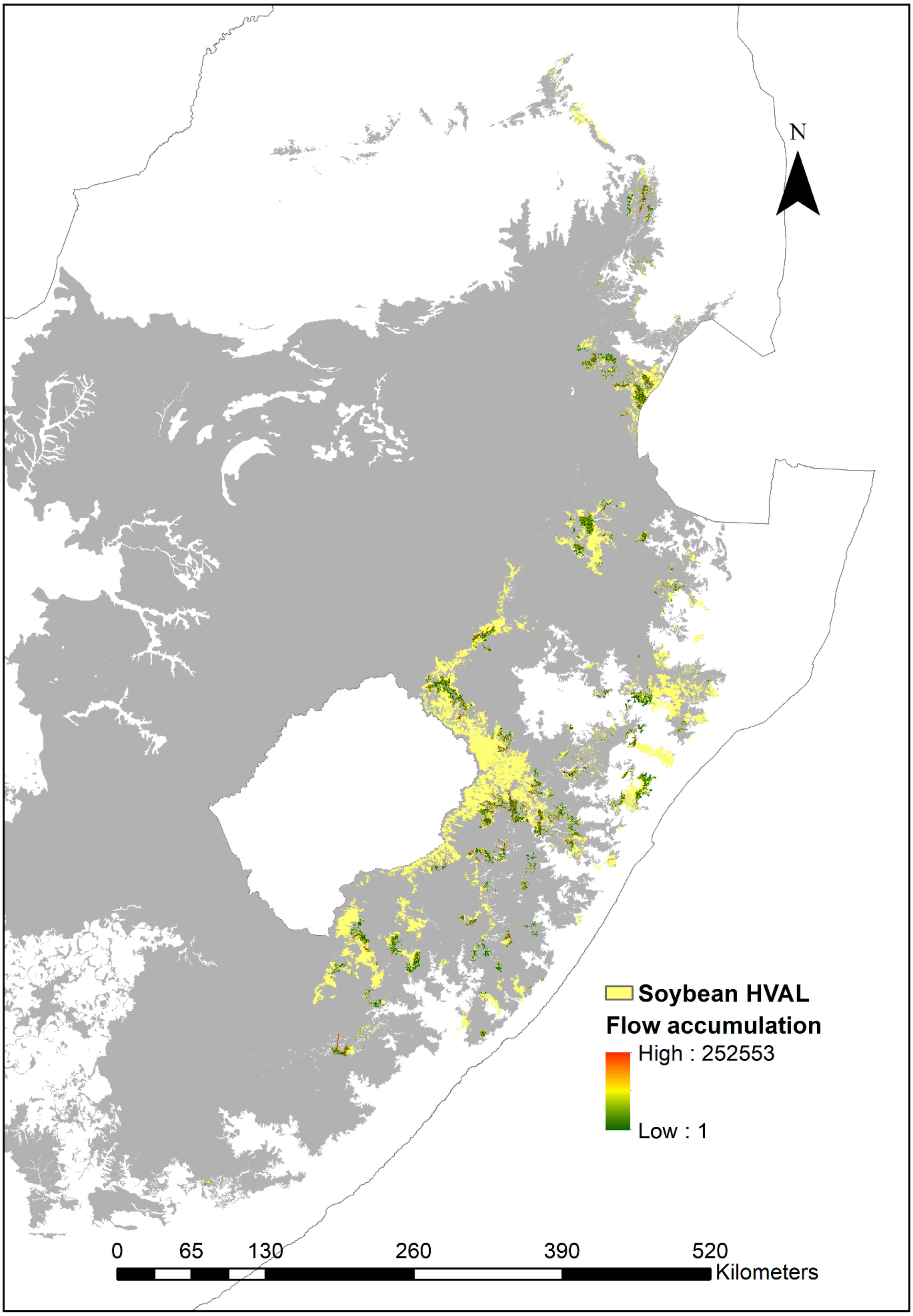

Figure S1.3. Water source pollution risk by mines, for soybean high-value agricultural land (HVAL) water sources, in South Africa's grasslands.

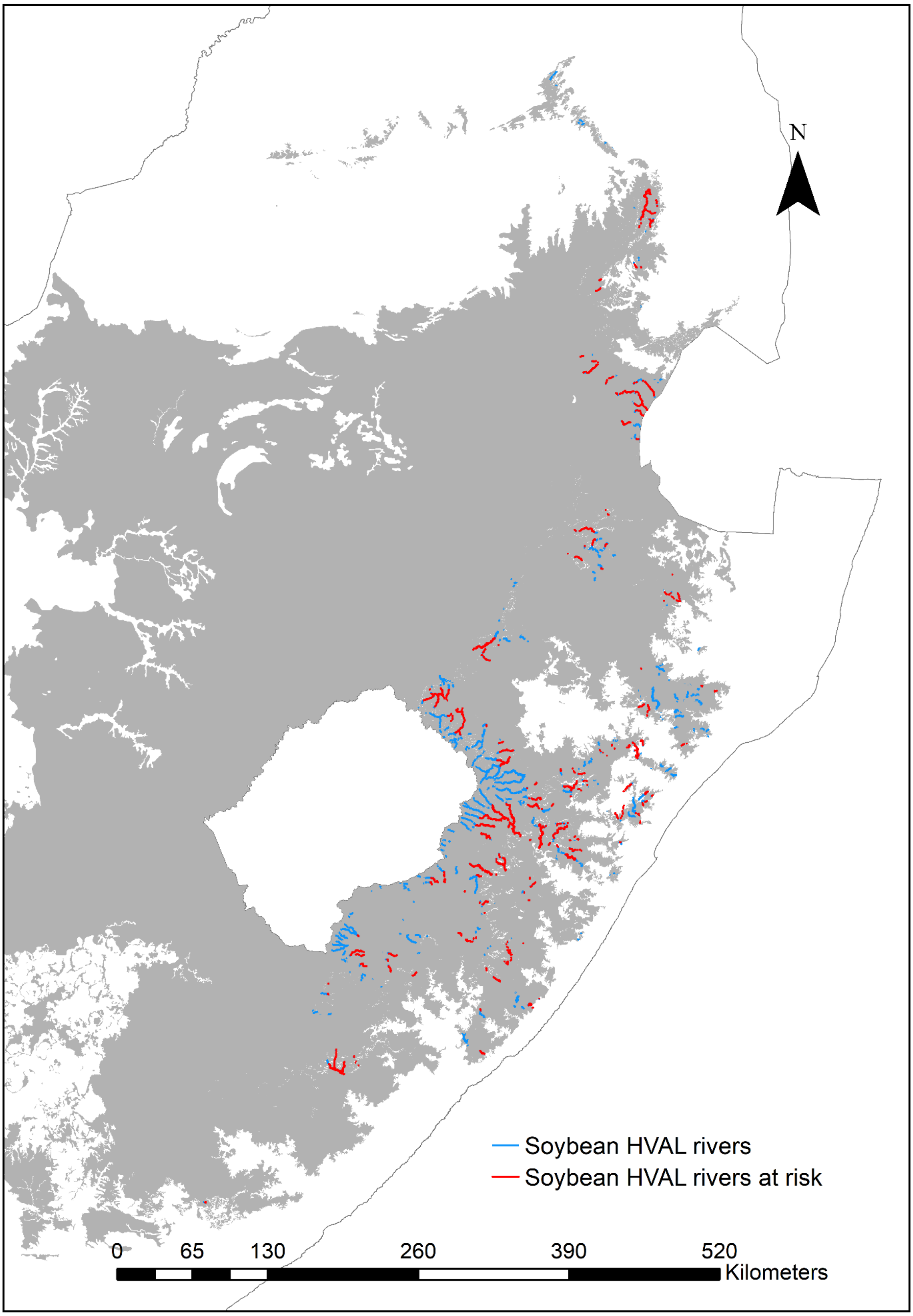

Figure S1.4. River courses at risk of pollution by mines, for soybean high-value agricultural land (HVAL) river courses, in South Africa's grasslands. Blue lines indicate all river courses within the HVAL of soybean, whilst the red lines indicate the soybean HVAL river courses potentially at risk of water pollution from mining.

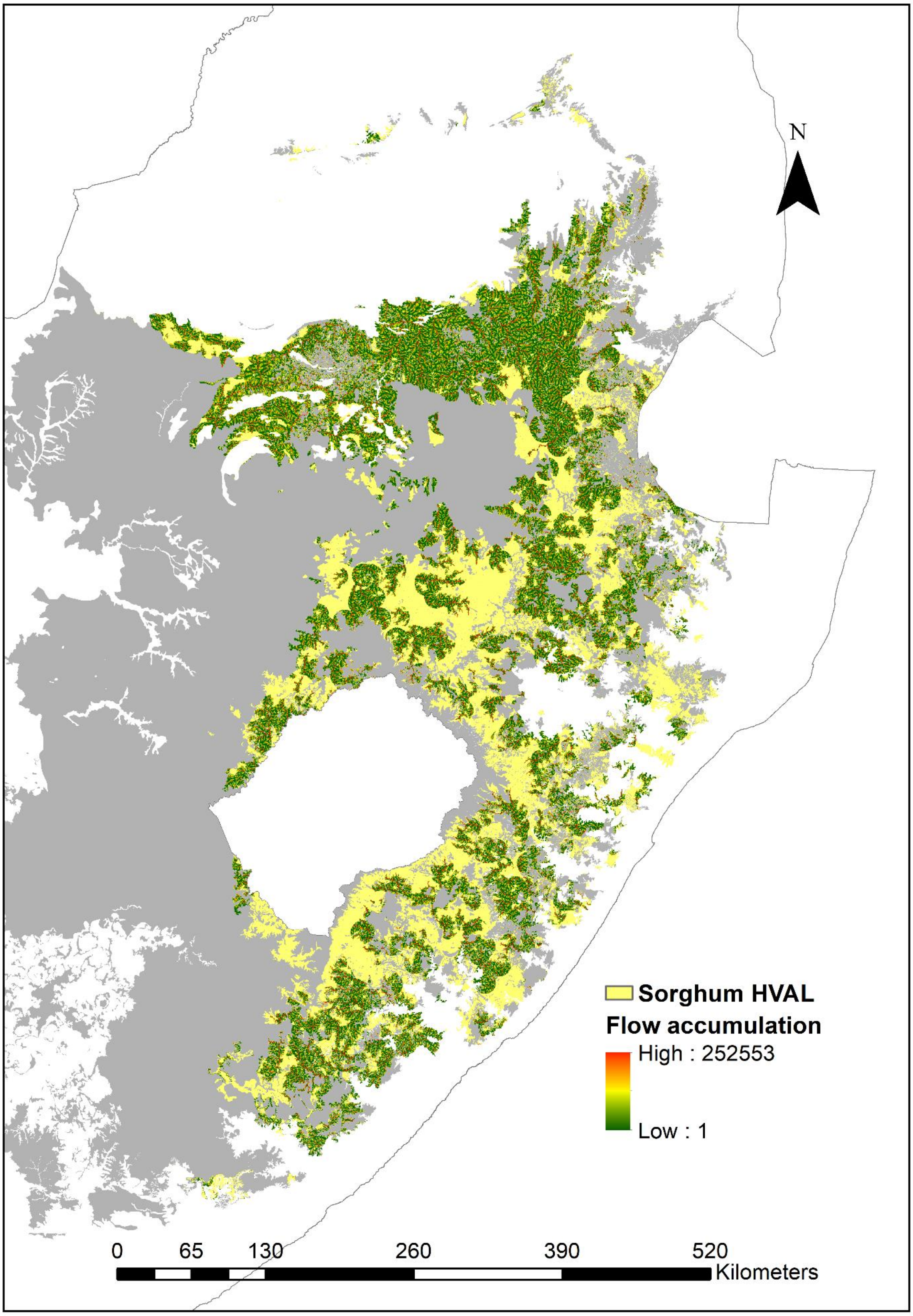

Figure S1.5. Water source pollution risk by mines, for sorghum high-value agricultural land (HVAL) water sources, in South Africa’s grasslands.

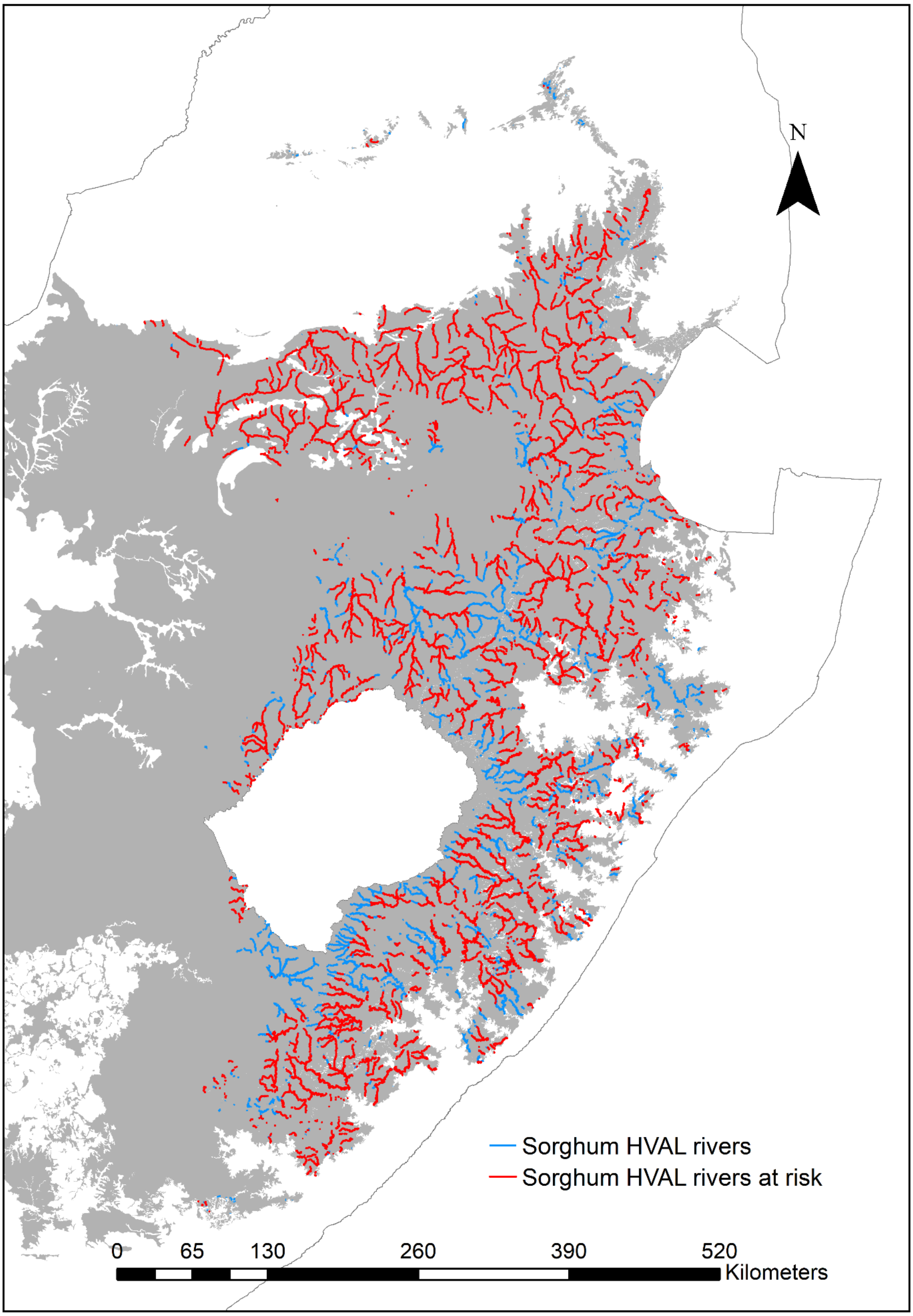

Figure S1.6. River courses at risk of pollution by mines, for sorghum high-value agricultural land (HVAL) river courses, in South Africa’s grasslands. Blue lines indicate all river courses within the HVAL of sorghum, whilst the red lines indicate the sorghum HVAL river courses potentially at risk of water pollution from mining.

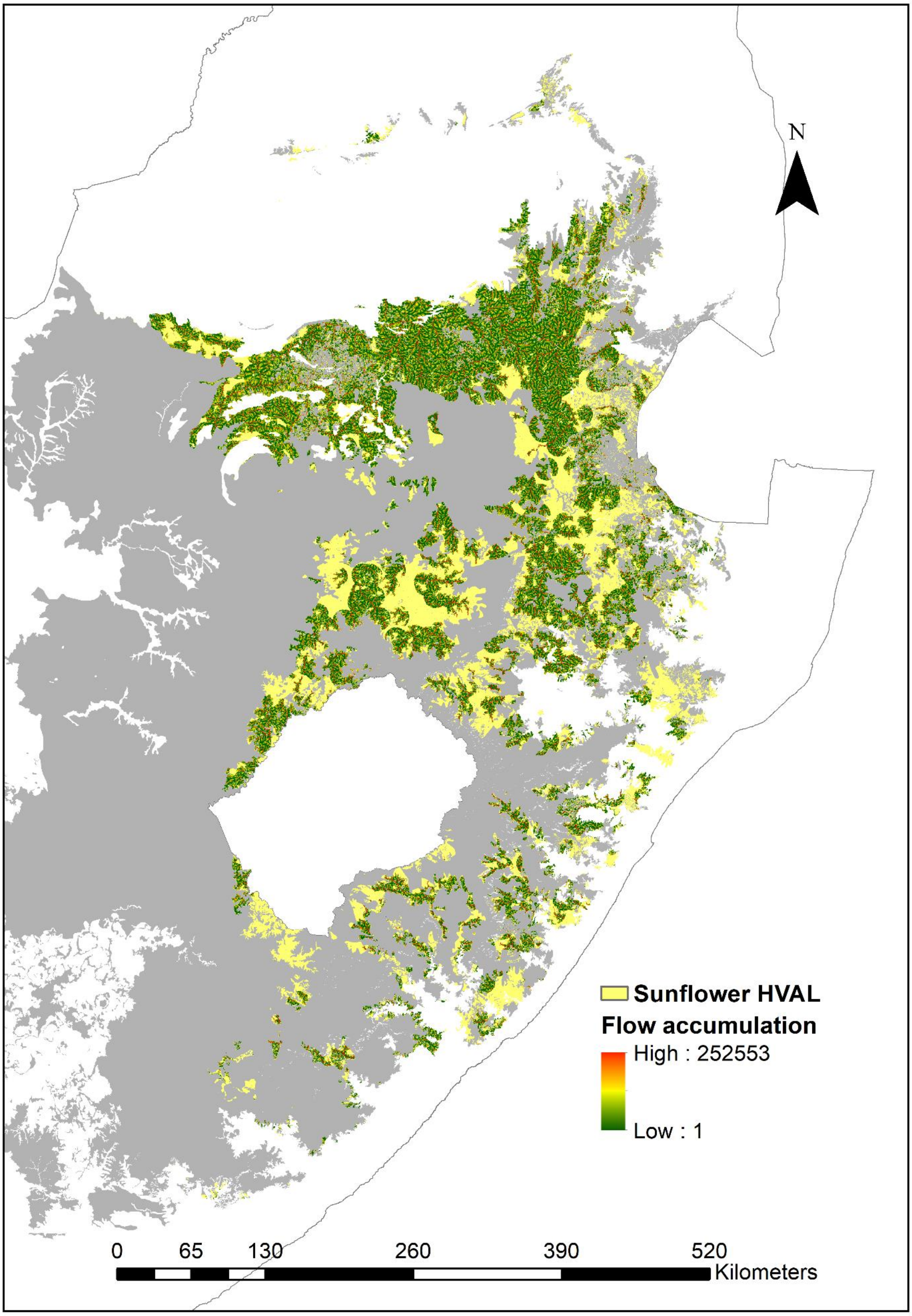

Figure S1.7. Water source pollution risk by mines, for sunflower high-value agricultural land (HVAL) water sources, in South Africa's grasslands.

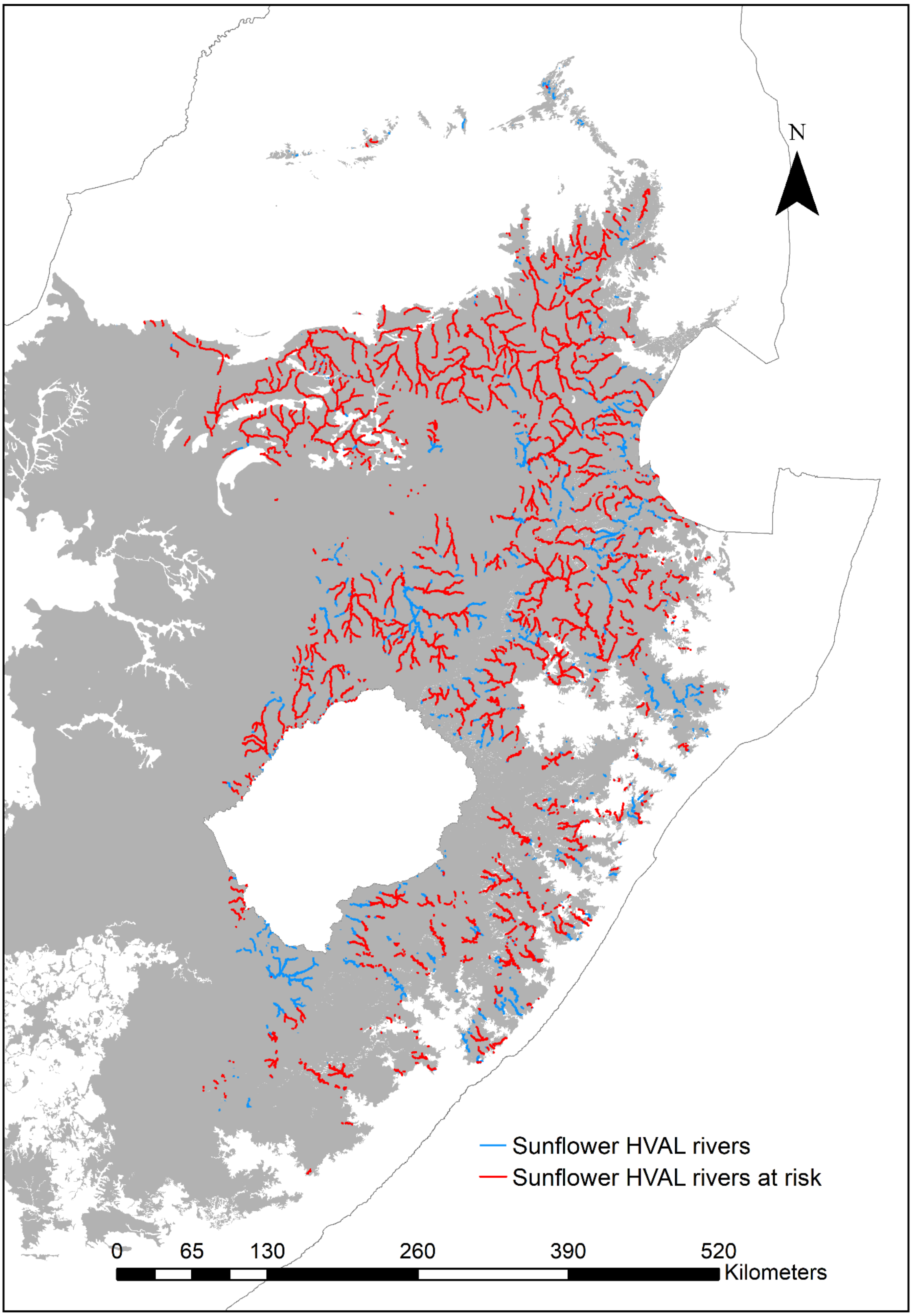

Figure S1.8. River courses at risk of pollution by mines, for sunflower high-value agricultural land (HVAL) river courses, in South Africa's grasslands. Blue lines indicate all river courses within the HVAL of sunflower, whilst the red lines indicate the sunflower HVAL river courses potentially at risk of water pollution from mining.

2. Chapter 2 supplementary material

Table S2.1. Untransformed land classes, as defined in the 1990 and 2018 National Land Cover Maps (GeoTerraImage, 2014, 2019).

1990 National Land Cover Map	2018 National Land Cover Map
Indigenous forest	Contiguous Forest
Dense bush, thicket & tall dense shrubs	Contiguous Low Forest & Thicket
Woodland and open bush	Dense Forest & Woodland
No direct equivalent – best is Grassland	Open Woodland
Low shrubland (other)	Low Shrubland (other regions)
Low shrubland (fynbos)	Low Shrubland (Fynbos)
No direct equivalent – best is Low shrub Other	Low Shrubland (Succulent Karoo)
	Low Shrubland (Nama Karoo)
Grassland	Sparsely Wooded Grassland
	Natural Grassland
No direct equivalents – best is a combination of water-seasonal or water -permanent	Natural Rivers
	Natural Estuaries & Lagoons
	Natural Lakes
	Natural Pans (flooded)
Wetlands	Herbaceous Wetlands (currently mapped)
	Herbaceous Wetlands (previous mapped extent)
No direct equivalent – best is Dense bush / thicket	Mangrove Wetlands
No direct equivalent – best is Bare non vegetated	Natural Rock Surfaces
	Dry Pans
No direct equivalent – best is Bare non vegetated	Sand Dunes (terrestrial)
	Coastal Dunes & Beach Sand
	Bare Riverbed Material

Table S2.2. The observed mine area, and the mean expected mine area (based on 1000 randomisations of mine polygons), within protected areas, important bird areas, and wetlands, and their respective 0-1 km, 1-5 km, and 5-10 km surrounding buffers, within South African grasslands. Significant differences between the observed and expected mine areas are indicated in bold. CIs = confidence intervals, generated from 1000 randomisations.

Biological region	**Buffer**	**Observed mining area (km2)**	**Mean expected mining area (km2) +- CIs**
Protected areas	**Within**	**19.88**	**39.04 ± 1.24**
	0-1 km	**29.65**	**38.86 ± 0.96**
	1-5 km	**189.73**	**165.35 ± 2.41**
	5-10 km	**341.89**	**233.77 ± 2.81**
Important bird areas	**Within**	**47.72**	**99.24 ± 2.22**
	0-1 km	**16.86**	**33.95 ± 0.92**
	1-5 km	**51.28**	**101.31 ± 2.09**
	5-10 km	**86.21**	**106.79 ± 2.04**
Wetland areas	**Within**	**25.77**	**29.13 ± 0.51**
	0-1 km	**752.89**	**367.03 ± 3.01**
	1-5 km	**417.64**	**462.7 ± 3.27**
	5-10 km	**59.54**	**192.34 ± 2.83**

Table S2.3. The mining area, endemic species richness (ESR), and ESR weighted according to the remaining natural area, for each South African grassland vegetation type, based on records in the ‘Vegetation of South Africa’ (Ladislav Mucina & Rutherford, 2006).

Grassland Vegetation Type	Mining Area (km2)	ESR	ESR/Area
Drakensberg-Amathole Afromontane Fynbos	**0**	**14**	**109.25**
Northern Escarpment Afromontane Fynbos	**0**	**3**	**32.38**
Drakensberg Afroalpine Heathland	**0**	**16**	**22.9**
Northern Escarpment Quartzite Sourveld	**0.56**	**38**	**11.91**
Woodbush Granite Grassland	**0**	**4**	**11.49**
Long Tom Pass Montane Grassland	**0.08**	**16**	**8.69**
Northern Zululand Mistbelt Grassland	**0.3**	**6**	**8.44**
uKhahlamba Basalt Grassland	**0**	**83**	**7.22**
Amathole Mistbelt Grassland	**0.01**	**9**	**7.14**
Soutpansberg Summit Sourveld	**0**	**5**	**6.59**
Wolkberg Dolomite Grassland	**0**	**7**	**6.3**
Northern Escarpment Dolomite Grassland	**4.75**	**8**	**5.8**
Barberton Montane Grassland	**0.25**	**12**	**5.46**
Strydpoort Summit Sourveld	**0.08**	**3**	**3.87**
Bloemfontein Karroid Shrubland	**0.54**	**1**	**2.99**
Midlands Mistbelt Grassland	**1.06**	**10**	**1.92**
Sekhukhune Montane Grassland	**6.43**	**4**	**1.46**
Amathole Montane Grassland	**0.31**	**26**	**1.17**
Lesotho Highland Basalt Grassland	**0.11**	**28**	**0.92**
Rand Highveld Grassland	**105.72**	**8**	**0.59**
Ithala Quartzite Sourveld	**0.11**	**4**	**0.59**
Lydenburg Thornveld	**2.23**	**3**	**0.58**
Wakkerstroom Montane Grassland	**0.87**	**5**	**0.56**
KaNgwane Montane Grassland	**2.32**	**4**	**0.45**
Southern KwaZulu-Natal Moist Grassland	**0.46**	**1**	**0.44**
Southern Drakensberg Highland Grassland	**0.49**	**17**	**0.44**
KwaZulu-Natal Highland Thornveld	**6.22**	**3**	**0.44**
Basotho Montane Shrubland	**1.46**	**2**	**0.29**
Paulpietersburg Moist Grassland	**1.55**	**1**	**0.27**
Drakensberg Foothill Moist Grassland	**1.71**	**8**	**0.25**
Karoo Escarpment Grassland	**0.54**	**10**	**0.22**
Northern Drakensberg Highland Grassland	**0.1**	**2**	**0.18**
East Griqualand Grassland	**2.18**	**4**	**0.17**
Northern KwaZulu-Natal Moist Grassland	**11.62**	**1**	**0.09**

Low Escarpment Moist Grassland	**0.07**	**1**	**0.06**
Besemkaree Koppies Shrubland	**3.24**	**4**	**0.06**
Zastron Moist Grassland	**1.48**	**1**	**0.05**
Stormberg Plateau Grassland	**0.98**	**1**	**0.04**
Xhariep Karroid Grassland	**7.1**	**3**	**0.04**
Carletonville Dolomite Grassland	**37.9**	**1**	**0.03**
Vaal-Vet Sandy Grassland	**83.64**	**1**	**0.03**
Aliwal North Dry Grassland	**1.98**	**0**	**0**
Amersfoort Highveld Clay Grassland	**2.97**	**0**	**0**
Bedford Dry Grassland	**0.11**	**0**	**0**
Bloemfontein Dry Grassland	**8.92**	**0**	**0**
Central Free State Grassland	**42.98**	**0**	**0**
Dry Coast Hinterland Grassland	**1.19**	**0**	**0**
Eastern Free State Clay Grassland	**3.91**	**0**	**0**
Eastern Free State Sandy Grassland	**2.63**	**0**	**0**
Eastern Highveld Grassland	**467.35**	**0**	**0**
Egoli Granite Grassland	**10.67**	**0**	**0**
Frankfort Highveld Grassland	**2.01**	**0**	**0**
Income Sandy Grassland	**4.99**	**0**	**0**
Klerksdorp Thornveld	**17.78**	**0**	**0**
Leolo Summit Sourveld	**0**	**0**	**0**
Mabela Sandy Grassland	**0.58**	**0**	**0**
Moist Coast Hinterland Grassland	**1.01**	**0**	**0**
Mooi River Highland Grassland	**0.32**	**0**	**0**
Mthatha Moist Grassland	**3.62**	**0**	**0**
Northern Free State Shrubland	**0**	**0**	**0**
Queenstown Thornveld	**1.83**	**0**	**0**
Senqu Montane Shrubland	**0**	**0**	**0**
Soweto Highveld Grassland	**128.97**	**0**	**0**
Steenkampsberg Montane Grassland	**7.2**	**0**	**0**
Tarkastad Montane Shrubland	**0.6**	**0**	**0**
Tsakane Clay Grassland	**21.28**	**0**	**0**
Tsomo Grassland	**1.88**	**0**	**0**
Vaal Reefs Dolomite Sinkhole Woodland	**29.96**	**0**	**0**
Vredefort Dome Granite Grassland	**0.71**	**0**	**0**
Waterberg-Magaliesberg Summit Sourveld	**0.01**	**0**	**0**
Western Free State Clay Grassland	**19.67**	**0**	**0**
Western Highveld Sandy Grassland	**14.71**	**0**	**0**
Winburg Grassy Shrubland	**2.51**	**0**	**0**

Table S2.4. The ten vegetation types containing the highest total mining area (km^2) within their borders, as well as their respective endemic species richness (ESR), and red list weighted ESR (RWESR) per 100 km^2 remaining natural area.

Grassland vegetation type	Mining area (km2)	ESR	RWESR per 100 km2
Eastern Highveld Grassland	**467.35**	**0**	**0**
Soweto Highveld Grassland	**128.97**	**0**	**0**
Rand Highveld Grassland	**105.72**	**8**	**0.59**
Vaal-Vet Sandy Grassland	**83.64**	**1**	**0.03**
Central Free State Grassland	**42.98**	**0**	**0**
Carletonville Dolomite Grassland	**37.9**	**1**	**0.03**
Vaal Reefs Dolomite Sinkhole Woodland	**29.96**	**0**	**0**
Tsakane Clay Grassland	**21.28**	**0**	**0**
Western Free State Clay Grassland	**19.67**	**0**	**0**
Klerksdorp Thornveld	**17.78**	**0**	**0**

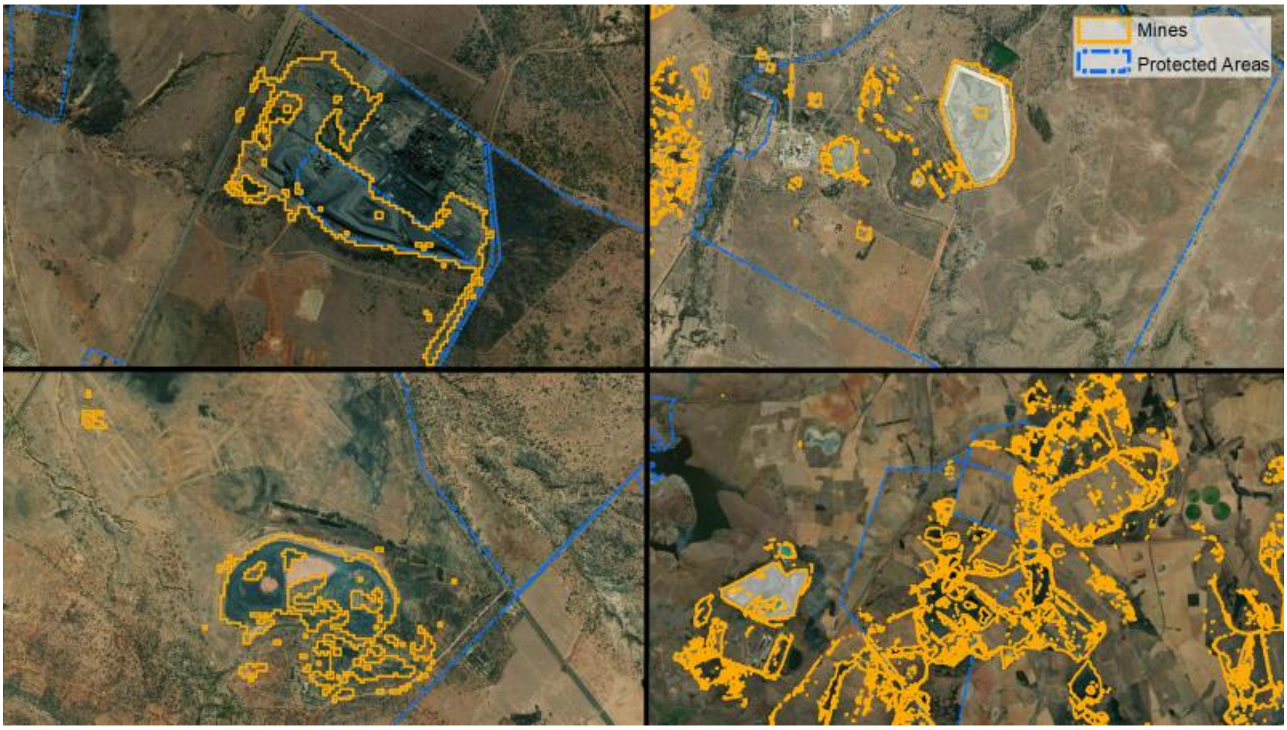

Figure S2.1. Examples of mines within protected area boundaries in 2018, in the (a) Lydenburg Nature Reserve, (b) H. J. Joel Private Nature Reserve, (c) Balele Game Park, (d) Heyns Private Nature Reserve, in the South African grasslands.