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Perspective

Global drivers of change across tropical savannah ecosystems and insights into their management and conservation

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ABSTRACT

All tropical savannahs are experiencing extensive transformation and degradation, yet conservation strategies do not adequately address threats to savannahs. Here, using a recently published ecosystem intactness metric, we assess the current condition of tropical savannahs across Earth, finding that <3 % remain highly intact. Moreover, their overall levels of protection are low, and of the protected savannahs, just 4 % can be considered highly intact while the majority (>60 %) are in poor condition. In order to address the clear mismatch between the decline in tropical savannah ecosystems' condition and the response to manage and conserve them, we reviewed the current drivers that lead to tropical savannah degradation and identified conservation approaches being used to address them. Many successful conservation approaches address multiple drivers of change but are applied across small areas. We argue these approaches have the potential to be up-scaled through integrated land-use planning.

1. Introduction

The increasingly intensive human influence across tropical savannah ecosystems has led to extensive degradation, resulting in profound declines of biodiversity and ecosystem services (IPBES, 2019; Millennium Ecosystem Assessment, 2005a, 2005b; Romero-Ruiz et al., 2012; Shukla et al., 2019; Western et al., 2009; Woinarski et al., 2010). Recent rates of tropical savannah loss are among the highest of all ecosystem types (Hoekstra et al., 2005; Williams et al., 2020b), and their overall protection is disproportionately low (Watson et al., 2016a). This is concerning as they are important components of global biodiversity (Baillie et al., 2004), and for people through the many ecosystem services they provide (Bond and Parr, 2010; Grace et al., 2006; Greiner et al., 2009; Kundhlande et al., 2000; Millennium Ecosystem Assessment, 2005b; Russell-Smith and Sangha, 2018). Rapid and severe biodiversity declines are being reported in tropical savannahs from every continent (Etter et al., 2017; Mustin et al., 2017; Mwangi and Ostrom, 2009; Vargas et al., 2015; Woinarski et al., 2011), yet there appears to be a large research bias towards understanding how biodiversity is being affected in other ecosystems (such as tropical forest), even though tropical savannahs are being impacted just as much, if not more, by a range of drivers and support outstanding values of biodiversity and ecosystem services (Fig. 1) (Murphy et al., 2016). Additionally, there is some evidence that savannah conservation is sometimes overlooked in national

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planning initiatives and global policy discussions (Bonanomi et al., 2019; Parr et al., 2014).

Many countries have been recently targeting tropical savannah regions for agricultural development, and have even incentivized their development (Australian Government, 2015; Clements and Fernandes, 2013; Departamento Nacional de Planeación, 2019; Morán-Ordóñez et al., 2017; Moreno, 2000; Williams et al., 2020a). It was the 'green revolution' in the 1960s and its dramatic advancement of agricultural production technologies, that led human influence to intensify across savannah ecosystems worldwide, because nutrient-poor tropical savannah soils could be improved and farmed more easily (Blaustein, 2008). This development has at times been described in high-profile international reports as having a low environmental cost due to high aboveground carbon gains and low biodiversity impacts (Alexandratos and Bruinsma, 2012; Edenhofer et al., 2011; Lewis et al., 2019; Searchinger et al., 2015). Such misconceptions, and the expected future increases in food production demand to meet a growing global population (Davis et al., 2016), coupled with other influences such as human-caused climate change (IPCC, 2018), constitute major management challenges across the tropical savannah biome.

While the drivers of change in tropical savannahs are slowly gaining recognition, key gaps remain in how best to ensure their long-term persistence as outlined in global conservation agendas such as the Convention for Biological Diversity (Secretariat of the Convention on Biological Diversity, 2020). In this Perspective, we first provide a contemporary, spatially-explicit assessment of the extent and current state (for the year 2013) of tropical savannahs using a global ecosystem intactness metric that reflects habitat area, quality and fragmentation effects (Beyer et al., 2019). We then identify the main drivers that threaten savannah-dependent tropical species by compiling data from the IUCN Threats Classification Scheme. Finally, we link these drivers to existing successful land management and conservation practices that address them from across the different savannah-containing continents. By drawing links between the state, drivers of change and successful conservation approaches, we highlight the immediate conservation actions to be taken and discuss how they can be scaled up for tropical savannah conservation.

1.1. Biodiversity and ecosystem service value of tropical savannahs

Tropical savannahs were once thought to be a form of degraded forests by many scientists (Pausas and Bond, 2019; Ratnam et al., 2011), but are now recognised as unique ecosystems (but see (Kumar et al., 2019)). They are comprised of a continuous layer of C4 grasses (C4 grasses refers to species that have additional steps in the photosynthetic cycle that make the photosynthetic pathways to capture carbon dioxide more suited to warmer conditions) inter-mixed with often thick-barked trees with an open canopy that is typically burnt frequently (Bond and Parr, 2010; Ratnam et al., 2011). They are broadly located in the tropics, found mainly in Africa, Australia and South America, and cover one-sixth of Earth's land surface according to our definition (Fig. 2). These ecosystems account for approximately 30 % of the primary production of all terrestrial vegetation, supporting a range of important provisioning, regulating and cultural ecosystem services to both local people and the global community (Fig. 2) (Grace et al., 2006; Greiner et al., 2009; Kundhlande et al., 2000; Russell-Smith and Sangha, 2018).

The main provisioning services delivered are food and raw materials, including livestock, forestry, and other agricultural products (Greiner et al., 2009; Phalan et al., 2013). But they also provide vital drinking water to people, and provide sources of irrigation to support surrounding agricultural activities (Fig. 2; Greiner et al., 2009; Kundhlande et al., 2000; Mora-Fernández et al., 2015; Strassburg et al., 2017). In addition, they provide cultural, knowledge, spiritual, aesthetic and recreational services to a diversity of people (Russell-Smith and Sangha, 2018; Sangha et al., 2017). Tropical savannahs are associated with several of Earth's nomadic and Indigenous peoples, many of whom rely on these ecosystems for subsistence, identity, and spirituality (Fig. 2; De Groot et al., 2013; Howlett and Lawrence, 2019).

Savannahs are also highly biodiverse (Morales-Martínez et al., 2018). They contain approximately 8000 vertebrate species, of which about 2500 are threatened and 1000 are endemic to the tropical savannah environment (Baillie et al., 2004). Their vertebrate species diversity is comparable to that of tropical forests. For example, globally they have a mean richness of 78 mammals and 284 birds 10 km^{-2}, while the tropical and subtropical moist broadleaf forest biome has a mean richness of 87 mammals and 296 birds 10 km^{-2} (Murphy et al., 2016). In the Llanos savannahs of Colombia bat species richness has been shown to be similar, or even higher, than Choco and Amazon forests (Morales-

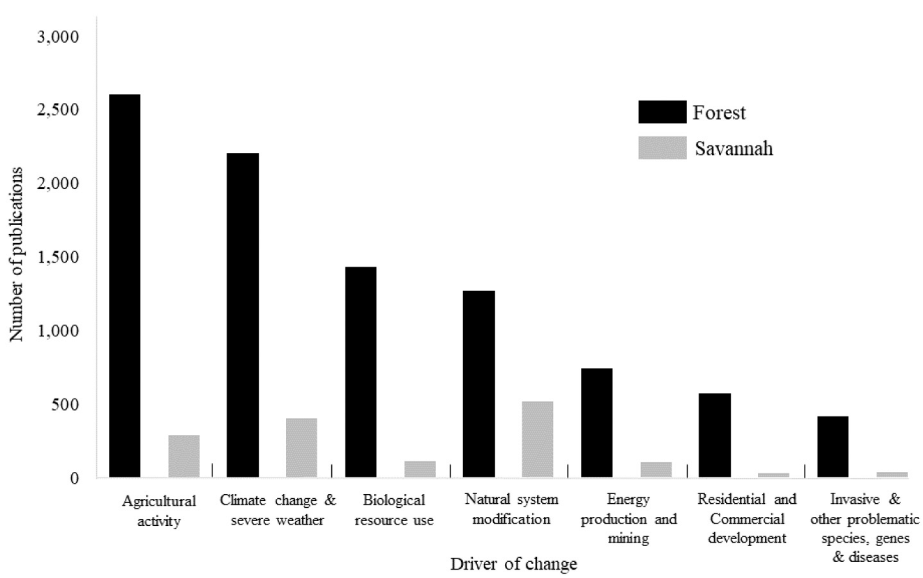

Fig. 1. Search conducted within the ISI Web of Science database for terms relating to drivers of change within tropical savannah and tropical forest (1978–2020). Despite tropical savannah and tropical forest ecosystems both being extensive (approximately 19,575,000 and 24,348,000 km^2, respectively), and species diversity being comparable (see "Biodiversity and ecosystem service value of tropical savannahs" section), there are 6 times more publications for forests overall. Drivers included were those that threatened the highest number of tropical savannah species (see Fig. 5). Search terms included the respective driver of change, 'tropical', ecosystem types: 'forest*', 'savanna*', and biodiversity: 'biodiversity', 'fauna*', 'amphibia*', 'mammal*', 'bird*', 'invertebrate*', 'reptil*', 'flora*', 'vegetation*'. The Boolean operator 'AND' was used between the respective driver of change, tropical, biodiversity search terms and ecosystem type. The Boolean operator 'OR' was used between biodiversity-related terms. This search established the general trend that there is a bias in the peer-reviewed literature towards forest ecosystems across all drivers. While not a systematic literature review, this figure is representative of the broader academic publishing trends. This figure is standalone and does not reflect the methods of other sections of this manuscript. See Supplementary methodology for further detail on search terms and Table A1 for the specific literature returned.

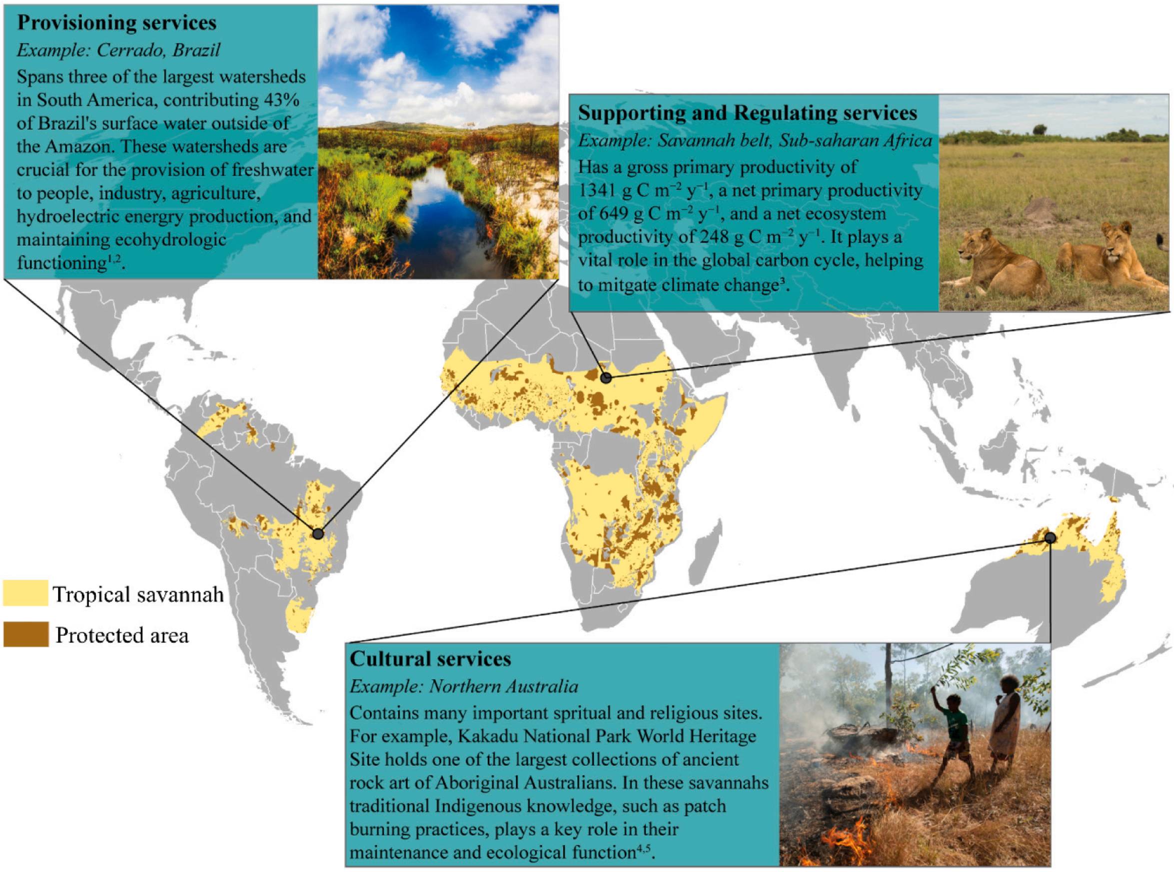

Fig. 2. Examples of ecosystem services that tropical savannahs provide across the continents. The global distribution of tropical savannah (as defined by (Olson et al., 2001) – see Supplementary methodology) is shown in yellow, and protected areas shown in brown. For protected area data, we use the World Database on Protected Areas dataset (UNESCO 2020). References: 1. (Strassburg et al., 2017) 2. (Oliveira et al., 2014) 3. (Bombelli et al., 2009) 4. (Sangha et al., 2017) 5. (Russell-Smith and Sangha, 2018). Photographs: Cerrado, Brazil - © Alisson Gontijo; Sub-Saharan Africa - ©Alexander Braczkowski Jr; Northern Australia - ©Ted Wood. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Martínez et al., 2018). Floristic diversity is typically lower than forests, but varies greatly between different types of tropical savannahs (Baruch, 2005; Mora-Fernández et al., 2015). In some locations, it can be extreme with existing records as high as 230 species in a 0.1 ha plot within the Cerrado of Brazil (recorded by (Eiten, 1978)), which rivals some tropical forests. For example, the Amazon rainforest can have 281 to 311 plant species per hectare (Martinez and Phillips, 2000).

The few assessments of the state of biodiversity and ecosystem services across tropical savannahs have frequently shown alarming outcomes. For example, a Red List of Ecosystems assessment in Colombia classified the Llanos piedmont savannahs as one of the 20 ecosystems in critical condition, with the loss of 86 % of its original extent which impacted biodiversity (Etter et al., 2017). Across Colombian savannahs, important terrestrial species habitats are being lost directly to agricultural, and riverine habitats to contamination by oil and rice activities (Lasso et al., 2010; Mora-Fernández et al., 2015). A long-term study in northern Australia's tropical savannahs revealed that small mammals are rapidly declining (Ibbett et al., 2018). Within African protected areas, large mammal population abundance has declined on average by 59 % between 1970 and 2005 (Craigie et al., 2010). These declines in biodiversity have occurred alongside declines in ecosystem services. Some savannah landscapes that people rely on to support agricultural activities are becoming unproductive due to poor management such as over-grazing or cultivation of inappropriate crops (Chianu et al., 2004; Dekkers et al., 2016; Eze, 2018; Fernandes et al., 2016; Hempson et al., 2017; Skarpe, 1991). In the Llanos of Colombia, rice crops in the southwest that decreased in productivity were displaced to the northwest of the region, where they affected highly diverse ecosystems of seasonally-flooded savannahs reducing their capacity to provide water for human use (Romero-Ruiz et al., 2012). Unsustainable practices and other anthropogenic activities such as mining have also altered hydrological processes, compromising the provision of water for human use and consumption (King et al., 2015).

1.2. The current condition of tropical savannahs and their levels of protection

We overlayed tropical savannah ecoregions (ecologically distinct geographical units that reflect the distributions of a broad range of fauna and flora across the entire planet (Olson et al., 2001)) with a recently developed continuous global intactness metric (Beyer et al., 2019), which reflects habitat area, quality and fragmentation effects by considering distance between cells, the quality of cells, and the number of cells within a spatial region - on a scale from 0 to 1 (where 0 is extremely degraded and 1 is a pristine environment; Fig. 3). Specifically, intactness (Q) is calculated as:

$$Q = \frac{\sum_{i=1}^{N}\sum_{j=i}^{N}\left(w_i w_j\right)^{Z} exp\left(-\beta d_{ij}\right)}{\sum_{i=1}^{N}\sum_{j=i}^{N} exp\left(-\beta d_{ij}\right)}$$

where d_{ij} is the distance between cells i and j (km), w (range 0–1) is a measure of the cell quality, z is an exponent that scales the product of two qualities, and N is the number of cells within a spatial unit (such as an ecoregion). Parameter β determines how the combined value of pairs of cells diminishes as a function of the distance between them. The

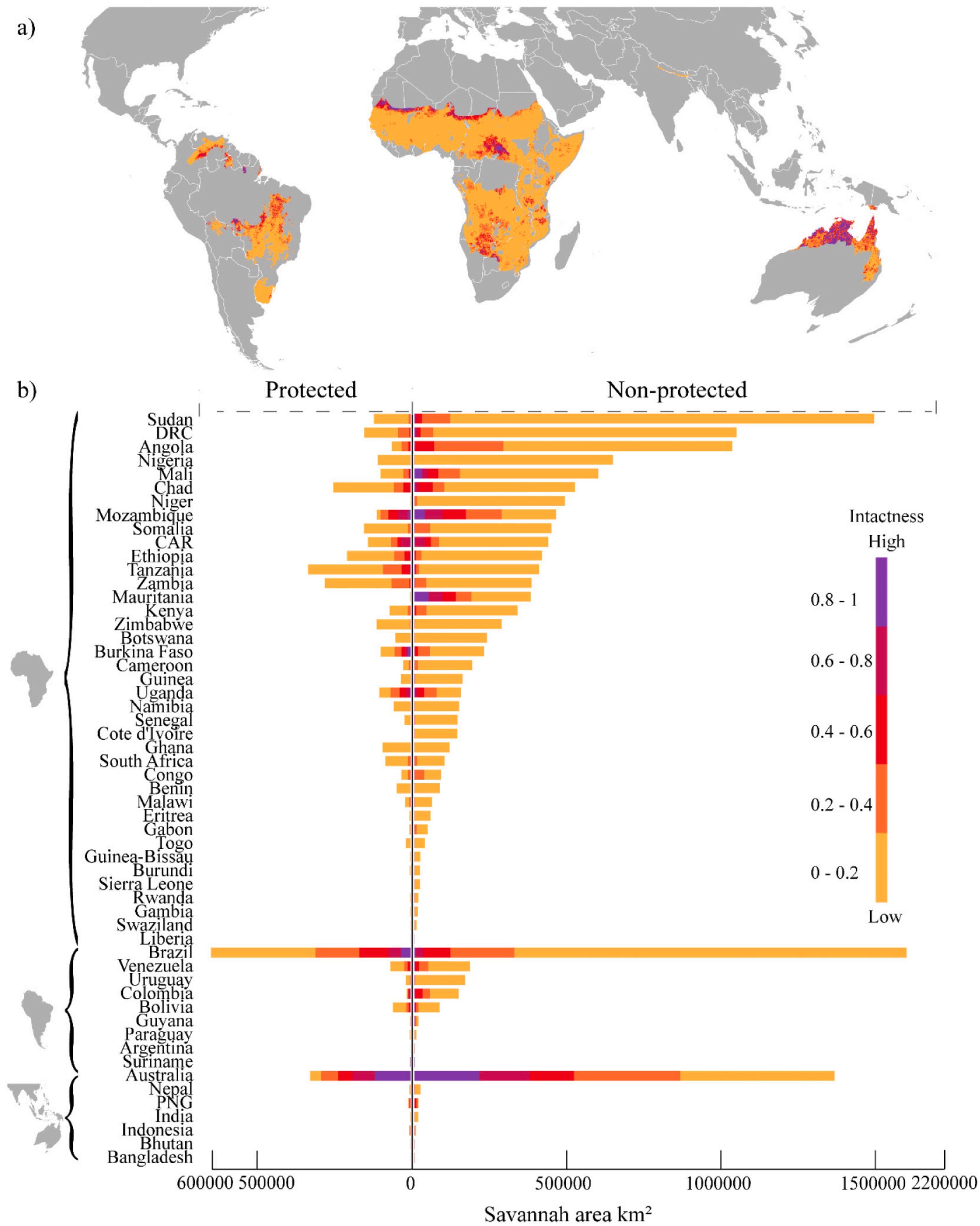

Fig. 3. The global distribution of tropical savannahs (as defined by Olson et al., 2001) (a), and the area within each country containing tropical savannah that is protected and (b) non-protected. The colours (a and b) represent 5 equal interval categories of intactness, defined by (Beyer et al., 2019), from lowest (orange) to highest intactness (purple). Protected areas are those delineated by the World Database on Protected Areas (UNEP-WCMC and IUCN, 2019). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

denominator standardizes the metric so that current state is relative to a hypothetical situation in which no degradation has occurred (all habitat weights equal one). The metric is parameterized to: (a) be proportional to habitat area when there is no habitat fragmentation; (b) to decline monotonically as fragmentation increases and to be sensitive to both the number of patches and the separation between patches; and (c) to be proportional to habitat quality for a given total area of habitat and degree of fragmentation. These criteria were achieved with values of $\beta = 0.2$ and $z = 0.5$. For more details on the intactness metric please see Beyer et al., 2019. Use of this metric requires a relative measure of habitat quality among cells. Quantifying the loss of ecosystem condition and function is challenging. In lieu of direct scientific surveys of ecological condition, proxy indicators can inform initial assessments for planning purposes (Watson and Venter, 2019). We adopted a generic measure of habitat quality to assess global intactness, using a the most up to date version (for the year 2013) of the Williams et al. (2020b) human pressure index. We note that there are other measures of human pressure available such as Ellis et al. (2021), Kennedy et al. (2019), and Jacobson et al. (2019), but we have used the human pressure index as it is the most widely used (for example see (Grantham et al., 2020; Jones et al., 2018; Pérez-Hämmerle et al., 2022; Watson et al., 2016b)) and is listed as an indicator in the First Draft of the Post-2020 Global Biodiversity Framework (Secretariat of the Convention on Biological Diversity, 2021). Most of the other measures have been found to be comparable to the human pressure index (Riggio et al., 2020), while some are starkly different (Grantham et al., 2022; Plumptre et al., 2021).

The human pressure index (Venter et al., 2016a; Williams et al., 2020b) includes pressures, all of which are well known to reduce ecosystem condition, on (1) the extent of built human environments (Aronson et al., 2014; Tratalos et al., 2007), (2) population density (Brashares et al., 2001; Burney and Flannery, 2005), (3) electric infrastructure (Guetté et al., 2018), (4) crop lands (Luck and Daily, 2003), (5) pasture lands (Kauffman and Krueger, 1984), (6) roadways (Trombulak and Frissell, 2000), (7) railways (Laurance et al., 2009), and (8) navigable waterways (Harvolk et al., 2015). Across several studies it has been shown to be a robust measure for assessing the consequences of human pressure for species and ecosystem processes (Crooks et al., 2017; Di Marco et al., 2019, 2018; Hill et al., 2021; Main et al., 2020; Tucker et al., 2018). It has the same limitations that are inherent to all cumulative pressure mapping efforts - it is not possible to fully account for all human pressures such as climate change and CO_2 fertilisation. Additionally, the latest year available for the dataset is 2013. These two factors mean that this assessment is likely an underestimation of tropical savannah degradation, given the global ecosystem decline since that time (IPBES, 2019). We note that many pressures not mapped within the tropical savannah ecosystem, such as invasive species (Spear et al., 2013) and poaching (Shaffer and Bishop, 2016), are associated with many of the pressures that are included in the human pressure index (e.g., roads, population density, and access) and their lack of direct inclusion may not strongly affect the overall results (Spear et al., 2013; Venter et al., 2016b).

We divided the continuous global intactness metric into 5 equal interval bins and described the area and protection status of savannah within these categories through spatial intersection with a protected area and tropical savannah spatial layer. Data on protected area location, and boundary were obtained from the August 2020 version of the World Database on Protected Areas dataset (Fig. 2; UNESCO 2020). We followed the methods of previous studies and excluded protected areas from the WDPA database that have a status of 'Proposed' or 'Not Reported', and points and polygons designated as 'UNESCO MAB Biosphere Reserves' as they do not meet the IUCN definition of a protected area and it is best practice to exclude them (Maxwell et al., 2020; UNEP-WCMC, 2021). For point data, we created polygons in accordance with their specified area and merged these with the main protected area dataset.

A standardised map of tropical savannah extent is still lacking, with various delineations available and little consensus within the scientific community of which one is the most appropriate (Hills, 1965; Lehmann et al., 2011; Parr et al., 2014). The lack of a standardised map for savannah is problematic as it can impede targeted efforts to conserve this important ecosystem type. For example, if a savannah (which requires management as savannahs have evolved alongside human influence (Harris, 1980)) is incorrectly classified as a degraded forest it could be left open for development or managed inappropriately, such as being excluded from natural fire regimes (Parr et al., 2014), or being relegated in importance in impact assessment frameworks such as that mandated by the International Finance Corporation (World Bank Group, 2021). Providing policy makers with an effective map of natural tropical savannahs is an important challenge that must be overcome. The lack of differentiation between savannah and grassland is also problematic, as the conversion of savannah to grassland is a key indicator of ecosystem degradation (Hoffmann and Jackson, 2000).

We utilise a broad delineation of tropical savannah, taking ecoregions that are classified as savannah (or to contain savannah) from the map of Terrestrial Ecoregions of the World (Olson et al., 2001; see Supplementary methodology for complete list). The consequence of using this delineation is that it may lead to over-estimation of tropical savannah extent as some woodland or forest-savannah mosaic ecoregions may only partially be savannah. Given the alternative definitions that exist for tropical savannah, we provide an alternative analysis using a more conservative definition of tropical savannah (5,386,444.6 km^2 less tropical savannah area), and discuss the limitations of our approach, in the Supplementary methodology. The main difference between the two definitions we present is that the Dixon et al. (2014) dataset used in the Supplementary methodology employed a set of criterion to exclude certain ecosystem types which were considered as grassland or savannah in both the Faber-Langendoen et al. (2014) and Olson et al. (2001). For example, there was a criterion for trees to have no >40 % cover in tropical ecoregions, and be typically no >8 m in height. By integrating the International Vegetation Classification of grassland types (Faber-Langendoen et al., 2014) with the map of Terrestrial Ecoregions of the World (Olson et al., 2001) the World Grassland Types dataset (Dixon et al., 2014) depicts a more comprehensive global biogeographical characterization of the Earth's grassland types than either of the datasets alone.

We found that there are very few areas remaining that can be classed as highly intact, with 469,211 km^2 (2.52 %) of all tropical savannah falling within the highest intactness category (score of >0.8). The largest expanse is in Australia 314,714 km^2, followed by Mauritania 44,784 km^2 (Fig. 3). These two countries alone contain 76.6 % of Earth's remaining highly intact tropical savannah. The majority (>50 %) of tropical savannah falls within the lowest intactness category (score of <0.2) in nearly every country ($n = 49$), with the exceptions of Namibia, Indonesia, Central African Republic, Australia, Papua New Guinea, and Suriname.

In all but a few countries the levels of protection of savannahs are low, and the majority of protected lands cover areas of low intactness (Fig. 3). The Convention on Biological Diversity's Aichi Target 11 calls for 17 % of terrestrial Earth to be protected, and for this protection to be ecologically representative (Secretariat of the Convention on Biological Diversity, 2011). Today 19.7 % of tropical savannahs globally are protected. On average countries have 20.8 % of their tropical savannah protected; however, protection levels range from 0 to 96 % with 32.4 % of all protected areas being located in Brazil, Tanzania, and Australia (Fig. 3). Globally, only 4.1 % of protected tropical savannah area falls within the highest intactness category (Fig. 3b). We note that we refer to intactness and degradation in the context of the intactness metric used, a relative and proxy measure for realised ecological condition. For a summary of the limitations of this approach and future research needs see Supplementary Methodology.

2. Drivers of tropical savannah loss and degradation

In this section, we summarise the major current and future drivers of change that are the greatest threats to the tropical savannah biome, and highlight examples of where they are occurring across Earth in Table A2. A driver is any natural or human-induced factor that directly or indirectly causes a change in an ecosystem (Nelson, 2005). The patterns and the information summarised in the following sections, were identified through a narrative review of published literature. We searched for recent literature in relation to tropical savannah drivers of change, conservation and management. Given the many drivers of change in tropical savannahs we focused on those that have been identified by the International Union for Conservation of Nature (IUCN) as threats to the most tropical savannah species (acknowledging this is only one facet of biodiversity) (IUCN, 2020; Fig. 5). We employed this method as we did not intend to provide an exhaustive coverage of the literature or identify a bias in literature, but rather to investigate global trends based on expertise such as those elucidated from IUCN assessments, and identify the gaps in linking drivers of change to conservation and management actions. To extract the threats and number of species from the IUCN database we used the rredlist R package (Chamberlain, 2017) which accesses the IUCN Red List API to first create a subsetted list of species within tropical realms (Afrotropical, Australasian, Indomalayan, Neotropical and Oceanian) that occurred within savannah (habitat type number 2) (IUCN, 2017). Using this list, we then created a frequency table in R (R Core Team, 2020) counting each time a threat was listed for one of these species (Fig. 5).

We found that agricultural activity is by far the biggest threat to tropical savannahs, threatening 3594 species across cropping (1941 species), livestock farming (1322), and wood and pulp plantations (331). Much of the world's recent savannah loss is attributed to a spike in crop cultivation, driven by the demand for sugar cane, maize, wheat and soy, with expansion predicted to increase (Kehoe et al., 2017; Romero-Ruiz et al., 2012; Zabel et al., 2019). Some savannah environments were traditionally unsuitable for farming, but recent advances in technology, and the application of petroleum-derived fertilizers have overcome these limitations (Pacheco, 2012). Tropical savannahs also contain much of Earth's rangelands and livestock (Scholes and Archer, 1997), approximately 20 % in the year 2010 according to our delineation (Gilbert et al., 2018). The extensive, predominantly grassy component of tropical savannahs provides a feed base for grazing livestock and, therefore, facilitates the production of cattle from which meat and dairy products can be derived (Boval and Dixon, 2010). Beyond direct clearing of native vegetation, grazing can affect tropical savannahs through the replacement of native herbivores with domestic animals (Hempson et al., 2017), overgrazing (Dekkers et al., 2016), and in turn land degradation, often leaving soils fragile and infertile with very low levels of organic matter (Eze, 2018; Skarpe, 1991).

Another expanding land-use, afforestation, is often viewed as a win-win for forestry industries and the environment due to the carbon sequestration benefits that come with growing trees within perceived low carbon environments (Castiblanco et al., 2013; Veldman et al., 2015). Afforestation, the process of planting trees at high densities (forests or for commercial purposes including oil palm, eucalyptus, and pine) in areas where they do not occur naturally, is often confused with reforestation, which is the process of re-planting trees where they once naturally occurred. Many natural tropical savannahs have been listed as suitable sites for high density plantings (i.e. afforestation), even though they have never been forested (Bastin et al., 2019; Laestadius et al., 2011; Veldman et al., 2015; World Resources Institute, 2014). With the rise of climate-change mitigation initiatives promoting the process (Bond et al., 2019; Putz and Redford, 2010), scientists are increasingly warning of the potential negative consequences (Parr et al., 2014; Stickler et al., 2009; Veldman et al., 2019). These agricultural industries are likely to increase in extent and intensity if global consumption continues to rise (Díaz et al., 2019; Kearney, 2010). This may have a disproportionate impact in regions that are seeking to improve their economic footing and standard of living through use of their natural resource base for internal consumption or global trade (Davis et al., 2016; Kehoe et al., 2017). With this increasingly globalised trade for a range of commodities, tropical savannahs are being affected by socio-economic factors thousands of kilometres away (Wesz, 2016). This is particularly concerning given agricultural expansion into savannahs is frequently unregulated (Vargas et al., 2015), incentivized (Australian Government, 2015; Departamento Nacional de Planeación, 2019; Morán-Ordóñez et al., 2017), seen as a win-win for development and environmental goals (Castiblanco et al., 2013), and impacts go undocumented (Clements and Fernandes, 2013; Müller et al., 2015) when compared to other ecosystem types.

Globalised trade also affects biological resource use, which is the threat affecting the second largest number of savannah species according to IUCN assessments (1733). Biological resource use includes logging and wood harvesting, gathering terrestrial plants, and hunting. Some species are hunted for their value in the global wildlife trade, such as the illegal ivory trade, which can lead to substantial loss of important functional characteristics such as seed dispersal and regeneration and ecological interactions (Rija et al., 2020). Residential and commercial development is in itself an important driver of species decline (Cabral et al., 2011) - threatening 1268 savannah species through land cover change and increased pressure on biological resources (Butz, 2013; Temudo et al., 2020; van Velden et al., 2018). Across Africa it has been suggested that any relief from pressure on habitats from the current rural–urban migration patterns that are occurring may be overtaken by the increased demand for food and other natural resources from the rapidly growing African cities (Güneralp et al., 2017). The globalised trade in extractive resources, such as oil, iron, uranium, and precious metals are also long-standing drivers of change (threatening 628 species), particularly in Australian (Woinarski et al., 2013), and African tropical savannahs (Weng et al., 2013) which in turn can exacerbate climate change.

Climate change and associated increases in temperature, precipitation, CO_2 concentration, and fire frequency and intensity are all predicted to affect tropical savannahs (Hoffmann et al., 2002; IPCC, 2018; Shanahan et al., 2016). Natural systems modifications (such as fire suppression) threatened 1068 species, which interacts with climate change (already affecting 503 species). Savannah ecosystems are a product of frequent fire, but there is still contention around what constitutes an ideal fire regime. However, much can be learned from Indigenous fire management, which includes small, low intensity burns early in the dry season mixed with unburnt patches, which creates heterogeneity at a landscape scale (Andersen et al., 2012). This pyrodiversity (variation in local fire characteristics) is also important for biodiversity, particularly in wetter regions (Beale et al., 2018), while higher intensity fires are important for preventing and reversing woody encroachment (Smit et al., 2016). Climate change threatens these complex processes as radically altered fire regimes can, in the worst case, result in ecosystem transition to other states (Shanahan et al., 2016). A recent review found that modification of fire regimes is a threat for the highest percentage of threatened species in savannahs, compared to any other habitat type (Kelly et al., 2020).

Climate change can also affect the savannah environment via other biophysical pathways. While the growth of savannah grass types is stimulated by higher temperatures, higher concentrations of CO_2 favour woody vegetation (Bond and Parr, 2010), which may affect vegetation composition. Increasing fire frequencies may favour grasses over wooded vegetation; however, it is still unclear how these forces will change the boundaries of forest and savannah ecosystems across different regions (Staver et al., 2011). Woody encroachment can be exacerbated by domestic grazers, which at high densities can create favourable conditions for woody plants and aid in seed-dispersal if the seed dispersal mechanism is endozoochoric, making livestock management a key mitigation strategy (Sharp and Whittaker, 2003; Tews et al., 2004). Tropical savannahs naturally contain woody vegetation but extensive woody encroachment, the increase in woody biomass, stem densities or woody cover, is a symptom of an alteration in ecosystem processes (Stevens et al., 2017). Already, there is extensive encroachment of woody vegetation occurring across tropical savannahs worldwide attributed to elevated atmospheric CO_2 levels and fire suppression (Stevens et al., 2017), with subsequent impacts on biodiversity (Beale et al., 2013). In addition to the direct impacts of climate change on tropical savannah ecosystems, indirect impacts already occurring include altered patterns of human migration that drive further land conversion (Turner et al., 2010), and afforestation initiatives (Putz and Redford, 2010).

Climate change is expected to grow as a major driver of change across all savannah environments in the near future (IPBES, 2019; IPCC, 2018), and potentially exacerbate and synergistically interact with other threats including degradation, altered fire regimes, and invasive species to further alter the savannah environment (Fig. 4; Andersen et al., 2012). The consequences for people will likely be greatest in Africa, where there are already challenges in meeting domestic demands for food due to degraded natural resources which is further complicated by low levels of political stability, government effectiveness, regulatory quality, rule of law, and control of corruption (Table 1; Hoffman and Vogel, 2008).

Threats to species within the tropical savannah environment (Fig. 5) are different to global analyses which report biological resource use,

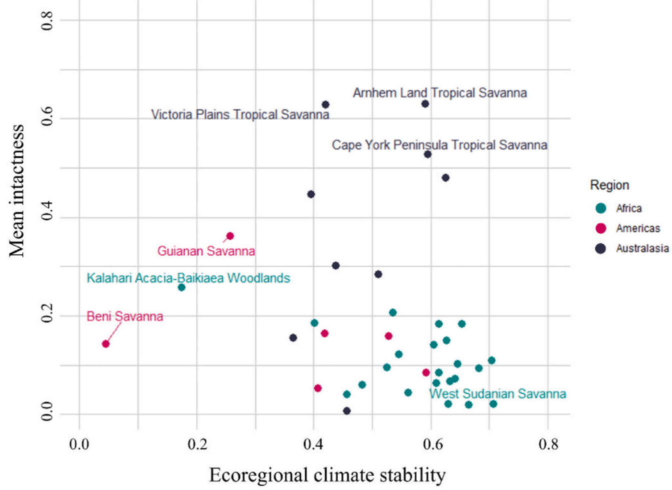

Fig. 4. The spatial relationship between future climate stability, defined as a measure of how similar the future climate of a ecoregion will be to the present climate (on a scale of 0–1 with 1 being highly stable and 0 being unstable, using data from Watson et al., 2013), and the mean intactness (on a scale of 0–1 with 1 being highly intact and 0 being degraded; using data from Beyer et al., 2019) of each tropical savannah ecoregion (as defined by Olson et al., 2001). All tropical savannahs are expected to experience some climate instability, but the ecoregions particularly at risk are those with lower intactness (predominately on African and American continents), as climate change negatively interacts with habitat loss synergistically contributing to the degradation of biological diversity (Watson et al., 2013). For exact values see Table A4.

agricultural activity, urban development, and invasive species and disease to be the biggest threats to species (Maxwell et al., 2016). In tropical savannahs agricultural activity is a relatively bigger threat to species, along with natural system modifications (e.g., fire and fire suppression), when compared to global trends. The implications of this are that conservation strategies in tropical savannahs may need to place greater emphasis on fire management and stemming agricultural impacts when compared to other systems. For examples of where each driver is affecting tropical savannahs across Earth see Table A2.

3. Successful conservation approaches and the need for integrated land management

Despite widespread degradation across many of Earth's tropical savannahs, there are some examples of conservation and land management approaches that are resulting in successful long-term outcomes for savannah biodiversity and people (Fig. 6). In this section we discuss the ways humans benefit tropical savannah ecosystems, the conservation approaches we discuss were selected because they can address the main drivers of change affecting tropical savannahs and can be implemented more broadly through integrated systematic planning (Fig. 6). In many savannah landscapes area-based conservation efforts such as protected areas have historically excluded local people, and benefits have been inequitable or inaccessible to local communities (Anderson and Grove, 1987; Neumann, 1998; Newmark and Hough, 2000). Well-placed and managed protected areas that move beyond this colonial legacy are an important tool for tropical savannah conservation, and can bring socio-economic benefits to a region, such as business opportunities through tourism and the provision of ecosystem services to local people (Gray et al., 2016; Naughton-Treves et al., 2005). Of all tropical savannahs globally 19.7 % are protected, 32.4 % of which are in Australia, Tanzania and Brazil (Fig. 3b). Many protected areas in tropical savannahs are of low intactness (Fig. 3b), and experiencing declines in biodiversity (Pringle, 2017; Western et al., 2009). Therefore, protection alone cannot mitigate all threats and are not enough to safeguard Earth's tropical savannahs.

Maintaining the condition of savannahs beyond the boundaries of designated protected areas is also essential for successful conservation outcomes. Large expanses of savannah allows species to track their preferred climate conditions and facilitate fundamental ecological processes (such as seasonal migrations) to operate unimpeded by boundaries (Kauffman et al., 2021; Ward et al., 2020). For example, in some parts of Kenya wildlife and livestock are still free to move unimpeded by fencing as they have done for thousands of years (Mwangi and Ostrom, 2009; Waithaka and Njoroge, 2018). Here, the co-existence between domestic and wild grazers plays a key role in biodiversity conservation outside of protected areas (Mwangi and Ostrom, 2009; Waithaka and Njoroge, 2018). These areas exemplify how concepts like 'other effective area-based conservation measures' (OECMs), that are being discussed and implemented as part of the Convention on Biological Diversity's global biodiversity framework, could work in the future (Dudley et al., 2018; Waithaka and Njoroge, 2018; Yılmaz et al., 2019). Enhancing

Table 1
Metrics table depicting geo-political differences between countries with the most intact* and most degraded expanses of tropical savannah (Brazil belongs to both categories). Variables where unit is not specified is on a scale of −2.5 (weak) to 2.5 (strong). For information on all savannah-containing countries, more relevant geo-political metrics, and sources, see full table in Table A3.

Country	Percentage of Earth's tropical savannah	Savannah area (km^2) in the highest intactness category (0.8–1)	Savannah area (km^2) in the lowest intactness category (0–0.2)	Total pop. in 2018 (million people)	Annual pop. growth % 2018	GDP 2018 (billion USD)	GDP per capita 2018 (USD)	Political stability and absence of violence/terrorism 2018	Government effectiveness 2018	Regulatory quality 2018	Rule of Law 2018	Control of corruption 2018
Australia*	8.91	314,714	533,914	24.98	1.54	1,434	57,374	0.98	1.60	1.93	1.72	1.81
Mauritania*	2.03	44,784	192,990	4.40	2.78	5	1,189	−0.67	−0.73	−0.81	−0.69	−0.81
Central African Republic*	3	38,315	186,455	4.67	1.52	2	476	−2.28	−1.72	−1.37	−1.69	−1.23
Brazil*	11.7	30,922	1,578,826	209.47	0.78	1869	8,921	−0.36	−0.45	−0.31	−0.28	−0.42
Mali*	3.68	24,649	516,743	19.08	3.01	17	900	−2.05	−1	−0.55	−0.8	−0.7
Democratic Republic of Congo	6.34	2424	1,082,359	84.07	3.23	47	562	−2.12	−1.55	−1.47	−1.78	−1.5
Sudan	8.59	4	1,476,088	41.8	2.39	41	977	−1.84	−1.62	−1.63	−1.12	−1.43
Angola	5.84	0	770,525	30.81	3.28	106	3,432	−0.32	−1.05	−1	−1.05	−1.14
Nigeria	3.98	0	738,527	195.87	2.59	397	2,028	−2.19	−1.02	−0.88	−0.88	−1.04

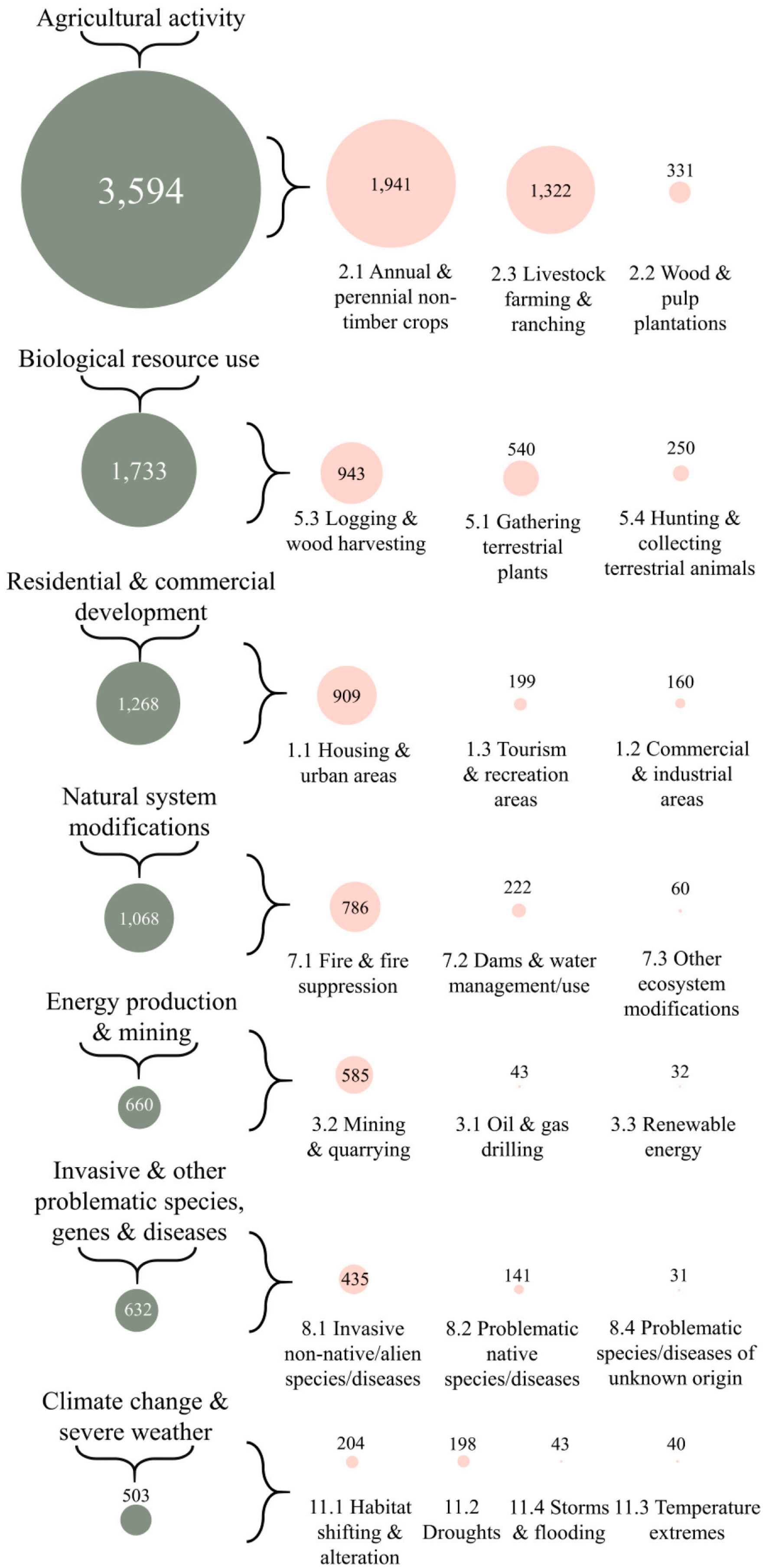

Fig. 5. The threats according to the IUCN (IUCN Red List API) that threaten the most tropical savannah species. Selected species were listed by the IUCN as being associated with a "savannah" habitat, and located within Afrotropical, Australasian, Indomalayan, Neotropical, and Oceanian regions (excluding those which occurred exclusively within Nearctic, Palearctic regions) (IUCN, 2020). Figure shows only those threats with >30 species listed. Other threats not shown in this figure were under the *Invasive & other problematic species, genes & diseases* category, 8.5 Viral/prion-induced diseases (22 species), 8.6 Diseases of unknown cause (2 species), 8.3 Introduced genetic material (1 species), and under *the Climate change & severe weather category*, 11.5 Other impacts (18). Another 994 savannah species were listed as threatened by *Pollution, Transportation & service corridors, Human intrusions & disturbance, Other threats*, and *Geological events*. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

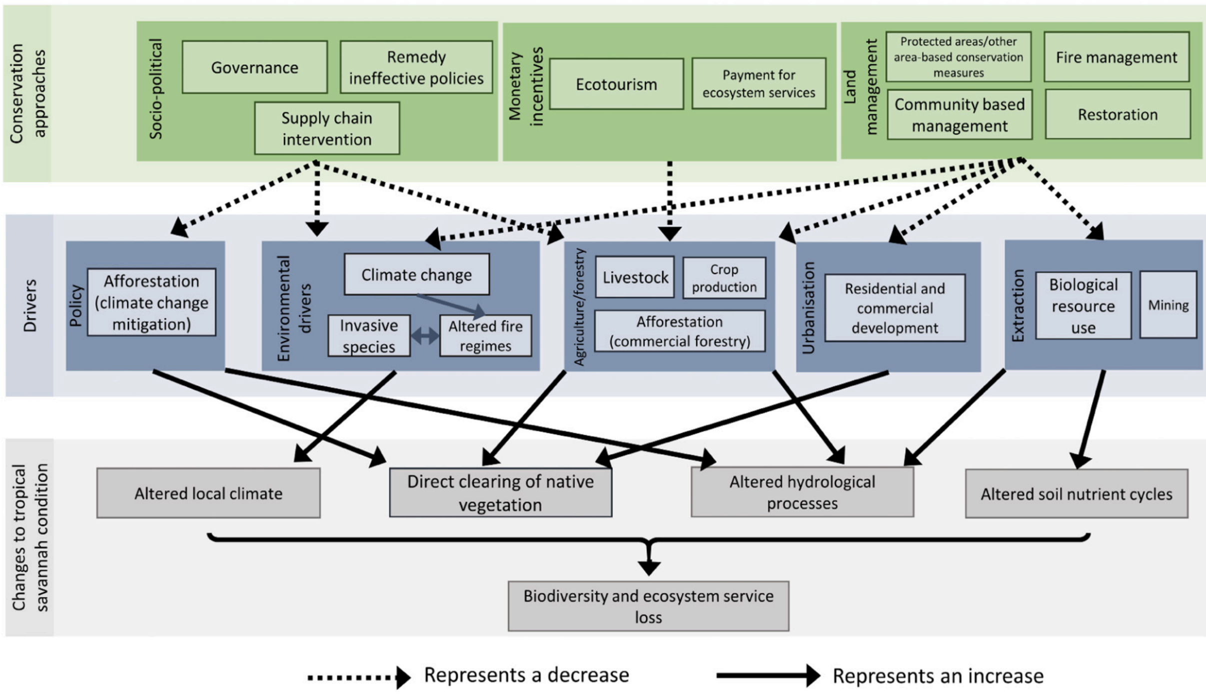

Fig. 6. Key links (as described throughout the paper) between conservation approaches (green), overarching drivers of change (blue), and tropical savannah condition, which can be guided through integrated systematic planning based on summaries presented in this perspective. A dotted line represents a decreased effect (or mitigation), while a solid line represents an increased effect. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

effectiveness and expanding area-based conservation measures (protected areas and OECMs) should remain a key objective in savannah ecosystems all over the world (Pringle, 2017); however, this strategy must be complemented by land-use policies that enable sustainable use of retained natural and semi-natural savannahs (Maron et al., 2018) to ensure future tropical savannah persistence and achieve multiple objectives.

While many savannahs are fire-dependent ecosystems, high-intensity wildfires within tropical savannahs contribute to global carbon emissions (Van Der Werf et al., 2017). There are examples of successful fire initiatives in the savannah environment that achieve positive outcomes for both environmental and human wellbeing objectives. One such initiative is the Western Arnhem Land Fire Abatement project (WALFA), the world's first fully operational savannah burning carbon offset scheme, abating 100,000 t of CO_2 per year over 17 years at $10 AUD per tonne (Lipsett-Moore et al., 2018). By burning controlled fires early in the dry season, fuel loads are reduced, and emissions from uncontrolled wildfire in the latter part of the dry seasons are mitigated. In turn, the money from the carbon credits *supports Indigenous people in returning to, remaining on and managing their land, and the protection of biodiversity*. The authors suggest that globally this is an under-utilized opportunity, identifying 37 other countries as having similar suitable conditions to implement such a program (Lipsett-Moore et al., 2018). However, Laris (2021) recently critically suggested that this example is unique to Australia, and a shift to earlier fires would cause an increase in methane emissions due to increased burning of uncured fuels in Africa. Laris argues that changing fire is not a matter of simply increasing early fire, but a complex management goal requiring strategies to maximize the beneficial aspects of fire while minimizing the potentially harmful impact in accordance with local desires (Laris, 2021).

There are many other successful initiatives within tropical savannahs that sustainably support livelihoods. Examples include incentives for sustainable livestock production practices such as adopting silvopastoral systems as a means of intensification rather than expansion (Nair et al., 2011), compensating farmers for a portion of land not to be grazed (Carriazo et al., 2019), and ecotourism business ventures (Hoogesteijn and Hoogesteijn, 2010). Likewise, incentivizing community-based management can lead to environmental benefits, as well as economic benefits such as employment opportunities (Altman et al., 2018). In northern Australia, these successful community-based approaches, such as the Martuwarra Fitzroy River Council (MFRC) established by traditional owners as a collective governance model to maintain the spiritual, cultural and environmental health of the catchment, have been facilitated by the legislated recognition of the Indigenous estate (Australian Government, 1993; Northern Territory of Australia, 1976), and by increased Indigenous engagement in land-use planning and decision-making (Holmes, 2012; Poelina et al., 2019).

However, there is also a need to remedy existing initiatives that have had perverse implications for tropical savannahs. These include afforestation-based climate mitigation initiatives (Murphy et al., 2016; Parr et al., 2014), and those that incentivise poorly planned and managed intensive agricultural development over natural savannah lands (Clements and Fernandes, 2013; Moreno, 2000). There should also be a focus on restoration and rehabilitation of degraded lands (Buisson et al., 2019), rather than abandonment. Degraded savannah lands should be priorities for natural savannah restoration if there is potential for biodiversity benefits or high natural carbon sequestration (Syktus and McAlpine, 2016) and for land rehabilitation for agricultural production otherwise. These degraded lands present an opportunity for efficient land-use allocation where demands for production can be met, or carbon can be sequestered without further damage to natural lands and biodiversity (Birhane et al., 2017; Brown et al., 2011; Jama et al., 2011; Mekuria and Aynekulu, 2013).

We have highlighted conservation and management approaches which could be implemented more broadly in tropical savannahs; however, the enabling conditions for effective management among savannah-containing countries are disparate. With the exception of Australia, governance metrics across tropical savannah-containing countries are relatively low (Table 1; Table A3), which can hinder the success of conservation approaches (Smith et al., 2003). Low governance capacity also limits the feasibility of successful transboundary conservation (Mason et al., 2020), which is particularly problematic for highly mobile savannah species in sub-Saharan Africa (Plumptre et al., 2007). We found some of the highest levels of savannah degradation (Fig. 3) occurred within the borders of countries with the lowest levels of governance effectiveness, regulatory control, and control of corruption (Table 1; Table A3). As such, conservation-focused policies or strategies must be designed through inter-disciplinary approaches, and may include targeted action towards reducing corruption or enhancing governance (Plumptre et al., 2007; Smith et al., 2003).

Clearly there are trade-offs among competing goals within the tropical savannah environment, hence not all land-use objectives (such as agricultural production, carbon sequestration, and biodiversity conservation) can be maximised simultaneously (Williams et al., 2020a). Finding a reasonable and sustainable balance among objectives requires integrated systematic planning, as any policy or planning that focuses on one objective is unlikely to achieve land management that performs well against others. Many planning frameworks have been developed to help guide land-use allocation decisions, and have been implemented in the savannah environment to provide decision makers with transparent consequences of different scenarios by assessing the trade-offs between objectives in Australia (Adams et al., 2016; Morán-Ordóñez et al., 2017), Africa (Estes et al., 2016), and Latin America (Kennedy et al., 2016; Williams et al., 2020a). These frameworks are crucial for best allocating lands to balance and achieve multiple objectives but must be coupled with innovative initiatives and policy mechanisms that balance environmental goals with other societal objectives such as meeting national and international demands for resources and economic growth, while at the same time aligning with the aspirations and values held by affected local communities and rights holders.

We have identified a range of successful conservation approaches but implementing these at larger scales and across different regions is difficult. While many integrated systematic planning frameworks exist, understanding how to design landscapes that support both production and other ecosystem services in the tropical savannah environment remains a challenge. Development of such landscapes is impeded by a lack of knowledge and available data about the complex relationship between drivers of change and savannah biodiversity. Once analyses that generate effective solutions have been carried out, barriers to their implementation must be overcome. This is further complicated by the disparities in capacities of tropical-savannah containing countries to conserve and manage the environment (see socio-development metrics Table 1; Table A3), as social and economic development imperatives are likely to take precedence (Steinberg, 2005). With economic advancement being a key objective, much of the development that occurs within these countries is driven by external international entities that are interested in the savannah lands of countries where the price of land is lower and there are fewer environmental restrictions. Examples include China, Brazil, and Australia's influence over development projects in sub-Saharan Africa (Clements and Fernandes, 2013; Edwards et al., 2014; Eisner et al., 2016; Laurance et al., 2015; Scoones et al., 2016). Therefore, these countries need to be supported in developing conservation, land management, and development plans that enable economic development but not at the expense of the environment. Integrated systematic planning is one mechanism for facilitating this, as it can be used to represent the interests of stakeholders at multiple scales (Kukkala and Moilanen, 2013).

4. Conclusion

Tropical savannahs are under threat, with <3 % globally remaining highly intact. We find that 19.7 % of tropical savannahs globally are protected but of those areas only 4.1 % falls within the highest intactness category. Without intervention, they risk ongoing erosion of their condition and conversion to more intensive land-uses. We find that the main threats to them are agricultural activity, biological resource use, residential and commercial development, natural system modifications, energy production & mining, invasive and other problematic species, genes, and diseases, and climate change and severe weather. Notably, agricultural activity and natural system modifications (e.g., fire and fire suppression), is a relatively bigger threat to species in tropical savannahs when compared to other major ecosystem types. Conservation approaches that can directly address these threats include fire management, improved governance, protected areas, other area-based conservation measures, payments for ecosystem services, restoration, ecotourism, community-based management, supply chain intervention, and to remedy existing ineffective policies. In tropical savannahs, particular emphasis should be placed on fire management and ensuring other conservation interventions adequately address agricultural activity. We argue (and recommend) that while complex socio-political obstacles need to be navigated, the majority of these conservation approaches can be up-scaled through integrated systematic planning. It is crucial that tropical savannahs are protected, sustainably managed, and restored to ensure they continue to simultaneously deliver production, biodiversity, and ecosystem service goals into the future.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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