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Restoration thinning reduces bush encroachment on freehold farmlands in north-central Namibia

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Bush encroachment affects ~45 million ha of Namibia and, without appropriate restoration measures, it negatively affects rangeland productivity and biodiversity. Thinning is a common method to counteract bush encroachment. The thinning strategy applied in north-central Namibia was assessed to examine how effective it has been in reducing bush encroachment. Trees/shrubs were selectively thinned manually, targeting all height classes, except individuals with stem diameters ≥18 cm. We investigated the effects on the vegetation and soil properties using surveys on three freehold farms (in 2016 and 2017) in bush-encroached and previously thinned habitats. Our results revealed significant differences in the mean total nitrogen (TN) content between the treatments; thinned areas had higher TN content which would be beneficial for fast-growing grasses. In the thinned plots, the occurrence probability of red umbrella thorn (*Vachellia reficiens* Warwa) was significantly reduced, indicating that it was the most harvested species; and umbrella thorn (*Vachellia tortilis* (Burch.) Brenan spp. *heteracantha*) was increased, indicating that it favoured reduced densities of dominant species. Natural regeneration was rapid; the tree/shrub abundance in the 0–1-m height class in the thinned area surpassed those in the non-thinned by 34 per cent, ~7.2 years since thinning. Thinning significantly reduced tree/shrub abundances of the 1–3- and >3-m height classes, which was still evident 7.2 years since thinning. Based upon the generalized linear mixed-effects model, tree/shrub counts between treatments may equalize in ~14 and 15 years for the 1–3- and >3-m height classes, respectively. Thinning was effective in reducing tree/shrub abundances and can be used to restore wildlife habitat on the Namibian farmland: however, post-thinning management is required to maintain an open savannah vegetation structure as the 0–1-m height class cohort will eventually grow into mature trees/shrubs.

Introduction

Globally, vegetation thinning is a common practice that is used to reduce fire risks, bush encroachment and insect infestations and to enhance biodiversity, increase timber growth and yield and for herbaceous production (Ritchie and Skinner, 2014; West, 2014; Smit, 2014a; Brown *et al.*, 2019). In Namibia, thinning is typically applied to counteract bush encroachment – the increase in density and biomass of woody vegetation that affects ~45 million ha of land (SAIEA, 2016). A variety of factors cause bush encroachment, including elevated atmospheric carbon dioxide concentrations, nitrogen pollution, fire suppression, loss of mega herbivores, inherent soil and climatic conditions, overgrazing, injudicious stocking rates and other poor rangeland management practices (de Klerk, 2004; Sankey, 2012). The impacts of bush encroachment include loss of grazing carrying capacity, poor soil water infiltration, loss of suitable habitat, reduced visibility for ecotourists and decreased hunting efficiency for predators, such as cheetah (*Acinonyx jubatus*) (Meik *et al.*, 2002; Muroua *et al.*, 2002; Muntifering *et al.*, 2006; Gray and Bond, 2013; Buyer *et al.*, 2016; Groengroeft *et al.*, 2018). Consequentially, this leads to a reduction in biodiversity as well as declines in farm production and profitability, affecting the livelihoods of the local inhabitants (MAWF, 2012; SAIEA, 2016).

Positive impacts of bush encroachment include the use of excess biomass for the production of charcoal, firewood, construction materials, browse for domestic and wild herbivores, habitat for biodiversity, carbon sequestration, nutrient and water cycling, nitrogen fixing and erosion control (de Klerk, 2004; Ludwig *et al*., 2004; Hagos and Smit, 2005; Wiegand *et al*., 2005; Ridolfi *et al*., 2008; Belay and Kebede, 2010; Dwivedi and Soni, 2011; Buyer *et al*., 2016; SAIEA, 2016). Some technologies proven effective for controlling bush encroachment include manual, semi-mechanized, mechanical, chemical and biological controls (de Klerk, 2004; MAWF, 2012; DAS, 2017). The manual control involves selectively thinning tree/shrubs using handheld tools (e.g. axes) and is locally common in small-scale harvest operations (Trede and Patt, 2015). This method is advantageous because it allows harvesters to control the selection of species and size, minimizes disturbance to soils and the environment and requires low investment costs (Trede and Patt, 2015; Birch *et al*., 2016). Limitations of this method include lower productivity (0.19 ha per man a day (ha/man-day)) and labour intensiveness (Leinonen, 2007). The estimated cost ranges from 1000 to 3000 Namibian dollars (N$) per hectare (DAS, 2017).

Semi-mechanical control involves selectively thinning tree/shrubs using small, powered hand tools (e.g. brush cutters, chainsaws and trolley saws). This method is more cost-effective than manual control because productivity is doubled (0.42 ha/man-day) and unit costs are reduced (Leinonen, 2007; DAS, 2017). Limitations of this method include required training in operating the equipment and exposure of the operator to danger from thorns and pollution (smoke and noise) generated during the process (Leinonen, 2007; DAS, 2017). The estimated cost ranges from 1500 to 2000 N$ per hectare (DAS, 2017).

Mechanical control involves selectively thinning tree/shrubs by means of self-propelled powered machines (e.g. skid steer and tractors) fitted with cutting equipment such as a rotary saw or harvester head. This method is ideal for large-scale operations due to its high productivity (2.48 ha/day) (Leinonen, 2007). Limitations of this method include reduced control over the selection of tree/shrub species and size and increased disturbance to the soils and the environment (Trede and Patt, 2015; Birch *et al*., 2016). The estimated costs range from 750 to 4000 N$ per hectare (DAS, 2017).

Chemical methods involve selectively poisoning tree/shrubs using arboricides (e.g. picloram and glyphosate), that inhibit the photosynthetic ability of tree/shrubs, on freshly cut stems, foliar and soil applications near stems. This method is applied as a primary control, especially in areas where it is impractical to implement other methods due to high tree/shrub densities, or as a follow-up treatment to suppress regrowth following treatment (de Klerk, 2004). Non-targeted application can be potentially dangerous to the ecosystem. Thus, adequate training on chemical use and adherence to precautions should be considered (DAS, 2017). Studies undertaken in Namibia and the north-west and northern Cape provinces of South Africa have shown that if applied selectively, ideal woody vegetation structure and grass layers could be achieved (Harmse *et al*., 2016; SAIEA, 2016). The estimated costs range from 500 to 2600 N$ per hectare and this depends on the tree/shrub densities and type of species present (DAS, 2017).

Browsing pressure is a type of biological bush control. Domestic goats (*Capra aegagrus hircus*) at high stocking rates in Namibia exhibit browsing pressure that causes significant reduction (88.8 per cent) of sickle bush densities (de Klerk, 2004). Limitations of this method include a significant reduction (76.5 per cent) of palatable bushes, no effect on densities of the black-thorn acacia, an increased effort required to manage livestock especially at high stocking rates and inability of goats to control mature dense vegetation or that beyond their browsing heights (>1.5 m). Therefore, it would be more effective if applied as a follow-up strategy to control regrowth or sapling establishment following thinning (de Klerk, 2004).

Fungi is another type of biological control that causes mortality especially among the black-thorn acacia (de Klerk, 2004). Attacks by fungal species (e.g. *Cytospora chrysosperma*, *Phoma cava*, *Phoma eupyrena* and *Phoma glomerata*) cause extensive leaf chlorosis and wood decay, which increases vulnerability to environmental stress conditions such as drought (de Klerk, 2004; DAS, 2017). Although promising, the collection and distribution of fungi spores into the natural environment by human interventions remain underexplored (de Klerk, 2004; SAIEA, 2016).

Harvesting of biomass may alter the population structure or dynamics of the targeted species by influencing the survival, growth and reproduction rates (Ticktin, 2004; Martins and Shackleton, 2017). Also, by thinning the dense target populations, the growth rates of retained trees/shrubs may exceed those of non-harvested areas due to the release from competition pressures (Smit, 2014b; Brown *et al*., 2019). Selective thinning, rather than clear harvesting, is the recommended method to control bush encroachment as it reduces the risk of removing non-target species or harvesting of mature trees (≥18 cm stem diameters) or protected species, minimizes disturbance of sensitive habitats and compaction by heavy machinery or the infliction of other injuries on wildlife (SAIEA, 2016).

In southern Africa, few studies have reported the response of bush-encroaching species to different methods, such as unselective mechanized clearing, selective mechanized thinning, mechanical clearing or selective and non-selective chemical bush control (e.g. Harmse *et al*., 2016; SAIEA, 2016). Also, few studies have focused on the changes in soil properties resulting from manual/non-mechanized bush thinning (e.g. Smit, 2014a; Buyer *et al*., 2016; Zimmermann *et al*., 2017). Our study differs from past research in that we monitored five target encroacher species on treated plots that had different ages since thinning; and trees/shrubs were manually thinned as opposed to mechanically thinned or cleared. Also, restoration was done in a farmland matrix with integrated livestock and wildlife management with the aim of restoring habitat for biodiversity.

Here, we report the results of a vegetation and soils study conducted in bush-encroached and previously thinned areas located on the north-central freehold farmlands of Namibia. We examined the responses of bush-encroaching species to the thinning strategy and the impacts on soil properties. We focused the research on three questions: (1) how do thinned and non-thinned areas differ in the physical and chemical properties of the soil; (2) how effective is thinning at counteracting bush encroachment and (3) and in what ways does thinning affect the natural regeneration patterns of the encroaching species?

Bush thinning has the potential to alter the tree/shrub population structure and soils; consequentially, this may affect the habitats for local biodiversity or affect the success of the restoration. We hypothesized that thinning would not disturb the soil's properties and that thinning would reduce the abundance of the encroaching vegetation. We sought to use our findings to provide stakeholders with credible information detailing how thinning

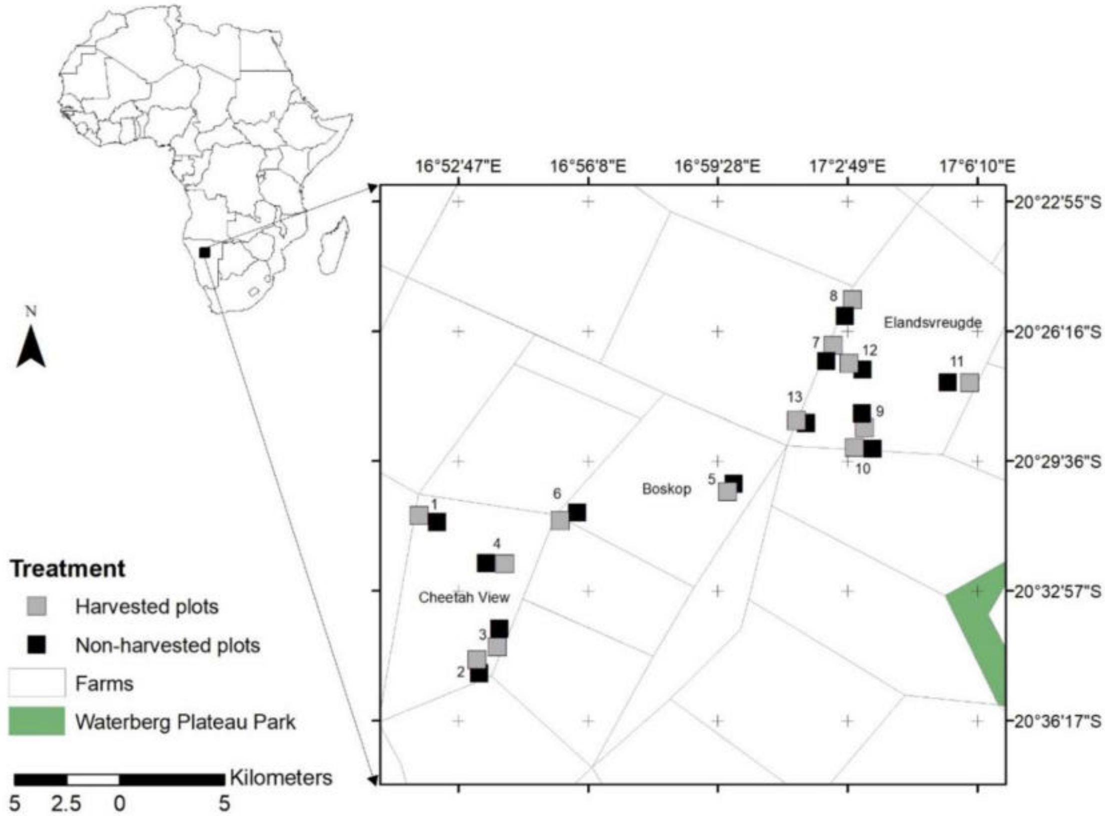

Figure 1 Map of the study area showing the location of blocks (plot pairs) with thinned (grey square) and non-thinned (black square) plots on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367), in north-central Namibia, 2017.

influences the thornbush species and soils. The results could be a reference for future studies and could be included in post-thinning management plans.

Material and methods

Study area

The study was conducted on three farms: Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367), in north-central Namibia (central coordinates: -20.477299° S, 17.024623° E) (Figure 1). Several semi-permanent water sources exist on these farms and provide year-round drinking water for livestock and wildlife. The region is characterized by a semi-arid climate with three main seasons: hot-wet (January–April), cold-dry (May–August) and hot-dry (September–December). The area receives average (± standard deviation (sd.)) annual rainfall of 401 (±257.4) mm, concentrated mainly in the hot-wet season (Figure 2). Mean (±sd.) annual temperature is 19.2°C (±2.4 sd.), and mean (±sd.) daily maxima is 22.7°C (±0.7) in January and 13.4°C (±0.7 sd.) in July (Fick and Hijmans, 2017). Elevation is ~1580 m (±31 sd.) (Fick and Hijmans, 2017). Soils are classified as Eutric Regosols and Chromic Cambisols (Mendelsohn *et al.*, 2003). The vegetation is broadly classified as thornbush shrubland (Mendelsohn *et al.*, 2003). Dominant genera of *Boscia*, *Combretum*, *Dichrostachys*, *Senegalia*, *Terminalia*, *Grewia* and *Vachellia* characterize the woody vegetation structure of the area. The Forest Stewardship Council (FSC™) has certified harvest operations since 2005 (certificate: FSC-C004580).

Thinning of target species

Since 2002, thinning operations were carried out on farms to reduce the densities of five species known to cause bush encroachment: sickle bush (*Dichrostachys cinerea* (L.) Wight & Arn subspecies spp. *africana*), black-thorn acacia (*Senegalia mellifera* (Vahl) Benth. subspp. *detinens*), blade thorn (*Senegalia cinerea* Schinz), red umbrella thorn (*Vachellia reficiens* Warwa) and umbrella thorn (*Vachellia tortilis* (Burch.) Brenan spp. *heteracantha*). Sickle bush is found mainly in the central, northern and north-eastern Namibia and regenerates strongly from root suckers (Mannheimer and Curtis, 2009; Mudzengi *et al.*, 2013). Black-thorn is widespread in Namibia and regenerates both from seeds and vegetatively (Coates, 1993; Mannheimer and Curtis, 2009). Blade-thorn is mainly found in the north-central, northern and north-eastern regions of Namibia and prefers to grow on sandy soils, although its presence on rocky soils has been reported, and it can easily encroach on areas due to overgrazing (likely from seed dispersal) (Coates, 1993;Curtis and Mannheimer, 2005; Mannheimer and Curtis, 2009). Red umbrella thorn is mainly distributed in the north-west, central-west, central highlands of Namibia and the Karstveld, and it

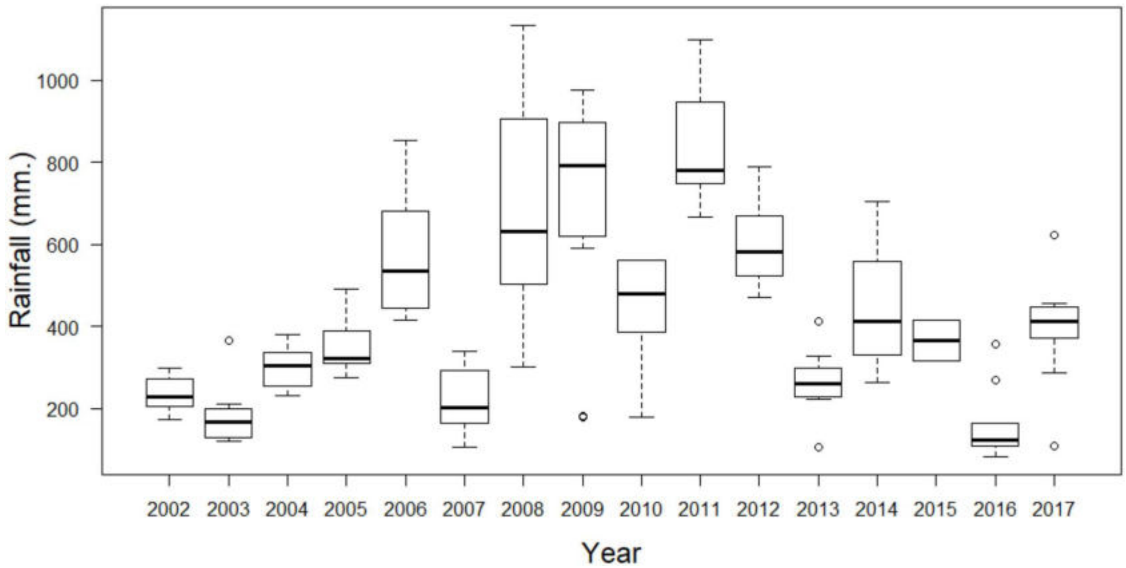

Figure 2 Annual rainfall (mm) received on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia during 2002–2017. The boxplot shows the median rainfall, whiskers are extremes (upper and lower) and open circles are outlier datapoints.

regenerates easily from seed (Curtis and Mannheimer, 2005; Mannheimer and Curtis, 2009). Umbrella thorn is widespread in Namibia, and it occurs on different soil types with a preference for sandy loam soils and regenerates easily from seed (Coates, 1993; Mannheimer and Curtis, 2009).

Modelling the desired open vegetation structure to be achieved by thinning dense bush was based on studies that quantified habitats highly frequented by cheetahs and their prey (e.g. Muntifering *et al.*, 2006; Marker *et al.*, 2008). Such areas typically have low tree/shrub density, higher habitat sighting visibility (≥50 m.) and dense grass cover (>80 per cent) than bush-encroached areas. Also, the government-recommended woody densities per hectare, expressed as tree equivalents (TE), served as a general guideline regarding thinning intensity. TE densities are standardized to account for height differences (1 TE = 1.5 m high tree/shrub) (Smit, 2014a). Densities (TE/ha) equivalent to 1.5 times the long-term annual rainfall of the area is recommended (MAWF, 2017). Our study area falls within the 400–450 mm rainfall region, therefore, as a general guideline, densities of ~600–750 TE/ha were optimum (MAWF, 2017). Prior to thinning, harvester teams of 5–7 members were trained on plant identification of target and protected species, size classes and how to minimize negative impacts, such as pollution or accidental fires in the area.

Tree/shrubs were manually thinned using handheld tools (axes and machetes). Thinning aimed to leave a heterogeneous mix of tree/shrub heights among the encroacher species. Thinning intensity was higher (40–60 per cent) among the small- and medium-sized (≤4 m height) individuals that were the most abundant. Taller trees/shrubs (>4 m) with stem diameters ≤18 cm were thinned minimally (≤10 per cent) unless most individuals present were taller. Mature trees/shrubs (≥18 cm stem diameter) or any individual with confirmed presence of active wildlife breeding sites were avoided. As a general guideline, field harvesters could harvest any tree/shrub of approved species whose stem diameter could fit within the grasp of their hand. Also, some tree/shrub clumps of the encroaching species were left undisturbed to provide browse, shelter or habitat for biodiversity.

Study design and data collection

We used a combination of blocked and split-plot study designs to assess the effect of vegetation thinning. This included 13 blocks (also called plot pairs), each with two plots, one for each treatment (Figure 1, Table 1). For the collection of vegetation data, circular subplots with a 6 m radius (113.1 m^2) were placed within each plot. A total of 295 subplots were surveyed.

The study plots were established on sites that had been manually thinned once from 2002–2013 with time since thinning equal to 3 years or more in 2016 (Table 1) so that multiple seasons followed thinning. Sites targeted for thinning were selected based on the area accessibility and severity of encroachment. We selected the plots opportunistically for several reasons: (1) harvesting was previously conducted for restoring bush-encroached areas and plots were not solely for experimental research; (2) thinning was done on a small scale and over multiple years, which limited the number of available sites; (3) selected sites had adequate area size for multiple subplot replications (plot width and length > 100 m) and (4) sites thinned less than 3 years prior to the survey were avoided. The plots had different ages since thinning (min: 3, max: 14), area (min: 2.4 ha, max: 29.7 ha) and location; these decisions were made prior to the start of our study. Plot sizes varied for several reasons: (1) thinning restoration aimed to create a mix of different sized open patches to increase habitat heterogeneity and (2) sizes of each plot thinned were determined by accessibility as well as quantity of biomass.

A single thinning cycle was conducted, and bush densities were reduced by ~50 per cent. Since 2005, aftercare treatment to prevent regrowth was carried out using picloram and triclopyr

Table 1 List of study plots, dates of thinning treatment, time since thinning, number of replicate plots surveyed and number of plots with aftercare treatment applied on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2017.

Block	Thinned plots						Non-thinned plots			Overall		
	Plot	Thinning	Time since thinning (years)	Subplots		Aftercare treatment	Plot	Subplots		Plot	Subplots	
(Plot pair)	Area (ha)	Completed		Count	Area (ha)		Area (ha)	Count	Area (ha)	Area (ha)	Count	Area (ha)
1	10.5	31 May 2005	12	15	0.17	No	10.5	15	0.17	21	30	0.34
2	5.2	26 September 2012	5	8	0.09	Yes	5.2	8	0.09	10.4	16	0.18
3	10.6	01 June 2013	4	13	0.15	Yes	10.6	15	0.17	21.2	28	0.32
4	4	28 February 2013	4	7	0.08	Yes	4	7	0.08	8	14	0.16
5	2.5	30 September 2010	7	4	0.05	Yes	2.5	5	0.06	5	9	0.1
6	11.2	14 April 2012	5	10	0.11	Yes	11.2	9	0.1	22.4	19	0.21
7	28.6	31 March 2007	10	15	0.17	Yes	28.6	15	0.17	57.2	30	0.34
8	29	31 March 2003	14	20	0.23	Yes	29	15	0.17	58	35	0.4
9	11.2	14 February 2008	9	15	0.17	No	11.2	15	0.17	22.4	30	0.34
10	16.8	23 February 2009	8	14	0.16	Yes	16.8	14	0.16	33.6	28	0.32
11	17.6	16 February 2010	8	14	0.16	Yes	17.6	14	0.16	35.2	28	0.32
12	10.4	15 May 2013	4	8	0.09	Yes	10.4	8	0.09	20.8	16	0.18
13	6.5	14 February 2014	3	6	0.07	Yes	6.5	6	0.07	13	12	0.14
Overall	164.1	–	Mean (±sd.) 7.2 (±3.47)	149	1.69	Yes = 11, no = 2	164.1	146	1.65	–	295	3.34

active arboricide on freshly cut stumps. Eleven previously thinned plots had been treated with arboricide and two plots that had been thinned earlier had not been treated. The control treatment plots were established on sites where thinning had not been conducted. These sites were near the thinning treatment, located at a mean (±sd.) distance of ~0.71 km (±0.17) in the same habitat that was affected by bush encroachment, with similar tree/shrub species and management history (Figure 3).

Boundaries of all plots (blocks) were clearly delineated physically with flagging tape, and their coordinates were marked using a handheld GPS (Garmin Etrex 30, Garmin international, Inc, Olathe, Kansas, US). We used Geographic Information System (GIS, ArcGIS 9.3 ESRI, 2008) software to create a regular grid with multiple 100 m^2 subplots overlaid on each block (plot). All subplots were assigned geographic coordinates (longitude and latitude) and were uploaded onto a GPS unit for location in the field. The centres of subplot were located with a GPS unit and a 6-m radius subplot (113.1 m^2) was demarcated around each centre point and the boundaries were clearly marked with flagging tape. All control and thinned plots were equal in geometry (Table 1). The number of subplots (replicates) within the blocks were approximately equal and the effect of plot pair (block pair), plot and subplot were considered as random factors during the modelling.

We used a systematic random plot sampling method to locate the vegetation subplots, which were 100 m apart within a plot. Vegetation data were collected from April to August 2017. In each subplot, vegetation was quantified as follows: (1) counts of individuals for all target species and (2) tree/shrub heights of all individuals were measured using an extendable marked polyvinyl chloride (PVC) pipe (3 m length), a standard ruler (30 cm) and a measuring tape (30 m). We classified the condition of tree/shrub as alive or dead.

Soil samples were collected from June to July 2016. Samples were collected to a depth of 20 cm, at eight randomized locations per plot. A composite sample was made by mixing all samples from the same plot. This resulted in a total of 26 samples for the entire study area. Samples were collected using a spade and, thereafter, were stored in individual transparent plastic bags and labelled with the subplot number. All samples were sent to the Namibian Ministry of Agriculture, Water and Forestry (MAWF) Soil Laboratory for data extraction using the Agri Laboratory Association of Southern Africa (AgriLASA, 2004) procedures. Soils were analysed for total nitrogen (TN), available phosphorus (P) and potassium (K), organic carbon (OC), sodium (Na), soil pH, available calcium (Ca) and magnesium (Mg), carbonate (CO_3^{-2}), electrical conductivity (EC), per cent organic matter (OM) and identification of the soil texture (sand, silt and clay fractions; sand and silt were determined by the pipette method).

Figure 3 Representative photographs of non-thinned (left) and previously thinned (right) plots showing differences in vegetation structure on farm Elandsvreugde (#367) in north-central Namibia in 2020. Non-thinned plots have higher tree/shrub density, tree/shrub cover, limited habitat visibility and less grass cover than thinned plots.

Soil samples were dried at a temperature not greater than 35°C and were sieved to 2 mm to obtain a fine earth fraction for analysis. Extractable cations (Ca, Mg, K and Na) were obtained by extraction with 1 M ammonium acetate at pH 7 and were measured by atomic absorption spectroscopy. TN was determined by introducing the samples to the furnace containing only pure oxygen, resulting in a rapid and complete combustion (oxidation). Nitrogen present was oxidized to NOx, respectively. The NOx gases were passed through a reduction tube filled with copper to reduce the gases to N and onto a thermal conductivity cell (TC) utilized to detect the N2. Available P was determined using the Olsen method: extraction was done with sodium bicarbonate; phosphate was measured spectrophotometrically using the phosphomolybdate blue method.

Soil texture and particle size analysis (sand, silt and clay) were determined by sieving to retain >53-μ fractions. The United States Department of Agriculture (USDA) classification system was used to derive textural classes. OC was determined using the Walkley–Black method (sulphuric acid–potassium dichromate oxidation). A factor was included in the calculations to consider incomplete oxidation. The OM content was calculated as OC × 1.74. The pH was measured in a 1:2.5 soil-to-water ratio suspension on a mass-to-volume basis. EC (soluble salt content) measurements were carried out in the supernatant of the 1:2.5 soil-to-water suspension prior to the measurement of pH. Elevated results that indicated possible salinity hazard were repeated on the extract of a saturated soil paste. Dry soil was treated with 10 per cent hydrochloric acid and effervescence-estimated carbonate.

We regarded pH, EC, OM, the nutrient contents of TN, P, K, Ca, Mg, CO_3^{-2}, Na and texture (sand silt and clay) as important soil properties. In comparison with the non-thinned areas, the soils of the thinned areas were expected to be equally able to contribute to plant growth. Thus, desired soils would have: (1) a pH range (5.5–7) compatible with optimal nutrient uptake and plant growth (Weil and Brady, 2017); (2) higher, but uncritical EC levels (<0.7 dS^{-1}) for sufficient transfer and retention of cations (Ca, K, Mg and Na) (Hasanuzzaman *et al.*, 2014; Olorunfemi *et al.*, 2016; Moral and Rebollo, 2017); (3) a constant OM content that is related to soil texture (e.g. more sand *vs* less OM) and (4) no significant loss of available soil nutrients (e.g. TN, P, K, Ca and Mg) as a result of thinning.

Data analysis

Mean differences in the soil properties between treatments were investigated using a paired sample *t* test ($P \le 0.05$). We log-transformed the distributions of soil properties (EC, P, K, Ca and Mg) due to their larger standard deviations in comparison with the mean. This was done because the sample size was small, and the *t* test may be affected by the skewness. A binary logistic mixed-effect model via penalized quasi-likelihood (glmmPQL) was used to investigate the effect of treatment type on the occurrence of target species in the study area (package MASS; Venables and Ripley, 2002):

$$y_{kji} \sim Bernoulli\left(p_{kji}\right)$$
$$\ln\left(\frac{P_{kji}}{1-P_{kji}}\right) = \mathbf{x}'_{kji}\beta + b_k + c_{kj} + d_{kji}$$

where y_{kji} is the binary response (presence or absence) of tree/shrub species occurrence, p_{kji} is the probability of tree/shrub species occurrence; $\boldsymbol{x}'_{kji}\boldsymbol{\beta}$ includes the effects of fixed factors species, treatment and the species – treatment interactions; $b_k + c_{kj} + d_{kji}$ includes the nested normally distributed random effects for plot pair (block), plot and subplot, respectively.

Tree/shrub heights were grouped into three distinct classes, namely, 0–1 m, >1–3 m and >3 m, which were analysed separately. We used a generalized linear mixed-effects model (GLMM) with Poisson distribution and log link function (package lme4, Bates *et al.*, 2015) to estimate how the vegetation structure was influenced by treatment and to also estimate the regeneration of thinned treatment:

$$E\left(y_{ki}\right) = \exp\left(\mathbf{x}_{ki}\boldsymbol{\beta} + b_k + e_{ki}\right),$$

where $E\ (y_k)$ = mean of the predicted distribution of number of tree/shrubs; $\boldsymbol{x}'_{ki}\boldsymbol{\beta}$ includes the effects of fixed factors treatment, arboricide application and centred time since thinning, (expressed as time since thinning – 7.2, where 7.2 is the mean time since thinning over all treated plots) and $b_k + e_{ki}$ includes the nested random effects for block and observation, respectively. For the non-thinned treatment, the centred time since thinning was set to zero (0). Plot and subplot were excluded in the final

Table 2 Mean differences in soil composition in the thinned and non-thinned treatment sites on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2016.

Description	Unit	Thinned		Non-thinned		Mean difference	t	df	P
		Mean (±sd.)		Mean (±sd.)					
pH (water)	–	6.32	±0.2	6.27	±0.11	0.05	0.97	12	0.35
EC	μS/cm	186.15	±73.42	151.1	±16.97	35.08	1.76	12	0.104
OM	%	0.842	±0.144	0.836	±0.155	0.01	0.18	12	0.863
Phosphorus (P)	mg/kg	1.12	±1.49	1.26	±2.09	−0.14	−0.3	12	0.769
Potassium (K)	mg/kg	181.92	±68.85	142.2	±43.45	39.69	1.7	12	0.114
Calcium (Ca)	mg/kg	448.92	±287.07	443	±179.07	5.92	0.08	12	0.936
Magnesium (Mg)	mg/kg	108.46	±53.36	108.2	±39.16	0.23	0.02	12	0.983
Sodium (Na)	mg/kg	0.46	–	0	–	0.46	1	12	0.337
TN	%	0.023	±0.013	0.019	±0.013	0.003	2.31	12	0.04
Sand	%	77.51	±4.63	77.41	±3.99	0.10	0.12	12	0.907
Silt	%	10.99	±3.44	10.82	±2.73	0.18	0.21	12	0.834
Clay	%	11.53	±2.78	11.77	±2.41	−0.24	−0.36	12	0.724
Bases	$cmol_{(+)}$/kg	36.15	18.40	34.82	11.86	1.33	0.30	12	0.769

models due to close-to-zero variances in the random factor levels. Aftercare application with arboricide was included in the estimation because 11 previously thinned plots were treated and 2 had not been treated. We included observation-level random effects as a strategy to overcome the overdispersion observed during the initial fitting (Mehtatalo and Lappi, 2020). We used a Wald $\chi 2$ test to test the significance of the model coefficients from the final model (Anova and lht of package car; Fox and Weisberg, 2011). All analyses were carried out with R version 3.5.2 (R Core Team, 2017) at the 0.05 per cent significance level. Summary statistics for height classes (mean ± standard deviation) were calculated per treatment, species and height classes.

Results

Soil properties

Mean differences in soil composition are presented in Table 2 (see Appendix A for overall correlations). For both treatments, the soil pH was slightly acidic, Ca exhibited the greatest concentration and OM content was generally low. Comparisons between treatments revealed statistically significant differences for TN content (t = 2.31, df = 12, P-value = 0.04). The TN measured in the thinned areas was on average 21.8 (±38.1 sd.) per cent higher than the non-thinned. Differences between treatments for other soil properties were not statistically significant.

Observed tree/shrub individuals

Overall, the total number of trees/shrubs recorded was 4223, from five encroacher species: sickle bush (45.5 per cent), black-thorn acacia (22 per cent), blade-thorn (15.6 per cent), red umbrella thorn (11.9 per cent) and umbrella thorn (5 per cent) (Table 3). In total, 92.5 per cent trees/shrub were alive and 7.5 per cent were dead. Of the dead, 76.9 per cent (n = 233) were in the non-thinned study plots and 23.1 per cent (n = 70) were in the thinned study plots (Table 3).

The baseline category (β_0) for the glmmPQL model shows that the occurrence probability of sickle bush was 0.93 in the non-thinned area (Table 4). The negative values for the effects of other tree/shrub species show that they were less abundant than the baseline category in the same treatment. The occurrence probabilities for the other species were 0.64 for black-thorn acacia, 0.64 for blade-thorn, 0.63 for red umbrella thorn and 0.10 for umbrella thorn. The occurrence probability of sickle bush in the thinned plots ($\beta 5$) was estimated to equal 0.93. Occurrence probabilities were reduced in the thinned areas; 0.55 for black-thorn acacia, 0.60 for blade-thorn and 0.39 for red umbrella thorn. Thinned areas had a greater occurrence probability (0.29) than non-thinned areas for umbrella thorn.

The *post hoc* tests of treatments by tree/shrub species showed significant differences in the occurrence probabilities for red umbrella thorn and umbrella thorn (Table 5). The probability of red umbrella thorn occurrence was significantly less in the thinned area but, for umbrella thorn, was higher than in the non-thinned areas (Tables 4 and 5).

Effect of treatment on vegetation structure and regeneration

The baseline category (β_0) of the GLMM model for height class 0–1 m (Tables 6 and 7) showed that the expected number of trees/shrubs in the non-thinned area was, on average, 2.36 individuals, when the centred time since thinning was set at zero (0) and when arboricide aftercare was not applied. For the same height class and time since thinning in the thinned plots at 7.2 years, the expected number of trees/shrubs was 1.34 times more abundant than in the non-thinned area. This led to a mean count of 3.17 tree/shrubs per subplot, which was 34 per cent greater than the amounts in the non-thinned area. The 0–1 m cohort was dominated by sickle bush and black-thorn acacia (Table 3).

Table 3 Observations of tree/shrub species showing the number of individuals counted (alive, dead) in the thinned and non-thinned plots per three height classes, 0–1, 1–3 and >3 m, on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2017.

Height class/species	Thinned				Non-thinned			
	Alive	%	Dead	%	Alive	%	Dead	%
0–1 m	1412	67.9	29	40.3	469	25.6	31	12.8
Sickle bush	547	26.3	6	8.3	264	14.4	23	9.5
Black-thorn acacia	574	27.6	20	27.8	81	4.4	3	1.2
Blade thorn	138	6.6	3	4.2	73	4.0	4	1.6
Red umbrella thorn	60	2.9	–	0.0	42	2.3	1	0.4
Umbrella thorn	93	4.5	–	0.0	9	0.5	–	0.0
1–3 m	602	29.0	39	54.2	1005	54.9	193	79.4
Sickle bush	294	14.1	31	43.1	520	28.4	158	65.0
Black-thorn acacia	28	1.3	–	0.0	82	4.5	13	5.3
Blade thorn	155	7.5	5	6.9	207	11.3	7	2.9
Red umbrella thorn	36	1.7	3	4.2	191	10.4	15	6.2
Umbrella thorn	89	4.3	–	0.0	5	0.3	–	0.0
>3 m	64	3.1	4	5.6	356	19.5	19	7.8
Sickle bush	7	0.3	2	2.8	57	3.1	12	4.9
Black-thorn acacia	17	0.8	1	1.4	106	5.8	5	2.1
Blade thorn	17	0.8	–	0.0	49	2.7	–	0.0
Red umbrella thorn	15	0.7	1	1.4	137	7.5	2	0.8
Umbrella thorn	8	0.4	–	0.0	7	0.4	–	0.0
Total	2078		72		1830		243	

Table 4 Logistic regression analysis showing the effects of treatment on the occurrence of encroacher species on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2017.

Variable	Parameter	Value	Std. error	Occurrence probability
Fixed effects				
(Intercept)	β_0	2.558	0.342	0.928 $(1/(1+\mathrm{EXP}\ (-(\beta_0))))$
Black-thorn acacia	β_1	-1.967	0.352	0.644 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_1))))$
Blade thorn	β_2	-1.967	0.352	0.644 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_2))))$
Red umbrella thorn	β_3	-2.029	0.351	0.629 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_3))))$
Umbrella thorn	β_4	-4.726	0.409	0.103 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_4))))$
Treatment-thinned	β_5	-0.004	0.489	0.928 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_5))))$
Black-thorn acacia	β_6	-0.402	0.500	0.546 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_1+\beta_5+\beta_6))))$
Blade thorn	β_7	-0.199	0.501	0.596 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_2+\beta_5+\beta_7))))$
Red umbrella thorn	β_8	-0.965	0.501	0.392 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_3+\beta_5+\beta_8))))$
Umbrella thorn	β_9	1.271	0.547	0.289 $(1/(1+\mathrm{EXP}\ (-(\beta_0+\beta_4+\beta_5+\beta_9))))$
Random effects				
Block (plot pair)	0.0003			
Plot	0.542			
Subplot	<0.001			

The baseline category (β_0) of the GLMM model for height class 1–3 m (Tables 6 and 7) showed that the expected number of trees/shrubs in the non-thinned area was, on average, 5.68 individuals, when the centred time since thinning was set at zero (0) and when arboricide aftercare was not applied. For the same height class and time since thinning in the thinned plots at 7.2 years, the number of trees/shrubs was 0.576 times the abundance recorded for the non-thinned area. This led to a mean count of 3.27 individuals per subplot, equivalent to 57.6 per cent of the amounts in the non-thinned area. Overall, the 1–3 m height class constituted most of the measured trees/shrubs. In addition, a greater number of trees/shrubs were present in the 1–3 m class in the non-thinned areas, resulting in a dominant height structure composed of medium-sized sparse vegetation.

The baseline category (β_0) of the GLMM model for height class >3 m (Tables 6 and 7) showed that the expected number of trees/shrubs in the non-thinned area was, on average, 1.66 individuals, when the centred time since thinning was set at zero

Table 5 Hypothesis tests of the estimated logistic regression model coefficients showing overall treatment effects and differences in the occurrence of encroacher species in thinned and non-thinned treatment on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2017.

	Chisq	df	*P*-value
(Intercept)	56.160	1	<0.001
Species	143.242	4	<0.001
Effect of treatment by tree/shrub species			
Overall	32.34	1	<0.001
Sickle bush	0.0001	1	1
Black-thorn acacia	1.554	1	1
Blade thorn	0.387	1	1
Red umbrella thorn	8.846	1	0.015
Umbrella thorn	10.642	1	0.006

Note: The *P*-values for the effect of treatment by tree/shrub counts were adjusted for multiple testing using the Bonferroni method.

Table 6 GLMM showing effects of fixed factors (treatment, centred time since thinning and arboricide aftercare) on the tree/shrub counts in height classes, 0–1, 1–3 and >3 m, on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2017.

Variable		0–1-m Height class			1–3-m Height class			>3-m Height class		
	Parameter	Estimate	Std. error	*P*-value	Estimate	Std. error	*P*-value	Estimate	Std. error	*P*-value
Fixed part										
Intercept	β_0	0.86	0.145	<0.001	1.737	0.001	<0.001	0.509	0.168	0.002
Treatment-thinned	β_1	0.293	0.353	0.406	−0.551	0.001	<0.001	−2.342	0.619	<0.001
Centred time since thinning	β_2	0.119	0.046	0.009	0.083	0.001	<0.001	0.29	0.087	0.001
Arboricide aftercare applied	β_3	0.913	0.387	0.018	−0.327	0.001	<0.001	0.393	0.64	0.54
Random part										
	Name	Variance	SD	–	Name	Variance	SD	Name	Variance	SD
Observation	Intercept	0.558	0.747	–	Intercept	0.513	0.716	Intercept	0.644	0.802
Plot pair	Intercept	0.174	0.417	–	Intercept	0.108	0.329	Intercept	0.218	0.467

Table 7 Expected number of tree/shrubs in the non-thinned and thinned treatment estimated from the GLMM model coefficients on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2017.

Treatment/height class	Model coefficients (β)	Sum (β)	Expected tree/shrub count (exp(sum(β)))	±95% Confidence level	Density, ha^{-1}	±95% Confidence level
Non-thinned						
0–1 m	$\beta0+\beta1(0)$	0.86	2.364	1.328	208.980	117.427
1–3 m	$\beta0+\beta1(0)$	1.737	5.682	1.002	502.375	88.602
>3 m	$\beta0+\beta1(0)$	0.509	1.664	1.391	147.146	122.981
Thinned						
0–1 m	$\beta0+\beta1(1)$	1.153	3.167	2.651	280.058	234.367
1–3 m	$\beta0+\beta1(1)$	1.186	3.274	1.000	289.511	88.411
>3 m	$\beta0+\beta1(1)$	-1.833	0.160	0.413	14.143	36.544

(0) and when arboricide aftercare was not applied. For the same height class and times since thinning at 7.2 years in the thinned plots, the expected number of trees/shrubs was 0.096 times the abundance recorded for the non-thinned area. This led to a mean count of 0.16 individuals per subplot, which was 90.4 per cent less than the amounts in the non-thinned area. Overall, fewer trees/shrubs were encountered in this height class.

The effect of time since thinning (β_2) on the number of trees/shrubs per plot across all height classes shows that increasing this variable by a unit (1 year) caused a 11.9, 8.3 and 29 per cent increase in the tree/shrub numbers among the 0–1-, 1–3- and >3-m height classes, respectively. Centred time since thinning showed significant positive associations with tree/shrub numbers across all three height class models

(Table 6). The effect of aftercare treatment (β_3) was significantly associated with a 91.3 per cent increase and 32.7 per cent decline of tree/shrub counts in the 0–1- and 1–3-m height classes, respectively (Table 6).

The variability in the mean tree/shrub numbers between the paired plots was greater in the 0–1- and >3-m height classes than in the 1–3-m height class (Table 6). There was also wide variability in the observation-level random effects, especially in the 0–1- and >3-m height class models, which were used to model overdispersion in the data (Table 6). The differences in the average number of trees/shrubs counted between the treatments were statistically significant in the height classes: 1–3 ($P = <0.001$) and >3 m ($P = <0.001$). This showed that there were less tree/shrub counts in the 1–3- and >3-m classes in the thinned compared with the non-thinned treatment.

The period in which the 0–1-m height class would be equal in both treatments was predicted to be ~5 years. For the other height classes, equal counts between treatments would take ~14 and 15 years after thinning treatment for the 1–3- and >3-m height classes, respectively.

Discussion

Soil properties

A key finding of our study showed that the effect of treatment (thinned *vs* non-thinned) was not statistically significant among most soil properties. We observed significant differences in the mean TN content between the treatments; thinned areas had a higher TN content than the non-thinned. Moreover, thinned areas had slightly higher EC and K concentrations among all other soil properties, however, differences between treatments were not statistically significant.

The reduction of tree/shrub densities by thinning increases the surface temperatures and moisture retention of the soils; consequently, soil OM decomposition is enhanced and would increase the release rate of soil nutrients (Ludwig *et al*., 2004; Palviainen *et al*., 2004; Belay and Kebede, 2010; Groengroeft *et al*., 2018; Zhang *et al*., 2018). This was one possible explanation because the harvested tree/shrub stumps and root left intact in the soil would be higher in the thinned plots. In a separate study, the same thinned areas had more wildlife activity than the non-thinned (Nghikembua *et al*., 2020). Thus, the import of nutrients through dung and urine depositions would result in soil fertilization (van der Waal *et al*., 2011; Zimmermann *et al*., 2017). The high TN content in thinned plots is beneficial since it would give fast-growing grasses a competitive advantage over woody seedlings (Kraaij and Ward, 2006). Also, the slightly elevated EC values may be due to soluble salts accumulating on the soil surface, especially because the reduced canopy cover increases surface temperatures and higher evaporation which can mobilize soluble salts to the upper soil surface (Hatton *et al*., 2003; Hasanuzzaman *et al*., 2014).

Observed tree/shrub individuals

The study area supports a greater abundance of sickle bush than any other encroaching species. A greater abundance of black-thorn acacia was anticipated since it occurs at greater densities regionally (Smit *et al*., 2015; Birch and Middleton, 2017). An elevated abundance of sickle bush may be explained by its preference for deep sandy loamy soils (Orwa *et al*., 2009), which comprise the majority (65.3 per cent) of soils in the study area (Table 2). Black-thorn acacia is known to prefer hard-surfaced, sandy, rocky substrates or loamy soils (Orwa *et al*., 2009), which are not widespread in the study area.

There were significant differences in the occurrence probabilities of red umbrella thorn and umbrella thorn between the treatments (Tables 5 and 6). A possible explanation for this occurrence of red umbrella thorn is the harvester preferences for larger stem sizes, as observed in other studies (e.g. Neke *et al*., 2006), because of its growth form. In the case of umbrella thorn, the increased occurrence probability in the thinned areas is interesting. Other studies have reported successful recruitment of species that were less dominant following the reduction of tree/shrub densities (e.g. Smit, 2004; Haussmann *et al*., 2016). Therefore, thinning possibly favoured the recruitment of umbrella thorn due to release from competition for resources (e.g. water and nutrients) with the other dominant species (Kambatuku *et al*., 2011). Also, umbrella thorn is easily propagated from seeds, and the pods are consumed by livestock and game that use the thinned plots (Coates, 1993; Mannheimer and Curtis, 2009). Thus, with the reduction of tree/shrub densities in the thinned areas, recruitment would be high.

Effect of treatment on the vegetation structure and regeneration

Counts per plot in the 0–1-m height class were not significantly different between the treatments. Regeneration consisted mainly of sickle bush and black-thorn acacia. This pattern is similar to other studies (e.g. O'Connor, 2017) where regeneration was related to species abundance. Interestingly, at 7.2 years and with aftercare applied, tree/shrub counts in the thinned areas surpassed the non-thinned areas by 34 per cent. Also, our predictions show that it took ~5 years for counts in the 0–1-m height class to equalize between treatments. One possible explanation for this occurrence is the amount of rainfall received.

We observed above average (>444 mm) rainfall in the study area between 2008 and 2012 (Figure 2), and the end of that period was separated from our field observations by ~5 years, which corresponds closely to our prediction. Therefore, it is possible that mass recruitment took place during this rainfall period (2008–2012). Another possible explanation is competition release due to the reduced tree/shrub densities, allowing seedlings to establish in areas where mature tree/shrubs were present. Other studies in different regions have also reported increased growth and recruitment following thinning (e.g. Rautiainen and Suoheimo, 1997; Dwyer *et al*., 2010; Smit, 2014a; Dwyer and Mason, 2018). Factors such as increased light intensity, release from competition, increased structural growth and increased reproductive output in the retained trees/shrubs have been reported by these studies due to thinning. Reseeding, especially from ungulates moving through thinned plots, might also have contributed to recruitment because seedpods from bush-encroaching species are consumed by livestock and ungulates (Mannheimer and Curtis, 2009) and may be distributed via dung deposition.

Thinning was effective in reducing the abundance of trees/shrubs as shown by the significant differences in the mean counts among the 1–3- and >3-m height classes. It was evident that in the 7.2 years following thinning, and with no follow-up aftercare applied, counts per subplot in the 1–3- and >3-m height classes were significantly less than the number of trees/shrubs recorded in the non-thinned areas. Our predictions are that counts between treatments may equalize in ~14 and 15 years for the 1–3- and >3-m height classes, respectively. In general, encroacher species experience slow growth rates, elevated mortality rates (e.g. from drought) and limitations in recruitment events (de Klerk, 2004; Joubert et al., 2013, 2017; Cunningham and Detering, 2017). Therefore, a longer growth period would be expected, and the transition to the upper height classes may be delayed. Herbivory is another factor that could delay this transition. Other studies have reported that megaherbivores, such as elephants (*Loxodonta africana*), could cause tree mortality (e.g. O'Connor, 2017), although elephants were absent in our study area. The common browser species present (e.g. kudu *Tragelaphus strepsiceros*, eland *Tragelaphus oryx*) would not exert a significant effect on bush encroachment because of low population densities and foraging habits that lead to the selection for other plant species (de Klerk, 2004).

Fire could also reduce recruitment by killing seedlings and sapling populations, whereas a crown fire could potentially cause mortality or reduce the canopy of tree/shrubs (Joubert *et al.*, 2008; Lohmann *et al.*, 2014). No veld fire was recorded in the study area following the start of thinning operations. Also, veld fire occurrences in the surrounding farmland were low due to the ongoing fire suppression.

Past research suggests that it may take ~10–15 years for regrowth to produce biomass equivalent to the pre-thinning levels which was close to our estimates (e.g. Smit *et al.*, 2015; SAIEA, 2016). Mean height growth rates for black-thorn acacia in our study area were expected to be greater (7.09 cm/annum) than those in the Highland savannah of Namibia (3.19 cm/year) due to greater rainfall and deeper soils (Joubert *et al.*, 2017). Also, sickle bush can potentially exhibit annual height increments of 6–8 cm/year (Orwa *et al.*, 2009). The objective to reduce bush encroachment was achieved, although, this may be short-lived in the absence of a post-thinning management plan.

The thinning strategy we applied has the potential to restore herbaceous production, open savannah habitat suitable for grazers and hunting efficiency for predators (e.g. cheetah) that rely on long sight lines when hunting (Muntifering *et al.*, 2006; Marker *et al.*, 2008; Stafford *et al.*, 2017). Consequentially, this would increase rangeland productivity, farm production, profitability and improve livelihoods of the local inhabitants (MAWF, 2012; SAIEA, 2016). Past research has shown that large mammal predators in the same study area responded positively and significantly to the thinning treatment by increasing their activity due to high ungulate abundances, high grass cover for concealment while hunting and improved hunting sight lines (Muroua *et al.*, 2002; Muntifering *et al.*, 2006; Nghikembua et al., 2016; Nghikembua *et al.*, 2020). Therefore, without appropriate mitigation measures, thinned areas could potentially become hotspots for human wildlife conflict if the livestock present in these areas are not adequately protected. Also, high ungulate activity in thinned areas may pose a risk of overgrazing due to continuous utilization, specifically, if thinning is done on a smaller scale. These potential threats could be addressed by restoring larger areas of the bush-encroached farmland matrix. This approach could reduce the predatory risk by spreading it over larger areas, and in this manner, there would be more options for animals to escape or avoid habitat edges that are highly frequented by predators and to utilize increased grazing capacity (Smit *et al.*, 2015; Stafford *et al.*, 2017; Nghikembua *et al.*, 2020).

Conclusions and management recommendations

Our results indicate that the effects of treatment (thinned *vs* non-thinned) were minimal on the soil properties within the study area. We only found significant differences in TN content, which was greater in the thinned area. Other soil properties were also slightly higher in thinned areas (except P), but differences between the treatments were insignificant. These conditions could potentially promote herbaceous production for grazing animals. The missing or lower effect for other soil properties may result from a thinning intensity that did not cause significant ecosystem disruptions, and perturbations may have declined with time to the 7.2 years' time since thinning mark.

We found significant differences in the occurrence probabilities for red umbrella thorn and umbrella thorn. This suggests that local distributions of target species could be substantially altered by thinning operations. Consequently, some preferred species (e.g. red umbrella thorn) may be reduced, whereas low-density species (e.g. umbrella thorn) may be increased in thinned areas. The higher frequencies of young cohorts in thinned areas are a potential risk for re-encroachment if no post-thinning management is applied. Overall, thinning was effective in reducing bush encroachment as shown by the significant low counts of mature trees/shrubs in the 1–3- and >3-m height classes at the 7.2 years since thinning mark.

The findings from this study will provide a baseline for future research related to thinning effects on soils, vegetation structure and the rates of natural regeneration. Also, harvest protocols and post-thinning management plans could incorporate these findings by taking into consideration the importance of thinning intensity, monitoring of soils and aftercare. The results were limited to the study period; therefore, longer-term data are required to fully evaluate the recovery of the soils and vegetation. For future research and assessment, we recommend: (1) control plots for all thinning activities; (2) ongoing monitoring of the condition and recovery of soils, temperature, rainfall and browsing/-grazing pressure; (3) monitoring of the growth rates of different species to predict future biomass yields and optimal periods to harvest; (4) harvester training on selection of tree/shrub sizes, species and minimum densities required for retention and (5) thinning of the established 0–1 m recruits to maintain open habitats.

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Conflict of interest statement

None declared.

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Data availability statement

Data used in deriving the results of this study are available from the first author, upon reasonable request.

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Appendix A Spearman correlation coefficients (*r*) matrix showing relationships between soil properties. Significance: 0.05–0.01∗; 0.01–0.001∗∗; <0.001∗∗∗ on farms Cheetah View (#317), Boskop (#324) and Elandsvreugde (#367) in north-central Namibia, 2016.

	pH(W)	EC	OM	P	K	Ca	Mg	Na	TN	Sand	Silt	Clay
pH (water)												
EC	0.20											
OM	-0.02	0.64∗∗										
P	-0.12	-0.15	−0.13									
K	0.15	0.66∗∗	0.50∗	0.05								
Ca	0.562∗∗∗	0.54∗	0.51∗	−0.21	0.47							
Mg	0.408∗∗	0.47	0.35	0.09∗	0.48∗	0.81∗∗∗						
Na	0.28	0.15	0.04	−0.19	0.25	0.23	0.31					
TN	0.14	0.45	0.59∗	−0.14	0.19	0.16	0.15	0.296				
Sand	-0.47∗	-0.51	−0.59∗∗∗	0.29	−0.39	-0.78∗∗∗	−0.67∗∗∗	-0.333∗	−0.54∗			
Silt	0.49∗	0.35	0.47	−0.40∗	0.15	0.64∗∗	0.37	0.093	0.57	-0.83∗∗∗		
Clay	0.15	0.58	0.63∗∗	−0.04	0.49∗	0.56∗	0.65∗∗∗	0.334∗∗	0.43∗	-0.67∗∗∗	0.28	
Bases	0.50∗∗∗	0.59∗	0.54∗	−0.10	0.58∗	0.97∗∗∗	0.89∗∗∗	0.253	0.16	-0.75∗∗∗	0.52∗∗	0.64∗∗