Kraals or bomas increase soil carbon and fertility across several biomes

Mia Momberg[1] , Anna Jean Haw[2] , Perushan Rajah[3], Jacques van Rooyen[2,4] and Heidi-Jayne Hawkins[3,5]*

[1] *Department of Zoology and Entomology, University of Pretoria, Pretoria, South Africa*
[2] *Herding 4 Health Programme, Africa Field Division, Conservation International, Nairobi, Kenya*
[3] *Conservation South Africa, Cape Town, South Africa*
[4] *Department of Veterinary Tropical Diseases, University of Pretoria, Pretoria, South Africa*
[5] *Department of Biological Sciences, University of Cape Town, Cape Town, South Africa*
** Correspondence: hhawkins@conservation.org; heidi.hawkins@uct.ac.za*

Knowledge about how pastoralism and kraaling may contribute to desired global objectives, such as soil fertility, is in danger of being lost. We tested whether short duration kraaling increases soil fertility across various biomes and countries via a meta-analysis (random effects model, *n* = 12 studies). Kraaling approximately doubled soil concentrations of carbon (C), nitrogen (N), phosphorus (P), potassium (K), and slightly increased pH compared to non-kraaled areas ($p \leq 0.0158$, all meta-analyses). Results support the idea of persistent nutrient hotspots post kraal abandonment as a generalizable phenomenon. Anecdotes from a case study, the Herding 4 Health Model, supported findings. However, inconsistency scores ($I^2 \geq 90\%$) indicated that while the average effect size was positive, in some cases the true outcome may in fact be negative. Kraal age did not predict soil fertility in our analysis, possibly due to coarse time intervals. Some studies nevertheless found kraal age to be important, with relatively immobile elements such as P persisting over time while N and K decreased. Using kraals to achieve 'desirable states' such as wildlife-livestock coexistence, land restoration, and crop fertilisation will require monitoring, and maintenance of fertility within ecological bounds, ideally with inputs from scientists and pastoralists alike as part of global partnerships.

Keywords: herding, meta-analysis, nutrient hotspots, pastoralism, rangelands

Introduction

Open ecosystems including rangelands cover 45% of the global terrestrial surface, providing food and livelihoods for millions of people while also making available critical habitat for wildlife species (Herrero et al. 2013). The importance of rangelands was recently emphasised by the United Nations (UN) when they declared this the decade (2021–2030) of ecosystem restoration, and placed particular importance on restoration and recognition of African rangelands as important open ecosystems (UN 2019). Here restoration is used in the sense of rehabilitation to a desired state, or where known, a native historical state. The consequences of rangeland mismanagement, often involving livestock, can be seen in rangeland resource degradation (e.g. soil erosion, loss of plant productivity), land use conflicts and decisions that favour short-term, piecemeal responses (UNCCD 2008). Yet in many of the arid and semi-arid rangelands of Africa, livestock are often the only viable production alternative to crops, and play a critical role in the cultural practices of pastoralist communities (Krätli et al. 2013) while partially fulfilling a niche in the absence of wildlife. "Optimal grazing", as one pathway in natural climate solutions, may provide opportunities to restore rather than degrade soil and vegetation (Griscom et al. 2017; Griscom et al. 2020). One overlooked tool in "optimal grazing" and rangeland restoration may be the use of traditional, short duration and overnight kraals.

Kraals (corrals or bomas) are barriers used by pastoralists to confine livestock overnight or for longer periods and are made of either natural material (scrub or logs), fencing, or synthetic mobile sheeting (Augustine 2003). Kraaling activity dates to ancient pastoral societies who herded their livestock alongside wild animals, often following similar migration routes and seasonal foraging patterns to wild herbivores (Fynn et al. 2016). Thus, a primary reason for pastoralists to herd and kraal livestock was to protect them from predation, i.e. for favourable animal production. However, these activities may have multi-faceted gains or 'desired effects' for the ecosystem compared to fenced or unattended livestock, e.g. herding and planned grazing may maintain plant biomass production and diversity, while animal confinement during kraaling potentially enables nutrient enrichment for plant growth as well as multiple actions such as animal observation, veterinary treatment, and milking.

While very little quantitative and comparative information exists on the effect of herding on ecosystems, more information is available for kraaling. Over the last decade, the concept and implementation of kraaling as an active restoration method has received increased attention in east and southern Africa, e.g. Hawkins et al. (2022); Huruba et al. (2018); Huruba et al. (2022). Kraals have long been known as sites of nutrient enrichment (Porensky and

Veblen 2015; Veblen 2012). Livestock spend many hours inside these kraals, and large dung and urine deposits may alter soil properties over time (Porensky and Veblen 2015) with knock-on effects on vegetation structure and biodiversity (Jamsranjav et al. 2018). Once abandoned, these sites can have higher grass diversity and biomass than the surrounding areas (Sibanda et al. 2016), and become important foraging areas for wild herbivores (Huruba et al. 2022). However, soil and plant responses differ between biomes (Hawkins et al. 2022) and soil nutrients may accumulate to levels that are undesirable ecologically and even for agricultural purposes (Augustine 2003). Therefore, it is important to know whether on balance and across biomes soil nutrients accumulate on abandoned kraals, so that this information can be used to make management decisions including restoration. This is especially urgent because pastoralism and associated activities such as kraaling are in decline due to, e.g. inadequate remuneration for herders and land tenure issues (Basupi et al. 2017; Manzano et al. 2021).

Therefore, we conducted a meta-analysis on the impact of short duration kraaling on soil nutrient accumulation and factors that moderate this. We centred our analysis around two main hypotheses (H1 and H2). We hypothesised that kraaling livestock would result in soil nutrient hotspots, the persistence of which would depend on the soil element concerned (H1). Specifically, we anticipated that less mobile elements, such as phosphorus (P), would have a higher residence time in topsoil compared to mobile cations, such as potassium (K), and may greatly exceed concentrations in native soils. We also expected that the volatile or easily leached elements in dung or urine would be ephemeral, e.g. nitrogen (N) in the form of ammonia, nitrate, or urea, and carbon (C) as carbon dioxide, but that total soil N and C would still increase with kraaling. We also expected that livestock characteristics (density, duration in kraals, time since kraaling) and the biome would moderate the soil responses to kraaling (H2).

The meta-analysis approach provided a way to measure the relative effect size for each of the response variables of interest from various studies, as well as an overall effect, so that the magnitude of the treatment effect (kraaling) could be quantified across studies (Osenberg et al. 1999). We discuss our findings relative to both the ecological and agricultural contexts, 'desired states', and a case study called the Herding for Health Model. We also discuss how kraaling and pastoralism relates to the Sustainable Development Goals of production (SDG 2), climate (SDG 13), and livelihoods (SDG 1), and how these may be supported by a blending of indigenous knowledge and science (SDG 17, Global Partnerships for the goals).

Methods

General approach

The meta-analysis tested whether short duration kraals altered soil chemical properties [soil organic carbon (C), total nitrogen (N), available phosphorus (P), available potassium (K), and pH] compared to adjacent non-kraaled areas based on data available from the literature. Both kraaled and non-kraaled areas were natural/native rangeland (except for one case in shrublands; Table 1) and while the condition thereof may have varied between studies it was comparable within studies. In addition, we aimed to test the effect of statistical moderators, i.e. factors affecting the strength or direction of the kraaling effect such as livestock characteristics (density, duration in kraals, time since kraaling) and the biome. Studies included in the meta-analyses had to meet the criteria of being peer-reviewed articles in journals and have sufficient sample size to determine both a mean and variance. These data were extracted directly from the article, or were provided by the authors, or extracted using WebPlotDigitizer (version 4.5; freeware available at https://automeris.io/WebPlotDigitizer). All analyses and plots were performed using R version 4.2.0 (R Core Team 2020), with additional functionalities from the *metafor* (Viechtbauer 2010) and *weightr* (Vevea and Hedges 1995) packages.

Literature search and data extraction

A systematic keyword-based search was conducted on Scopus (searching for the same terms on Clarivate Web of Science yielded no additional results). Peer-reviewed articles or reviews were considered when the term *kraal* or *corral* or *boma*, and at least one of the terms *nutrient* or *hotspot* or *grassland* or *savanna* were present in the abstract of the article. This resulted in a total of 92 articles. After reading the articles, only those relevant to the topic, and those that included measurements of soil chemical properties from within a kraal and a comparable control (non-kraal) site were included in the analysis. In addition, studies were also identified through 'snowballing', i.e. reference lists of acquired studies were searched for additional relevant studies. Based on the selection criteria, 12 studies were used in the meta-analysis (Table 1) with an additional three providing qualitative data (Kizza et al. 2010; Muchiru et al. 2009; McManus et al. 2018). Within study data was separated based on kraal age resulting in several datasets depending on the soil element (C: n = 15; N: n = 14; P: n = 17; K: n = 19; pH: n = 12). Where authors classified multiple age groups together, the median value of this grouping was used (e.g. kraals between 5 and 10 years old were categorised as 7.5 years old), and where kraals were all older than a particular age that number was used (e.g. equal or greater than 45 years was set to 45 years).

Data harmonisation

Studies measuring soil characteristics at 0–10 or 0–15 or 0–20 cm soil depth were considered comparable. In one case (Gonçalves et al. 2021) results from analyses at 0–10 cm and 10–20 cm were averaged to provide the 0–20 cm soil depth. Although methods of elemental analysis differed between studies, kraal and control treatments from which relative differences were calculated always used the same method, and therefore any differences in values between methods was not considered to affect comparisons. Specifically, total soil organic carbon (C) in the studies was measured using dry combustion or the Walkley-Black procedure where these methods yield similar results (Shamrikova et al. 2022). If loss on ignition was reported, the resulting estimate of soil organic matter (SOM) was converted to soil organic carbon by assuming

50% carbon in SOM (Pribyl 2010). Total soil nitrogen (N) was measured using dry combustion or Kjeldahl digestion. Available soil phosphorus (P) was measured using Bray II / citric acid, or the Olsen method for calcareous soils with a pH above 7.4. Available soil potassium (K) was measured using inductively couple plasma spectroscopy, and soil pH was measured in KCl, $CaCl_2$ or water extracts. Soil physical characteristics and other soil chemical characteristics (cation exchange capacity and ion concentrations of other elements) were considered but not reported in enough studies to be used in the analysis.

Meta-analysis models

The effect of kraaling relative to not kraaling, or effect size, was measured using the standardised mean difference estimator or SMD (Hedges 1981). The SMD takes the difference in means between the kraaled and non-kraaled group for any measure and divides this by a pooled standard deviation from the two groups, hence standardising it and allowing direct comparison between different studies. The SMD is more generalizable with similar statistical power compared to mean difference (Takeshima et al. 2014). In *metafor*, the function "*SMD*" corrects for the positive bias in SMD, yielding Hedges' *g*. The SMD of each study was also averaged to provide an overall effect size for all the studies. The SMDs (hereafter "effect sizes") were calculated for soil total organic C and total N (%), soil available P (ppm), soil available K (ppm), and soil pH from each study and used in the meta-analysis via a random-effects (RE) model. Thereafter, a second model called a mixed-effects meta-regression random effects model was run for each response variable where kraal age was included as a moderator in the model. The RE model aligns with our assumption that the reported studies represent all relevant studies that could be reasonably found about the phenomenon of interest but are likely a random sample from a larger population of studies that were or will be conducted. As such, RE models make unconditional inferences about the overall outcome and can be generalised beyond the set of studies included. This contrasts with the fixed-effects model where the outcomes are conditional in referring only to the included studies, or the equal-effects model which works in the same way as the fixed-effects but assumes homogeneity between studies (Viechtbauer 2010), neither of which applied to our hypothesis.

Model checking

We assessed model fit, statistical heterogeneity and bias, tested for influential studies, and conducted sensitivity analyses using *metafor*.

Model fit was reported using P values. Several measures were used to assess heterogeneity because no one measure is adequate especially for small meta-analyses (Olkin et al. 2012): between-study variance (τ^2), Cochran's Q-test and its transform I^2 or Inconsistency (Higgins et al. 2003). If high heterogeneity was detected ($\tau^2 > 0$), a prediction interval for the true outcomes was provided (Riley et al. 2011). Where $I^2 > 75\%$ (indicative of subgroups), kraal age was included in RE models. The rank correlation test (Begg and Mazumdar 1994) and the regression test (Sterne and Egger 2006) were used to check for funnel plot

Table 1: Details of articles in the meta-analysis including kraal age, kraal size, type of livestock, biome, country, and soil variables. Abbreviations: C: carbon, N: nitrogen, P: phosphorus, K: potassium

Author(s) & year	Kraal age (years since abandonment)	Kraal size (ha)	Livestock	Biome	Country	Soil variables
Augustine (2003)	1.5, 12–24, 30–39	0.4	Cattle	Savanna	Kenya	C, N, P, K
Chikorowondo et al. (2017)	22	1	Cattle	Savanna	Zimbabwe	C, P, K, pH
Gonçalves et al. (2021)	6 to 18	NA	Cattle	Savanna	Brazil	N, P, K, pH
Hawkins et al. (2022)	0 to 2	0.005	Cattle, sheep, goats	Grassland	South Africa	C, N, P, K
Huruba et al. (2022)	2	0.7	Cattle	Savanna	Zimbabwe	N, P, K
Porensky and Veblen (2015)	0, 1.5	0.02	Cattle	Savanna	Kenya	C, N, P, K, pH
Porensky and Young (2016)	3	0.02	Cattle	Savanna	Kenya	C, N, P, K, pH
Rahman et al. (2019)	0	0.03	Sheep, goats	Savanna	Ghana	C, N, P, K, pH
Saatkamp et al. (2021)	1–40, 100–150, 1500–2000	0.1, 0.064, 0.025	Sheep	Grassland	France	C, N, P, K, pH
van der Waal et al. (2011)	37	0.008–0.07	Cattle	Savanna	South Africa	P, K, pH
Veblen (2012)	< 42, > 42	0.25–1	Cattle	Savanna	Kenya	C, N, P, K, pH
Vinograd et al. (2019)	5–10, 15–20	0.2–0.5	Sheep	Shrubland – planted forest	Israel	K

asymmetry or small-study bias by using the standard error of the observed outcomes as a predictor. Publication bias was estimated by creating a selection model that increased the weight of studies that are assumed to be less likely to be published (because their p values were > 0.05), while decreasing the weight of studies assumed to be more likely to be published (because their p-values were < 0.05). The selection model further compared adjusted and unadjusted effect size values using a Likelihood Ratio Test where any p-value less than 0.1 was indicative of publication bias.

Influential studies and potential outliers were identified using the studentised residuals test as well as the commonly used Cook's distances (*influence* function). Studies with a Cook's distance larger than the median plus six times the interquartile range were considered disproportionately influential. In addition, a combinational meta-analysis was used to identify clusters of influential studies by performing a series of meta-analyses based on all possible combinations of study subsets. For example, if there are 12 studies, then $2^n–1$ analyses were performed where n is the number of studies, resulting in 4095 analyses. The latter was visualised using GOSH plots, or graphical display of study heterogeneity plots (Olkin et al. 2012). In a homogeneous set of studies (ones with similar results), the distribution in GOSH plots should be roughly symmetric, contiguous, and unimodal. Multimodal distributions indicated heterogeneity due to influential studies, distinct subgroupings of studies, and outliers (that were influential or not). Suspected influential studies were named and described but retained in the analysis due to limited studies. A sensitivity analysis was performed using a leave-one-out analysis (*leave1out* function). This analysis leaves one study out for n repetitions of the meta-analysis to indicate whether this changes the overall effect size.

Results

Short duration kraaling approximately doubled soil organic C, total N, available P, and available K, and slightly increased pH compared to non-kraaled areas, based on meta-analyses ($p \leq 0.0158$; Figures 1 to 5). Studies in the meta-analyses spanned various biomes (savanna, grassland, shrubland) and continents including Africa (Ghana, Kenya, South Africa, Zimbabwe), South America (Brazil), Europe (France), and Asia (Israel). As expected for small meta-analyses, there was high between-study heterogeneity in all cases as measured by both Q and I^2 (e.g. $I^2 \geq 90\%$; Figures 1–5). This means that while the average effect size was positive, in some cases the true outcome may in fact be negative. High I^2 can indicate that subgroups exist within the studies, e.g. different kraal ages since abandonment. However, soil fertility was not predicted by kraal age in our study (Table 2). Other potential moderators of soil fertility (livestock density, kraaling duration, biome type) were specified in too few studies or had too low a sample size to be tested. As expected, there was evidence of small-study bias in all cases ($p < 0.05$) except pH where neither the rank correlation nor the regression test indicated any funnel plot asymmetry ($p = 0.1160$ and $p = 0.0740$, respectively). The selection models showed no evidence of publication bias ($p > 0.1$) in any of the meta-analyses.

Specifically for soil organic C, the overall effect size based on the RE model was positive (2.74, $p = 0.0025$; Figure 1a) with the same result (2.74) from the leave-one-out analysis and a similar result (2.82) from the combinational meta-analysis (GOSH plot, Figure 1b). This means that kraaling more than doubled the soil organic carbon. While most (80%) effect sizes were positive, the heterogeneity scores for the RE model were significant (Q (df = 14) = 152, $p < 0.0001$, $\tau^2 = 11$, $I^2 = 97\%$). Thus the true range of effect sizes is –4.13 to 9.61 based on 95% prediction intervals. Based on studentised residuals, studies by Rahman et al. (2019) and Porensky and Veblen (2015) had values (+ 2.93) identifying them as potential outliers, while their relatively Cook's distance values also indicated they were disproportionately influential. The GOSH plot (Figure 1b) revealed that the latter study contributed to a bimodal distribution (red histogram and dots in kernel density estimate) compared to when it was excluded (blue histogram and dots) in the combinational meta-analysis. This study and Rahman et al. (2019) increased the overall effect size and were responsible for a large amount of the heterogeneity.

Similar results were obtained for soil total N (Figure 2a) where the overall effect size of kraaling was 1.96 ($p = 0.0009$), i.e. kraaling approximately doubled total soil N compared to no kraaling. A similar result was found with the leave-one-out analysis (1.97) and combinational meta-analysis (2.04). Most estimates were positive (79%). High heterogeneity values for the RE model (Q (13) = 99, $p < 0.0001$, $\tau^2 = 4.12$, $I^2 = 94\%$) again indicated that we should consider 95% prediction intervals for the true overall effect size (–2.19 to 6.11). The Porensky and Veblen study (2015, adjusted kraal age of 1.5 years) was identified as both an influential study and potential outlier from

Table 2: Results from the mixed-effects meta-regression random effects model of all soil variables, where kraal age was included as a moderating factor. QE = test for residual heterogeneity and associated p-values, I^2 = residual heterogeneity in percent, df = degrees of freedom, QM = test of moderator (kraal age) and associated p-values.

Soil variable	QE	p-value	I^2 (%)	df	QM	p-value
Carbon	138.64	< 0.0001	97.40	14	0.92	0.34
Nitrogen	90.23	< 0.0001	93.43	13	1.37	0.24
Phosphorus	103.56	< 0.0001	95.73	16	1.48	0.22
Potassium	194.16	< 0.0001	99.91	18	0.22	0.64
pH	75.82	< 0.0001	92.08	11	0.57	0.45

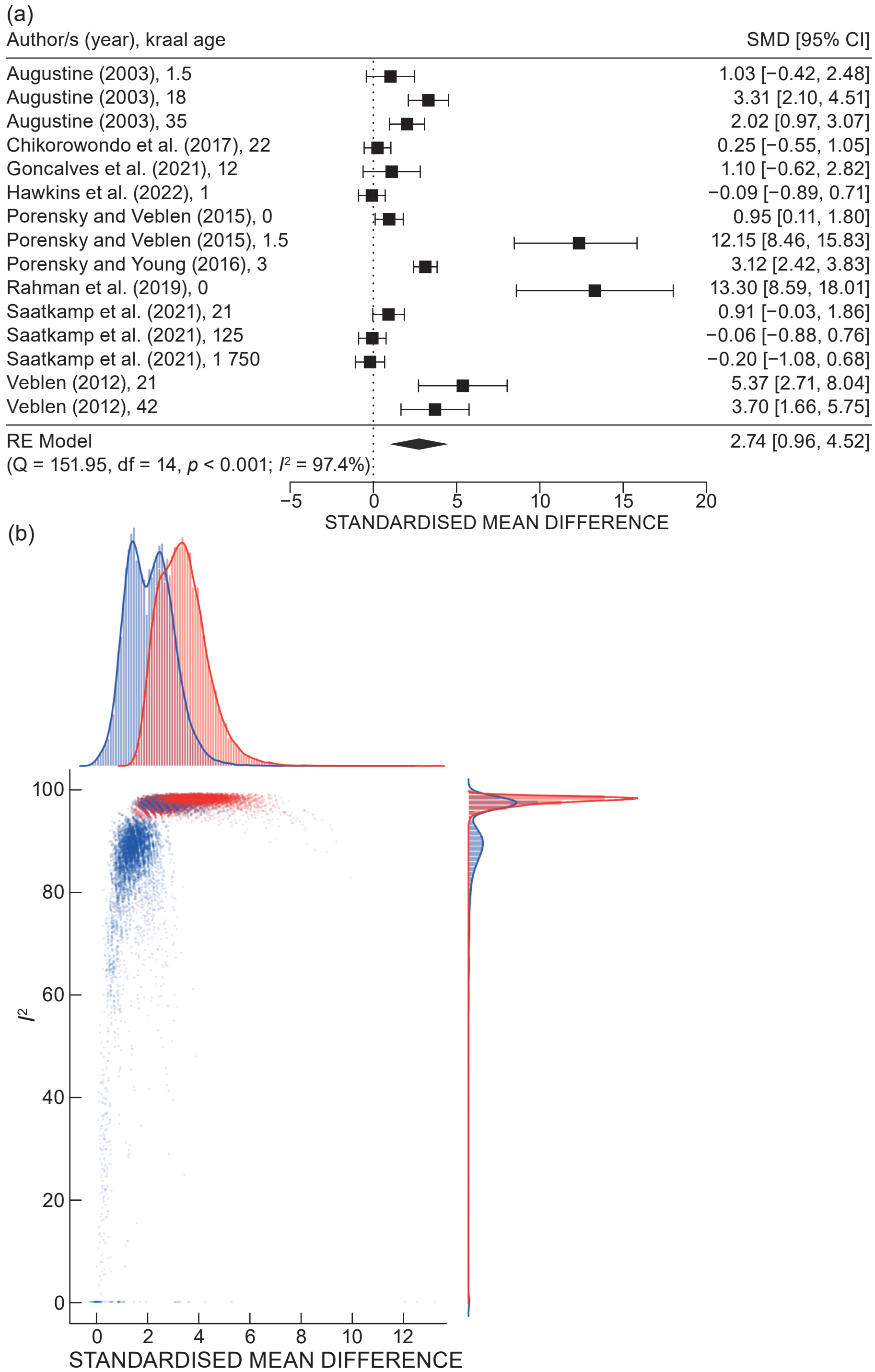

Figure 1: Effect size of kraaling on soil organic carbon showing a juxtaposed Forest plot (a) and associated graphical display of study heterogeneity (GOSH) plot (b). The standardised mean difference method was used within a random effects meta-analysis model without moderating factors. In the Forest plot, the effect size from each study is indicated by the mean (■) and the 95% confidence intervals (lines through means). The overall effect size from all the studies is indicated at the bottom of the plot by a diamond (♦). The dashed vertical line at zero indicates no effect. Studies are ordered alphabetically as Authors, Year, Kraal age. Q refers to model heterogeneity with associated degrees of freedom and *p*-value. I^2 is another measure of heterogeneity in percent. Red or blue histograms and dots in the GOSH plot include or exclude influential studies, respectively

studentised residuals values (+3.33) and Cook's distance values. The GOSH plot (Figure 2b) further revealed that this study was part of a subgroup that contributed to a bimodal distribution (red histogram and dots) compared to when it was excluded. These studies increased the overall effect size and had amongst the highest heterogeneity values.

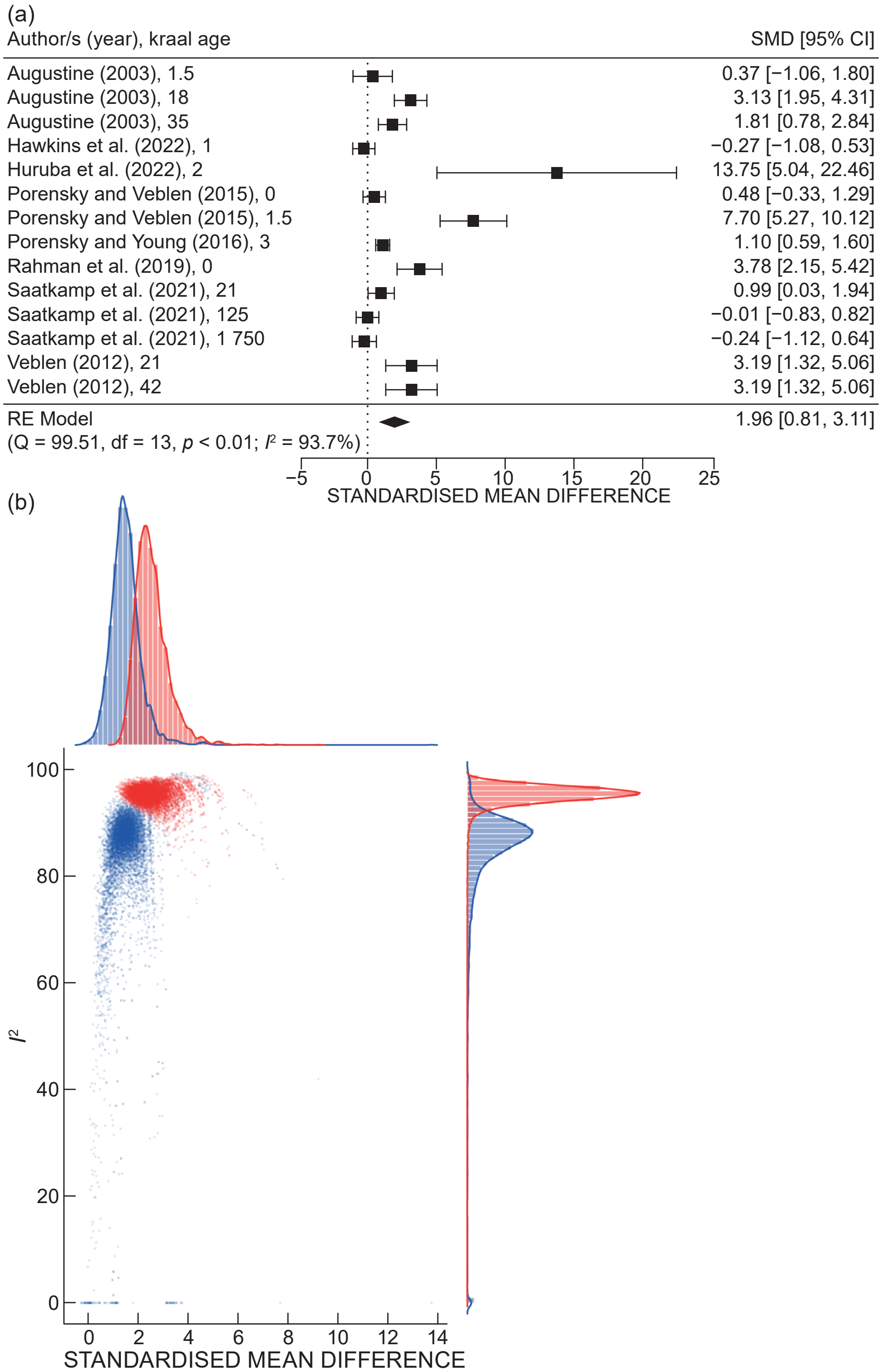

Figure 2: Effect size of kraaling on soil total nitrogen showing a juxtaposed Forest plot (a) and associated GOSH plot (b) using the standardised mean difference method and random effects meta-analysis model as described in Figure 1

In the soil P meta-analysis (Figure 3a), many studies (88%) reported an increase in available P with kraaling versus non-kraaling, as evidenced by the overall positive effect size from the RE model (2.56, p = 0.0002), with the same result from the leave-one-out analysis and a similar result from the combinational meta-analysis (2.59). Due to high heterogeneity between studies (Q (16) = 119, p < 0.0001, τ^2 = 7.13, I^2 = 96%) we can apply a 95% prediction interval to estimate the true effect sizes (–2.84 to 7.97). The Porensky and Veblen (2015; kraal age of 1.5 years) study was again identified as a potential outlier and disproportionately influential study (studentised residual of +4.09). The GOSH plot for available P (Figure 3b) indicated that this study was part of a subgroup that had

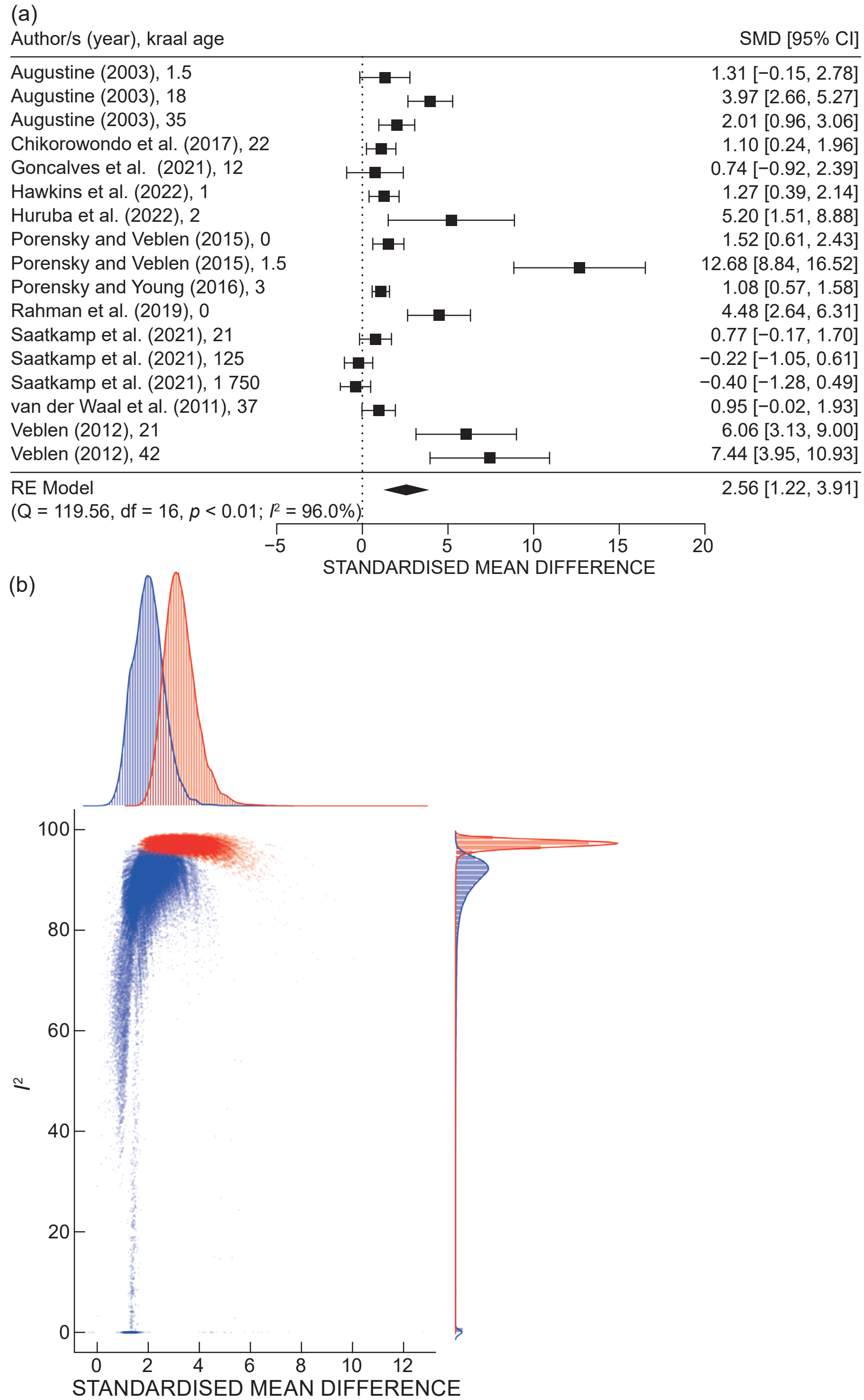

Figure 3: Effect size of kraaling on soil available phosphorus showing a juxtaposed Forest plot (a) and associated GOSH plot (b) using the standardised mean difference method and random effects meta-analysis model as described in Figure 1

a slightly higher effect size estimate and heterogeneity than other subgroup combinations. The subgroup with low heterogeneity and effect size (blue dots with I^2 tending towards zero; Figure 3b) referred to study combinations that were different but not overly influential.

Most (95%) studies reported an increase in available K with kraaling versus non-kraaling (Figure 4a) with an overall positive effect size (3.09, $p < 0.0001$) from the RE model. However, the leave-one-out analysis (7.39) and combinational meta-analysis (9.45) reported values more

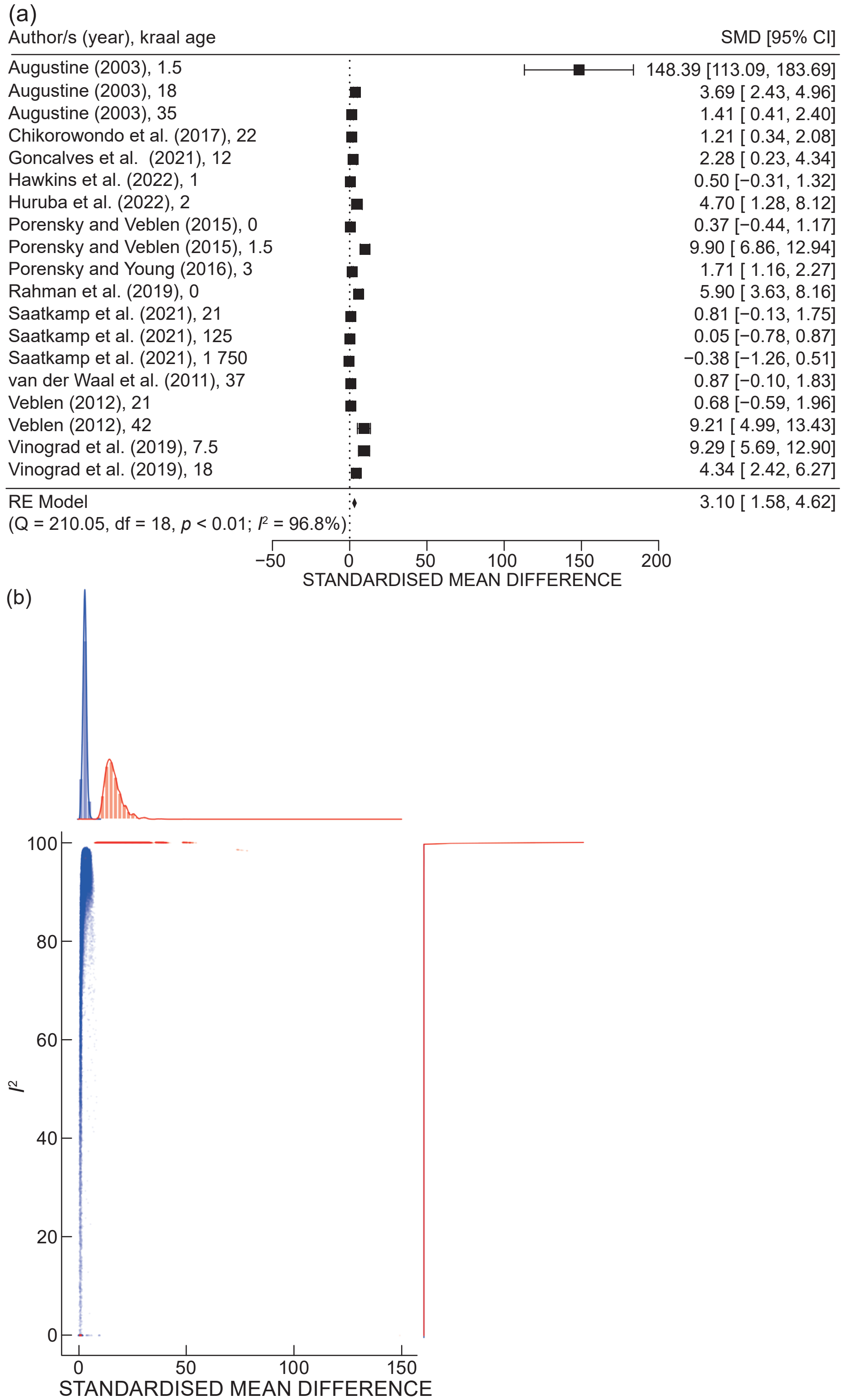

Figure 4: Effect size of kraaling on soil available potassium showing a juxtaposed Forest plot (a) and associated GOSH plot (b) using the standardised mean difference method and random effects meta-analysis model as described in Figure 1

than double the RE model. Due to high heterogeneity between studies ($Q(18) = 210$, $p < 0.0001$, $\tau^2 = 9.94$, $I^2 = 97\%$) we can apply a 95% prediction interval to estimate the true outcomes, which could vary between –3.26 to 9.46.

One study (Augustine 2003) with an adjusted kraal age of 1.5 years had an effect size two orders of magnitude larger than the others, and was identified as an outlier (+7.98) and influential study. The GOSH plot (Figure 1b) confirmed that this study was part of a subgroup combination that contributed to a positive overall effect size (Figure 4a and b) and had a high heterogeneity (Figure 4b).

Over half (67%) of the studies reported that kraaling increased soil pH (Figure 5a) to values approaching neutrality (pH 7) with an overall effect size of 1.28 (p = 0.0158), which was similar to the value obtained from the leave-one-out analysis (1.27) and the combinational meta-analysis (1.30). Due to high heterogeneity between studies (Q(11) = 80, p < 0.0001, τ^2 = 2.98, I^2 = 92%) the true overall effect size could vary between –2.25 and 4.80. No single study disproportionately influenced the outcome of the meta-analysis for pH. Supporting this, even when the study with the highest Cook's distance (van der Waal et al. 2011) was made an outgroup in the GOSH plot (Figure 5b), the histogram of effect size values was nearer to unimodal with more similarity in I^2 values between subgroups compared to the meta-analyses of other soil elements.

Three relevant studies (Kizza et al. 2010; Muchiru et al. 2009; McManus et al. 2018) had to be rejected because

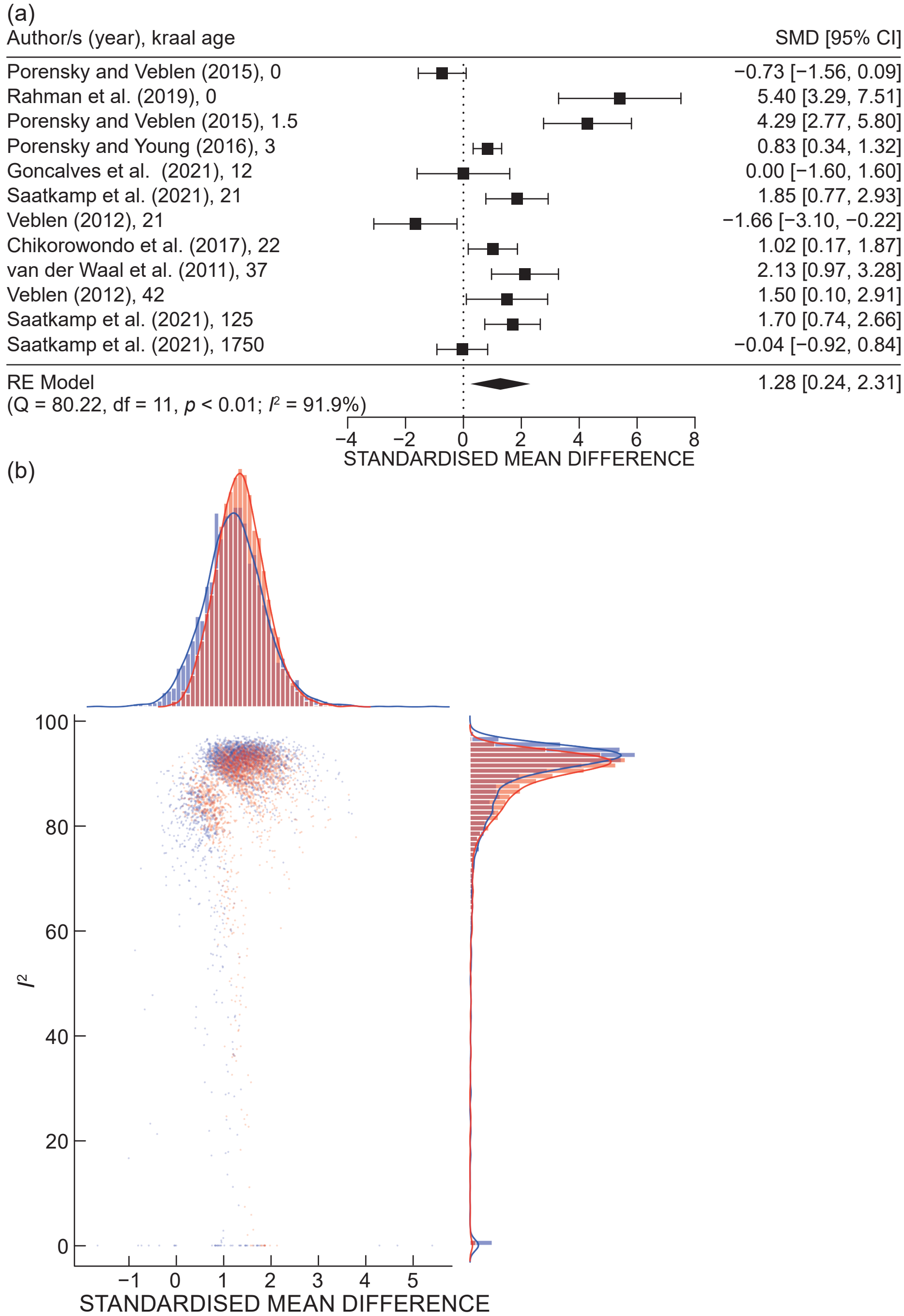

Figure 5: Effect size of kraaling on soil pH showing a juxtaposed Forest plot (a) and associated GOSH plot (b) using the standardised mean difference method and random effects meta-analysis model as described in Figure 1

they did not meet the meta-analysis criteria. The responses of soil properties to kraaling could nevertheless be counted from these studies along with the meta-analysis studies by categorising them into groups of negative, neutral, or positive (increasing) effects (Figure 6). Here, more than half of the studies reported that kraaling increased N, P, and pH, while over half of the studies reported neutral effects for C, and most studies reported neutral effects for K even though the overall effect sizes from the meta-analysis were positive. Very few studies reported decreasing effects of kraaling. Overall, these counts reflect the meta-analysis findings.

Discussion

The development of nutrient hotspots and palatable grasses that attract livestock and wild herbivores after short duration kraaling gained attention from studies such as those in the savanna of Kenya (Veblen 2012; Porensky and Young 2016; Riginos et al. 2012). Our meta-analyses demonstrated that the persistence of these nutrient (C, N, P, K) hotspots and increased pH occurs more generally, i.e. across biomes and continents, supporting H1. Because some studies showed kraals to have no or even decreasing effects on soil nutrients that may be linked to unmeasured moderating factors, our findings will be improved by more studies besides those available.

We expected that less mobile soil elements such as P would persist longer than more mobile elements such as K (part of H1), but we could not show this via the meta-analysis. The age of kraals and biome, the only potential drivers of kraal nutrients we could measure, did not resolve as drivers of the soil nutrient concentrations in kraals (H2 was not supported). This is likely due to studies being mostly in the savanna with only two being in the grassland biome and one in a shrubland/planted forest, i.e. a larger sample size across biomes was needed. Soil nutrients were similar in, e.g. 21, 125, and 1 750 year-old kraals in Roman ruins (Saatkamp et al. 2021) but it is possible that changes in soil nutrients happen over the first few months versus years after abandonment and studies have not captured this. Besides overall effect sizes, some individual studies found that nutrient enrichment of kraals was indeed time dependent and differed depending on the soil element. For example, one year after kraal abandonment in mesic grasslands, relatively immobile soil P remained raised while N and K differences were absent (Hawkins et al. 2022), and 45 years after kraal abandonment, P, Ca, and Mg persisted or reduced more slowly over time compared to N (Kizza et al. 2010).

As we hypothesised in H1, total soil organic C and total N accumulated in kraals relative to non-kraaled areas, despite the likelihood that most of the inorganic N in dung and urine (NO_3^- and NH_4^+) was leached or volatilised, respectively. Indeed several studies report an increased inorganic N that disappears after several years (Vinograd et al. 2019; Kizza et al. 2010) or with increasing distance from the kraaling station (Meglioli et al. 2017). While we could only analyse one of the soil cations (K), increased Mg, Ca (Kizza et al. 2010; Muchiru et al. 2009; Porensky and Young 2016), Na (Muchiru et al. 2009; Porensky and Young 2016) and cation exchange capacity (Meglioli et al. 2017; Kizza et al. 2010; Porensky and Young 2016) with kraaling has also been reported. Too few studies measured soil bulk density to include in the meta-analysis but of these a few studies report that short duration kraaling did not influence bulk density (Hawkins et al. 2022; Kizza et al. 2010) or soil texture (Kizza et al. 2010) but long-term kraaling did (Tate et al. 2004) as one might expect. Generally the effects

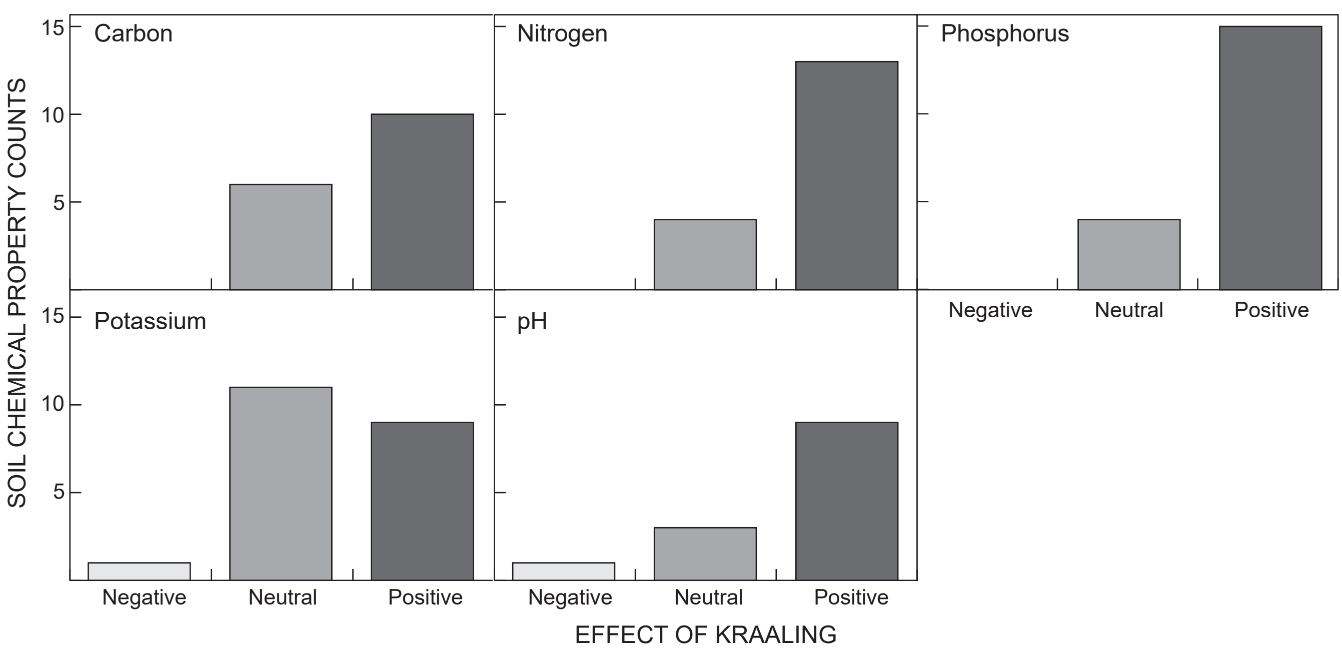

Figure 6: Counts of studies showing a positive, neutral, or negative effect of kraaling on soil chemical properties. Results were sourced from studies included in the meta-analysis as well as others from which data required in the meta-analysis could not be extracted

of kraals on soils can be expected to be higher in upper versus lower soil layers (Meglioli et al. 2017).

Nutrient hotspots left by kraals are known to increase the N, P and K in forage, which attracts domestic and wild ungulates alike. Soil pH increase (towards pH 7) was also an overall effect of kraaling in our study, implying that many nutrients would not only be more abundant but also more available for plant uptake (Marschner 1995). Changes in soil due to kraaling often drive changes in plant biodiversity and cover. Veblen (2012) first reported that kraal sites persisted for 40 years or more as nutrient-enriched, treeless 'glades' on black cotton soils of Kenya. Here, repeated grazing by herbivores maintained these glades, and slowed recruitment of a later successional grass, i.e. altered plant community dynamics. Jamsranjav et al. (2018) reported that cover of unpalatable annual forbs and palatable grasses increased and decreased respectively with increasing proximity to kraals in steppe of Mongolian rangelands. In the arid Karoo vegetation of South Africa, kraaling decreased the density and diversity of shrubs although grasses were not affected (McManus et al. 2018). In the savanna of South Africa, van der Waal et al. (2011) found that kraaling altered the grass-tree competition, probably by increasing soil nutrients, attracting herbivores, and reducing tree recruitment in a negative feedback loop. Similarly, Huruba et al. (2022) reported that elephants were attracted to old kraal sites and damaged trees within them. However, high grazing intensity near active kraals decreased plant diversity and cover of perennial graminoids while increasing woody invasive plants in the sandy grasslands of South Africa (Shezi et al. 2021). In mesic high altitude grasslands of South Africa, short duration kraaling increased grass cover if basal cover was low or degraded (<50% grass, <10% forbs) but decreased it if the grass sward was intact (Hawkins et al. 2022) making clear that the use of kraals as a restoration tool is biome and context specific.

It can generally be expected that after kraal abandonment, plant species composition will go through a successional change from ruderal to vegetation that is similar to native adjoining areas. This was clearly illustrated by a study in the savanna in Kenya (Muchiru et al. 2009; Veblen 2012) and semi-arid Mediterranean shrubland in Israel (Vinograd et al. 2020). However, kraaling may irreversibly change the vegetation as evidenced from ancient Roman ruins of sheep kraals that were abandoned two millennia ago but have not yet reverted to the native Mediterranean grassland (Saatkamp et al. 2021). The duration for which a particular kraal was used, as well as the density of the animals that were placed within the kraal, may affect the recovery period of the vegetation. Changes in vegetation structure as well as increased soil fertility will inevitably affect the abundance and biodiversity of other organisms, including soil microbes (Vermeire et al. 2021) and arthropods (Thoresen et al. 2021). For example arthropod abundance decreased with increasing soil fertility (C, N, K, Mg, Ca) and proximity to kraals in subtropical forest of Brazil (Pestana et al. 2020).

While kraaling can result in soil nutrient hotspots and increased plant biomass that can have desired effects for livestock, wildlife, or even restoration at the local scale, it should be clearly acknowledged that pastoralist settlements and kraals will likely alter the native nutrient and patch dynamics of an area over decadal (Muchiru et al. 2009; Meglioli et al. 2017; Saatkamp et al. 2021; Veblen 2012), century and even millennial timescales (Saatkamp et al. 2021). Thus, kraaling over an excessive area or time may result in soil characteristics outside of the local ecological bounds. Soil P being relatively immobile in all but the most sandy soils, may accumulate excessively in soils and thus increase in plant leaves to levels above those recommended for livestock (Augustine 2003), i.e. may even be outside of desired agricultural bounds. Spatial dynamics should also be considered because local enrichment in kraals may mean depletion at larger scales. For example, local enrichment at the kraal scale resulted in net losses outside the kraal in woodlands of the Monte desert, Argentina (Meglioli et al. 2017). These authors speculate that losses outside the kraals were due to net nutrient losses (through NO_3^- leaching, NH_4^+ volatilisation, denitrification, organic matter oxidation due to trampling, manure exports, soil erosion) being relatively higher than the nutrient imports from dung and urine. While kraals have the potential to contribute to the structural heterogeneity of savannas and other ecosystems (van der Waal et al. 2011), Chikorowondo et al. (2017) recommend that conservation monitoring plans be put in place for nutrient hotspots created by kraals since they can influence foraging behaviour, especially in dystrophic soils.

Impacts of kraals on climate change via greenhouse gas emissions is also a concern. Besides losses of CO_2 from dung and urine, abandoned kraals are sources of CH_4 and N_2O via anaerobic decomposition of dung. Butterbach-Bahl et al. (2020) found that kraals are sources of N_2O even 40 years after abandonment. These authors scaled their measurements to the continent and estimated that kraals contribute ca. 5% of the current estimate of total anthropogenic N_2O emissions for all of Africa. Accordingly, certain carbon standards (e.g. Verra Verified Carbon Standards) stipulate that dung should not be allowed to accumulate and cover more than 50% of a kraaled area. Thus, while our findings show that kraals are hotspots of soil organic C (carbon sequestration), a measure of net removal and emissions reduction in CO_2 equivalents under a range of kraaling durations and livestock densities would provide a fuller picture of whether kraaling can be used as a climate-smart restoration tool across biomes. Meanwhile, initiatives in Africa such as Herding 4 Health potentially provide us with a natural experiment to test this and other questions.

Herding 4 Health case study

The Herding 4 Health (H4H) Programme supports the implementation of the H4H Model (van Rooyen 2016) in 15 sites across six countries in African savanna and grassland. The H4H Model is novel because it focuses on a few interventions (strategic herding and kraaling, Figure 7) to address a multitude of challenges facing rangelands and pastoral communities, including loss of soil and biodiversity, livestock-wildlife conflict, herd health, illegal wildlife trade (Heermans et al. 2021) and loss of rural livelihoods. Through this model, herders are professionalised and given community-wide responsibilities to emphasise the importance of herding for entire communities and their natural resources. Since 2019, H4H has supported the

Figure 7: Example of traditional permanent kraals or bomas (a) and mobile short-duration kraal in Mozambique within the Herding 4 Health Model (b). The inset in (b) shows an active boma (left) next to an abandoned one (right), both of which have visibly darker soil surfaces compared to the surrounding landscape

training of over 500 herders through various partners, starting in and around Mozambique's Limpopo National Park (LNP). It is presently operational in seven villages and involves nearly 14 000 cattle (more than 85% of the cattle in these villages). There are currently 40 full-time employed skilled herders and 72 volunteers who ensure controlled animal movement, planned grazing, and mitigation of wildlife-livestock conflict over 204 600 ha of rangelands. The need for farmers to extract wood resources to construct traditional, non-mobile kraals (Figure 7a) is negated by mobile kraals (or bomas as they are called in H4H) constructed from lightweight steel/composite fibre and canvas sheets (Figure 7b). Within-kraal soil is easily distinguishable from its surrounds after movement of a kraal (Figure 7b inset). Sixteen of these predator-proof mobile bomas (Figure 7b) are currently being used in the LNP and each one is moved approximately every 4 to 7 d, depending on the soil conditions.

The premise of the model is that judicious movement of livestock, including kraaling, across rangelands can lead to multiple desired outcomes for people and environment. Some effort to test this premise has begun (Heermans 2021) but the multi-faceted impact of herding is under-researched and has yet to be reviewed in a quantified way. To date, monitoring within H4H comprises anecdotal information but also counts of farmer participation and animal losses. For example, the use of these mobile bomas has been extremely successful in stopping the historically high predation of livestock, especially of calves (Jacques van Rooyen, pers. comm. 4 May 2022). Data from LNP indicates that farmers lost up to 20 animals per month to predators prior to their use. Losses have reduced to one incident since the beginning of 2020 according to LNP and H4H records. Besides livestock protection, the H4H bomas apparently play a major role in encouraging farmers to combine herds and graze livestock in a coordinated manner for improved rangelands and livestock productivity, e.g. in Botswana and Zambia. Cattle can now be moved more strategically to take advantage of different water sources at different times of the year, thus facilitating the optimal utilisation of available natural resources by aligning the distribution of grazing and grazers across seasons. This strategic herding minimises the distance cattle need to walk, which in turn should improve the condition of the cattle while distributing the impact of cattle across rangelands. In non-H4H sites of Mozambique, cattle have been measured to walk between 20 to 30 km a day from the village in search of grazing in the dry season. Being part of H4H has dramatically reduced these walking distances as mobile kraals are stationed near water and grazing while avoiding degraded areas near villages. In this way, herders aim to allow degraded areas to recover while ensuring energy efficient livestock husbandry. Mobile bomas (Figure 7b) also limit bi-directional movement between stationary kraals (Figure 7a) and grazing areas, where the latter can lead to multiple cattle paths, vegetation trampling, and eventually piospheres and soil erosion. In addition, mobile kraals and herders are trained to avoid contact with disease-carrying livestock and wildlife such as buffalo. This plays an important role in limiting the transmission of transboundary and zoonotic diseases, such as foot-and-mouth disease and bovine tuberculosis, especially since vaccination coverage in H4H herds are much higher than others in the same general area. Besides livestock protection and development of nutrient hotspots, the mobile bomas facilitate the observation, recording and reporting on herd health, and subsequent decision making on e.g. bull selection for improved conception rates.

Anecdotally, the incorporation of organic matter into the soil of the boma area has resulted in the rapid regrowth of grass that appears to have relatively higher biomass and palatability once the boma is removed, encouraging wild ungulates to select for these sites. Free-roaming wildlife (elephant, zebra, buffalo, and other grazers) have been observed to target regrowth in boma sites across H4H sites in the Okavango Delta and Mozambique. While this anecdotal evidence supports the findings from the meta-analysis that kraaling improves soil fertility, further analyses are needed. H4H offers an excellent opportunity to better understand the benefits of short-duration kraaling. The large sample size of kraals in the LNP alone (16 bomas moved weekly), plus multiple mobile kraals across other H4H sites, provides researchers with the opportunity to supplement the results obtained from the current meta-analysis.

Conclusion

Short duration kraals (or bomas) have potential as a cost-effective restoration tool if used judiciously because they rely on passive reestablishment of soil nutrients and ground cover post animal impact versus active actions such as seeding. Clearly, there are trade-offs and synergistic (facilitative) pathways in how kraals can be used, ones that impact the Sustainable Development Goals of production (SDG 2), climate (SDG 13) and livelihoods (SDG 1). An understanding of these pathways may be supported by a blending of indigenous knowledge and science (the SDG 17, of Global Partnerships for the goals) leading to various 'desirable states'. For instance, pastoralists and other rangeland managers can make judicious use of nutrient hotspots post kraal abandonment to facilitate coexistence between livestock and wild herbivores (and potentially other biodiversity), while ideally staying within both ecological and agricultural bounds of that ecosystem. Also, the placement of kraals onto existing fallow cropland prior to planting should avoid eutrophication of native rangelands while fertilising crops at no cost. It may even be possible to mitigate bush encroachment by attracting mega herbivores to old kraal sites. Because certain biomes and plant habits are evidently more sensitive to kraaling than others, e.g. high altitude grassland versus savanna, kraaling should preferentially occur on degraded grassland versus intact grassland, and similar nuances may apply in other biomes, possibly dependent on rainfall-fertility gradients. Generally, we recommend that the suitability of kraaling be evaluated per vegetation type, local context and 'desired state', as informed by science, best practice, and indigenous or local knowledge.

Acknowledgements — We thank the authors who supplied their raw data upon request. We also acknowledge the funding from Lui-Walton Innovators Fellowship via Conservation International supplied to JvR that supported this study.

ORCIDs

Mia Momberg: http://orcid.org/0000-0002-8901-9271
Anna Jean Haw: http://orcid.org/0000-0002-5904-2753
Perushan Rajah: http://orcid.org/0000-0002-8436-3727
Jacques van Rooyen: http://orcid.org/0000-0002-4062-1384
Heidi-Jayne Hawkins: http://orcid.org/0000-0001-9334-0669

References

Augustine D. 2003. Long-term, livestock-mediated redistribution of nitrogen and phosphorus in an East African savanna. *Journal of Applied Ecology* 40: 137–149. https://doi.org/10.1046/j.1365-2664.2003.00778.x.

Basupi LV, Quinn CH, Dougill AJ. 2017. Historical perspectives on pastoralism and land tenure transformation in Ngamiland, Botswana: What are the policy and institutional lessons? *Pastoralism* 7: 24–37. https://doi.org/10.1186/s13570-017-0093-1.

Begg C, Mazumdar M. 1994. Operating characteristics of a rank correlation test for publication bias. *Biometrics* 50: 1088–1101. https://doi.org/10.2307/2533446.

Butterbach-Bahl K, Gettel G, Kiese R, Fuchs K, Werner C, Rahimi J, Barthel M, Merbold L. 2020. Livestock enclosures in drylands of Sub-Saharan Africa are overlooked hotspots of N_2O emissions. *Nature Communications* 11: 4644. https://doi.org/10.1038/s41467-020-18359-y.

Chikorowondo G, Muvengwi J, Mbiba M, Gandiwa E. 2017. Influence of abandoned cattle enclosures on plant assemblages and herbivory in a semi-arid savanna. *Ecological Research* 32: 1023–1033. https://doi.org/10.1007/s11284-017-1522-8.

Fynn RWS, Augustine DJ, Peel MJS, de Garine-Wichatitsky M. 2016. Strategic management of livestock to improve biodiversity conservation in African savannahs: a conceptual basis for wildlife-livestock coexistence. *Journal of Applied Ecology* 53: 388–397. https://doi.org/10.1111/1365-2664.12591.

Gonçalves LVC, Pinho RC, Ayres MIC, Ticona-Benavente CA, Pereira HS, Junior AFN, Alfaia SS. 2021. Influence of the Caiçaras on soil properties in the Savanna Region of Roraima, Northern Amazon. *Sustainability* 13: 11354. https://doi.org/10.3390/su132011354.

Griscom BW, Adams J, Ellis PW, Houghton RA, Lomax G, Miteva DA, Schlesinger WH, Shoch D, Siikamaki JV, Smith P, et al. 2017. Natural climate solutions. *Proceedings of the National Academy of Sciences USA* 114: 11645–11650. https://doi.org/10.1073/pnas.1710465114.

Griscom BW, Busch J, Cook-Patton SC, Ellis PW, Funk J, Leavitt SM, Lomax G, Turner WR, Chapman M, Engelmann J, et al. 2020. National mitigation potential from natural climate solutions in the tropics. *Philosophical Transactions of the Royal Society B* 375: 20190126. https://doi.org/https://doi.org/10.1098/rstb.2019.0126.

Hawkins H-J, Mgwali N, Vetter S. 2022. Effects of short-duration kraaling depend on initial conditions in a mesic grassland. *African Journal of Range & Forage Science*. https://doi.org/10.2989/10220119.2021.2012716.

Hedges L. 1981. Distribution theory for Glass's Estimator of effect size and related estimates. *Journal of Educational Statistics* 6: 107–128. https://doi.org/10.3102/10769986006002107.

Heermans B. 2021. A One Health assessment of a "Herding for Health" project at the wildlife-livestock interface in western Ngamiland, Botswana. PhD thesis, University of Pretoria, South Africa.

Heermans B, van Rooyen J, Fynn R, Biggs D, Lewis M, McNutt J. 2021. Husbandry and Herding: A community-based approach to addressing illegal wildlife trade in northern Botswana. *Frontiers in Conservation Science* 2: 675493. https://doi.org/10.3389/fcosc.2021.675493.

Herrero M, Havlik P, Valin H, Notenbaert A, Rufino MC, Thornton PK, Blummel M, Weiss F, Grace D, Obersteiner M. 2013. Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences USA* 110: 20888–20893. https://doi.org/10.1073/pnas.1308149110.

Higgins JP, Thompson SG, Deeks JJ, Altman DG. 2003. Measuring inconsistency in meta-analyses. *British Medical Journal* 327: 557–560. https://doi.org/10.1136/bmj.327.7414.557.

Huruba R, Mlambo T, Mundy PJ, Sebata A, MacFadyen DN. 2018. Short duration overnight cattle kraaling in natural rangelands: implications for grass composition, quality, above ground biomass, species diversity and basal cover. *Agriculture, Ecosystems & Environment* 257: 144–151. https://doi.org/10.1016/j.agee.2018.02.004.

Huruba R, Ndlovu C, Mundy PJ, Sebata A, MacFadyen DN. 2022. Short duration overnight cattle kraaling in natural rangelands leads to increased tree damage by elephants. *Journal of Tropical Ecology* 38: 1–8. https://doi.org/10.1017/S0266467421000353.

Jamsranjav C, Reid RS, Fernandez-Gimenez ME, Tsevlee A, Yadamsuren B, Heiner M. 2018. Applying a dryland degradation framework for rangelands: the case of Mongolia. *Ecological Applications* 28: 622–642. https://doi.org/10.1002/eap.1684.

Kizza S, Totolo O, Perkins J, Areola O. 2010. Analysis of persistence soil nutrient status in abandoned cattle kraals in a semi arid area in Botswana. *Scientific Research and Essays* 5: 3613–3622. https://hdl.handle.net/20.500.11951/214.

Krätli S, Huelsebusch C, Brooks S, Kaufmann B. 2013. Pastoralism: A critical asset for food security under global climate change. *Animal Frontiers* 3: 42–50. https://doi.org/10.2527/af.2013-0007.

Manzano P, Burgas D, Cadahía L, Eronen JT, Fernández-Llamazares Á, Bencherif S, Holand Ø, Seitsonen O, Byambaa B, Fortelius M, et al. 2021. Toward a holistic understanding of pastoralism. *One Earth* 4: 651–665. https://doi.org/10.1016/j.oneear.2021.04.012.

Marschner H. 1995. *Mineral nutrition of higher plants*. London: Academic Press.

McManus J, Goets SA, Bond WJ, Henschel JR, Smuts B, Milton SJ. 2018. Effects of short-term intensive trampling on Karoo vegetation. *African Journal of Range & Forage Science* 35: 311–318. https://doi.org/10.2989/10220119.2018.1529706.

Meglioli PA, Aranibar JN, Villagra PE, Riveros CV. 2017. Spatial patterns of soil resources under different land use in *Prosopis* woodlands of the Monte desert. *Catena* 149: 86–97. https://doi.org/10.1016/j.catena.2016.09.002.

Muchiru AN, Western D, Reid RS. 2009. The impact of abandoned pastoral settlements on plant and nutrient succession in an African savanna ecosystem. *Journal of Arid Environments* 73: 322–331. https://doi.org/10.1016/j.jaridenv.2008.09.018.

Olkin I, Dahabreh IJ, Trikalinos TA. 2012. GOSH – a graphical display of study heterogeneity. *Research Synthesis Methods* 3: 214–223. https://doi.org/10.1002/jrsm.1053.

Osenberg CW, Sarnelle O, Cooper SD, Holt RD. 1999. Resolving ecological questions through meta-analysis: Goals, metrics, and models. *Ecology* 80: 1105–1117. https://doi.org/10.1890/0012-9658(1999)080[1105:REQTMA]2.0.CO;2.

Pestana LFA, Souza ALT, Tanaka MO, Labarque FM, Soares JAH; Pestana LFdA. 2020. Interactive effects between vegetation structure and soil fertility on tropical ground-dwelling arthropod assemblages. *Applied Soil Ecology* 155: 103624. https://doi.org/10.1016/j.apsoil.2020.103624.

Porensky LM, Veblen KE. 2015. Generation of ecosystem hotspots using short-term cattle corrals in an African savanna. *Rangeland Ecology & Management* 68: 131–141. https://doi.org/10.1016/j.rama.2015.01.002.

Porensky LM, Young TP. 2016. Development of edge effects around experimental ecosystem hotspots is affected by hotspot density and matrix type. *Landscape Ecology* 31: 1663–1680. https://doi.org/10.1007/s10980-016-0344-3.

Pribyl DW. 2010. A critical review of the conventional SOC to SOM conversion factor. *Geoderma* 156: 75–83. https://doi.org/10.1016/j.geoderma.2010.02.003.

R Core Team. 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing. http://www.R-project.org.

Riginos C, Porensky LM, Veblen KE, Odadi WO, Sensenig RL, Kimuyu D, Keesing F, Wilkerson ML, Young TP. 2012. Lessons on the relationship between livestock husbandry and biodiversity from the Kenya Long-term Exclosure Experiment (KLEE). *Pastoralism: Research, Policy and Practice* 2: 10. https://doi.org/10.1186/2041-7136-2-10.

Riley R, Higgins J, Deeks J. 2011. Interpretation of random effects meta-analyses. *British Medical Journal* 342: d549. https://doi.org/10.1136/bmj.d549.

Saatkamp A, Henry F, Dutoit T. 2021. Romans shape today's vegetation and soils: Two millennia of land-use legacy dynamics in Mediterranean grasslands. *Ecosystems* 24: 1268–1280. https://doi.org/10.1007/s10021-020-00581-w.

Shamrikova EV, Kondratenok BM, Tumanova EA, Vanchikova EV, Lapteva EM, Zonova TV, Lu-Lyan-Min EI, Davydova AP, Libohova Z, Suvannang N. 2022. Transferability between soil organic matter measurement methods for database harmonization. *Geoderma* 412: 115547. https://doi.org/10.1016/j.geoderma.2021.115547.

Shezi TA, O'Connor TG, Witkowski ETF. 2021. Impact of livestock grazing intensity on plant diversity of montane grassland in the northern Drakensberg, South Africa. *African Journal of Range & Forage Science* 38: 67–79. https://doi.org/10.2989/10220119.2020.1837956.

Sibanda P, Sebata A, Mufandaedza E, Mawanza M. 2016. Effect of short-duration overnight cattle kraaling on grass production in a southern African savanna. *African Journal of Range & Forage Science* 33: 217–223. https://doi.org/10.2989/10220119.2016.1243580.

Sterne J, Egger M. 2006. Regression methods to detect publication and other bias in meta-analysis. In: Rothstein H, Sutton A, Borenstein M (eds.), *Publication bias in meta-analysis: Prevention, assessment and adjustment*. New Jersey: Wiley. pp 99–110.

Takeshima N, Sozu T, Tajika A, Ogawa Y, Hayasaka Y, Furukawa TA. 2014. Which is more generalizable, powerful and interpretable in meta-analyses, mean difference or standardized mean difference? *BMC Medical Research Methodology* 14: 30. https://doi.org/10.1186/1471-2288-14-30.

Tate KW, Dudley DM, McDougald NK, George MR. 2004. Effect of canopy and grazing on soil bulk density. *Rangeland Ecology & Management* 57: 411–417. https://doi.org/10.2111/1551-5028(2004)057[0411:EOCAGO]2.0.CO;2.

Thoresen J, Vermeire M-L, Venter Z, Wolfaard G, Krumins JA, Cramer M, Hawkins H-J. 2021. Fire and herbivory shape soil arthropod communities through habitat heterogeneity and nutrient cycling in savannas. *Global Ecology and Conservation* 25: e01413. https://doi.org/10.1016/j.gecco.2020.e01413.

UN. 2019. General Assembly resolution 73/284, United Nations Decade on Restoration (2021–2030): A/RES/73/284. Report no. 73. Geneva: United Nations.

UNCCD. 2008. *Human rights and desertification*. Germany: Secretariat of the United Nations Convention to Combat Desertification.

van der Waal C, Kool A, Meijer SS, Kohi E, Heitkönig IMA, de Boer WF, van Langevelde F, Grant RC, Peel MJS, Slotow R, et al.. 2011. Large herbivores may alter vegetation structure of semi-arid savannas through soil nutrient mediation. *Oecologia* 165: 1095–1107. https://doi.org/10.1007/s00442-010-1899-3.

van Rooyen J. 2016. Livestock production and animal health management systems in communal farming areas at the wildlife-livestock interface in southern Africa. PhD thesis, University of Pretoria, South Africa. http://hdl.handle.net/2263/60128.

Veblen KE. 2012. Savanna glade hotspots: plant community development and synergy with large herbivores. *Journal of Arid Environments* 78: 119–127. https://doi.org/10.1016/j.jaridenv.2011.10.016.

Vermeire ML, Thoresen J, Lennard K, Vikram S, Kirkman K, Swemmer AM, Te Beest M, Siebert F, Gordijn P, Venter Z, et al. 2021. Fire and herbivory drive fungal and bacterial communities through distinct above- and belowground mechanisms. *Science of the Total Environment* 785: 147189. https://doi.org/10.1016/j.scitotenv.2021.147189.

Vevea JL, Hedges LV. 1995. A general linear model for estimating effect size in the presence of publication bias. *Psychometrika* 60: 419–435. https://doi.org/10.1007/BF02294384.

Viechtbauer W. 2010. Conducting meta-analyses in R with the metafor package. *Journal of Statistical Software* 36: 1–48. https://doi.org/10.18637/jss.v036.i03.

Vinograd A, Zaady E, Kigel J. 2019. Dynamics of soil nutrients in abandoned sheep corrals in semi-arid Mediterranean planted forests under grazing. *Journal of Arid Environments* 164: 38–45. https://doi.org/10.1016/j.jaridenv.2019.02.007.

Vinograd A, Zaady E, Kigel J. 2020. Abandoned corrals: colonization and vegetation recovery of ephemeral habitats in silvo-pastoral systems. *Journal of Plant Ecology* 13: 722–731. https://doi.org/10.1093/jpe/rtaa062.