NATURE'S *letters*



Issue 3: April 2026



Journal's focus and scope

Nature's Letters is a multi-disciplinary journal aimed at documenting, describing and critically reflecting on applied ecological and/or social (e.g. historical) research topics that contribute to understanding and conserving Nature's Valley and its surrounds.

The journal is published online and in print format by the NVT TSITSIKAMMA, a not-for-profit company that focuses on conservation, conservation education, research and community engagement in the Western Cape, South Africa.

Manuscripts for review can be submitted by anyone working on or who has worked on any social and/or ecological topics in and around Nature's Valley within the past five years.

Guidelines for submissions are available at https://nvt.org.za/natures-letters/.

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FROM THE EDITOR'S DESK

Last year, NVT crossed a threshold that few organisations in our field ever reach: twenty-five years of dedicated conservation. This milestone was not simply a celebration of longevity, but a testament to resilience, vision, and the collective will of countless individuals who have carried the mission forward. It was a year of profound achievement: the launch of the KuCoNa Create and Connect Centre in Kurland, designed to weave the social fabric of conservation into our daily practice; the publication of *Nature's Valley: History and Conservation Efforts*, a book reminding us that today's ecological challenges are deeply rooted in the past; and the strengthening of our scientific capacity with the appointment of three PhD-level staff, alongside the mentorship of nineteen interns and students.

These achievements underscore a truth we hold dear: conservation is not static. It is a living, evolving endeavour, shaped by history, science, and people. Exotic plantations introduced more than a century ago to relieve pressure on indigenous forests now pose pressing challenges to fynbos and catchment systems, threatening biodiversity and water security. Such legacies remind us that conservation must be both retrospective and forward-looking - learning from the past while shaping a sustainable future.

When I initiated *Nature's Letters* two years ago, it was with a simple aim: to share our research with the Nature's Valley community, partners, and stakeholders, and to strengthen communication in a landscape where our work is sometimes invisible. I could not have foreseen how swiftly the journal would grow - attracting researchers, expert reviewers, and an ever-widening readership. With this third issue, we add to more than twenty scientific outputs produced in 2025 alone, each one a building block in the architecture of knowledge that sustains our mission.

Since my last letter, our team - together with students and partners - has continued to explore the Valley's biodiversity across ecosystems. This issue features studies on baboons, black stinkwood, dung beetles, and leopards, three of which represent entirely new research frontiers for NVT. Each study is a fragment of a larger picture, helping us piece together the intricate puzzle of life in this extraordinary landscape.

Among these, the Nature's Valley Baboon Project stands out. Now entering its third year, and led with dedication by Drs Kris Sabbi and Margaret Bryer, the project builds on earlier work published in our inaugural issue. New insights into behaviour and foraging patterns are not only advancing science but also guiding us toward a long-term, evidence-based solution for human-baboon coexistence - a challenge emblematic of the delicate balance between people and nature.

For the first time, this issue also embraces the social-ecological dimensions of conservation. Contributions examine a new community-based conservation model underpinning NVT's Behavioural Change Programme; unpack the Drivers, Pressures, State, Impacts, and Responses (DPSIR) shaping Plettenberg Bay's marine and coastal systems; and explore the carrying capacity of protected areas such as Robberg Marine Protected Area. The latter, submitted by the Transdisciplinary International Learning Lab (TILLie) students from South Africa, Germany, and Canada, exemplifies the power of collaborative, transdisciplinary work.

These studies remind us that conservation is not only about species and habitats. It is equally about people - how they think, feel, and behave toward nature. At its core, conservation is about safeguarding ecosystems from the intended and unintended consequences of human action. This holistic, systems-based approach is the foundation of NVT's work, and it is increasingly clear that resilience resides as much in social systems as in ecological ones.

Together, the studies in this issue illuminate the profound interconnectedness of species, habitats, and people. They affirm that effective conservation must embrace complexity, and that rigorous research is indispensable to understanding and protecting what we hold dear. For NVT, the path forward is clear: to strengthen the foundations of conservation in Nature's Valley and its surrounding landscapes, ensuring their vitality for generations to come.

The KuCoNa Centre, now fully operational, is already generating valuable insights into community engagement and community-based conservation practice. Future issues of *Nature's Letters* will increasingly highlight work emerging from the Centre, in collaboration with our partners, and will likely showcase contributions in health and education alongside environmental conservation.

I extend my deepest gratitude to the researchers, our editorial team, and external reviewers whose expertise and commitment have brought this third issue to life. Their work is not only scientific but profoundly human - an offering to the Valley, its people, and its future.

I hope you find this issue both enlightening and inspiring. I warmly invite your feedback, insights, and suggestions as we continue to grow *Nature's Letters* into a strong regional platform for researchers and practitioners.

Sincerely,



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SCHOLARLY PAPERS

To follow are nine papers submitted by former interns, staff and research partners. They explore a diverse range of topics, including baboons, black stinkwoods, dung beetles, leopards and social-ecological topics.

Together, these papers contribute to our ever increasing knowledge base related to nearly every habitat found in the Valley - estuaries, forests, fynbos and rivers, along with our previously unexplored species and topics.

To pique your interest, here's a brief overview of what you can expect to learn from each paper, organised by subject:

Baboons

1. Gabrielle Ralphe, a former NVT intern from the University of Wisconsin-Madison, together with colleagues, examined grooming equitability among female chacma baboons (*Papio ursinus*) in Nature's Valley, following a rare group fusion. Grooming equitability measures the balance of grooming exchanged within dyads, offering insight into the stability and reciprocity of social bonds. The study compared pre-fusion data (2023) with post-fusion observations (November 2024 - June 2025), analysing 410 grooming bouts across nine female dyads. Results showed no statistically significant changes in grooming equitability at the group level, suggesting overall stability in female social bonds despite the disruption of fusion. However, individual dyads varied considerably: some became more reciprocal, while others declined sharply. The findings highlight that while group-level grooming equitability remained stable, individual females adjusted their grooming strategies in response to shifts in rank, aggression, and integration into a new hierarchy. Some females may have redirected grooming toward new partners or concentrated effort into fewer but stronger bonds as a coping mechanism against stress. The study underscores the resilience and adaptability of female baboon social networks, showing that established bonds can buffer disruption and help facilitate integration during fusion events. From a conservation perspective, these results illustrate how primates can maintain social cohesion under demographic and ecological pressures, reinforcing the importance of social bonds in enabling survival and persistence in changing environments.
2. Yukun (Sarah) Zhang, also a former NVT intern from Boston University, together with colleagues, investigated seasonal food availability for chacma baboons using phenology transects and iNaturalist records. The transect data collected in winter (June - August 2025) showed that overall food availability remained relatively stable, with leaves (both young and mature) providing the bulk of resources, while flowers and fruits contributed less. Certain species such as black witch hazel (*Trichocladus crinitus*) and black ironwood (*Olea capensis macrocarpa*) consistently offered high food availability, whereas species like Cape beech (*Rapanea melanophloeos*) and hard pear (*Olinia ventosa*) contributed little due to low density. The iNaturalist dataset revealed clear seasonal variation: leaf flush was concentrated in warmer months

for most species, though evergreen species like assegai (*Curtisia dentata*) and forest elder (*Nuxia floribunda*) maintained leaves year-round. Flowering and fruiting showed distinct peaks, with most species producing ripe fruit in late summer to autumn, while a few species such as num-num (*Carissa bispinosa*) and candlewood (*Pterocelastrus tricuspidatus)* provided fruit over extended periods. Importantly, ripe fruits may be quickly consumed by birds and other frugivores, limiting baboon accessing more than expected during peak fruiting times. From a conservation perspective, this seasonal fluctuation in food supply has implications for baboon behaviour, as reduced food availability may drive them toward Nature's Valley village and other human settlements like the Rest Camp in search of alternative foods, increasing the risk of human-wildlife conflict. Long-term monitoring of food availability, integrating both systematic phenology and citizen-science data, will be essential for predicting baboon movements and designing proactive conservation strategies that balance ecological needs with conflict mitigation.

3. Isabelle Anderson, another former NVT intern from the University of Wisconsin-Madison, together with colleagues, explored baboon feeding ecology across forest, fynbos, and human settlement habitats. After nearly 250 hours of observation, the team found that baboons devote almost half of their feeding time to indigenous plant species, while spending far less time on invasive alien plants (9.84%) and non-invasive alien plants (0.80%). Human-related food accounted for less than 2% of their diet. Importantly, the baboons targeted key indigenous species such as black witch hazel (*Trichocladus crinitus*), forest grape (*Rhoicissus tomentosa*), and assegai (*Curtisia dentata*), and showed seasonal variation in feeding patterns. While baboons are often viewed negatively, this study highlights their ecological importance: they may disperse seeds of indigenous plants, thereby supporting forest regeneration, and appear to play only a limited role in spreading alien invasives. Nature's Valley looks the way it does in part thanks to its baboons, whose foraging behaviour likely contributes directly to the resilience and restoration of Afrotemperate forest ecosystems.

Black stinkwoods

4. Dr Rudi Swart, NVT's Forest Programme Manager and a Research Associate at Nelson Mandela University, assessed the population health of black stinkwood (*Ocotea bullata*) in Hebron Private Nature Reserve. A total of 314 individuals were surveyed, revealing a broadly inverse J-shaped size-class distribution (many saplings and fewer large trees), though seedlings were notably scarce. Mature trees were relatively small compared to reference forests, with maximum heights of 22m and diameters under 29cm, suggesting historical timber extraction may have removed larger individuals. Health assessments showed that most trees were healthy to very healthy, but intermediate size classes (large saplings and small mature trees, 4 - 9.9m) were more moisture-deficient than either smaller saplings or larger mature trees. This pattern may reflect physiological trade-offs, where intermediate trees invest more in height growth than root development, leaving them vulnerable to drought stress. Despite these stresses, the population appears stable, with demographic patterns consistent with long-term persistence. The study highlights that while legacy effects of past harvesting remain evident in reduced maximum tree sizes, the population of *O. bullata* in HPNR is healthy and resilient. The scarcity of seedlings, however, points to potential recruitment challenges that warrant monitoring.

From a conservation perspective, protecting existing individuals, ensuring habitat integrity, and monitoring moisture stress will be critical for sustaining this endangered species.

Community-based conservation

5. Dr Sandra du Plessis (NVT's Behavioural Change Programme Manager and Research Associate at Stellenbosch University), myself, and Prof Werner Nell of North-West University, introduce the "*Community Conservation Cycle-of-Praxis*" (version 1.0) - a new model designed to strengthen the social outcomes of community-based conservation (CBC). Developed in the communities of Kurland and Covie, the model responds to common shortcomings in CBC initiatives, particularly weak social processes that undermine conservation effectiveness. The model integrates community well-being frameworks, socio-psychological theories, and conservation practice into an eight-step iterative cycle. These steps include identifying community ideals, challenges and barriers, community assets, developing solutions, creating pathways and roadmaps for change, implementing pragmatic actions, and monitoring progress. By prioritising community needs and leveraging locally held assets, the model aims to foster behavioural change, strengthen social foundations, and support community-driven transformation. Importantly, while version 1.0 focuses on social outcomes, environmental considerations are incorporated through stewardship-oriented actions, with a planned version 2.0 to explicitly address ecological outcomes.

   Applied in Kurland and Covie, the model revealed community ideals ranging from food security and skills development to safety, healthcare, and economic opportunities (in other words, exactly the aspects we are focusing on at the KuCoNa Centre). This model demonstrates that effective CBC must go beyond ecological goals to address the social realities of communities. By embedding conservation within frameworks of community wellness, equity, and behavioural change, the cycle-of-praxis strengthens local stewardship and long-term sustainability. It provides a practical roadmap for NVT and practitioners to design interventions that improve livelihoods, reduce conflict, and build the social conditions necessary for conservation success.

Dung beetles

6. Carla van Hasselt, a former NVT intern from the University of Glasgow, with Prof Francois Roets of Stellenbosch University and Dr Rudi Swart provide the first pilot assessment of dung beetle assemblages in the Nature's Valley region, comparing communities across burnt fynbos (≈2 years post-fire), dense fynbos (≈26 years), and Afrotemperate forest habitats. Using baited pitfall traps, the study recorded 9 species and 640 individuals. Results showed that abundance differed significantly among habitats, with dense fynbos supporting far higher numbers of dung beetles than burnt fynbos or forest. In contrast, species richness and diversity did not differ significantly, indicating that similar numbers of species were present across habitats but with different dominance structures. Community composition analyses revealed that forest assemblages were distinct from both fynbos sites, while burnt and dense fynbos showed partial overlap. Certain species displayed clear habitat associations: *Onthophagus deterrens* and *Sisyphus muricatus* were restricted to dense fynbos, *Odontoloma pygidiale* and *Onthophagus*

hyaena occurred only in burnt fynbos, and *Bohepilissus subtilis* was found exclusively in forest. From a conservation perspective, these findings highlight that vegetation age and structure strongly influence dung beetle abundance and community composition, with mature fynbos providing more favourable conditions for dung beetle activity. Forest habitats, while supporting fewer individuals, hosted distinct assemblages, underscoring their ecological uniqueness. Dung beetles, as key ecosystem engineers, can therefore serve as valuable indicators of habitat condition and post-fire recovery, linking fire ecology, vegetation succession, and insect-mediated ecosystem services. This baseline contributes to biodiversity monitoring and informs fire-management strategies in the Cape Floristic Region, where prescribed burning and habitat restoration are central to conservation planning.

Leopards

7. Naomi Talbot-Ponsonby, a former NVT intern from the University of Exeter, and Dr Rudi Swart, provide a preliminary assessment of leopard (*Panthera pardus*) presence and distribution in Nature's Valley, using a combination of camera traps and citizen science records. Camera traps captured only one leopard photograph during the study period, but community-reported and verified signs (tracks, scat, sightings, and one kill site) expanded spatial coverage, yielding 29 records between 2020 and 2025. Density mapping revealed a core activity area around Kalanderkloof, while additional dispersed records across the wider landscape suggested a central high-use zone embedded within a broader mosaic of movements. A nearest-neighbour analysis indicated no significant departure from a random spatial distribution. Importantly, leopard signs were found near human-use areas, including within Nature's Valley village itself, highlighting both persistence in human-modified landscapes and potential biases in citizen science data toward areas frequented by people. The study confirms that leopards persist in the Tsitsikamma region despite low densities and high human pressures. Their presence across forest, fynbos, plantations, farmland, and even village boundaries underscores the importance of maintaining habitat connectivity and prey populations. Integrating systematic surveys with citizen science offers a cost-effective way to monitor leopard activity, expand spatial coverage, and foster local stewardship. However, future work should deploy larger camera-trap networks to identify movement corridors and reduce biases inherent in human-reported data. From a conservation perspective, these findings reinforce the leopard's role as a flagship predator and emphasise the need for proactive conservation strategies to ensure coexistence in mosaic landscapes.

Marine and coastal Drivers, Pressures, State, Impacts and Responses (DPSIR) in Plettenberg Bay

8. Dr Sandra du Plessis, Dr Alejandra Vargas (NVT's Marine & Coastal Programme Manager and a Research Associate at Nelson Mandela University), and I applied the DPSIR framework to explore the marine and coastal-related socio-ecological dynamics of Plettenberg Bay, South Africa. The analysis categorises user groups into extractive (fishers, bait collectors, marine harvesters) and non-extractive (beach and water users, land-based nature enthusiasts, and

commercial boat operators). Key drivers include economic necessity, tourism growth, and cultural traditions, which generate pressures such as overfishing, pollution, habitat disturbance, and noise impacts. Estuarine systems show varying states of health, ranging from urbanisation-driven eutrophication to relatively pristine but vulnerable conditions. The impacts include biodiversity loss, declining fish stocks, reduced ecosystem resilience, and risks to tourism and local livelihoods. The study identifies responses such as stricter regulations, habitat restoration, adaptive management, community co-management, and innovative tourism models. From a conservation perspective, the findings highlight the interconnectedness of ecological and socio-economic systems in Plettenberg Bay. Effective conservation requires integrated governance strategies that balance biodiversity protection with sustainable livelihoods.

Social and physical carrying capacity for protected areas

9. The final paper in this issue by Dr César Soares from the Global Institute for Water Security at the University of Saskatchewan and his colleagues, introduces a visitor-based tool for assessing social and physical carrying capacity in protected areas, piloted at Robberg Nature Reserve in South Africa. The tool was developed through literature review, expert consultation, stakeholder engagement, and pilot testing with 134 visitors, alongside field audits. Validation confirmed a five-factor structure: parking/restrooms, crowding, trail safety, width/maintenance, and signage/seating - with acceptable reliability and coherence. External validity analyses showed that perceptions varied predictably with visitor origin and visit frequency, aligning with theoretical expectations: local visitors rated parking more positively than international visitors, and first-time visitors reported lower perceived crowding than repeat visitors. The findings demonstrate that the tool reliably captures visitor perceptions of both social carrying capacity (crowding, safety, visitor interactions) and physical carrying capacity (infrastructure, facilities, trail conditions). Results highlight areas of strength, such as well-maintained trails and positive visitor experiences, as well as areas needing attention, including restroom availability and signage clarity. By offering a concise, evidence-based dashboard, the tool enables managers to detect thresholds of crowding or infrastructure strain and intervene proactively. This visitor-based tool provides a scalable, replicable method for monitoring tourism impacts in protected areas, particularly in under-resourced contexts. By integrating visitor perceptions with infrastructure audits, managers can balance ecological integrity with visitor satisfaction. Applied more broadly, the tool can strengthen adaptive management across South Africa's protected areas, helping safeguard biodiversity while sustaining tourism benefits.

Grooming equitability in female chacma baboons (*Papio ursinus*) following group fusion in Nature's Valley, South Africa

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How to cite this article:

Ralphe, G., Kurilla, S.E., Bryer, M.A.H., van Hasselt, C., Coetzee, H., Swart, R.C. & Sabbi, K.H. 2026. Grooming equitability in female chacma baboons (*Papio ursinus*) following group fusion in Nature's Valley, South Africa. *Nature's Letters* **3**:6-14. URL: https://nvt.org.za/natures-letters/

Abstract

Social bonds among philopatric female primates are central to social stability as well as individual fitness. Grooming equitability measures the balance and mutual investment within these relationships and offers insight into their resilience under social disruption or environmental pressures. This study examined changes in grooming equitability among adult female chacma baboons (*Papio ursinus*) following a rare group fusion event in Nature's Valley, South Africa, in October of 2024. We compared Grooming Equitability Index scores among five original adult females across nine dyads before fusion and during three post-fusion time windows. Although individual dyads showed marked increases or decreases in reciprocity, no statistically significant group-level changes in grooming equitability were detected. This suggests that despite individual-level restructuring and social instability associated with integration, overall reciprocity among original females remained stable. The results indicate that female bonds may function as a stabilising mechanism during major demographic disruption, highlighting the resilience and flexibility of primate social networks under rare fusion events.

Introduction

In many primate species, particularly those with female philopatry (in which females remain in their natal groups with close matrilineal kin), establishing long-term social bonds along matrilineal lines is critical (Le Roux *et al*., 2011). These bonds not only play a central role in maintaining group cohesion, but also promote individual fitness (Wittig *et al*., 2008) via increased infant survival, enhanced longevity, and greater lifetime reproductive output (Silk *et al*., 2003; Campos *et al*., 2020). For these reasons, bonds between females are often considered a central organising principle in female philopatric primate social systems, shaping both immediate group dynamics and long-term evolutionary trajectories.

Social grooming is a well-studied mechanism that reflects the quality of primate social bonds across primates and plays a central role in both building alliances and reconciling relationships after internal conflict, processes that are critical for individual fitness and group-level persistence (Sambrook *et al.*, 1995). In addition, grooming provides critical health benefits (Barrett *et al*., 1999). Beyond removing ectoparasites, grooming lowers stress hormones, like cortisol, and thus functions as a tool for stress reduction, a process linked to stronger immune function and increased longevity

(Wittig *et al*., 2008). Grooming relationships are therefore invaluable for learning about individual primates and the bonds they have with their conspecifics.

Grooming is easily observed, recorded, and quantified in wild primates, allowing researchers to describe the frequency and duration of grooming bouts of different dyads (Silk *et al*., 2013). While frequency and duration of grooming interactions are useful, grooming equitability, defined as the balance of grooming exchanged within dyads, provides a clearer picture of fairness and mutual investment in maintaining a relationship by measuring the reciprocity of a grooming relationship (Silk *et al*., 2010; Barrett *et al*., 2002). Equitable relationships are more stable over time and more predictive of fitness outcomes than one-sided interactions (Silk *et al*., 2010). Grooming equitability has been particularly useful in assessing resilience under conditions of stress or disruptions, such as increased competition and reduced resources (Barret *et al*., 2002). In this sense, measurement of equitability offers a window into the stability of social relationships in the midst of demographic or environmental stress.

Chacma baboons (*Papio ursinus*) provide an ideal system for examining the resilience of female social bonds. They typically live in multi-male, multi-female groups ranging from 20 to over 80 individuals, with males dispersing at sexual maturity and females remaining in their natal groups to form life-long relationships with kin and nonkin conspecifics (Seyfarth, 1976; Barret *et al*., 2002). Like other female philopatric species, female ties, particularly between kin, form the backbone of group cohesion, while male immigration and often associated infanticide can destabilise social order (Campos *et al*., 2020). Although baboon home ranges may overlap in range with neighbouring groups, leading communities to experience intergroup encounters where more than one group spends time in either friendly or antagonistic association with another, fusions, where two groups join together more permanently, are rare (Altmann, 1980; Strum, 2019), suggesting primate group fusions occur only under unusual ecological or demographic pressures (Jaffe *et al*., 2010).

As traditionally female philopatric and matrilineal societies, such a fusion represents a dramatic form of social disruption that has only rarely been documented in primates despite decades of dedicated scientific observation. One study of Japanese macaques (*Macaca fuscata*) reported that two groups fused in response to habitat change (Sugiura *et al*., 2002). In vervet monkeys (*Chlorocebus pygerythrus*) two groups fused, following demographic collapse (Jaffe & Isbell, 2010). Finally, olive baboons (*Papio anubis*) fused in the face of either heightened predation or resource scarcity (Strum, 2023).

These events are assumed to be stressful and risky as females used to socialising exclusively with kin and natal group members integrate with unfamiliar females. At the same time, fusions provide a unique natural experiment to explore questions about the flexibility of primate social systems. How do long-standing affiliative bonds adapt to the sudden influx of new group members? Do they weaken, collapse, or instead provide stability in the face of disruption? Understanding responses to rare fusion events has broad implications for primate behavioural ecology, as social resilience is central to survival in changing environments. By testing the limits of bond stability, such events shed light on how primates adapt to habitat fragmentation, human disturbance, and demographic instability (Strum, 2023; Byrne *et al*., 1990).

Here we investigated grooming behaviour among adult female chacma baboons before and after a rare group fusion in Nature's Valley, South Africa. Grooming equitability is used as a proxy for the symmetry and strength of affiliative relationships (Silk *et al*., 2010; Barrett *et al*., 2002). By comparing equitability scores from the pre- and post-fusion periods, we examined whether original female bonds persisted, weakened, or were restructured following the integration of two previously separate groups. These findings contribute to our understanding of how affiliative relationships respond to large-scale social disruptions, providing insight into the flexibility and resilience of primate social networks.

Methods

This study was conducted in Nature's Valley, South Africa, where a habituated group of wild chacma baboons (n = 21) has been under continuous observation since 2023. The focal group underwent a rare group fusion event in October 2024, during which a small group (n = 8) consisting of five adult females and three juveniles merged with a neighbouring group containing one resident adult male, four adult females, two juveniles, and four infants (n = 11). Behavioural data were collected on all adult females from the original group (n = 5), spanning both the pre-fusion and post-fusion periods. During 10-minute focal follows, observers recorded detailed records of grooming interactions including the identity of grooming partners, direction of grooming, and duration of all grooming behaviours.

To determine the balance of grooming investment within each adult female dyad (n = 9), we calculated the Grooming Equitability Index following Silk *et al.* (2010). The index was calculated using the formula:

$$1 - abs[(Dij - Dji)/ (Dij + Dji)]$$

where Dij represents the grooming rate from individual i to individual j (measured as minutes spent grooming divided by the total hours the dyad was observed), and Dji represents the reverse. A value of 1 indicates perfectly reciprocal grooming, while values approaching 0 reflect grooming relationships that are entirely one-sided (Silk *et al*., 2010). These scores were calculated using grooming data post-fusion event and compared to pre-fusion scores previously calculated from 2023 to provide insight into how such an event may have affected female grooming relationships. Scores were calculated over three different time windows following the fusion event (November 2024; November 2024 to February 2025; November 2024 to June 2025) to evaluate how dyadic reciprocity may have changed in the short term versus the longer- term following the fusion event.

To assess whether grooming equitability changed following the group fusion, we used a Wilcoxon signed-rank test to compare pre- and post-fusion equitability scores. This non-parametric paired test was selected due to the non-normal distribution commonly associated with behavioural data and the matched structure of dyadic observations across time periods. The test evaluates whether the median difference in equitability scores before and after the fusion deviates significantly from zero, indicating whether grooming relationships became more or less reciprocal following the social reorganisation.

Results

From November 2024 to June 2025, we observed 410 grooming bouts where both groomer and receiver were identified as one of the original five adult females of the study

group. Grooming equitability scores were calculated for 9 adult female dyads as part of the post-fusion phase to be compared to equitability scores calculated prior to the fusion based on grooming data collected in 2023.

Across three time windows no statistically significant changes in grooming equitability were detected. In the immediate aftermath of the fusion (November 2024), the mean dyadic grooming equitability score decreased from 0.686 pre-fusion to 0.437 post-fusion ($V = 9$, $p = 0.8125$). Similarly, although equitability decreased from 0.686 to 0.420 in the short-term window (November 2024 to February 2025), this change was also not significant ($V=10$, $p =0.625$). In the longer-term window (November 2024 to June 2025), equitability scores remained nearly unchanged, with a pre-fusion average of 0.686 and a post-fusion average of 0.609 ($V = 29$, $p = 0.4961$). This longer-term stability is reflected in the distribution of dyadic equitability scores, which show wide variability but similar medians before and after fusion (**Figure 1**).

Despite a lack of significance, specific adult female dyads showed dramatic changes in equitability scores between phases, with some increasing and others decreasing in reciprocity (**Figure 2**). At the most extreme, AF4;AF5 had a decrease from a pre-fusion score of 0.816 to 0.012 (change of -0.803) while AF1;AF3 saw an increase from 0 to 0.848.

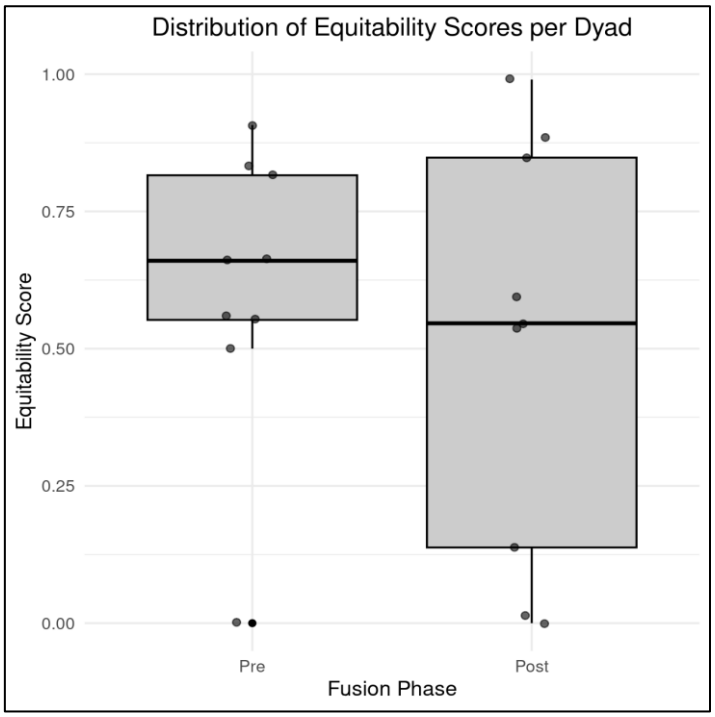


Figure 1: Distribution of grooming equitability scores for adult female dyads from pre- and post-fusion phases. Boxplots show the median, interquartile range, and overall spread of dyadic equitability scores, with points representing individual dyads. Pre-fusion scores were more tightly clustered at moderate-to-high reciprocity, whereas post-fusion scores showed greater variability, including both highly reciprocal and highly unbalanced dyads.

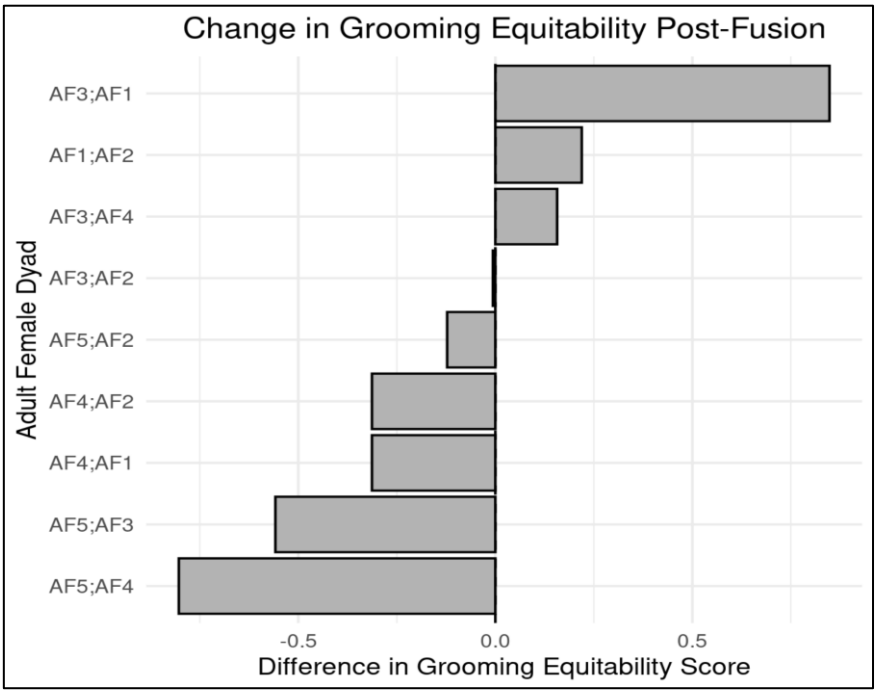


Figure 2: Change in grooming equitability scores for adult female dyads before and after group fusion (time period: November 2024–June 2025). Positive values indicate an increase in reciprocity post-fusion, while negative values indicate a decrease. Dyads are ordered by magnitude of change. Although individual dyads varied in the direction and extent of change, no consistent pattern emerged across the group.

Looking at specific individuals and their grooming relationships also revealed trends. Overall, AF5 was of specific interest as this individual experienced a period of isolation over the course of the fusion as a result of aggressive attacks by her original group members before eventually reintegrating. Prior to the fusion event, AF5 was the highest-ranking adult female of the original group. AF5's average equitability score pre-fusion was 0.678 while her average equitability score post-fusion was 0.493. However, the greatest decrease in average grooming equitability was AF4 who saw a decrease from 0.778 to 0.539. The individual with the largest increase in average grooming equitability scores was AF1 who went from an average score of 0.406 to 0.802 (**Figure 3**).

Discussion

In this study, we leveraged an extremely rare event - the joining of two sets of matrilineal chacma baboon hierarchies through group fusion - to examine how such a disruption may impact established social relationships between females. Specifically, we examined changes in grooming equitability among adult female chacma baboons compared to before the merger. While our results revealed no statistically significant post-fusion changes in grooming equitability among the original group members across three different time windows, individual baboons and specific dyads varied in whether and how much they changed across the course of the study. Overall, this pattern suggests group-level stability in the balance of grooming relationships before and after fusion, consistent with broader findings that grooming reciprocity reflects stable social bonds in female primates (Schino *et al*., 2008).

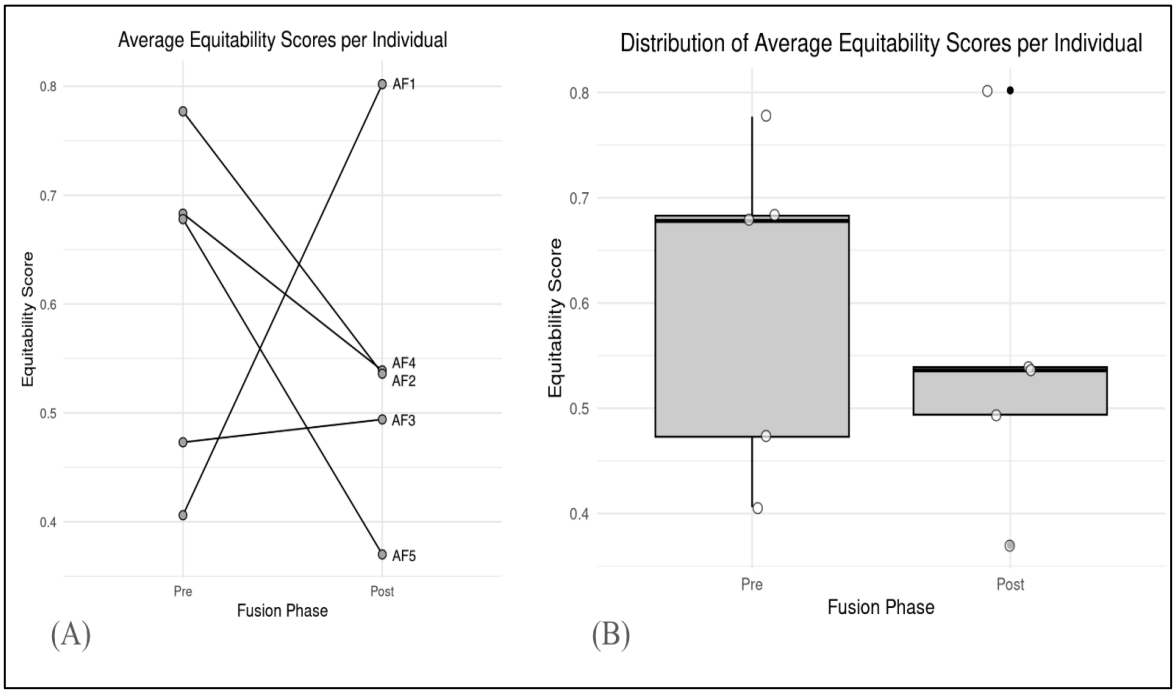


Figure 3: (**A**) Average grooming equitability scores for each adult female before and after the group fusion. Lines connect the same individual across phases to illustrate female change. (**B**) Distribution of average equitability scores across all adult females in the pre- and post-fusion periods, shown as boxplots. Overall, three of the five females experienced a decrease in average grooming equitability following the fusion, with the largest average decrease being -0.241 and largest average increase being 0.396.

Individual changes amongst dyads are nonetheless interesting and worth further exploration. One potential driver of a reorganisation of reciprocity could be the likely shifts of rank of the original females as they integrated into the other group's hierarchy. Amongst female philopatric primates such as baboons, females form strong linear hierarchies that affect access to resources, patterns of aggression, and the distribution of affiliative behaviours such as grooming reciprocity, especially as rank distance between females increases (Silk *et al*., 2006). The effects of fusion on female rank order is a topic for future investigations, especially considering the highest ranking individual pre-fusion was AF5, the same individual that faced severe aggression, a temporary group expulsion and one of the largest drops in mean grooming equitability scores. In the predator-induced group fusion of vervet monkeys on the Laikipia Plateau in Kenya, adult females of the smaller group assumed the lowest ranks in the newly fused group (Jaffe *et al*., 2010). Similarly, dominance instability, particularly for these five females in the chacma baboon study group, following the fusion may have disrupted previously balanced grooming relationships, as shifts in rank are known to reorganise coalition patterns and affiliative interactions in baboons (Bergman *et al*., 2003). AF5's decline in equitability may reflect both her social demotion and the flexibility of lower-ranking females adapting their grooming strategies within the restructured group.

Another factor to consider in the changes in reciprocity amongst dyads is the potential expansion of grooming networks to the new females of the post-fusion group. Female baboons are known to prioritise maternal kin in affiliative behaviours such as grooming but also coalitions and tolerance (Silk *et al*., 2010). Although the relatedness of the study

subjects at Nature's Valley is unknown, given the fact female baboons remain in their natal groups, their closest maternal and even paternal kin are most likely among the original adult females with the likelihood of kinship going down severely with the new females. That being said, it is possible that females who saw a decrease in reciprocity in some of their grooming relationships may have been shifting grooming investment towards these new females in an attempt to build new relationships and ease integration. Female primates have also been known to form relationships with nonkin in response to drastic social change (Kurihara *et al*., 2019). New females could be the gatekeepers of resources such as access to food as well as to their resident adult male as seen in female-dispersing species (Kahlenberg *et al*., 2008) and thus, investment of grooming effort towards these new social bonds could pay off in greater fitness benefits. If this is the case, this would represent resilience and strategic flexibility within an evolving social market, in which grooming functions as a tradable commodity shaped by supply and demand dynamics (Noë *et al*., 1995). Thus a decrease in equitability amongst original group members may be associated with greater fitness outcomes.

Decreasing grooming equitability amongst the original females may also be in part due to the addition of an adult male where there previously was none. The introduction of a reproductive male likely increased competition among females, and in female-resident species, heightened competition for access to males is associated with reduced grooming reciprocity among females (Hemelrijk *et al*., 1998). Within this changing social market, females may have shifted time spent maintaining their grooming network with fellow females towards the new adult male, particularly during times where a female is in oestrus, as grooming can function as a commodity exchanged for mating opportunities in primate social markets (Gumert, 2007). In this case females would be prioritising the potential offspring that could come from such copulations and thus clearly weighing the costs and benefits of their own fitness outcomes in the wake of a changing social landscape. Furthermore, the implications of pregnancy, lactation, and infant presence are also known variables that can and do disrupt grooming networks amongst female primates (Frank *et al*., 2009).

Given the shifts in grooming on the individual and dyadic level, another potential explanation of the results is that while grooming investment remained within the five adult females it redistributed from a relatively diffuse grooming network pre-fusion to a more concentrated one. As stated previously, social bonds are known to offer physiological benefits in the form of stress reduction. The degree of benefits is not correlated to the amount of partners, but rather the quality of such relationships. Females with fewer but stronger grooming partners are known to experience a lower spike in glucocorticoid levels in the face of social disruption than females with more diffuse networks and see their glucocorticoid levels return back to normal more quickly (Crockford *et al*., 2008; Wittig *et al*., 2008). Group fusions provide a lot of opportunities for stress: for example, an increase in aggression and a drop in rank as seen in vervet monkey group fusion, is most severe for females from the smaller group (Jaffe *et al*., 2010). An increase in glucocorticoid levels in response to threats is an important part of allostasis and adaptability, however chronic stress can be detrimental (Wittig *et al*., 2008). Females may have concentrated their grooming efforts into a smaller number of strong social bonds to combat such disruption.

Preliminary observations of male grooming patterns highlight how individual characteristics may influence affiliative

behaviour. One juvenile male from the original group who experienced a severe injury that led him to lose both a hand and foot, received substantially more grooming than he gave, suggesting that grooming may serve not only to reinforce social bonds but also to express tolerance, care, or protection toward vulnerable group members (Silk, 2007). Future work that includes males and juveniles could therefore provide a broader understanding of the social dimensions of grooming beyond adult female relationships.

It is important to note several limitations of this study. The small sample size and exclusion of dyads with incomplete grooming data reduced statistical power and may have masked more subtle patterns of change. Additionally, equitability captures only one dimension of relationship quality. Frequency, diversity, tenor, tension, predictability, and stability are also integral dimensions by which we can analyse social relationships (Silk *et al*., 2013) and may also offer meaningful insights into how females navigated the fusion. Future studies could incorporate these additional metrics and examine factors such as kinship, reproductive status, and dominance rank to better understand the drivers of these post-fusion social strategies. Another focus of future studies would be to examine the grooming relationships between the original females and those of the other group to gain a clearer picture of reciprocity and the formation of new affiliative bonds. This would provide insight into whether females are actively redistributing grooming effort to facilitate integration and establish stability within the newly fused group.

Despite these limitations, this study provides rare empirical insight into how established female bonds respond to major social disruption. Documented cases of group fusion in cercopithecoid primates are rare, with only a handful of natural events reported in the literature (e.g., Jaffe *et al*., 2010), making this an unusual opportunity to assess the flexibility of social networks under natural conditions. While group fusions likely carry substantial costs including intensified competition, dominance instability, and elevated stress responses associated with social upheaval (Jaffe *et al*., 2010; Sapolsky, 2005), fusions can also offer profound benefits such as greater predation protection, an increased access to resources through potential range expansion and reduced intragroup competition (Kurihara *et al*., 2019) and in this case, the critical opportunity to reproduce. These are consequential advantages for the fitness and persistence of these original adult females. The overall stability in grooming equitability observed here suggests that females may have relied on their existing social relationships as a coping strategy, buffering stress and preserving key relationships even amidst the upheaval of integration. In this light, the successful fusion documented here represents more than just a demographic shift but a striking example of resilience and adaptability. This capacity to sustain affiliative balance during social disruption not only makes fusion possible but also highlights the central role of social bonds in enabling primates to navigate and survive major ecological and demographic challenges.

References

Altmann. J. 1980. Baboon mothers and infants. Chicago: University of Chicago Press.

Barrett, L., Gaynor, D. & Henzi, S. P. 2002. A dynamic interaction between aggression and grooming reciprocity among female chacma baboons. Animal Behaviour, 63(6), 1047–1053.

Barrett, L., Henzi, S. P., Weingrill, T., Lycett, J. E. & Hill, R. A. 1999. Market forces predict grooming reciprocity in female baboons. Proceedings of the Royal Society of London. Series B: Biological Sciences, 266(1420), 665-670.

Bergman, T. J., Beehner, J. C., Cheney, D. L. & Seyfarth, R.

M. 2003. Hierarchical classification by rank and kinship in baboons. Science, 302(5648), 1234-1236.

Byrne, R. W., Whiten, A. & Henzi, S. P. 1990. Social relationships of mountain baboons: leadership and affiliation in a non-female-bonded monkey. American Journal of Primatology, 20(4), 313-329.

Campos, F. A., Villavicencio, F., Archie, E. A., Colchero, F. & Alberts, S. C. 2020. Social bonds, social status and survival in wild baboons: A tale of two sexes. Philosophical Transactions of the Royal Society B: Biological Sciences, 375(1811), 20190621.

Crockford, C., Wittig, R. M., Whitten, P. L., Seyfarth, R. M. & Cheney, D. L. 2008. Social stressors and coping mechanisms in wild female baboons (*Papio hamadryas ursinus*). Hormones and Behavior, 53(1), 254-265.

Frank, R. E. & Silk, J. B. (2009). Grooming exchange between mothers and non-mothers: the price of natal attraction in wild baboons (*Papio anubis*). Behaviour 146(7), 889-906.

Gumert, M. D. 2007. Payment for sex in a macaque mating market. Animal Behaviour, 74(6), 1655-1667.

Hemelrijk, C. K. & Luteijn, M. 1998. Philopatry, male presence and grooming reciprocation among female primates: a comparative perspective. Behavioral Ecology and Sociobiology, 42, 207-215.

Jaffe, K. E. & Isbell, L. A. 2010. Changes in ranging and agonistic behavior of vervet monkeys (*Cercopithecus aethiops*) after predator-induced group fusion. American Journal of Primatology: Official Journal of the American Society of Primatologists, 72(7), 634-644.

Kahlenberg, S. M., Thompson, M. E., Muller, M. N. & Wrangham, R. W. 2008. Immigration costs for female chimpanzees and male protection as an immigrant counterstrategy to intrasexual aggression. Animal behaviour, 76(5), 1497-1509.

Kurihara, Y., Nishikawa, M. & Mochida, K. 2019. Behavioral responses to changes in group size and composition: a case study on grooming behavior of female Japanese macaques (*Macaca fuscata yakui*). Behavioural Processes, 162, 142-146.

Le Roux, A., Beehner, J. C. & Bergman, T. J. 2011. Female philopatry and dominance patterns in wild geladas. American Journal of Primatology, 73(5), 422-430.

Noë, R. & Hammerstein, P. 1995. Biological markets. Trends in Ecology & Evolution, 10(8), 336-339.

Sambrook, T. D., Whiten, A. & Strum, S. C. (1995). Priority of access and grooming patterns of females in a large and a small group of olive baboons. Animal Behaviour, 50(6), 1667-1682.

Sapolsky, R. M. 2005. The influence of social hierarchy on primate health. Science, 308(5722), 648-652.

Schino, G. & Aureli, F. 2008. Grooming reciprocation among female primates: a meta-analysis. Biology letters, 4(1), 9-11.

Seyfarth, R. M. 1976. Social relationships among adult female baboons. Animal Behaviour, 24(4), 917-938.

Silk, J. B., Alberts, S. C. & Altmann, J. 2003. Social bonds of female baboons enhance infant survival. Science, 302(5648), 1231–1234.

Silk, J. B. 2007. The adaptive value of sociality in mammalian groups. Philosophical Transactions of the Royal Society B: Biological Sciences, 362(1480), 539-559.

Silk, J. B., Beehner, J. C., Bergman, T. J., Crockford, C., Engh, A. L., Moscovice, L. R., Wittig, R. M., Seyfarth, R. M. & Cheney, D. L. 2010. Female chacma baboons form strong, equitable, and enduring social bonds. Behavioral Ecology and Sociobiology, 64(11), 1733–1747.

Silk, J., Cheney, D. & Seyfarth, R. 2013. A practical guide to the study of social relationships. Evolutionary Anthropology: Issues, News, and Reviews, 22(5), 213-225.

Strum, S. C. 2019. Why natural history is important to (primate) science: a baboon case study. International Journal of Primatology, 40(6), 596-612.

Strum, S. C. 2023. Confessions of a baboon watcher: from inside to outside the paradigm. Primates, 64(4), 393-406.

Sugiura, H., Agetsuma, N. & Suzuki, S. (2002). Troop extinction and female fusion in wild Japanese macaques in Yakushima. International Journal of Primatology, 23, 69-84.

Wittig, R. M., Crockford, C., Lehmann, J., Whitten, P. L., Seyfarth, R. M. & Cheney, D. L. 2008. Focused grooming networks and stress alleviation in wild female baboons. Hormones and Behavior, 54(2), 170-177.

Seasonal food availability for chacma baboons (*Papio ursinus*) in Nature's Valley's forests, South Africa: Integrating transect phenology and iNaturalist data

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How to cite this article:

Zhang, Y., Swart, R.C., van Hasselt, C., Sabbi, K.H., Anderson, I. & Bryer, M.A.H. 2026. Seasonal food availability for chacma baboons (*Papio ursinus*) in Nature's Valley, South Africa: Integrating transect phenology and iNaturalist data. *Nature's Letters* **3**:15-33. URL: https://nvt.org.za/natures-letters/

Introduction

Food availability, defined as the presence and accessibility of essential plant resources such as leaves, flowers, and fruits, is a key factor influencing primate behaviour (Altmann & Muruthi, 1988). It is often assessed through plant phenology, which is the study of recurring biological events such as leaf flush, flowering, and fruiting, and how these events change over time or seasons (Tang *et al*., 2016). By recording the timing and intensity of these events, researchers can estimate when and where food resources are most abundant for primates. Phenological patterns therefore create substantial temporal and spatial variation in the availability of plant foods and consequently primate diets, with important implications for primate grouping, ranging, and social behaviour (Marshall & Wich, 2013).

Investigating food availability is crucial because it enables researchers to understand how primates adapt to seasonal or other changes in food resource distribution. Phenology provides clues to potential periods of nutritional stress for primates and can provide valuable insights for wildlife management. This knowledge is especially important in regions where humans' and baboons' habitats overlap, as it can help mitigate conflict (Fehlmann *et al*., 2017).

Chacma baboons (*Papio ursinus*) are an excellent species with which to examine these dynamics because they exhibit remarkable dietary flexibility and behavioural adaptability across diverse environments in southern Africa (Hill & Dunbar, 2002). Their feeding and ranging patterns are influenced by seasonal fluctuations in food availability, often leading them to adjust travel distances, alter activity budgets, or exploit anthropogenic food sources during periods of scarcity (Barton *et al.*, 1992). Therefore, linking phenological data to baboon foraging behaviour provides valuable insight into how environmental seasonality shapes primate ecology and human-wildlife interactions. To explore this relationship within the local context of Nature's Valley, South Africa, this study integrates plant phenology and citizen science data to assess baboon food availability over time.

The objective of this study was to assess food availability by integrating phenology data collected from one forest transect between June and August 2025 (to calculate food availability index per plant food part, FAI), complemented by year-round iNaturalist data to construct an annual pattern of baboon food availability. By combining transect and iNaturalist data, this study aims to determine food availability during the winter season (transect) and predict seasonal variations in leaf flush, mature leaves, flowers, unripe fruits, and ripe fruits throughout the year (iNaturalist). In the future. these findings can be combined with baboon behavioural and ranging data to better understand how these fluctuations may influence baboon activity and movement. This approach will help anticipate periods when baboons are more likely to enter human spaces, providing valuable insights for managing human-wildlife interactions and reducing potential conflicts.

Methods

Phenology transect

Transect phenology data were collected along a phenology transect located in Nature's Valley, South Africa. The survey area follows a tourist boardwalk within the Tsitsikamma section of the Garden Route National Park, Western Cape, and lies within the eastern portion of the Cape Floristic Region. Unlike the winter-rainfall regime characteristic of western parts of the Western Cape, this region experiences year-round rainfall with warm, productive summers and cooler, wetter winters (Mucina *et al*., 2006). The climate supports Afrotemperate forest that responds strongly to seasonal temperature changes, with four broad seasonal periods: winter (June–August), spring (September–November), summer (December–February), and autumn (March–May). The tree species included in the study (i.e., along the transect) were chosen because the Nature's Valley Baboon Project (NVBP) team has observed baboons consuming parts (e.g., fruits, leaves, flowers) of these tree species during ongoing behavioural data collection that began in 2023. Phenological data were collected along the transect by Y. Zhang twice per month from June-August 2025.

The transect was originally established in June 2025 by C. van Hasselt, Y. Zhang, I. Anderson, and M. Bryer (**Figure 1**) and the location was chosen for its accessibility and representative forest habitat. It was measured in segments using a 30-meter tape, resulting in a total length of 336 meters and a width of 8 meters (2688 m^2). Each baboon food tree was identified based on trunk and leaf characteristics, including colour, shape, and texture, and all trees were labelled with flagging tape on which was written the tree species and the number of the tree on the transect. The transect included the following tree species: Wild pomegranate (*Burchellia bubalina*), n = 1; assegai (*Curtisia dentata*), n = 2; white forest spikethorn (*Gymnosporia nemorosa*), n = 2; black witch hazel (*Trichocladus crinitus*), n = 2; forest saffron (*Elaeodendron croceum*), n = 5; forest elder (*Nuxia floribunda*), n = 1; Cape beech (*Rapanea melanophloeos*), n = 1; Outeniqua yellowwood (*Afrocarpus falcatus*), n = 1; hard pear (*Olinia ventosa*), n = 1; kamassi (*Gonioma kamassi*), n = 4; num-num (*Carissa bispinosa*), n = 2; candlewood (*Pterocelastrus tricuspidatus*), n = 1; and ironwood (*Olea capensis macrocarpa*), n = 2.





Figure 1: Nature's Valley Baboon Project team members (left to right) Isabelle Anderson (University of Wisconsin-Madison, USA), Yukun Zhang (Boston University, USA) Carla van Hasselt (University of Glasgow, UK) set up a transect and collect the first round of phenological data on baboon feeding trees. Photo by M. Bryer.

Researchers recorded the percentage cover of each baboon food component (young leaves, mature leaves, unripe fruit, and ripe fruit) for the selected and tagged baboon food trees along the transect using a 0-4 scale (0 = 0%, 1 = 1–25%, 2 = 26–50%, 3 = 51–75%, 4 = 76–100%) following the methods of Chapman *et al*. (1994, 2018). The target plant part was visually estimated as the proportion of all branches producing that part relative to the whole tree. This proportion in percentage was then converted into the 0-4 scale. A score of 0 indicated that the target plant part was not observed on any branch, whereas higher scores corresponded to increasing coverage across the tree's branches. For trees producing only a minimal number of flowers or fruits, the score consistently remained at 1, regardless of tree size.

Tree plant parts for phenological scoring were observed using binoculars. Young leaves (also referred to as leaf flush) were identified as newly emerged leaves, typically red, light green, or yellow depending on the species, whereas mature leaves were darker green and fully expanded. Flowers and fruits were identified based on clear visual cues such as colour, size, and texture, which also allowed distinction between ripe and unripe fruits. Plant species and their parts were identified and characterised using help from NVT researchers and using the book *Trees of the Garden Route: Mossel Bay to Storms River* (Venter, 2009) as a reference. This guide provided detailed descriptions of each species' leaf, flower, and fruit morphology, including colour, shape, and texture, which helped standardise identification and scoring.

Tree diameter at breast height (DBH) was measured during the first phenology survey when the transect was established in June 2025. The research team measured DBH by placing a standard measuring tape across the trunk at approximately 1.3 meters above the ground (breast height) and reading the width directly in meters. If the tree had multiple trunks, DBH was measured for each of its trunks and recorded. In temperate forests, where annual radial growth is relatively slow, DBH does not need to be measured during every phenology survey, as a single measurement can reliably represent tree size over short study periods (D'Orangeville *et al*., 2021).

FAI calculations

The food availability index calculation followed the method of Chapman *et al*. (1994).

Young leaves, mature leaves, flowers, unripe fruits, and ripe fruits were calculated separately for each availability index then combined to have one total FAI. Calculating FAI requires integrating information on individual tree size, species density, and phenology scores.

For each tree species along the transect, the mean basal area (BA) was calculated from diameter at breast height (DBH) measurements using:

$$\text{Basal area} = \pi \times \left(\frac{\text{DBH}}{2}\right)^2$$

If a tree had multiple trunks, the basal areas of all trunks were summed to represent the total basal area of that tree. The basal area is used as a proxy for the relative size and biomass of individual trees, which reflects their contribution to the total food availability.

Phenology data were collected twice per month for each tree. For each species and plant part (young leaves, mature leaves, flowers, unripe fruit, ripe fruit), the mean phenology score was calculated by averaging all individual scores across the two surveys. These scores quantify the proportion of the tree producing each food component.

The density values for all tree species used in the calculation were obtained from the SANParks report *Monitoring for Sustainable Indigenous Forest Management in the Garden Route National Park* (2014). Density estimates for three shrub species were derived from transect sampling in the study area and applied to the entire forest area. For each species, the basal area per hectare was calculated as the product of mean basal area per tree and species density.

The basal area per hectare for each species was calculated as:

$$BA_{\text{ha}} = BA \times \text{species density}$$

The FAI was calculated as:

$$FAI = BA_{\text{ha}} \times \text{mean phenology score}$$

iNaturalist phenological data

iNaturalist data were collected from the iNaturalist website and filtered by species on a month-by-month basis within the whole Western Cape area, South Africa. All photos included in the study were verifiable and research-grade, having been pre-reviewed by authorised scientists. The percentage of available food parts was calculated manually by the researcher (Y. Zhang): if a photo clearly showed fruits, flowers, or young leaves, it was counted as containing that food part. Each photo was counted separately within each plant-part category. The total number of photos showing fruits, flowers, or young leaves in a given month was divided by the total number of photos for that month to determine the percentage.

Results

Phenology transect results

Based on three months of phenology transect data collection, food availability indices (FAI) for leaf flush differed across species and months (**Table 1**). Black witch hazel (*Trichocladus crinitus*) consistently showed the highest values in all three months, followed by white forest spikethorn (*Gymnosporia nemorosa*) and black ironwood (*Olea capensis macrocarpa*). Some species, such as wild pomegranate (*Burchellia bubalina*), increased in August compared to June and July, while forest saffron (*Elaeodendron croceum*) decreased slightly in August. Other species, such as Cape

beech (*Rapanea melanophloeos*), had very low FAI values throughout.

Food availability indices for mature leaves showed a similar pattern of dominance by a few species (**Table 2**). Black witch hazel had by far the highest values across all three months, followed by white forest spikethorn and black ironwood. Moderate values were observed for forest saffron, Outeniqua yellowwood (*Afrocarpus falcatus*), and kamassi (*Gonioma kamassi*). Smaller contributions came from species such as wild pomegranate and num-num (*Carissa bispinosa*), and Cape beech had consistently low values. Some species, including black witch hazel (*Trichocladus crinitus*) and forest elder (*Nuxia floribunda*), showed no recorded unripe fruit during the survey period. However, this likely reflects detection difficulty rather than a true absence. Black witch hazel produces small, hairy ballistic capsules that are difficult to identify without prior experience, and forest elder produces tiny seeds contained in brown-tipped capsules that are similarly easy to overlook.

Food availability indices for flowers showed that only a few species contributed to flowering across the three months (**Table 3**). Wild pomegranate had consistently high values, while forest elder also contributed substantially, particularly in June. Forest saffron flowered in June only, and kamassi had flowers in July and August. All other species, including assegai, white forest spikethorn, and black ironwood, had no recorded flowering in these months.

Food availability indices for unripe fruit showed that only a few species produced unripe fruit across the three months (**Table 4**). Wild pomegranate had unripe fruit in June only, while white forest spikethorn had unripe fruit in June and July. Forest saffron produced unripe fruit in June and July as well, though at lower levels. Num-num had unripe fruit in all three months, with a peak in June. All other species, including assegai and black ironwood, had no recorded unripe fruit during this period.

Food availability indices for ripe fruit indicated that fruiting was limited to a few species and months (**Table 5**). White forest spikethorn had ripe fruit only in July, while forest saffron produced ripe fruit in June and July, with the highest availability in June. Num-num had ripe fruit across all three months, peaking in August. All other species, including wild pomegranate and black ironwood, had no recorded ripe fruit during this period.

The total Food Availability Index for the 13 major baboon food tree species was 4,026,275.88 in June, 4,055,429.65 in July, and 3,942,298.11 in August, showing a slight decline over the winter months. This indicates that overall food availability was relatively stable but gradually decreased during this short time period in South African winter. The slight decline may reflect reduced leaf flush or fruiting in some species, while mature leaves remained the dominant food component. Detailed FAI values for each species and plant part are provided in **Table 6**.

iNaturalist results

The analysis of leaf flush (young leaf) presence from iNaturalist photo data for multiple tree species in the Western Cape revealed clear seasonal and species-specific patterns (**Figures 2-13**). Some species, such as wild pomegranate (*B. bubalina*), exhibited a distinct seasonal pattern with leaf flush present mainly during the summer months (November to May) and absent in the winter months (June to October). In contrast, species like black witch hazel (*T. crinitus*) showed continuous leaf flush throughout the year, suggesting an

extended or overlapping leaf renewal period. Other species, including white forest spikethorn (*G. nemorosa*) and Cape beech (*R. melanophloeos*), displayed intermittent leaf flush presence, indicating possible multiple flush events or irregular leaf production. Additionally, black witch hazel (*T. crinitus*), forest elder (*N. floribunda*) and num-num (*C. bispinosa*) had leaf flush recorded in all months, consistent with evergreen leaf turnover. Species such as black ironwood (*O. c. macrocarpa*) showed an increase in leaf flush during the latter part of the year, suggesting seasonal leaf flushing patterns.

Flowering patterns varied widely among the studied tree species (**Figures 14-25**), with distinct seasonal peaks observed in several species. Wild pomegranate exhibited a long flowering season, peaking in winter through to summer (July to December, *B. bubalina*). In contrast, assegai (*C. dentata*) showed very limited flowering, mostly restricted to early summer months.

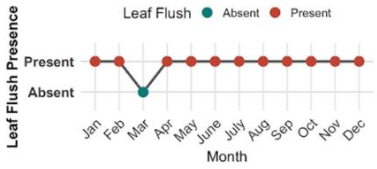


Figure 2. Leaf flush presence of *Gonioma kamassi*

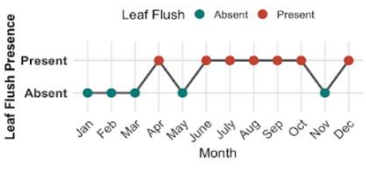


Figure 3. Leaf flush presence of *Olea capensis macrocarpa*

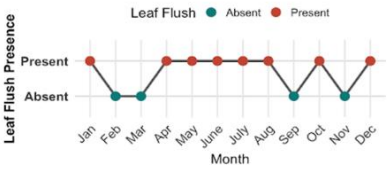


Figure 4. Leaf flush presence of *Elaeodendron croceum*

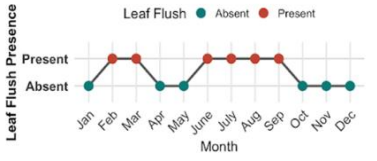


Figure 5. Leaf flush presence of *Rapanea melanophloes*

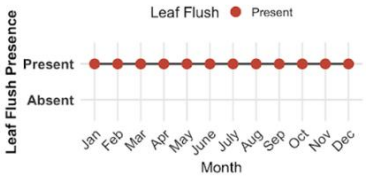


Figure 6. Leaf flush presence of *Carissa bispinosa*

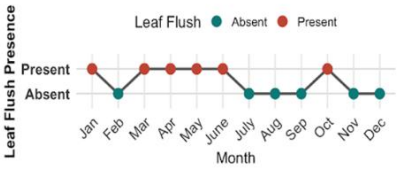


Figure 7. Leaf flush presence of *Gymnosporia nemorosa*

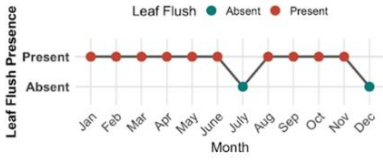


Figure 8. Leaf flush presence of *Curtisia dentata*

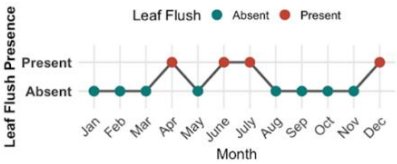


Figure 9. Leaf flush presence of *Olinia ventosa*

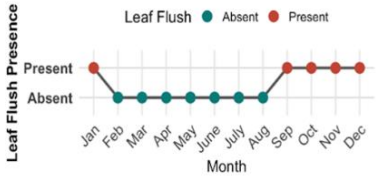


Figure 10. Leaf flush presence of *Afrocarpus falcatus*

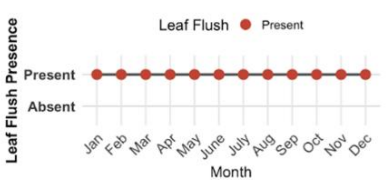


Figure 11. Leaf flush presence of *Nuxia floribunda*

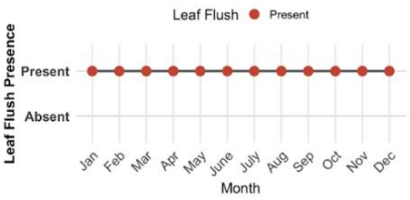


Figure 12. Leaf flush presence of *Trichocladus crinitus*

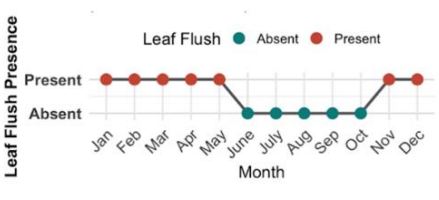


Figure 13. Leaf flush presence of *Burchellia bubalina*

White forest spikethorn (*G. nemorosa*) had low to moderate flowering in the first half of the year, with a gap in mid-winter and a small increase in late spring. Black witch hazel flowered mainly in summer and early autumn, with a secondary peak late in the year. Saffron (*E. croceum*) showed very low and irregular flowering throughout the year, but mostly flowered during spring, while forest elder (*N. floribunda*) had a concentrated flowering period from mid-autumn to early spring, peaking in late winter.

Other species such as Cape beech (R. *melanophloeos*) displayed low flowering activity with minor peaks in late winter and early spring. Hard pear (*Olinia ventosa*) began flowering in mid-winter, peaking in early spring before declining again. Kamassi (*G. kamassi*) showed scattered flowering with a marked peak in early summer. Num-num (*C. bispinosa*) flowered mainly from early spring through summer, peaking in October to December. Candlewood (*Pterocelastrus tricuspidatus*) exhibited low-level flowering year-round with a slight increase in spring and early summer. Finally, black ironwood (*O. c. macrocarpa*) showed a peak from spring through to summer, with reduced flowering in mid-winter.



Figure 14. Flowering pattern of *Burchellia bubalina*



Figure 15. Flowering pattern of *Curtisia dentata*



Figure 16. Flowering pattern of *Gymnosporia nemorosa*



Figure 17. Flowering pattern of *Trichocladus crinitus*



Figure 18. Flowering pattern of *Elaeodendron croceum*



Figure 19. Flowering pattern of *Nuxia floribunda*

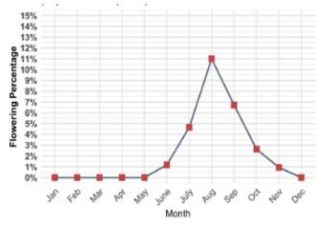


Figure 20. Flowering pattern of *Rapanea melanophloeos*

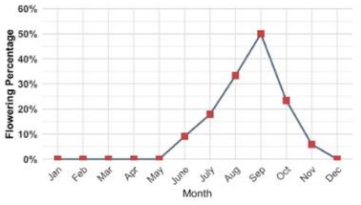


Figure 21. Flowering pattern of *Olinia ventosa*

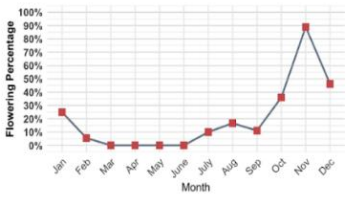


Figure 22. Flowering pattern of *Gonioma kamassi*

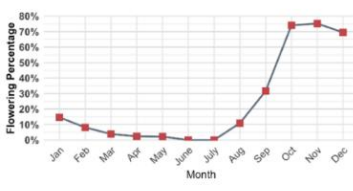


Figure 23. Flowering pattern of *Carissa bispinosa*

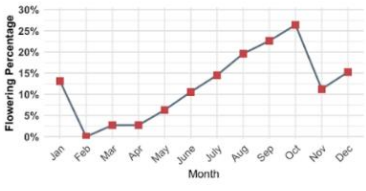


Figure 24. Flowering pattern of *Pterocelastrus tricuspidatus*

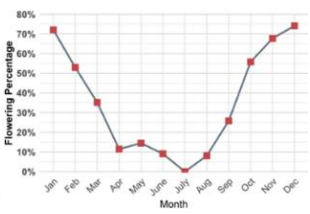


Figure 25. Flowering pattern of *Olea capensis macrocarpa*

Unripe fruit presence showed clear temporal patterns, with peaks generally occurring in specific months that varied between species (**Figures 26-38**). Several species, such as black ironwood (*O. c. macrocarpa*), assegai (*Curtisia dentata*) and hard pear (*Olinia ventosa*) displayed high percentages of unripe fruit during the winter months, followed by a decline towards spring. In contrast, species like candlewood (*P. tricuspidatus*) and white forest spikethorn (*G. nemorosa*) exhibited more extended fruiting periods, with moderate unripe fruit presence spread across multiple months. Some species, including black witch hazel (*T. crinitus*) and forest elder (*N. floribunda*), showed no recorded unripe fruit during the survey period.

As previously mentioned, Black witch hazel and forest elder capsules may be more challenging to identify. Because many iNaturalist photographs were taken by untrained public contributors, these subtle fruiting structures may have been missed. A general absence of unripe fruit across multiple species in August and September may also indicate a seasonal low in fruit development at the end of winter.

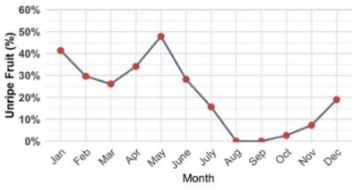


Figure 26. Unripe fruiting pattern of *Burchellia bubalina*

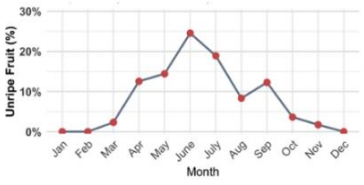


Figure 27. Unripe fruiting pattern of *Curtisia dentata*



Figure 28. Unripe fruiting pattern of *Trichocladus crinitus*

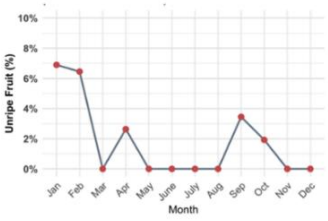


Figure 29. Unripe fruiting pattern of *Elaeodendron croceum*



Figure 30. Unripe fruiting pattern of *Gymnosporia nemorosa*



Figure 31. Unripe fruiting pattern of *Afrocarpus falcatus*



Figure 32. Unripe fruiting pattern of *Rapanea melanophloeos*

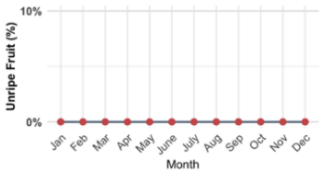


Figure 33. Unripe fruiting pattern of *Nuxia floribunda*

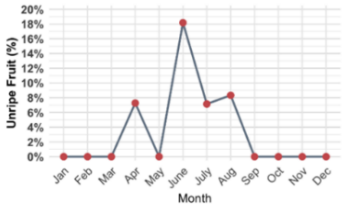


Figure 34. Unripe fruiting pattern of *Olinia ventosa*



Figure 35. Unripe fruiting pattern of *Gonioma kamassi*



Figure 36. Unripe fruiting pattern of *Carissa bispinosa*



Figure 37. Unripe fruiting pattern of *Pterocelastrus tricuspidatus*

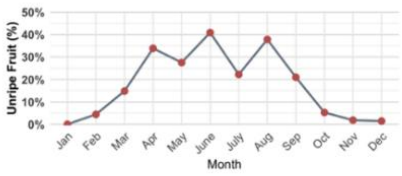


Figure 38. Unripe fruiting of *Olea capensis macrocarpa*

Ripe fruit presence was generally seasonal rather than continuous, with most species producing ripe fruit in specific months and showing long periods without ripe fruit (**Figures 39-49**). Several species, such as wild pomegranate (*B. bubalina*), Cape beech (*R. melanophloeos*), and black ironwood (*O. c. macrocarpa*), exhibited limited fruiting windows with low recorded percentages (<13%), while others, including assegai (*C. dentata*) and black witch hazel (*T. crinitus*) showed no ripe fruit during the observation period.

In contrast, num-num (*C. bispinosa*) and candlewood (*P. tricuspidatus*) were notable for producing ripe fruit year-round, although intensity varied, with peaks in August (27.27%) for num-num and July (40.32%) for candlewood. Distinct peak months were also evident for other species, including hard pear (*O. ventosa*), which reached its highest in June (31.82%) and August (29.17%), and Outeniqua yellowwood (*A. falcatus*), which peaked in August (12%) and September (11.27%).



Figure 39. Ripe fruiting pattern of *Burchellia bubalina*

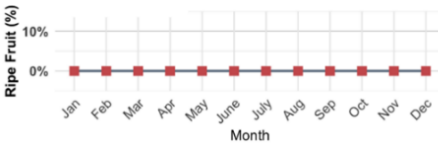


Figure 40. Ripe fruiting pattern of *Curtisia dentata*



Figure 41. Ripe fruiting pattern of *Trichocladus crinitus*



Figure 42. Ripe fruiting pattern of *Gymnosporia nemorosa*



Figure 43. Ripe fruiting pattern of *Rapanea melanophloeos*



Figure 44. Ripe fruiting pattern of *Olinia ventosa*

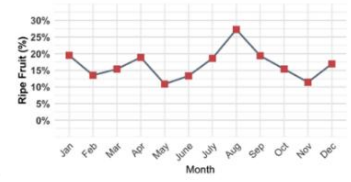


Figure 45. Ripe fruiting pattern of *Carissa bispinosa*

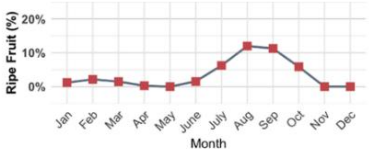


Figure 46. Ripe fruiting pattern of *Afrocarpus falcatus*

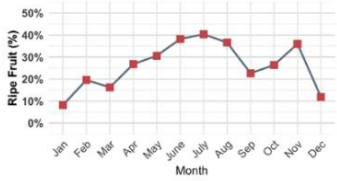


Figure 47. Ripe fruiting pattern of *Pterocelastrus tricuspidatus*

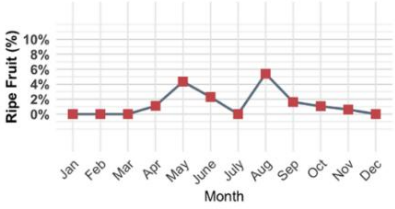


Figure 48. Ripe fruiting pattern of *Olea capensis macrocarpa*



Figure 49. Ripe fruiting pattern of *Gonioma kamassi*

Discussion

The total food availability index across the 13 baboon food tree species along the one transect remained relatively stable from June to August, suggesting a stable availability of natural food resources during the winter months, though longer-term phenology data needs to be collected from the transect in order to understand seasonal, interannual and annual fluctuations in baboon food availability. When looking at food availability during this short time window, young leaves and mature leaves were the main contributors to food availability for most tree species, while flowers, unripe fruits, and ripe fruits contributed less. This pattern reflects how leaf-based resources often dominate food availability during cooler or drier months in African forests (Hill & Dunbar, 2002). Species such as black witch hazel and black ironwood had consistently high FAI across the three months, mainly due to abundant mature leaves. The finding may also be partly explained by black witch hazel's relatively high density (1010 trees/ha) and black ironwood high basal area (2 463m²) in the forest. In contrast, species like hard pear and Cape beech had very low FAI. Given the similar level of phenology data, this could be mainly explained by the low density (hard pear 0.93 trees/ha, Cape beech 5.99 trees/ha) of the plants in the forest.

The short study period limited detection of the full seasonal range of phenological changes, particularly flowering and fruiting peaks, which often vary across years and are influenced by climatic cycles such as rainfall and solar radiation (Chapman *et al*., 1994). Many primate phenological studies are conducted in tropical forest habitats (because many wild primate species live in tropical zones), where phenological changes happen more rapidly than at temperate sites. However, our study offers preliminary results and points to the need for longer-term phenological research in relation to primate food availability in Afrotemperate forest habitat, which involves slower phenological processes than tropical forests.

To supplement the limited three-month phenology dataset collected during the internship period, iNaturalist observations were used to provide longer-term information on plant food availability. Although the study site, Nature's Valley, is located in the eastern portion of the Western Cape, where rainfall occurs throughout the year and summers are relatively warm and productive (Mucina *et al*., 2006; Rebelo *et al*., 2006), the iNaturalist dataset used in this study encompasses the entire Western Cape Province. Consequently, the dataset includes observations from regions with different climatic regimes, particularly the western parts of the province where a winter-rainfall system dominates. Despite this spatial heterogeneity, iNaturalist remains a valuable tool for identifying broad seasonal phenological trends across large temporal and geographic scales, especially when field-based data collection is limited (Jones *et al*., 2022). However, iNaturalist data can be biased because observations are contributed by general users. Sampling effort typically peaks during the summer tourist season, and less conspicuous tree species may be underrepresented. Because phenological percentages were calculated by dividing the number of photos showing the target plant part by the total photos for that month, uneven sampling effort may influence the estimated seasonal patterns (Barve *et al*., 2020).

Based on the iNaturalist findings, there is clear seasonal variation in leaf, flower, and fruit production of plants in the baboon diet. Leaf flush occurs mainly in the warmer months for most species, although some, like black witch hazel, num-num and forest elder, maintain leaf flush throughout the year. Such

evergreen behaviour is common among tree species in southern African temperate forests, allowing continuous photosynthesis under mild winter conditions (Phillips, Midgley, & Bond, 2003). Flowering often follows or overlaps with leaf flush. This pattern, where flowering closely follows leaf emergence, has been widely documented in African woody plants and may enhance pollination efficiency during optimal climatic periods (Chapman *et al*., 2018). However, our dataset shows no clear patterns in synchronized leaf flush and flowering.

Fruiting shows distinct seasonal peaks rather than constant production. Most species have few or no ripe fruits in winter (June to August), with fruit generally peaking in late summer to autumn (around March to May). Such peaks coincide with higher rainfall and temperature, which stimulate fruit maturation (Boyer, 1982). Some species, such as candlewood and num-num, fruit over longer periods but still show clear peaks. Extended fruiting periods are adaptive for maintaining seed dispersal opportunities and supporting frugivore populations, including primates, in variable environments (Van Schaik *et al*., 1993).

Overall, these patterns from the iNaturalist dataset align with the trends described in the *Trees of the Garden Route: Mossel Bay to Storms River* (Venter, 2009), where leaf flush and flowering occur primarily in warmer months, and fruiting peaks toward late summer to autumn (Phillips, Midgley & Bond, 2003). This agreement suggests that despite potential sampling bias, citizen-science platforms can capture broad phenological cycles consistent with field-based ecological observations (Jones *et al*., 2022).

One interesting pattern observed in the iNaturalist data is that, based on the percentages of unripe fruits produced by each tree species, a similar level of ripe fruit would be expected in the following month. However, the data show that fewer ripe fruits were actually recorded or available. This suggests that many of the ripe fruits preferred by baboons are likely highly attractive to birds and other frugivores, which consume them quickly. Such interspecific competition for ripe fruits has been widely documented in African forests, where avian frugivores often remove large portions of fruit crops soon after ripening (Bleher *et al*., 2003). As a result, baboons may have limited access to some ripe fruits, even during periods that would theoretically be peak fruiting seasons, leading them to rely more heavily on foods such as leaves or unripe fruits (Marshall & Wrangham, 2007).

Next steps in this project include NVBP examining the statistical relationship between food availability and what the baboons are observed eating. NVBP collects feeding behaviour data during ten-minute focal follows of individual baboons and therefore the project will examine how baboon feeding on particular parts and plant species map onto (or does not map onto) the availability of plant foods in the environment according to the iNaturalist dataset and according to future long-term phenology data on the baboon foods transect. One limitation of our current approach to measuring baboon food availability is that the phenology transect only captures (1) plant foods in the baboon diet (baboons are omnivores and eat insects, soil, mushrooms and other items) and (2) Afrotemperate forest foods the baboons eat (none of our food availability measures yet examine the availability of foods in their diet outside of the Afrotemperate forest; NVBP has observed the baboons eating parts of fynbos, for example). Other studies have examined the relationship between food availability and feeding behaviour in baboons at other sites. Segal (2008) found that during

the dry (winter) season, food sources were limited and chacma baboons primarily consumed leaves; during this lean period, baboons were more likely to leave the forest and enter nearby villages to access high-energy human foods, which can increase the likelihood of human–wildlife conflict. Mukuve (2024) similarly observed that baboons in winter were more active around human villages, particularly near schools, where they waited for opportunities to consume human food. Rahman (2023) also found seasonal dietary differences, with baboons consuming a significantly more varied diet in summer than in winter. When natural food availability declined during winter, baboons supplemented their diet with human-derived foods such as trash or crops, highlighting their adaptability to natural resource scarcity (Fehlmann *et al*., 2017). During months of high fruit availability at other sites, baboons have been found to largely rely on forest resources (Altmann & Muruthi, 1988; Hill & Dunbar, 2002).



These preliminary results and future directions with baboon food availability monitoring will help us understand ecological variables that affect baboon movements in Nature's Valley. By understanding these patterns of food availability, it is possible to some extent to predict the activity and movement of baboons in different seasons. Under certain food availability conditions, baboons may shift toward human settlements in search of alternative food sources, increasing the potential for human-wildlife conflict. Behavioural adjustments to resource fluctuations have been reported in chacma baboons living in anthropogenically modified habitats at other sites (Rahman, Hill & MacLarnon, 2025). This knowledge can help local managers and residents prepare in advance and develop mitigation strategies to reduce the risk of conflict between humans and baboons.

Table 1: Food availability index (FAI) for leaf flush across 13 species in June, July and August and the total food availability for all species.

Scientific name	Common name	June FAI	July FAI	Aug FAI
Burchellia bubalina	Wild pomegranate	2,146.28	2,146.28	4,292.56
Trichocladus crinitus	Black witch hazel	630,857.57	630,857.57	630,857.57
Gymnosporia nemorosa	White forest spikethorn	103,955.93	129,944.91	103,955.93
Curtisia dentata	Assegai	50,014.55	30,008.73	20,005.82
Elaeodendron croceum	Forest saffron	27,777.12	27,777.12	20,832.84
Nuxia floribunda	Forest elder	22,735.89	22,735.89	22,735.89
Rapanea melanophloeos	Cape beech	230.52	230.52	230.52
Afrocarpus falcatus	Outeniqua yellowwood	23,175.76	23,175.76	23,175.76
Olinia ventosa	Hard pear	1,145.30	572.65	572.65
Gonioma kamassi	Kamassi	33,326.30	39,991.57	39,991.57
Carissa bispinosa	Num-num	18,241.66	27,362.49	18,241.66
Pterocelastrus tricuspidatus	Candlewood	1,710.79	1,710.79	1,710.79
Olea capensis macrocarpa	Black ironwood	113,949.27	113,949.27	113,949.27
	Total	**1,029,266.95**	**1,050,463.54**	**1,000,552.82**

Table 2: FAI for mature leaves across 13 species in June, July and August and the total food availability for all species.

Scientific name	Common name	June FAI	July FAI	Aug FAI
Burchellia bubalina	Wild pomegranate	6,438.84	6,438.84	4,292.56
Trichocladus crinitus	Black witch hazel	1,892,572.70	1,892,572.70	1,892,572.70
Gymnosporia nemorosa	White forest spikethorn	311,867.79	285,878.81	311,867.79
Curtisia dentata	Assegai	30,008.73	50,014.55	60,017.46
Elaeodendron croceum	Forest saffron	41,665.69	41,665.69	48,609.97
Nuxia floribunda	Forest elder	22,735.89	22,735.89	22,735.89
Rapanea melanophloeos	Cape beech	691.57	691.57	691.57
Afrocarpus falcatus	Outeniqua yellow-wood	69,527.27	69,527.27	69,527.27
Olinia ventosa	Hard pear	1,145.30	1,717.95	1,717.95
Gonioma kamassi	Kamassi	73,317.87	66,652.61	66,652.61
Carissa bispinosa	Num-num	54,724.97	45,604.14	54,724.97
Pterocelastrus tricuspi-datus	Candlewood	5,132.36	5,132.36	5,132.36
Olea capensis macro-carpa	Black ironwood	341,847.82	341,847.82	341,847.82
	Total	**2,851,676.79**	**2,830,480.18**	**2,880,390.91**

Table 3: Food availability index (FAI) for flowers across 13 species in June, July and August and the total food availability for all species.

Scientific name	Common name	June FAI	July FAI	Aug FAI
Burchellia bubalina	Wild pomegranate	4,292.56	3,219.42	6,438.84
Curtisia dentata	Assegai	0	0	0
Gymnosporia nemorosa	White forest spikethorn	0	0	0
Trichocladus crinitus	Black witch hazel	0	0	0
Elaeodendron croceum	Forest saffron	3,472.14	0	0
Nuxia floribunda	Forest elder	22,735.89	11,367.94	11,367.94
Rapanea melanophloeos	Cape beech	0	0	0
Afrocarpus falcatus	Outeniqua yellowwood	0	0	0
Olinia ventosa	Hard pear	0	0	0
Gonioma kamassi	Kamassi	0	3,332.63	6,665.26
Carissa bispinosa	Num-num	0	0	0
Pterocelastrus tricuspidatus	Candlewood	0	0	0
Olea capensis macrocarpa	Black ironwood	0	0	0
	Total	**30,500.59**	**17,920.00**	**24,472.05**

Table 4: Food availability index (FAI) for unripe fruit across 13 species in June, July and August and the total food availability for all species.

Scientific name	Common name	June FAI	July FAI	Aug FAI
Burchellia bubalina	Wild pomegranate	2,146.28	0	0
Curtisia dentata	Assegai	0	0	0
Gymnosporia nemorosa	White forest spikethorn	51,977.96	51,977.96	0
Trichocladus crinitus	Black witch hazel	0	0	0
Elaeodendron croceum	Forest saffron	13,888.56	3,472.14	0
Nuxia floribunda	Forest elder	0	0	0
Rapanea melanophloeos	Cape beech	0	0	0
Afrocarpus falcatus	Outeniqua yellowwood	0	0	0
Olinia ventosa	Hard pear	0	0	0
Gonioma kamassi	Kamassi	0	0	0
Carissa bispinosa	Num-num	27,362.49	13,681.24	18,241.66
Pterocelastrus tricuspidatus	Candlewood	0	0	0
Olea capensis macrocarpa	Black ironwood	0	0	0
	Total	**95,375.29**	**69,131.35**	**18,241.66**

Table 5: Food availability index (FAI) for ripe fruit across 13 species in June, July and August and the total food availability for all species.

Scientific name	Common name	June FAI	July FAI	Aug FAI
Burchellia bubalina	Wild pomegranate	0	0	0
Curtisia dentata	Assegai	0	0	0
Gymnosporia nemorosa	White forest spikethorn	0	77,966.95	0
Trichocladus crinitus	Black witch hazel	0	0	0
Elaeodendron croceum	Forest saffron	10,416.42	5,208.21	0
Nuxia floribunda	Forest elder	0	0	0
Rapanea melanophloeos	Cape beech	0	0	0
Afrocarpus falcatus	Outeniqua yellowwood	0	0	0
Olinia ventosa	Hard pear	0	0	0
Gonioma kamassi	Kamassi	0	0	0
Carissa bispinosa	Num-num	9,120.83	4,560.41	18,241.66
Pterocelastrus tricuspidatus	Candlewood	0	0	0
Olea capensis macrocarpa	Black ironwood	0	0	0
	Total	**19,537.25**	**87,735.57**	**18,241.66**

Table 6: Total Food Availability Index (FAI) for 13 baboon food tree species in Nature Valley, South Africa, across three months (June–August). FAI values represent the combined availability of young leaves, mature leaves, flowers, unripe fruit, and ripe fruit for each species.

Scientific name	Common name	June FAI	July FAI	August FAI
Burchellia bubalina	Wild pomegranate	15,023.97	11,804.54	15,023.96
Trichocladus crinitus	Black witch hazel	2,523,429.27	2,523,429.27	2,523,429.27
Gymnosporia nemorosa	White forest spikethorn	467,801.68	545,768.63	415,823.72
Curtisia dentata	Assegai	80,023.28	80,023.28	80,023.28
Elaeodendron croceum	Forest saffron	97,239.93	78,023.16	69,442.81
Nuxia floribunda	Forest elder	68,207.67	56,839.72	56,839.72
Rapanea melanophloeos	Cape beech	922.09	922.09	922.09
Afrocarpus falcatus	Outeniqua yellow-wood	92,603.03	92,603.03	92,603.03
Olinia ventosa	Hard pear	2,290.60	2,290.60	2,290.60
Gonioma kamassi	Kamassi	106,644.17	109,976.81	113,309.44
Carissa bispinosa	Num-num	109,449.95	91,108.28	109,949.95
Pterocelastrus tricuspidatus	Candlewood	6,843.15	6,843.15	6,843.15
Olea capensis macrocarpa	Black ironwood	455,797.09	455,797.09	455,797.09
	All species	**4,026,275.88**	**4,055,429.65**	**3,942,298.11**

References

Altmann, J. & Muruthi, P. 1988. Differences in daily life between semi-provisioned and wild-feeding baboons. American Journal of Primatology, 15(3), 213–221.

Barton, R. A., Whiten, A., Strum, S. C., Byrne, R. W. & Simpson, A. J. 1992. Habitat use and resource availability in baboons. Animal Behaviour, 43(5), 831–844.

Barve, V., Brenskelle, L., Li, D., Stucky, B. J., Barve, N., Hantak, M. M. & Guralnick, R. P. 2020. Methods for broad-scale plant phenology assessments using citizen scientists' photographs. Applications in Plant Sciences, 8(1), e11315.

Bleher, B., Böhning-Gaese, K. & Fiedler, W. 2003. The role of birds in seed dispersal and natural regeneration of an Afrotropical forest. Journal of Tropical Ecology, 19(5), 615–626.

Boyer, W. D. 1982. Phenology of tree species in tropical and subtropical forests. Biotropica, 14(3), 252–263.

Chapman, C. A., Valenta, K., Bonnell, T. R., Brown, K. A. & Chapman, L. J. 2018. Solar radiation and ENSO predict fruiting phenology patterns in a 15-year record from Kibale National Park, Uganda. Biotropica, 50(3), 384–395.

Chapman, C. A., Wrangham, R. & Chapman, L. J. 1994. Indices of habitat-wide fruit abundance in tropical forest. Biotropica, 26(2), 160–171.

D'Orangeville, L., Duchesne, L., Houle, D. & Pederson, N. 2021. Drought impacts on the diameter growth of mature trees in temperate Eastern North America. Tree Physiology, 41(6), 853–865.

Fehlmann, G., O'Riain, M. J., Kerr-Smith, R. & King, A. J. 2017. Adaptive space use by baboons (*Papio ursinus*) in response to management interventions in a human changed landscape. Biological Conservation, 212, 1–9.

Hill, R. A. & Dunbar, R. I. M. 2002. Climatic determinants of diet and foraging behaviour in baboons. Evolutionary Ecology, 16(6), 579–593.

Jones, M. O., Allred, B. W., Naugle, D. E., Maestas, J. D., Donnelly, P., Metz, L. J., Karl, J., Smith, R., Bestelmeyer, B., Boyd, C., Kerby, J. & McIver, J. 2022. Harnessing citizen science for large-scale plant phenology monitoring. Global Change Biology, 28(4), 1403–1417.

Marshall, A. J. & Wich, S. 2013. Characterization of primate environments through assessment of plant phenology. In Primate ecology and conservation: A handbook of techniques (pp. 103–127). Oxford University Press.

Marshall, A. J. & Wrangham, R. W. 2007. Evolutionary consequences of fallback foods. International Journal of Primatology, 28(6), 1219–1235.

Mucina, L., Geldenhuys, C.J., Rutherford, M.C., Powrie, L.W., Lötter, M.C., von Maltitz, G.P., Euston-Brown, D.I.W., Matthews, W.S., Dobson, L. & McKenzie, B. 2006. Afrotemperate, subtropical and azonal forests. In: Mucina, L. & Rutherford, M.C. (eds), The vegetation of South Africa, Lesotho and Swaziland: 584-615. SANBI, Pretoria.

Mukuve, B. Z. 2024. Analysis of seasonal variation in dietary composition and behaviour of wild chacma baboons (*Papio ursinus*) in two habitats of different human influence around Windhoek, central Namibia (Doctoral dissertation, Department of Environmental Science, University of Namibia).

Phillips, O. L., Midgley, J. J. & Bond, W. J. 2003. Environmental constraints on leaf phenology and growth form in southern African forests. Journal of Ecology, 91(4), 765–776.

Rahman, M. 2023. Behavioural and physiological responses to anthropogenic food sources in chacma baboons (Doctoral dissertation, Durham University).

Rahman, M. M., Hill, R. A. & MacLarnon, A. M. 2025. Behavioural and physiological responses of chacma baboons (*Papio ursinus*) to human-induced mortality in anthropogenically modified habitat. People and Nature, 7(8), 1865–1878.

Rebelo, A. G., Boucher, C., Helme, N., Mucina, L. & Rutherford, M. C. 2006. Fynbos biome. In L. Mucina & M. C. Rutherford (Eds.), The vegetation of South Africa, Lesotho and Swaziland (Strelitzia 19, pp. 144–145). South African National Biodiversity Institute.

Segal, C. 2008. Foraging behaviour and diet in chacma baboons in Suikerbosrand Nature Reserve. University of the Witwatersrand, Johannesburg (South Africa).

Tang, J., Körner, C., Muraoka, H., Piao, S., Shen, M., Thackeray, S. J. & Yang, X. 2016. Emerging opportunities and challenges in phenology: a review. Ecosphere, 7(8), e01436.

Van Schaik, C. P., Terborgh, J. W. & Wright, S. J. 1993. The phenology of tropical forests: Adaptive significance and consequences for primary consumers. Annual Review of Ecology and Systematics, 24, 353–377.

What's on the menu? Exploring chacma baboon (*Papio ursinus*) consumption of indigenous and invasive plants and implications for Afrotemperate forest management

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How to cite this article:

Anderson, I., Sabbi, K.H., van Hasselt, C., Kurilla, S.E., Coetzee, H., Swart, R.C. & Bryer, M.A.H. 2026. What's on the menu? Exploring chacma baboon (*Papio ursinus*) consumption of indigenous and invasive plants and implications for Afrotemperate forest management. *Nature's Letters* **3**:34-45. URL: https://nvt.org.za/natures-letters/

Introduction

Herbivores in diverse ecosystems aid in the spread of plant species through seed dispersal (Janzen, 1984; Malo & Suárez, 1995; Chapman & Russo, 2007; Calviño-Cancela, 2011; Auffret & Plue, 2014). Indigenous plant seed dispersal by animals is key not only to maintaining existing forested landscapes but also to successful indigenous forest restoration in both tropical and temperate biomes (Wunderle, 1997; Selwyn *et al*., 2023). As alien invasive species are a major concern worldwide due to their detrimental effects on native plant communities (Vitousek *et al*., 1997; Asner *et al*., 2008), understanding how native fauna act as seed dispersers or seed destroyers of invasive plants is important for predicting effects of plant invasions on processes like forest restoration. Seed predation (eating and destroying instead of eating and passing) also has implications for positive alien invasive plant eradication or detrimental indigenous plant removal (Barnes, 2001; Kunz & Linsenmair, 2010).

In tropical forest habitats, wild primates are well characterised as both seed dispersers (Lambert & Garber, 1998; Chapman & Russo, 2007; Albert *et al*., 2013) and seed predators (Janzen, 1971; Norconk *et al*., 2013). Primates disperse seeds through multiple handling mechanisms, including through swallowing seeds whole and defecating the seeds away from the parent tree (Lambert & Garber, 1998). In addition to ingestion, several primate species also temporarily store fruits in cheek pouches for later processing and spit fruit seeds out away from the parent tree as they separate them from edible plant parts (Lambert & Garber, 1998). The germinating potential of seeds that are dispersed in these ways may differ by the way that they are handled as some seed species require gut-passage to germinate, whereas others are damaged during ingestion and digestion (Gross-Camp & Kaplin, 2011). Though well-studied in other ecosystems, these dynamics are less well understood in non-tropical-rainforest primate habitats. Generalist herbivore seed dispersal is also less well studied than dispersal by fruit-specialists (Howe, 1986; Jordano, 2000).

As large-bodied mammals with relatively large home ranges, baboons (*Papio* spp.) have been shown to play important roles as both seed dispersers (Lieberman *et al*., 1979; Tew *et al*., 2018) and

pre-dispersal seed predators (Barnes, 2001; Paterson, 2006). Chacma baboons (*Papio ursinus*) have diverse diets (Hamilton *et al*., 1978; Johnson *et al*., 2013; Pebsworth, 2020) that consist primarily of plant matter, including fruit, although they also feed on invertebrates and small vertebrates (Allan *et al*., 2022) and foods discarded by humans (Altmann & Muruthi, 1988; Johnson *et al*., 2013). This dietary flexibility allows baboons to thrive in a variety of different environments and at diverse levels of anthropogenic pressure. While chacma baboons are known to play a role as seed dispersers in some South African ecosystems (Tew *et al*., 2018; Pamla *et al*., 2024), little is known about their seed dispersal potential in Afrotemperate forest. One study in Hogsback in the Eastern Cape found that chacma baboons ranging in indigenous Afromontane Forest (Southern Mistbelt) consume the seeds of multiple alien plant species, suggesting that baboons may facilitate the seed dispersal of non-native plant species, with implications for forest management (Pamla *et al.*, 2024).

Nature's Valley, South Africa, is a mosaic habitat that includes Afrotemperate forest that is part of the Knysna-Tsitsikamma forest complex. This Afrotemperate forest is a rare ecosystem that supports high levels of endemic plant species, includes tropical remnants (Von Breitenbach, 1972), and is part of the broader Afromontane forest group (Swart *et al*., 2024). There is also a history of invasive plant species impacting this area, and local conservation groups like NVT have been working to manage or remove invasives (Coetzee, 2026). Understanding the diet of local baboons in this forest will help us understand their ecosystem roles as seed dispersers and seed predators, allowing for informed management decisions for indigenous Afrotemperate forest restoration and invasive species eradication efforts.

In this study, we examine the extent to which a troop of chacma baboons ranging in Afrotemperate forest, fynbos, and human settlement rely on indigenous versus non-invasive alien versus invasive alien plant foods. We aim to quantify the feeding time devoted to these categories of plants and investigate seasonal differences in their reliance on indigenous versus invasive plants, as this has implications for seed dispersal or seed destruction in the context of indigenous forest management, restoration, and invasive species eradication.

Methods

Study site and species

This research was conducted in Nature's Valley, South Africa, from July 2023 to July 2025. Nature's Valley is located on the eastern border of the Western Cape and is within the boundaries of the Tsitsikamma Section of the Garden Route National Park. This study area includes a unique intersection of ecosystems, with Afrotemperate forest (Mucina *et al*., 2006), fynbos, marine, and human settlement. Over the course of the study, the baboon study group included 8-21 individuals, and the size of their annual home range was 4.83 - 6.12 km^2.

Data collection

Researchers searched for and followed the troop of baboons 5-6 days per week and collected behavioural data via group scans and focal follows to collect information on movement and individual behaviour. The group scan datasheets were used to record spatial and *ad libitum* behavioural data on the baboons to give a cohesive view of how the troop moved and behaved together. We also collected ten-minute focal follows (Altmann, 1974) of all individually identifiable baboons (n = 8-16), excluding unweaned infants as their diets were primarily milk and therefore their impact as potential seed dispersers minimal. During each focal

follow, we recorded the focal individual's behaviour, including all of their feeding behaviours. For each feeding bout, we recorded the start and end time (to calculate feeding duration), as well as the species of plant and plant part that were consumed, or any detailed information on the non-plant food consumed. When a baboon was observed eating a plant or food item unknown to the observer, a photo was taken of the food item in the field. With the assistance of plant identification books and experts on NVT's team, we were able to identify many of the unidentified species being consumed. Behavioural observations were not collected when the baboons ranged within the limits of Nature's Valley village, as the Nature's Valley Baboon Project (NVBP) communicates with and collaborates with the Baboon Monitor Programme whereby baboon monitors herd the baboons away from residences.

Data analysis

First, we created a list of foods observed eaten by the baboons during the study period. We included the following specific information for each plant food item consumed: scientific name, common name(s), plant parts consumed, type of plant (e.g., tree, shrub, vine, fynbos). Most importantly to this project, we categorised each plant according to its origin as native, non-invasive alien, or alien invasive. Alien plants were identified using the SANBI PlantZAfrica database and NVT's *Alien Invasive Plants of Nature's Valley* (Arendse & Breen, 2018). We calculated time spent feeding for each of these plant categories based on ten-minute focal follow data.

All calculations and analyses were made using R (ver. 4.5.2, R Core Team 2025). We calculated the percent of time spent feeding on the following six categories: indigenous, non-invasive alien, invasive alien, human food, unidentified foods, and other foods (those that were eaten infrequently and did not fall under the plant categories). Foods were only categorised into the "unidentified" category if there were either no details recorded or not sufficient details recorded during the observation period to confidently identify the species. Once all of the feeding bouts were classified, the amount of time feeding within each category was divided by the total time spent eating to calculate the percent of time spent feeding in the given category.

We further investigated seasonal differences in feeding on each category of plant by dividing the data into feeding that occurred during spring (September - November), summer (December - February), autumn (March - May) and winter (June - August). We used Kruskal-Wallis tests using the package *stats* (ver. 4.5.2, R Core Team 2025), to identify significant differences between seasonal consumption of indigenous, non-invasive alien, invasive alien, and human foods. If there was a significant difference detected across the four seasons, a Dunn's post-hoc test using the package *FSA* (ver. 4.5.2, R Core Team 2025) with a Bonferroni correction was performed to determine which seasonal pairs differed significantly within each food category.


Credit: Adrien Vol

Results

We collected 243 hours of focal observations over the study period, including 79 hours of feeding. Over the course of the study, we observed the baboons feeding on 71 indigenous plant species, 8 non-invasive alien plant species, and 11 alien invasive plant species (**Table 1**). The baboons spent much of their feeding time on indigenous plant foods (45.91%), and less of their feeding time on non-invasive alien (0.80%) and invasive alien plant foods (9.84%). Human food (e.g., food from bins in picnic areas) made up 1.78% of feeding time, while unidentified foods comprised 37.89% of feeding time, and other (e.g., non-plant foods like mushrooms and insects) comprised 3.78% of feeding time. The top plant foods consumed by the baboons were the fruits and leaves of black witch hazel (*Trichocladus crinitus*), the fruit of forest grape (*Rhoicissus tomentosa*), and the fruit from the assegai (*Curtisia dentata*), all of which are indigenous to the Afrotemperate forest. The alien invasive species consumed the most during the study was kikuyu grass (*Cenchrus clandestinus*), of which they fed on the roots and shoots.

Of the food categories of particular interest in this study (plant foods that were indigenous, non-invasive alien, and invasive alien), indigenous was the only category that differed significantly across seasons ($H(3) = 84.574$, $p<0.01$). Post-hoc pairwise comparisons showed time spent feeding on indigenous foods in winter was significantly greater compared to other seasons ($p<0.01$). Time spent feeding on indigenous foods in summer was significantly greater than in autumn and spring ($p<0.01$). However, no significant difference was found for indigenous plant food consumption between autumn and spring or between summer and winter.

There was no significant difference in time spent feeding on non-invasive alien species ($H(3) = 3.443$, $p = 0.328$) or alien invasive plant species ($H(3) = 5.105$, $p = 0.164$) across seasons. Additionally, although not a primary category of investigation, no seasonal variation in consumption of human food was found ($H(2) = 1.051$, $p = 0.591$).

Table 1: All species of Indigenous, Non-Invasive Alien, and Alien Invasive plant species members of the baboon group were observed feeding on during the study. Information provided includes the plant species name, common name(s), plant type, and plant part(s) consumed.

Indigenous species

Species name	Common name	Plant type	Plant part
Afrocarpus falcatus	Outeniqua yellowwood	Tree	Seeds
Albuca nelsonii	Wild onion	Bulb	Bulb
Apodytes dimidiata	White pear	Tree	Fruit, leaves
Aspalathus ciliaris	Fringe cape gorse	Fynbos	Leaves
Asparagus setaceus	Plumosa fern	Fern	Leaves
Bobartia aphylla	Garden route rushiris	Bulb	Flowers

Burchellia bubalina	Wild pomegranate	Tree	Fruit, leaves
Canthium inerme	Turkey berry	Tree	Fruit
Carissa bispinosa	Num-num	Shrub	Fruit
Carpobrotus edulis	Sour fig	Succulent	Fruit, leaves
Cassine peragua	Bastard saffron	Tree/Shrub	Leaves
Cassine schinoides	Spoonwood	Shrub	Leaves
Celtis africana	White stinkwood	Tree	Fruit
Centella asiatica	Pennywort	Forb	Leaves
Chionanthus foveolatus	Pock-ironwood	Shrub	Leaves
Chrysophyllum viridi-folium	Fluted milkwood	Tree	Fruit
Cissampelos torulosa	Kidney-leaf	Vine	Leaves
Curtisia dentata	Assegai	Tree	Fruit
Cyathea dregei	Forest tree fern	Fern	Leaves
Cynanchum africanum	Cape buckhorn	Climber	Flowers
Cynanchum ellipticum	Monkey rope	Climber	Flowers
Dietes iridioides	Cape iris	Bulb	Seeds
Diospyros dichrophylla	Star apple	Tree	Fruit, bark
Dipogon lignosus	Cape sweet pea	Vine	Seed pods
Dovyalis rhamnoides	Cape cranberry	Shrub	Leaves
Elaeodendron croceum	Forest saffron	Tree	Leaves
Eragrostis racemosa	Narrow heart love grass	Grass	Spikelet
Eriocephalus africanus	Wild rosemary	Shrub	Not recorded
Eucleas sp.	Gwarries	Tree	Leaves
Ficus burtt-davyi	Scrambling fig	Tree	Fruit
Grammatotheca bergiana	Water lobelia	Forb	Leaves, stem
Grewia occidentalis	Crossberry	Shrub	Fruit

Gymnanthemum capense	Climbing hawthorn	Shrub	Leaves
Gymnosporia buxifolia	Common spikethorn	Shrub	Leaves
Gymnosporia nemorosa	White forest spikethorn	Shrub	Leaves
Halleria lucida	Notsung	Tree	Leaves
Helichysum aureonitens	Golden everlasting	Shrub	Leaves
Hippobromus pauciflorus	False-horsewood	Tree	Fruit, leaves
Hypolepis sparsisora	False bracken	Fern	Head
Hypoxis hemerocallidea	Wild potato	Tuber	Tuber
Ilex mitis	Cape holly	Tree	Leaves
Metalasia muricata	Bristlebush	Fynbos	Leaves
Nuxia floribunda	Forest elder	Tree	Leaves
Ocotea bullata	Stinkwood	Tree	Leaves
Olea capensis	Black ironwood	Tree	Fruit, leaves
Olea exasperata	Dune olive	Shrub	Leaves
Olinia ventosa	Hard pear	Tree	Leaves
Osteospermum moniliferum	Bitou bush	Shrub	Flowers
Osyris compressa	Cape sumach	Shrub	Fruit, leaves
Parmeliaceae	Old man's beard moss	Moss	Whole
Passerina corymbosa	Gonnas	Fynbos	Leaves
Pentarrhinum insipidum	African heartvine	Vine	Leaves
Polygala myrtifolia	September bush	Shrub	Leaves
Pterocelastrus tricuspidatus	Candlewood	Tree	Leaves
Pyrenacantha scandens	Climbing firethorn	Climber	Leaves

Rapanea melanophloeos	Cape beech	Tree	Fruit, leaves
Rhoicissus digitata	Baboon grape	Vine	Fruit
Rhoicissus tomentosa	Forest grape	Vine	Fruit
Rothmannia capensis	Wild gardenia	Tree	Fruit
Scolopia mundii	Redpear	Tree	Fruit
Scutia myrtina	Cat-thorn	Shrub	Fruit
Searsia lucida	Glossy currant	Tree	Leaves
Searsia pallens	Eastern kunibush, kunibush	Shrub	Leaves
Senecio angulatus	Cape ivy	Climber	Leaves
Senecio deltoideus	Climbing ragwort	Vine	Leaves
Sideroxylon inerme	Milkwood	Tree	Fruit, leaves
Sparrmannia africana	Cape hemp	Shrub	Flower bud, leaves
Stachys aethiopica	African stachys	Shrub	Leaves
Strelitzia alba	Cape wild-banana	Tree	Leaf base
Trichocladus crinitus	Black witch hazel	Tree/Shrub	Fruit, leaves
Virgilia divaricata	Blossom tree, Keurboom	Tree	Flower

Non-invasive alien species

Species name	**Common name**	**Plant type**	**Plant part**
Coix lacryma-jobi	Job's tears	Grass	Stem
Helichrysum bracteatum	Strawflower	Forb	Leaves
Hibiscus rosa-sinensis	Hibiscus	Shrub	Flower
Physalis peruviana	Cape gooseberry, groundcherry	Shrub	Fruit
Oxalis pes-caprae	Wild sorrel, Cape sorrel	Herb	Leaves, stems
Strelitzia reginae	Crane flower, bird of paradise	Ornamental plant	Roots
Syzygium paniculatum	Australian water pear, Australian brush cherry, magenta cherry	Tree	Fruit

Thunbergia alata	Black-eyed susan vine	Vine	Leaves

Alien invasive species

Species name	**Common name**	**Plant type**	**Plant part**
Acacia mearnsii	Black wattle tree	Tree	Leaves, seeds
Ipomoea cairica	Morning glory	Climber	Flowers, leaves
Cenchrus clandestinus	Kikuyu grass	Grass	Shoots, roots
Pinus radiata	Pine	Tree	Seeds
Polystichum munitum	Sword fern	Fern	Leaves
Rubus fruticosus	European blackberry	Shrub	Fruit
Stellaria media	Chickweed	Herb	Shoots, leaves
Taraxacum officinale	Dandelion	Forb	Flower
Tradescantia fluminensis	Wandering dude	Trailing plant	Whole
Trifolium angustifolium	Narrow-leaved clover	Forb	Leaves, stem, Flowers
Trifolium repens	Clover	Forb	Shoots

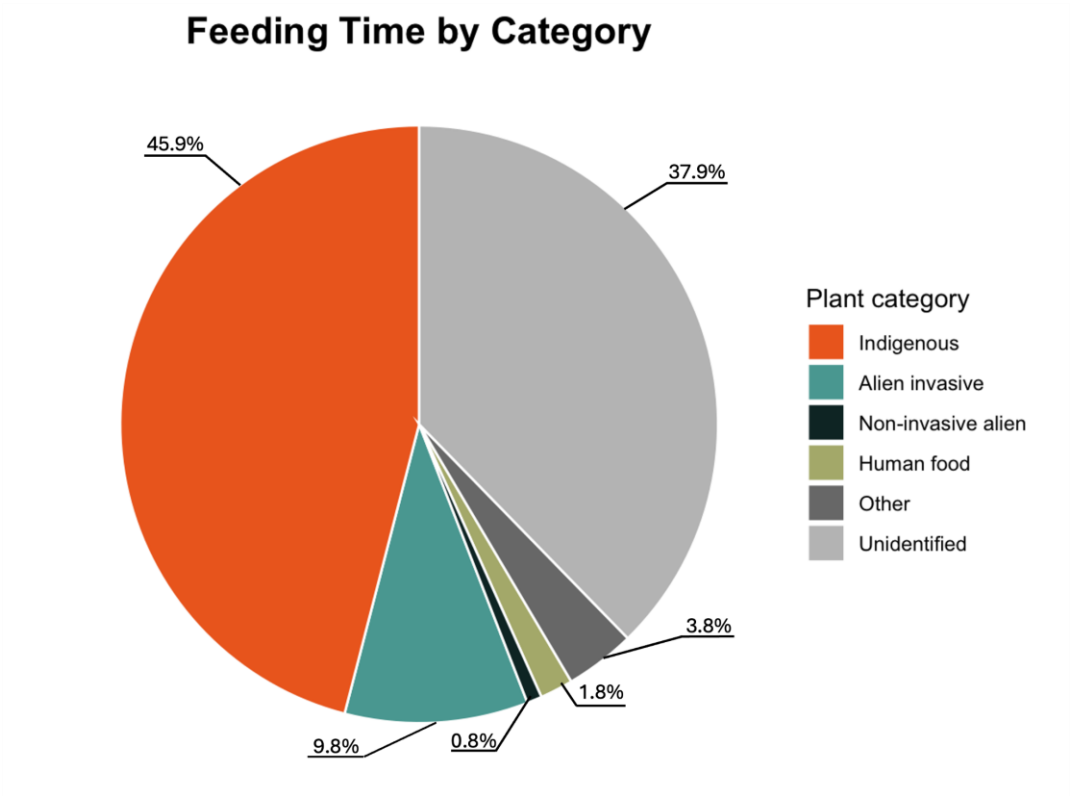


Figure 1: Percentage of total feeding time on six categories of foods. In addition to the three plant food categories of focus, here we include also Human foods, Other (e.g. non-plant items like fungi and insects) and Unidentified foods (observers could not identify the food due to size of food or because identification was not possible given minimal evidence collected via photographs, samples or descriptions).

Discussion

In this study, we followed a group of wild chacma baboons that resides around Nature's Valley, South Africa, for two years to better understand their role in the local ecosystem. We identified 90 plant species eaten by the baboons over this time including a mix of indigenous, non-invasive alien, and alien invasive plants. Plant parts that were eaten by the baboons included fruits, leaves, and flowers. Of those plants that we were able to positively identify, most were indigenous to the forest, whereas fewer alien species were eaten by the baboons. Further, although a higher proportion of their diet consisted of indigenous species during the summer compared to other seasons, baboons ate a higher proportion of indigenous compared to alien species year-round.

The heavy reliance on indigenous plant species in the Nature's Valley baboons' feeding time suggests not only that the baboons rely on indigenous plants but also that the baboons may play an important role in seed dispersal in Afrotemperate forest (**Figure 1**). Though beyond the scope of this study, anecdotally, the observation team has seen whole seeds in baboon faeces after observing defecation. Additionally, baboons in this study group have a known territory of at least 4 km^2 and can travel multiple kilometres per day. By consuming fruits and dispersing seeds through defecation, it is possible that the baboons are positively contributing to forest regeneration through the

dispersal of indigenous plants; baboons at other sites have been shown through faecal analyses to disperse seeds of indigenous plants (Tew *et al.,* 2018).

The top plant foods consumed by the baboons included *Trichocladus crinitus* (black witch hazel), *Rhoicissus tomentosa* (forest grape), and *Curtisia dentata* (assegai) throughout the year. While these plants fruit seasonally, the year-round reliance on them by local baboons suggests not only their importance to baboon nutrition but also that baboons may spread the seeds of these indigenous plants whenever they are fruiting. Accordingly, results show that the troop spends significantly more time feeding on indigenous species in the winter and summer months when compared to spring and autumn months. Early indications point to seasonal fruiting cycles as an explanation for this difference that we plan to investigate further with continued research.

In addition to consuming fruits and swallowing their seeds, we observed that the baboons can serve as seed predators. For example, the Nature's Valley baboons were observed extracting with their teeth, chewing and destroying pine nuts from pine cones. This was not necessarily plant-species specific, though, as with other sites, sometimes baboons simply swallowed seeds and other times they chewed and destroyed the seeds of the same species (Kunz & Linsenmair, 2008).

Consumption of alien invasive species was relatively low compared to indigenous food sources and this was consistent throughout the year, with no significant seasonal shifts. The alien invasive plant species the baboons fed on the most was *Cenchrus clandestinus* (kikuyu grass). This species can be spread by propagation or seed. However, as the seeds are small and delicate, they may be particularly susceptible to destruction within the digestive tract. This is an important avenue for future work including detailed investigation into whether baboons, specifically, function as seed predators or dispersers for this species.

Compared to indigenous plants that have evolved over time with local climate including seasonal change, alien plants may fruit or flower at differing times from indigenous plants (Gosper, 2004; Lediuk *et al*., 2014), providing important sources of nutrition when indigenous food availability is low. However, we found no obvious evidence for this, as baboons emphasised indigenous over alien plants during all seasons.

Overall, our results suggest that the baboons in Nature's Valley consume parts of indigenous plants, as well as non-invasive and invasive alien species. Of the species that we were able to identify, most were indigenous including multiple fruiting species, highlighting the potential for baboons to disperse seeds throughout local forests. Further, consumption of their top invasive plant food, kikuyu grass, involved eating roots and shoots which may be destructive to delicate seeds, though further examination is needed. See **Table 1** for the 11 species (and plant parts consumed by baboons) of invasive alien species.

Further work by the NVBP is ongoing in identification of seeds in Nature's Valley baboon faecal samples and in germination experiments at the NVT nursery. These faecal analyses and germination experiments are crucial to characterising seed dispersal potential, and this study of time spent feeding on different categories of plant species lays a foundation for that work. Passage of seeds through the animal gut has been previously shown in primates and in other taxa to improve seed germination via scarification of seeds and other mechanisms (Fuzessy *et al*., 2016; Traveset & Verdú, 2002). Further investigation is also

needed on the dynamics between seed dispersal potential and baboon consumption of human foods: human food consumption by other primates has been shown to reduce seed dispersal potential, especially though depositing seeds in locations unconducive to germination, like roads (Sengupta *et al.*, 2015). Understanding varied dietary patterns, including whether there is differential consumption of indigenous or alien species across baboon age and sex classes, will inform our understanding of individual variation in quantity, quality, and distance of dispersal of seeds, with implications for forest structure (Zwolak, 2018). NVT's multiple applied research programmes also facilitate ecosystem-level projects investigating seed dispersal and seed predation by multiple organisms (e.g., primary and secondary seed dispersal) in Afrotemperate forest. For applied conservation, these multiple lines of evidence regarding seed dispersal or destruction potential can then inform management decisions, including particularly urgent invasive alien eradication target species or positive seed dispersal roles to highlight in forest conservation education.

References

Albert, A., Hambuckers, A., Culot, L., Savini, T. & Huynen, M. C. 2013. Frugivory and seed dispersal by northern pigtailed macaques (*Macaca leonina*), in Thailand. International Journal of Primatology, 34(1), 170-193.

Allan, A. T., LaBarge, L. R., Howlett, C., Bailey, A. L., Jones, B., Mason, Z., ... & Hill, R. A. 2022. Patterns of predation and meat-eating by chacma baboons in an Afromontane environment. Folia Primatologica, 94(1), 13-36.

Altmann, J. & Muruthi, P. 1988. Differences in daily life between semi provisioned and wild feeding baboons. American Journal of Primatology, 15(3), 213-221.

Arendse, B. & Breen, C. 2018. Alien Invasive Plants of Nature's Valley. NVT, Nature's Valley, South Africa.

Asner, G. P., Hughes, R. F., Vitousek, P. M., Knapp, D. E., Kennedy-Bowdoin, T., Boardman, J., Martin, R.E., Eastwood, M. & Green, R. O. 2008. Invasive plants transform the three-dimensional structure of rain forests. Proceedings of the National Academy of Sciences, 105(11), 4519-4523.

Auffret, A. G. & Plue, J. 2014. Scale-dependent diversity effects of seed dispersal by a wild herbivore in fragmented grasslands. Oecologia, 175(1), 305-313.

Barnes, M. E. 2001. Seed predation, germination and seedling establishment of *Acacia erioloba* in northern Botswana. Journal of Arid Environments, 49(3), 541-554.

Calvino-Cancela, M. 2011. Seed dispersal of alien and native plants by vertebrate herbivores. Biological Invasions, 13(4), 895-904.

Chapman, C. A. & Russo, S. E. 2007. Primate seed dispersal. In C. J. Campbell, A. Fuentes, K.C. MacKinnon, M. Panger & S. K. Bearder (Eds.), Primates in Perspective (pp. 510-525). Oxford University Press.

Coetzee, H.C. 2026. Nature's Valley: History and Conservation Efforts. NVT, Nature's Valley, South Africa.

Fuzessy, L. F., Cornelissen, T. G., Janson, C. & Silveira, F. A. 2016. How do primates affect seed germination? A meta-analysis of gut passage effects on neotropical plants. Oikos, 125(8), 1069-1080.

Gosper, C. R. 2004. Fruit characteristics of invasive bitou bush, *Chrysanthemoides monilifera* (Asteraceae), and a comparison with co-occurring native plant species. Australian Journal of Botany, 52(2), 223-230.

Gross-Camp, N. D. & Kaplin, B. A. 2011. Differential seed handling by two African primates affects seed fate and establishment of large-seeded trees. Acta Oecologica, 37(6), 578-586.

Hamilton III, W. J., Buskirk, R. E. & Buskirk, W. H. 1978. Omnivory and utilization of food resources by chacma baboons, *Papio ursinus*. The American Naturalist, 112(987), 911-924.

Howe, H.F. 1986 Seed dispersal by fruit-eating birds and mammals. In: Murray, D.R. (ed.) Seed Dispersal. Academic Press, Sydney, Australia, pp. 123-190.

Janzen, D. H. 1971. Seed predation by animals. Annual Review of Ecology and Systematics, 2(1), 465-492.

Janzen, D. H. 1984. Dispersal of small seeds by big herbivores: foliage is the fruit. The American Naturalist, 123(3), 338-353.

Johnson, C. A., Raubenheimer, D., Rothman, J. M., Clarke, D. & Swedell, L. 2013. 30 days in the life: daily nutrient balancing in a wild chacma baboon. PLOS ONE, 8(7), e70383.

Jordano, P. 2000. Fruits and frugivory. In: Seeds: The Ecology of Regeneration in Plant Communities, 2nd edn (ed. M. Fenner). CABI, Wallingford, UK, pp. 125–166.

Kunz, B.K. & Linsenmair, K.E. 2008. The role of olive baboons (*Papio anubis*, Cercopithecidae) as seed disperser in a savanna-forest mosaic of West Africa. Journal of Tropical Ecology, 235-246.

Kunz, B. K. & Linsenmair, K. E. 2010. Fruit traits in baboon diet: a comparison with plant species characteristics in West Africa. Biotropica, 42(3), 363-371.

Lambert, J. E. & Garber, P. A. 1998. Evolutionary and ecological implications of primate seed dispersal. American Journal of Primatology, 45(1), 9-28.

Lediuk, K. D., Damascos, M. A., Puntieri, J. G. & Svriz, M. 2014. Differences in phenology and fruit characteristic between invasive and native woody species favor exotic species invasiveness. Plant Ecology, 215(12), 1455-1467.

Lieberman, D., Hall, J. B., Swaine, M. D. & Lieberman, M. 1979. Seed Dispersal by Baboons in the Shai Hills, Ghana. Ecology, 60(1), 65–75.

Malo, J. E. & Suárez, F. 1995. Herbivorous mammals as seed dispersers in a Mediterranean Dehesa. Oecologia, 104(2), 246-255.

Norconk, M. A., Grafton, B. W. & McGraw, W. S. 2013. Morphological and ecological adaptations to seed predation–a primate-wide perspective. Evolutionary biology and conservation of Titis, Sakis and Uacaris, 55-71.

Pamla, M., Vukeya, L.R. & Mokotjomela, T.M. 2024. The Potential of Foraging ChacmaBaboons (*Papio ursinus*) to Disperse Seeds of Alien and Invasive Plant Species in the Amathole Forest in Hogsback in the Eastern Cape Province, South Africa. Diversity, 16(3), 136.

Paterson, J. D. 2006. Aspects of diet, foraging, and seed predation in Ugandan forest baboons. In Primates of Western Uganda (pp. 75-92). New York, NY: Springer New York.

Pebsworth, P. 2020. Feeding ecology of chacma baboons (*Papio ursinus*) living in a human modified environment. African Journal of Ecology, 58(2).

R Core Team 2025. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing.

Selwyn, M., Pino, J. & Espelta, J. M. 2023. Disentangling the importance of intrinsic and extrinsic seed dispersal factors in forest restoration success: a global review. Restoration Ecology, 31(4), e13868.

Sengupta, A., McConkey, K. R. & Radhakrishna, S. 2015. Primates, provisioning and plants: impacts of human cultural behaviours on primate ecological functions. PLOS ONE, 10(11), e0140961.

Swart, R. C., Geerts, S., Geldenhuys, C. J., Pauw, J. & Coetzee, A. 2024. Weak latitudinal trends in reproductive traits of Afromontane forest trees. Annals of Botany, 133(5-6), 711-724.

Tew, E., Landman, M. & Kerley, G.I.H. 2018. The contribution of the chacma baboon to seed dispersal in the eastern Karoo, South Africa. African Journal of Wildlife Research, 48(2), 1-12.

Traveset, A. & Verdú, M. 2002. A meta-analysis of the effect of gut treatment on seed germination. In Seed dispersal and frugivory: ecology, evolution and conservation. Third International Symposium Workshop on Frugivores and Seed Dispersal, São Pedro, Brazil, 6-11 August 2000 (pp. 339-350). Wallingford UK: CABI publishing.

Vitousek, P. M., D'Antonio, C. M., Loope, L. L., Rejmanek, M. & Westbrooks, R. 1997. Introduced species: a significant component of human-caused global change. New Zealand Journal of Ecology, 1-16.

Von Breitenbach, F. 1972. Indigenous forests of the southern Cape. Veld & Flora, 58(1), 18.

Wunderle Jr, J. M. 1997. The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. Forest ecology and management, 99(1-2), 223-235.

Zwolak, R. 2018. How intraspecific variation in seed-dispersing animals matters for plants. Biological Reviews, 93(2), 897-91.



An assessment of the health and size class distribution of black stinkwood (*Ocotea bullata*), an endangered tree species, in Hebron Private Nature Reserve, Nature's Valley

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How to cite this article:

Swart, R.C. 2026. An assessment of the health and size class distribution of black stinkwood (*Ocotea bullata*), an endangered tree species, in Hebron Private Nature Reserve, Nature's Valley. *Nature's Letters* **3**:46-59. URL: https://nvt.org.za/natures-letters/

Abstract

Black stinkwood (*Ocotea bullata*) is an endangered, nationally protected tree species endemic to South Africa, restricted to evergreen forests. Hebron Private Nature Reserve (HPNR), a 15-hectare indigenous southern Cape platform forest declared in 1996, likely experienced historical timber extraction due to its accessibility and flat terrain. The current status of *O. bullata* within HPNR is unknown, yet the reserve has a mandate to ensure its persistence. This study aimed to determine (1) the size-class distribution and (2) the health status of *O. bullata* in HPNR. An inverse J-shaped population structure was predicted, indicating many young individuals and fewer large trees, and it was expected that mature trees will show better health and less moisture stress than smaller individuals. Mature trees were limited to heights under 22m and stem diameters under 29cm. The population broadly followed an inverse J-shaped distribution, although seedlings were scarce, partially supporting the first prediction. Contrary to expectations, intermediate and larger trees (>4 m) showed greater signs of moisture deficiency than smaller individuals (<4 m). Compared with reference forests, the largest trees in HPNR were relatively small, suggesting possible historical removal of larger individuals. Despite this, the population appears stable and intact, with demographic patterns consistent with long-term persistence over several decades. Estimated ages of associated canopy species and the likely history of exploitation further support the idea that past disturbance explains the reduced size of *O. bullata* trees relative to old-growth forests. Overall, *O. bullata* in HPNR shows a healthy, self-sustaining population structure, though legacy effects of past harvesting remain evident.

Introduction

Black stinkwood (*Ocotea bullata*) is an indigenous tree species to the southern Cape, and a South African endemic. Its distribution is from mountain forests in the southwestern Cape, along the southern and eastern coastlines towards the Limpopo Province (Venter, 2011; Swart *et al*., 2024). It is a strictly forest species, associated with evergreen, moist forests. Taxonomically, *O. bullata* is placed in the Lauraceae family, linking it phylogenetically with avocado, cinnamon, and bay laurel (Hammel, 1986). The common name, stinkwood, refers to the pungent smell of freshly cut wood. The scientific name, *bullata*, refers to the visible swellings (bullae) in the axils of the lower lateral veins of the leaves. Apart from making it easy to identify, these swellings, called domatia, can house small,

predatory mites, which are purported to play an anti-herbivory role (Pauw, 1992). The leaves are alternate, leathery and simple, with wavy margins. New foliage is often reddish-green.

The tree can grow into large specimens, often multi-stemmed with a massive base, able to reach heights of ca. 30m. In mature mountain forests, a multi-stemmed tree with a massive base is usually either black stinkwood or white alder (*Platylophus trifoliatus*) (Von Breitenbach, 1974; personal observation). A single tree can have flowers which are male, female or bisexual. Flowers are small, yellowish-green, and are borne in loose clusters in the axils of the leaf stalks near the tips of the branches. Typically, flowers are found on the higher branches of the tree and rarely found in the understorey or subcanopy layers (Venter, 2011). The main flowering period is between December and February (Venter, 2011), although earlier work has suggested that flowering occurs sporadically due to unknown abiotic triggers (Phillips, 1924). The flowering and subsequent pollination of *O. bullata* could be the topic for a whole doctorate thesis: John Phillips (1924) confidently stated that the flowers are not wholly insect pollinated. No honeybees, carpenter bees, wasps, flies or moths readily visit its flowers. Minute spiders, weevils and parasitic plant lice (psyllids) were observed on flowers but are unlikely to be important pollinators (Phillips, 1924). The observation of a range of flowers borne on any one tree, between functionally male (carrying stamens with pollen), functionally female (carrying pistils receptive of pollen) or bisexual remains unstudied. There are also hermaphroditic flowers, which contain stamens but rudimentary pistils only, or pistils but rudimentary stamens only. The exact proportions and potential fluctuations remain undetermined, and the ecological and evolutionary significance of this intriguing flowering system is not well understood and will remain unstudied without long-term canopy access facilities.

The fruit looks like an acorn, starting out yellowish green and turning purple-brown, often when lying on the forest floor. It contains a single, large soft seed that is about 20mm long. Fruits appear between March to June but can remain on the tree as green drupes for many months. Viable seed is not easy to come by, as fruits are often attacked by an undescribed, internal feeding moth species in the *Stathmopoda* genus (Swart, unpublished research). Moreover, Phillips (1924) noted that apart from Lepidopterous (i.e., moth) attack on fruits, a *Fusarium* fungus also causes damage to fruits. Reserves in the fruits include proteoses, peptones (proteins), non-reducing sugars and a heavy, clear, green, aromatic oil (Phillips, 1924). Most fruits fall immediately below the crown zone of the parent plant, with as many as 60 fruits per square meter under the crown zone of large individuals. Yet, recruitment is typically low beneath tree crowns, possibly due to a concentration and higher density of antagonistic fungi and insects, as mentioned, which is an archetypal negative density-dependence effect in ecology (Phillips, 1924; Janzen, 1970; Connell, 1971).

Ocotea bullata is adapted to a range of soil moisture conditions, varying from 25% - 70% soil moisture (Phillips, 1924). It was noted that regeneration of seedlings was highest in soil with 30%-40% soil moisture, and lowest in moister soils (Phillips, 1924). Severe drought periods, however, increase competition with neighbouring tree species, notably black witch hazel (*Trichocladus crinitus*), and can lead to growth retardation or in serious cases, complete dieback (Phillips, 1924). Yet, growth of *O. bullata* is better on drier soils compared to heavy, moist clays, based on increased aeration and decreased acidity (Phillips, 1924). The main dispersers of *O. bullata*

fruits are the African olive pigeon (*Columba arquatrix*) and Knysna turaco (*Tauraco corythaix*), with endocarps remaining undigested when moving through the gut and allowing for good regeneration where they are discarded (Phillips, 1924). Lastly, in terms of recruitment, it is estimated that only 0.5-1%, if not less, of all fruits produced eventually produce seedlings (Phillips, 1924), which are then themselves subject to competition to survive biotic and abiotic threats in a quest to reach the canopy.

Ocotea bullata has exceptional timber that is heavy and fine-textured. Due to past exploitation, it is classified as endangered and a nationally protected tree species. In the not-so-distant past, its timber was used for wagon-building, railway sleepers and ship building. Notably, George Rex's brig (a two-masted sailing vessel) was built entirely from *O. bullata* timber (Venter, 2011), and was in use from 1831 until 1844, when it ran aground at Crackington Haven, near Cornwall in England. The timber is not only durable, but also has lovely colour to it, ranging from light cream to black. As early as 1818, the Cape Government declared '*stinkwood timber in the forests adjacent to the Knysna, should be secured solely for Government purposes and on no account be permitted to be cut for the use of any individuals*' (King, 1939). A few decades later, in 1880, it was estimated to be 5-6 times more valuable than yellowwood timber, indicating the removal of vast quantities (King, 1939). Yield dropped substantially from the 1800s to the 1930s. Between 1931-1938, for example, estimates report 102 158 cubic feet (ca. 2 892 cubic metres) of *O. bullata* to have been removed (King, 1939). Yet, between ca. 1772 to 1939, an estimated 4 250 000 cubic feet (120 346 cubic metres) were removed. Lastly, in 1900 approximately 13% of *O. bullata* trees were mature enough for marketable timber; by 1928, it decreased to 1.7% (King, 1939).

Apart from its timber, the species is also much-used in traditional medicine and prone to bark harvesting (Vermeulen *et al*., 2012). Recent research has shown that stem bark contains active phytochemicals with diverse pharmacological potential, beneficial for diabetes and inflammation (Ogundajo *et al*., 2018). A comparison of assegai (*Curtisia dentata*), Cape beech (*Rapanea melanophloeos*) and *O. bullata* in terms of their response to bark harvesting, revealed that *O. bullata* showed the best bark regrowth, via phellogen edge development (Vermeulen *et al*., 2012). This ability is linked to the tree's resilience to fire (Vermeulen *et al*., 2012). Thus, sustainable bark harvesting is possible with correct management.

Hebron Private Nature Reserve (HPNR) was formally declared in 1996. It is a 15-hectare indigenous forest, classified as a southern Cape platform forest (Geldenhuys, 1993). The dominant tree species in HPNR are real yellowwood (*Podocarpus latifolius*), saffron (*Elaeodendron croceum*) and ironwood (*Olea capensis macrocarpa*) (internal report). The last potential significant timber extraction was around the early 1900's, when woodcutters and timber merchants extracted resources in the form of individual trees (i.e., not clear-cutting whole areas) (Ivan Donion, personal communication). Record keeping was non-existent. After 1920, the property which currently encompasses HPNR has been privately managed, and timber extraction was probably minimal, although it is not well known. Some of the oldest *Podocarpus latifolius* in HPNR are estimated to be at most 225 years old. Based on previous work at Witelsbos (Geldenhuys, 1994), this might indicate the last time the forest saw a fire. Regardless, it is difficult to discern exactly how much *O. bullata* specifically

was removed from the forest, due to it being a highly valuable timber species, and record-keeping lacking in the past.

It is not known what the status of *O. bullata* is in HPNR and, as an endangered and nationally protected tree species, the reserve has a mandate to ensure the species survives into the future. No study has been done on *O. bullata* in HPNR before, and in the broader southern Cape forests, species-focused demography studies are very rare (Midgley *et al*., 2024). This study aimed to determine 1) the size class distribution of *O. bullata* in HPNR, and 2) the health status of *O. bullata* in HPNR. It was firstly predicted that the size class distribution will follow an inverse J-shaped curve, with a high abundance of young individuals (seedlings and saplings) and a few large trees, as opposed to a bell-shaped curve. Inverse J-shaped curves typically develop when site conditions suit a species, with new cohorts of seedlings establishing regularly and only a few surviving through to maturity; this is typically expected for shade tolerant, interior forest species such as *O. bullata* (Seydack *et al*., 2007). Secondly, it was predicted that mature trees will be healthier and less moisture deficient compared to seedlings and small saplings. Mature trees have access to deeper water sources, are less vulnerable to desiccating top soils during drought conditions, and have less competition for resources, having been already established.

Materials and methods

Between the 19th of August 2025 and the 19th of December 2025, systematic surveys were done within the whole HPNR. Using satellite imagery and starting along the southern border of the reserve, researchers traversed different sections of HPNR, eventually covering the entire reserve. When an *O. bullata* individual was found, it was firstly GPS tagged. Subsequently, each individual was given a unique number, starting with OB1 (*Ocotea bullata* 1) and following chronologically as new individuals were added. Each individual tree received an aluminium tag with its code. Quantified for each individual were a) its height, b) its diameter at breast height (if >5cm only), and c) its health. Tree health ranged in number from 0 (dead) to 5 (very healthy) (**Table 1**). Added to health, if trees were showing signs of being moisture deficient, specifically via the presence of having dry, brittle, brown leaves, the individual was classified as being 'moisture deficient' (**Figure 1**). Using height, individuals were classified into seedlings (<0.2m), sapling small (0.2 – 1.9m), sapling medium (2m – 3.9m), sapling large (4m – 5.9m), small mature (6m – 9.9m), medium mature (10m – 15m) and large mature (>15m).

Table 1: Individual trees of *Ocotea bullata* were assigned to a health code, ranging from 0-5, based on the descriptions provided below.

Tree health	Code	Description
Dead	0	Clearly showing no signs of life, cambium (layer beneath the bark) is hard and woody. Leaves are dry, brown or absent
Very stressed	1	Most leaves are dry and brown. Cambium still green. No obvious signs of growth.

Stressed	2	Many leaves are dry and brown, some are green. Shoots forming at base of plant. Growing tip dying back.
Mildly stressed	3	Some leaves are brown, many are green. Shoots forming at base of plant.
Healthy	4	Most leaves are green. Tree showing signs of active growth.
Very healthy	5	Tree clearly growing well and is becoming well established. Many new shoots and leaves are green and vibrant.



Figure 1: *Ocotea bullata* individuals in Hebron Private Nature Reserve showing signs of moisture deficiency. Brown, brittle leaves are eventually shed.

Statistical analyses

To evaluate tree health's relation to size classes and height, an ordinal logistic regression model (also known as a proportional odds model) was employed. This approach was selected because the response variable, *health*, is measured on an ordinal scale (0–5), where the intervals between categories are not assumed to be equal. The predictor variables included *category* (a nominal factor referring to size class) and *height* (a continuous covariate referring to tree height). The *mature large* group was treated as the baseline reference for the categorical predictor. All analyses were performed in R (v4.3.2) using the *polr* function from the MASS package. Statistical significance for the coefficients was determined by calculating p-values using a two-tailed z-test based on the estimated standard errors and t-statistics. Model fit was assessed using Akaike Information Criterion (AIC) and residual deviance.

Results

A total of 314 individual *O. bullata* were found in HPNR (**Figure 2**). Large, mature trees clustered in the northeastern section, and a significant clustering of saplings was found along the southern border (**Figure 3**). Disregarding size, the density was 20.93 individuals per hectare. The breakdown for each size class is captured in **Table 2**.

Size and class distribution

Of the 314 tree individuals, 34 (10.83%) had a diameter at breast height (DBH) of over 5cm. Large mature trees (>15m in height) revealed an average DBH of 22.6cm (±7.15cm SE), medium mature trees (10m – 15m in height) had an average DBH of 11.4cm (±5.1cm SE), small mature trees (6m – 9.9m) had an average DBH

of 7cm (±1.94cm SE), large saplings (4m – 5.9m) had an average DBH of 5.75cm (±2.88cm SE) and medium saplings (2m – 3.9m) had an average DBH of 5.5cm (±3.89cm SE) (**Figure 4**). None of the small saplings (0.2m – 1.9m) had a DBH of over 5cm and were thus not measured.

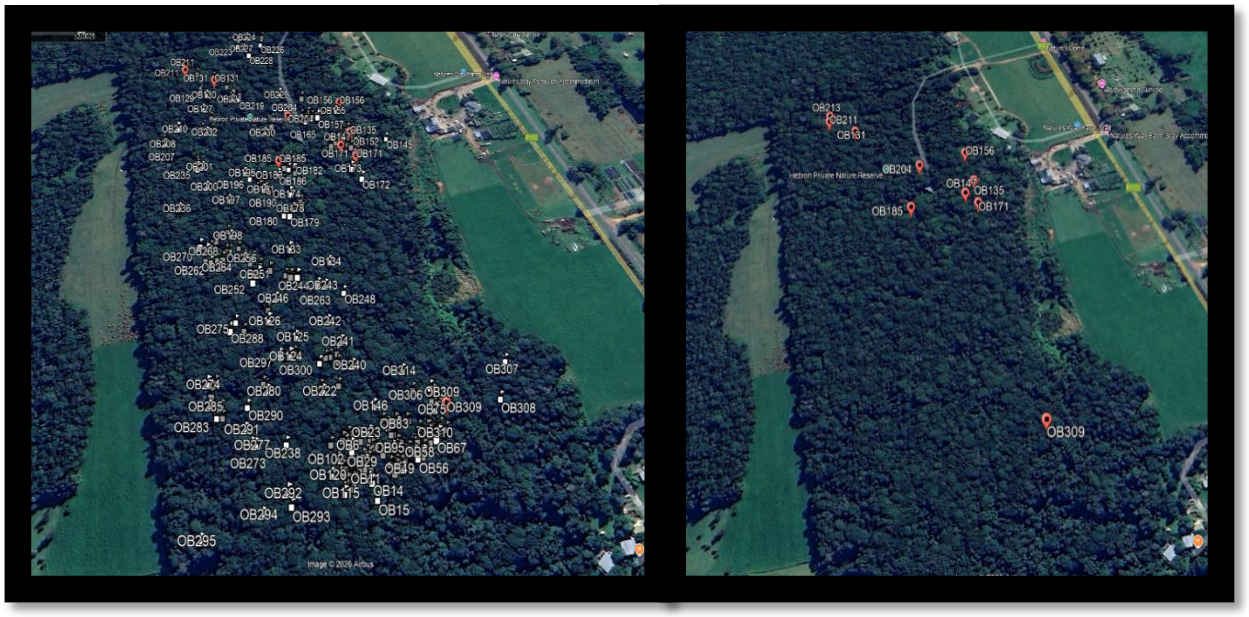


Figure 2: A total of 314 individual *Ocotea bullata* trees were encountered and surveyed in HPNR. Large, mature trees of over 15m are indicated with red markers.

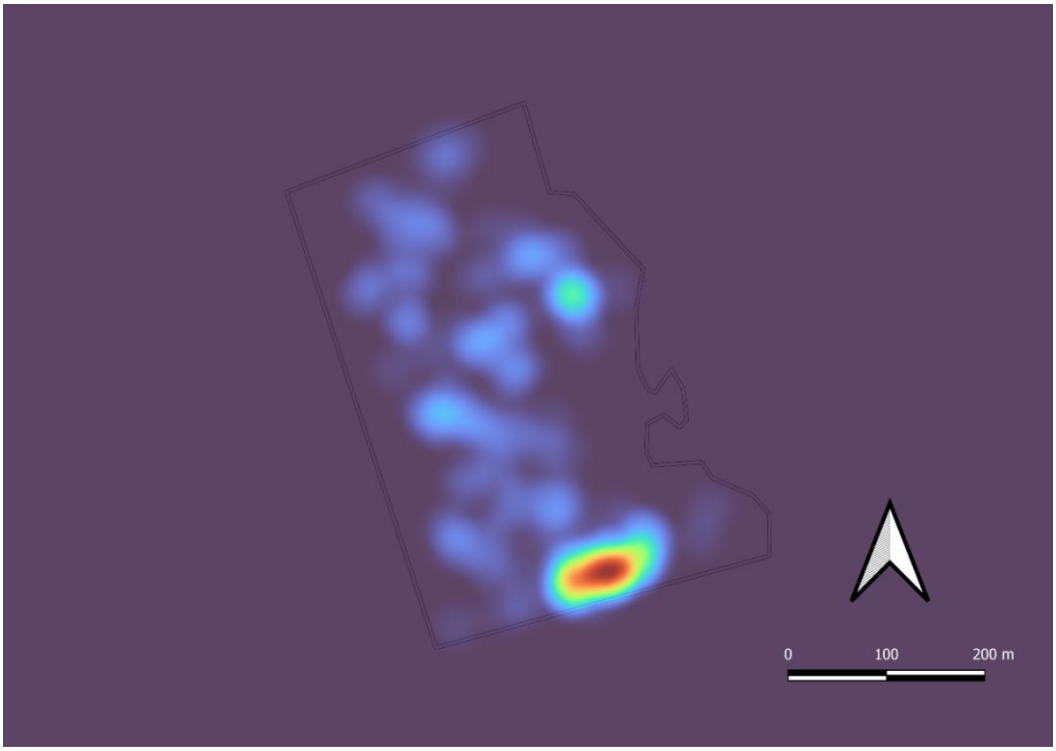


Figure 3: Kernel density heat map for *Ocotea bullata* indicating a significant cluster of saplings along HPNR's southern border, and a smaller cluster of large trees in the northeastern section.

Table 2: Density summary for size classes of *Ocotea bullata* in HPNR.

Size class	Height range	Density (per ha)
Seedling	<0.2m	0.13
Small sapling	0.2-1.9m	7.27
Medium sapling	2-3.9m	8
Large sapling	4-5.9m	2.33
Small mature	6-9.9m	1.87
Medium mature	10-15m	0.66
Large mature	>15m	0.66
All		**20.93**

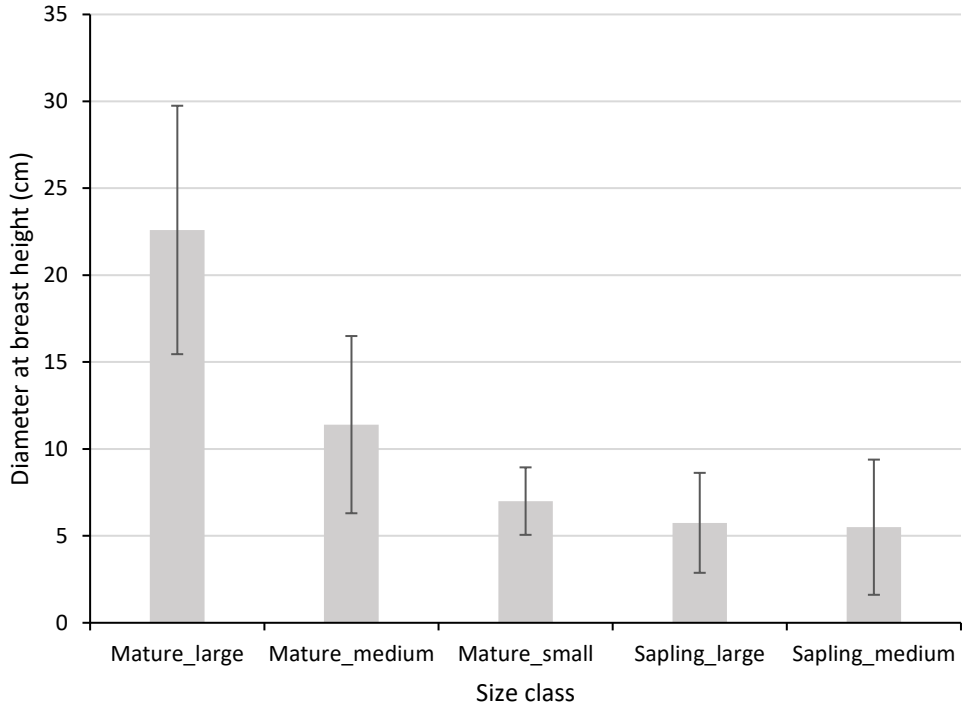


Figure 4: Average diameter at breast height (DBH) for *Ocotea bullata* surveyed in HPNR grouped into five size class categories.

The tallest tree recorded was 22m. Of the mature trees, 10 were taller than 15m (3.18%), 10 were between 10m – 15m (3.18%), and 28 were between 6m – 10m (8.92%). We found 35 large saplings (11.15%; 4m - 6m tall), 120 medium saplings (38.22%; 2m – 4m) and 109 small saplings (34.71%; 0.2m – 2m). Only 2 seedlings were observed (<0.2m). An inverse J-shaped curve was obtained, both for size classes (**Figure 5**) and for DBH (**Figure 6**), with one notable exception: a paucity of seedlings. Along the inverse J-shaped curve, the biggest drop in size classes was between medium saplings (120 individuals) and large saplings (35 individuals); thus, saplings rarely reached over 4m in height. Yet, the difference between seedlings observed versus small saplings observed was the largest, from 2 (seedlings) to 109 (small saplings).

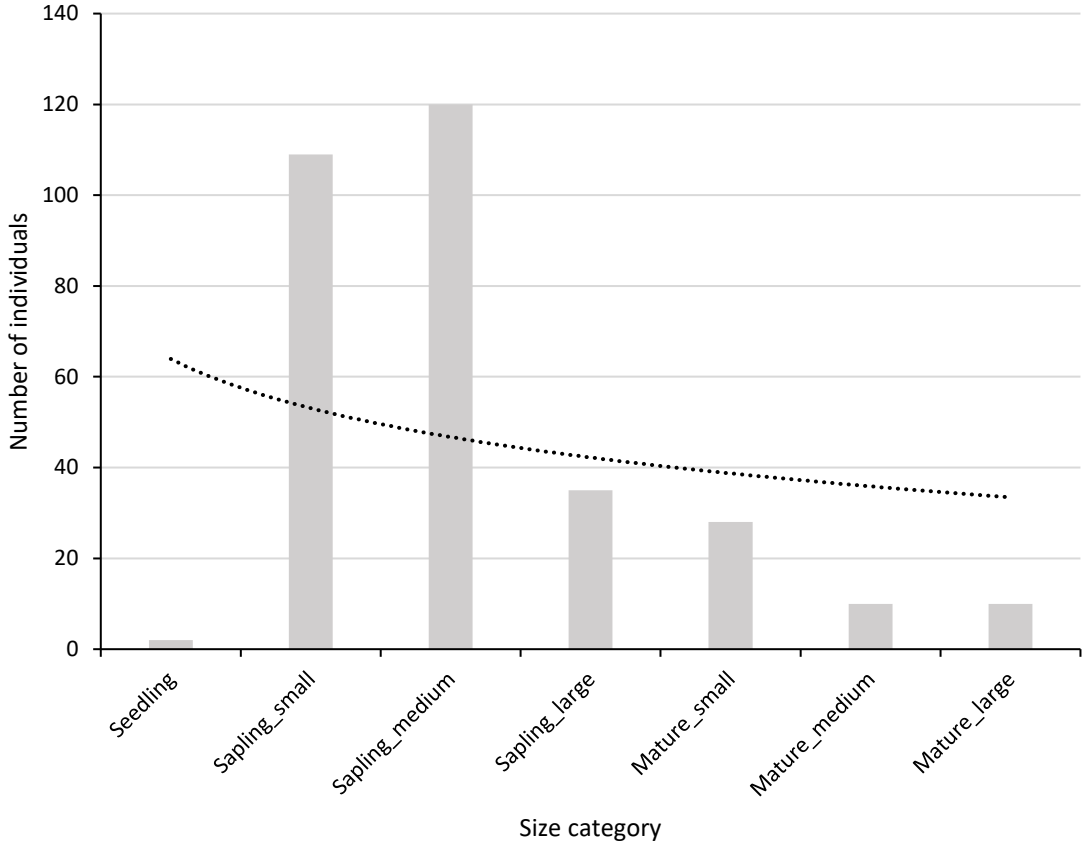


Figure 5: Size class distribution for *Ocotea bullata* surveyed in HPNR showing the number of individuals within each size class category ordered from small to large.

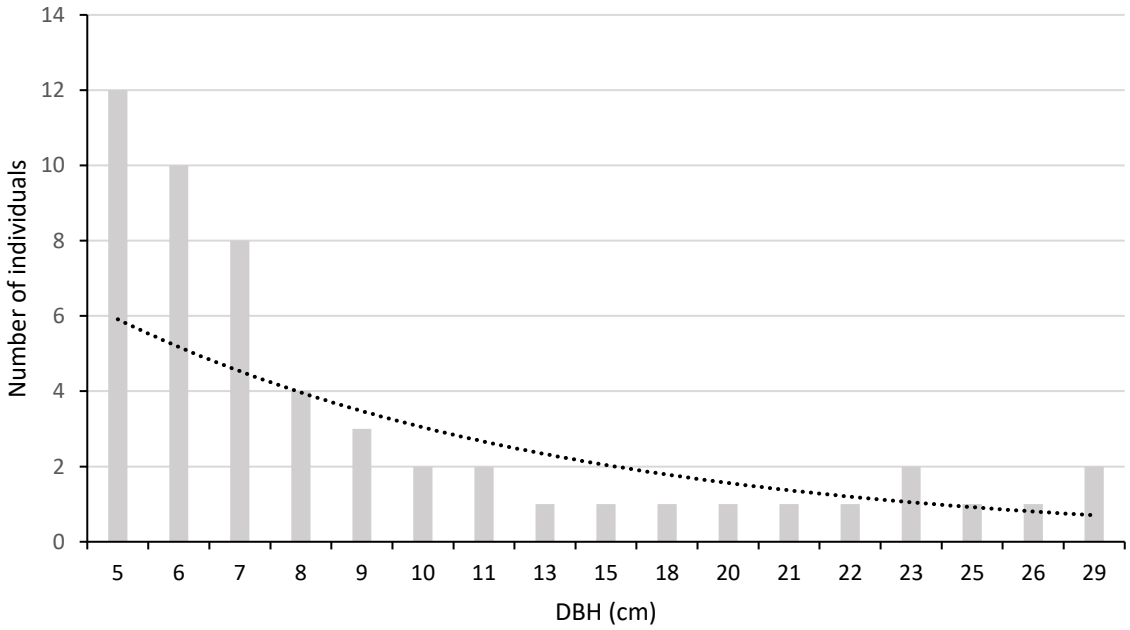


Figure 6: Diameter at breast height (DBH) distribution for *Ocotea bullata* surveyed in HPNR showing the number of individuals for each DBH class.

Health

The majority of *O. bullata* individuals (of all size classes pooled) were between healthy (category 4) and very healthy (category 5) (**Figure 7**). The breakdown of size classes for individuals that were mildly stressed (category 3, n = 44), showed that it was dominated by saplings, namely small saplings (23%), medium saplings (36%), and large saplings (25%). Small mature trees (14%) and medium mature trees (2%) accounted for the rest of the mildly stressed trees. The breakdown of size classes for individuals that were stressed (category 2, n = 5) showed that it consisted of one small sapling (20%), 2 medium saplings (40%) and 2 small mature trees (40%). Only one individual out of the 314 was dead, which was a medium sapling.

Only for trees in the large sapling (average health = 3.74) and small mature (average health = 3.67) categories, i.e., height range between 4m – 9.9m, did the average health fall below 4 (**Table 3**; **Figure 8**). Statistically, small mature trees and large saplings showed the lowest relative health (4m – 9.9m), which was marginally significant (**Table 3**).

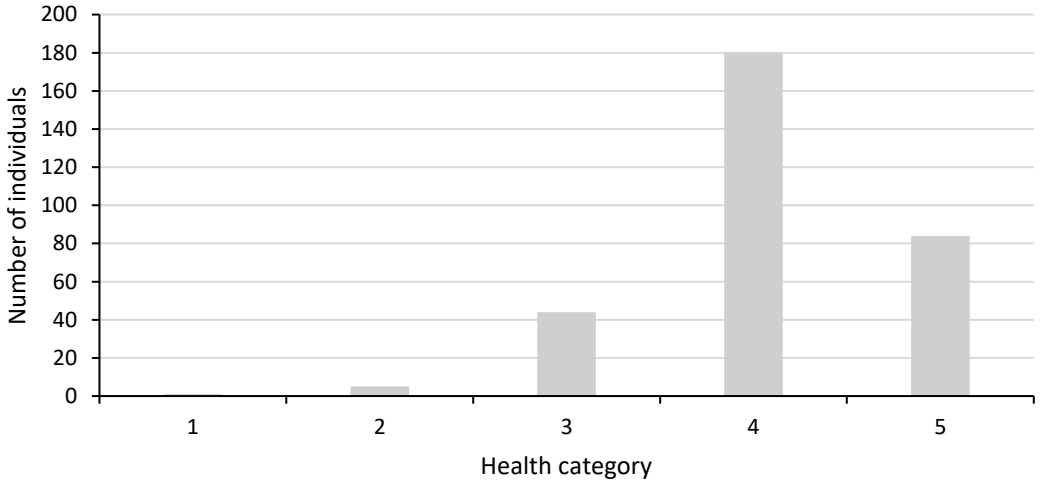


Figure 7: Number of *Ocotea bullata* individuals within each health category regardless of size class.

Table 3: Results of the ordinal logistic regression showing the relation between size class, height and tree health for *Ocotea bullata* surveyed in HPNR.

Predictor variable	Value	Std. Error	t value	p value
Mature_medium (10-15m)	-1.68	1.23	-1.36	0.17
Mature_small (6-9.9m)	-3.75	1.86	-2.02	**0.04***
Sapling_large (4-5.9m)	-4.27	2.19	-1.94	**0.05.**
Sapling_medium (2-3.9m)	-3.46	2.43	-1.42	0.15
Sapling_small (0.2-1.9m)	-3.35	2.65	-1.26	0.21
Seedling (<0.2m)	-2.97	3.10	-0.96	0.34
Height	-0.22	0.14	-1.55	0.12

.p<0.1;*p<0.05; **p<0.01; ***p<0.001

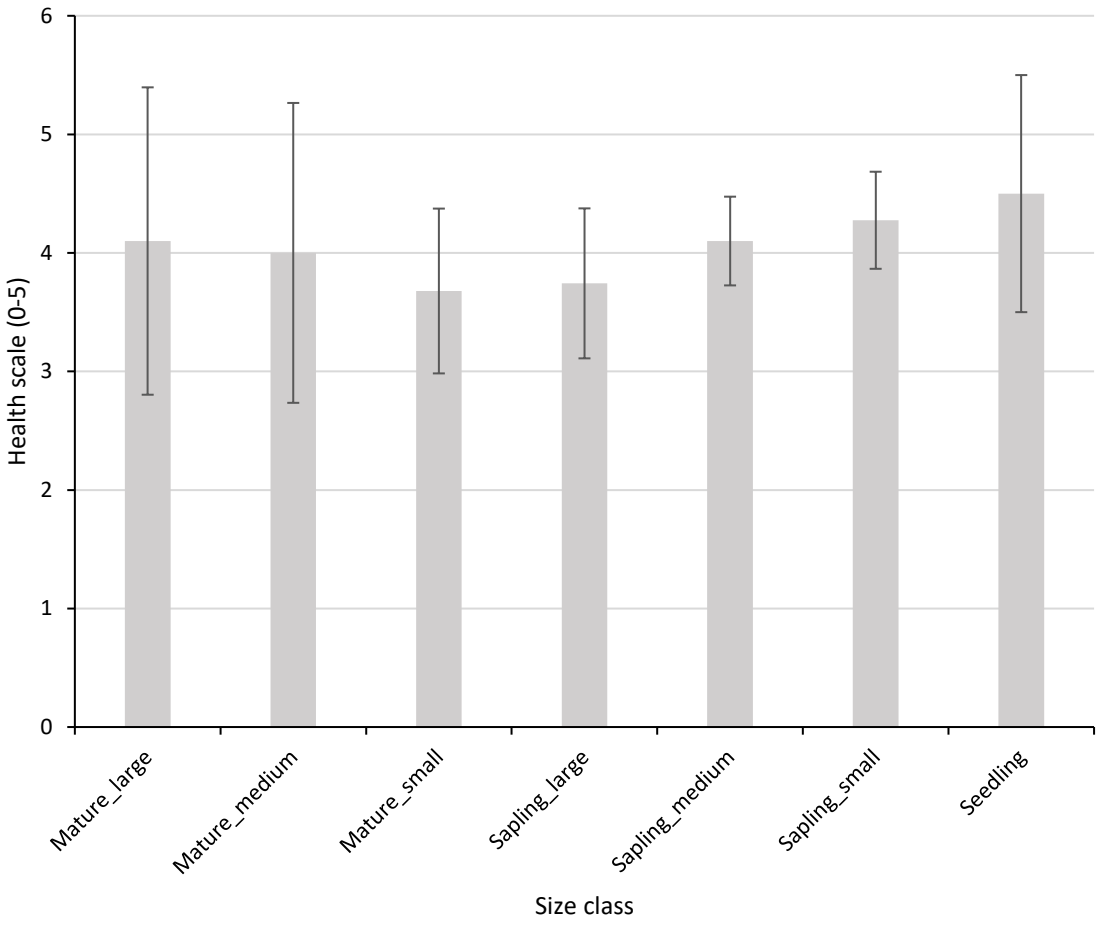


Figure 8: Average health status for different size classes of *Ocotea bullata* surveyed in HPNR.

When assessing the effect of moisture deficiency on different size class categories (**Table 4**), individuals of *O. bullata* had a higher chance to be moisture deficient in the *small mature category*, with 75% of tree individuals showing brittle, dry, brown leaves. In general, for tree individuals 4m and taller, between half and three-quarters of individuals showed signs of being affected by moisture deficiency. This is contrasted with the findings for saplings under 4m: trees had between 16.5% and 37.5% chance of showing signs of moisture deficiency, with none of the two seedlings showing browning of leaves (**Table 4**).

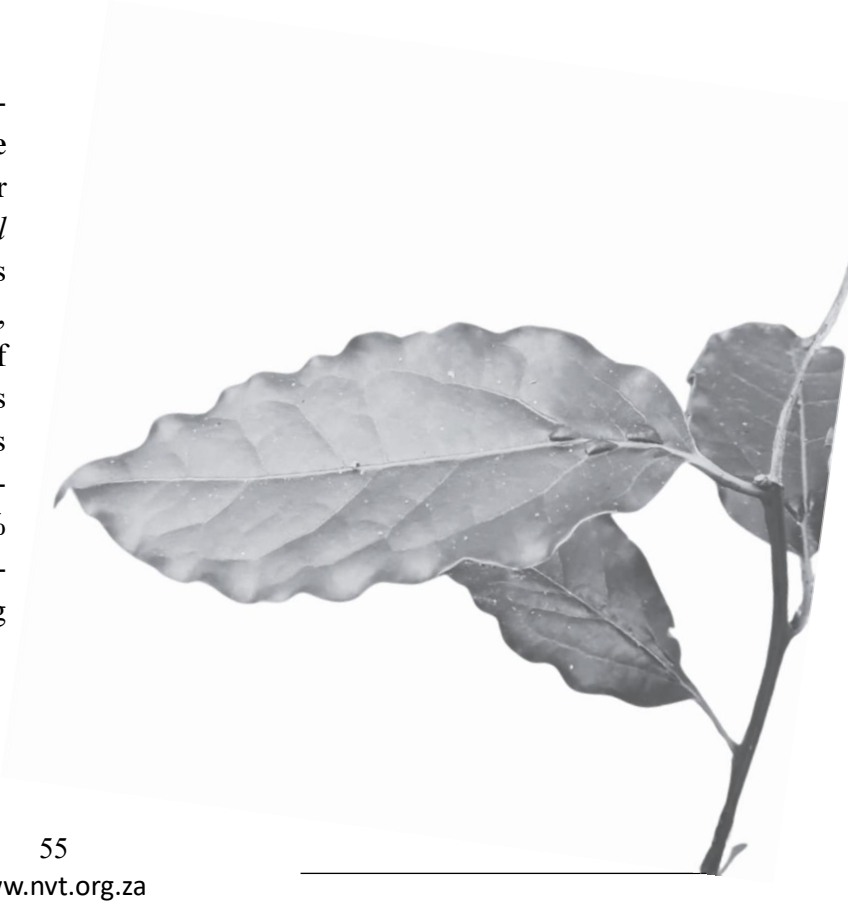


Table 4: Summary of the breakdown of diseased, moisture deficient or dead trees for the various size class categories, showing the number of trees within each category as well as the proportion per size class.

Size category	Height range	Total	Moisture deficient (n)	Moisture deficient (proportion)	Diseased (n)	Diseased (proportion)	Dead (n)	Dead (proportion)
Mature_large	>15m	10	5	50%	0	0%	0	0%
Mature_medium	10-15m	10	5	50%	0	0%	0	0%
Mature_small	6-9.9m	28	21	**75%**	0	0%	0	0%
Sapling_large	4-5.9m	35	21	**60%**	0	0%	0	0%
Sapling_medium	2-3.9m	120	45	**37.5%**	1	0.83%	1	0.83%
Sapling_small	0.2-1.9m	109	18	**16.51%**	0	0%	0	0%
Seedling	<0.2m	2	0	0%	0	0%	0	0%



Discussion

The findings presented here provide a rare assessment of *Ocotea bullata* population demographics in an indigenous, southern Afrotemperate forest. Mature individuals in HPNR are restricted to less than 22m in height and 29cm in stem diameter. The first prediction was met, in that the population followed an inverse J-shaped pattern if one disregards the paucity of seedlings. The second prediction was not met: intermediate - larger trees (>4m) seemed to be more affected by moisture deficiency compared to smaller trees (<4m).

The largest individual in HPNR was 22m tall, with a DBH of 29cm. To put the population in HPNR into perspective: at Diepwalle, north of Knysna, the largest *O. bullata* specimen had a DBH of 112cm (nearly four times larger – Phillips, 1924) and at Beervlei, northwest of Knysna, the largest specimen had a DBH of over 80cm (Lübbe & Geldenhuys, 1990). It is also known that *O. bullata* can grow up to 30m in height (Venter, 2011). Thus, the largest individuals in HPNR are relatively small when compared to reference forest sites, which opens up the possibility of larger individuals being extracted in the past. *Ocotea bullata* may take many decades to reach the forest canopy (Phillips, 1924). In a shaded forest (where most saplings and trees were recorded in the present survey), growth rates are slower than in sunlit gaps. If extraction of timber was happening ca. 100-120 years ago in the current borders of HPNR, the effects thereof could still be evident in species' size classes. *Ocotea bullata* is a slow-growing species, with previous research showing tree increments of between 0.5mm to 2.4mm diameter per annum (Seydack *et al*., 2007).

Due to factors like specific microhabitat requirements (e.g., in its pole sapling stage it is a light-demander, but in its seedling and small sapling stage it can be susceptible to damage from insolation – Phillips, 1924), fungal, insect and pathogen attack (e.g., *Phytophthora*, a root pathogen – Lübbe & Geldenhuys, 1990), not many seedlings survive through to the canopy layer. The earliest work on *O. bullata* was done by Phillips (1924), who further listed dry bergwinds as having negative effects on seedlings and young shoots of adult trees, which shrivel and die if crowns are over-exposed. If a major disturbance disrupted the historic size classes of a population in a stand or a forest, the natural antagonistic factors present in the system, such as fungi, insects, drought or dry winds, could significantly delay population recovery. Removing most mature trees through logging, causes the population to rely on seeds, seedlings, saplings or pole stages to mature first before new recruitment is possible. However, in the case of *O. bullata*, if the base is left, trees can resprout (Coert Geldenhuys, personal communication), with recent work showing an 85% survival rate post-fire of trees with an average DBH of 35cm, via resprouting (Giddey *et al*., 2022).

If seedlings are disregarded in the current study, a clear inverse J-shape pattern emerges. The inverse J-shaped pattern for *O. bullata* in HPNR was similarly found in a previous demography study near Diepwalle, north of Knysna, and in Lelievlei Forest, Gouna, where individuals of *O. bullata* became gradually less abundant as trees increased in size (Phillips, 1924; Midgley *et al*., 2024). An inverse J-shaped curve is typically indicative of stable, old-growth forest systems, wherein the number of individuals is greatest for smaller trees, sharply declining with increased size and then eventually reaching a plateau (Bastias *et al*., 2020). When mature trees reproduce, an even-aged cohort of seedlings is produced. When the site is suitable for the species, new cohorts will establish often and develop in

maturity, and due to natural senescence processes, an inverse J-shaped curve is formed (Geldenhuys *et al*., 2026).

However, in many past studies, the inverse J-shaped curve has not been observed. This could be due to small sampling sizes, or disturbance (Li *et al*., 2024). Contrasted to the inverse J-shaped curve, is the bell curve. Often, instead of a site suiting a species, we observe that suitable sites are widely spaced in time. For example, a disturbance in a forest causing a large canopy gap, allowing for sunlight penetration, could suit a species for a certain period in time (e.g., light demanding species). When this happens, new cohorts might establish and develop infrequently, leading to widely spaced, even-aged cohorts (i.e., bell shapes) (Geldenhuys *et al*., 2026). Within these even-aged cohorts, some stems will grow better than others, and some die off. This differential in growth and survival of even-aged cohorts forms a normal distribution curve around the mean of the cohort, i.e., forming a bell-shaped curve (Geldenhuys *et al*., 2026). Generally, however, mortality rates for younger individuals are highest, leading to observable patterns such as the inverse J-shaped curve.

The results pertaining to tree health are particularly interesting. A previous study on stinkwood health revealed that dead and/or dying trees were spread out across all size classes, and not restricted to larger or smaller tree individuals (Lübbe & Geldenhuys, 1990). The results presented here support this: although in terms of general health and not referring specifically to dead or dying trees. Trees in HPNR were healthy across size classes, however the two size classes that stood out are large sapling and small mature trees (intermediate in size), covering the height range of 4m – 9.9m. Trees in these categories had average health statuses of under 4, therefore approaching mildly stressed. Moreover, these same two categories were severely affected by moisture deficiency. It is possible that this tree size class (4m – 9.9m) focus more energy on growing taller than investing in roots compared to large mature trees (Grossnickle & MacDonald, 2018). With a low root:shoot ratio, leaves tend to lose more water via transpiration than the amount being taken up by the roots (Ding *et al*., 2025). Contrastingly, large, mature trees have deeper and more established roots, and are thereby able to reach moisture from a deeper and wider area (Lai *et al*., 2013). Compared to seedlings, it is possible that this intermediate tree size class (4m – 9.9m) has more water demand, with seedlings typically having only a few leaves and thus a lower water and energy demand (Grossnickle & MacDonald, 2018). The exact mechanisms at play remain unknown, and tagged trees will be monitored over time.

In this study, only two seedlings were found. This is not to say that seedlings are inherently rare in HPNR. Rather, it could be a flaw of the study design. Many hectares were covered by foot, and seedlings are easily overlooked given forest density. Saplings and mature trees were more easily spotted, given the wavy leaf margins and general characteristic leaf morphology of saplings, and the stem patterning of mature individuals. Regardless, it is suggested that specifically this aspect be looked at in more detail – if seedlings are indeed as rare as our survey method led us to believe, more focussed research is needed.

In conclusion, the current *O. bullata* population in HPNR is healthy and intact, following an expected inverse J-shaped curve of a stable population. Additionally, these demography patterns are expected from a population that has been stable for at least a few decades. Based on previous estimates on the age of the oldest *P. latifolius* in HPNR, and the known significance of *O. bullata* in the past, it is predicted that past disturbance explains the

relatively small size of HPNR's *O. bullata* trees compared to old-growth, reference forests.

References

Bastias, C. C., Truchado, D. A., Valladares, F., Benavides, R., Bouriaud, O., Bruelheide, H. & De la Cruz, M. 2020. Species richness influences the spatial distribution of trees in European forests. Oikos, 129(3), 380-390.

Connell, J. H. 1971. On the role of natural enemies in preventing competitive exclusion in some marine animals and in rain forest trees. Dynamics of populations, 298(312), 1-610.

Ding, R., Nóbrega, R. L. & Prentice, I. C. 2025. Global assessment of environmental and plant trait influences on root: shoot biomass ratios. Global Change Biology, 31(10), e70543.

Geldenhuys, C. J. 1993. Composition and dynamics of plant communities in the southern Cape forests. Report for DEA, 612, 57.

Geldenhuys, C. J. 1994. Bergwind fires and the location pattern of forest patches in the southern Cape landscape, South Africa. Journal of Biogeography, 21(1), 49–62.

Geldenhuys, C. J., Moll, E. J. & Swart, R. C. 2026. Evergreen forests in South Africa – their composition, biogeography, ecological dynamics and sustainable use management. South African Journal of Botany, 188, 63-98.

Giddey, B. L., Baard, J. A. & Kraaij, T. 2022. Fire severity and tree size affect post-fire survival of Afrotemperate forest trees. Fire Ecology, 18(1), 5.

Grossnickle, S. C. & MacDonald, J. E. 2018. Why seedlings grow: influence of plant attributes. New forests, 49(1), 1-34.

Hammel, B. E. 1986. Lauraceae. Selbyana, 9(1), 218-233.

Janzen, D. H. 1970. Herbivores and the number of tree species in tropical forests. The American Naturalist, 104(940), 501-528.

King, N. L. 1939. The Knysna forests and the woodcutter problem. Journal of the South African Forestry Association, 3(1), 7-15.

Lai, J., Yang, B., Lin, D., Kerkhoff, A. J. & Ma, K. 2013. The allometry of coarse root biomass: log-transformed linear regression or nonlinear regression? PLOS ONE, 8(10), e77007.

Li, Y., Li, J. & Wei, L. 2024. Stable reverse J-shaped diameter distribution occurs in an old growth karst forest. Journal of Forestry Research, 35(1), 107.

Lübbe, W. A. & Geldenhuys, C. J. 1990. Decline and mortality of Ocotea bullata trees in the southern Cape forests. South African Forestry Journal, 154(1), 7-14.

Midgley, J. J., Cramer, M. D. & Durrheim, G. 2024. Thirty years of stasis in the dynamics of the Knysna Afro montane forest, South Africa. Austral Ecology, 49(9), e13594.

Ogundajo, A. L., Adeniran, L. A. & Ashafa, A. O. 2018. Medicinal properties of Ocotea bullata stem bark extracts: phytochemical constituents, antioxidant and anti-inflammatory activity, cytotoxicity and inhibition of carbohydrate-metabolizing enzymes. Journal of Integrative Medicine, 16(2), 132-140.

Pauw, A. 1992. Mite-plant mutualism: leaf domatia of African plants house beneficial mites (Doctoral dissertation).

Phillips, J. 1924. The biology, ecology, and sylviculture of stinkwood, Ocotea bullata E. Mey: introductory studies. South African Journal of Science, 21(11), 275-292.

Seydack, A. H. W., Vermeulen, W. J., Baard, J. & Kok, H. R. 2007. Demographic Patterns of Stinkwood (Ocotea bullata) in the Forests at Weza, KwaZulu-Natal: Divergent Modes of Regeneration. South African Journal of Botany, 58(6), 505–512.

Swart, R. C., Geerts, S., Geldenhuys, C. J., Pauw, J., & Coetzee, A. 2024. Weak latitudinal trends in reproductive traits of Afromontane forest trees. Annals of Botany, 133(5-6), 711-724.

Venter, E. (2011). Trees of the Garden Route: Mossel Bay to Storms River. BRIZA Publications.

Vermeulen, W. J., Geldenhuys, C. J. & Esler, K. J. 2012. Response of Ocotea bullata, Curtisia dentata and Rapanea melanophloeos to medicinal bark stripping in the southern Cape, South Africa: implications for sustainable use. Southern Forests: a Journal of Forest Science, 74(3), 183-193.

Von Breitenbach, F. V. 1974. Southern Cape forests and trees. Pretoria: Government printer.

The "Community Conservation Cycle-of-Praxis" (version 1.0): A model aimed at improving the social outcomes in community-based conservation

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How to cite this article:

Du Plessis, S., Coetzee, H. & Nell, W. 2025. The "Community Conservation Cycle-of-Praxis": A model aimed at improving the social outcomes in community-based conservation. *Nature's Letters* 3:60-88. URL: https://nvt.org.za/natures-letters/

Abstract

To enhance positive social outcomes in Community-Based Conservation (CBC), we developed a praxis-oriented model aimed at improving community well-being and fostering behavioural change in two communities in South Africa's Western Cape, namely Kurland and Covie. Our "Community-Based Conservation Cycle of Praxis" (version 1.0) combines principles from community well-being, socio-psychological theory, and conservation practice into seven iterative steps: identifying community ideals, challenges and barriers, assets, solution development, pathway creation, and progress monitoring. By prioritising community needs, leveraging locally held assets, and incorporating environmental conditions through stewardship-oriented actions, the model strengthens the social foundations of conservation initiatives and supports community-driven change. While social outcomes remain central, environmental considerations are embedded through stewardship and monitoring. A subsequent iteration of the model (version 2.0) will explicitly focus on strengthening environmental outcomes.

Introduction

The Nature's Valley area, in the Western Cape Province of South Africa, along with many coastal communities globally, is confronted by complex social and environmental challenges such as poverty, unemployment, biodiversity loss, and climate change (Rice *et al*., 2021). Experts advocate for contemporary, people-centric approaches such as Community-based Conservation (CBC) (Esmail *et al*., 2023), particularly those led by Non-Governmental Organisations (NGOs) (Wachira *et al*., 2024), to improve these issues. However, a recent review of CBC's environmental and social impacts indicates that the shortcomings are predominantly social (Galvin *et al*., 2018) - a trend that is visible on a global level (Esmail *et al*., 2023). These social shortcomings are particularly significant given that CBC is premised on the interdependence between community well-being and conservation outcomes, where ineffective social processes ultimately undermine conservation practice and long-term environmental sustainability.

The literature indicates that several factors contribute to the ineffectiveness of CBC in producing concrete social outcomes. These factors encompass inadequate governance (Salerno *et al*., 2021), policy challenges (Rice *et al*., 2021), the framing of social dimensions (Galvin *et al.*, 2018),

the lack of a model providing actionable steps towards positive outcomes (Kaplan-Hallam & Bennett, 2017) and insufficient encouragement of behavioural change for lasting sustainability. Furthermore, insufficient integration of monitoring and evaluation processes often undermines the social efficacy of CBC initiatives (Mahajan *et al*., 2019). These limitations not only constrain social outcomes but also weaken the capacity of communities to engage meaningfully in conservation practice, comply with conservation objectives, and sustain stewardship-oriented behaviours over time.

To address some of these challenges, we set out to develop a praxis-of-change model which is aimed at producing more beneficial social outcomes, such as enhanced objective and subjective community well-being. Although the model is intentionally oriented towards strengthening social outcomes, it is designed to operate within, and contribute to, conservation practice by supporting the social conditions necessary for stewardship, compliance, and long-term conservation effectiveness. We reviewed various CBC models (e.g., Kaplan-Hallam & Bennett, 2017, Biedenweg *et al*., 2016; Britton & Coulthard, 2013), but found that individually, they often lack the detailed elements necessary to produce positive social outcomes and failed to offer the necessary practical steps to facilitate positive outcomes. In other words, although they provide valuable insights, they do not fully address the social outcomes that CBC aims to achieve. Consequently, we identified the community wellness model (Nelson & Prilleltensky, 2005; Prilleltensky & Nelson, 2002; Prilleltensky, 2001) and Totikidis' (2003) community wellness cycle-of-praxis as appropriate foundations for our model, for the following reasons: The first is designed to guide practitioners from theory to actionable outcomes, aiming to balance personal, relational and collective well-being, with an emphasis on integrating all three dimensions for true wellness, while the second, building on Prilleltensky's framework, is designed to accommodate diverse cultural contexts, facilitating its application across different community settings.

Community well-being

In the context of this article, and our chosen models, we adopted Breslow *et al*.'s (2018) definition of community well-being as: *"the state of being with others and the environment, which arises when human needs are met, when individuals and communities can act meaningfully to pursue their goals and when individuals and communities enjoy a satisfactory quality of life*". This definition is particularly relevant for CBC, as it explicitly situates human well-being within human–environment relationships, providing a conceptual bridge between social well-being and conservation practice.

In building our conceptual understanding of community well-being, we found Sirgy's dimensions of well-being and bottom-up spillover theory useful (Sirgy *et al*., 2010; Sirgy & Cornwell, 2002; Sirgy *et al*., 2000). Sirgy *et al*.'s (2010) dimensions of well-being highlight the interconnectedness of community services and conditions with an individual's overall life satisfaction and commitment to his/her community. Examples of some of these encompass family, neighbourhood, social, political, leisure, spiritual, educational, health, safety, environmental and financial well-being dimensions.

In addition, the bottom-up spillover theory, as proposed by Sirgy *et al*. (2010), merges global and domain-based views on well-being. It posits that satisfaction in specific life areas, such as community services and

basic living conditions, overflows into general satisfaction with community life. This satisfaction then extends to other life domains, such as social, family, work and spiritual well-being, thus enhancing overall life satisfaction (Sirgy *et al*., 2010; Sirgy & Cornwell, 2002). In essence, the theory highlights the interrelation of different life aspects, suggesting that enhancing certain life domains can have a positive effect on general well-being.

In a conservation setting, this spillover effect suggests that improvements in social and institutional domains may enhance community capacity and motivation to participate in conservation practice, even where environmental outcomes are not the immediate focus of intervention.

Building upon this framework, Breslow *et al*. (2018) characterised human well-being as a condition stemming from the fulfilment of human needs (which includes the meaningful pursuit of goals and a satisfactory quality of life), within the core constituents of well-being (encompassing connections, capabilities, conditions) and within overarching constituents like equity, justice, security, resilience and sustainability. To develop accurate well-being indicators, we have categorised Sirgy *et al*.'s (2010) well-being dimensions within Breslow *et al*.'s (2018) four primary constituents of human well-being, as outlined in **Table 1**.

Within a CBC setting, this framing allows conservation practice to be understood not as an external objective, but as one element embedded within the broader social conditions, capabilities and relationships that shape community well-being.

Table 1: Integration of Sirgy *et al*.'s (2010) well-being dimensions within Breslow *et al*.'s (2018) four primary constituents of human well-being.

Conditions
(Circumstances in which human needs are met)
• Economic well-being (Sirgy *et al*.'s financial well-being)
• Environmental health (Sirgy *et al*.'s environmental well-being)
• Safety and security (Sirgy *et al*.'s safety well-being)
• Human health (Sirgy *et al*.'s health well-being)
Connections
(Being with others and the environment)
• Social relationships (Sirgy *et al*.'s social well-being)
• Cultural values and identities (Sirgy *et al*.'s spiritual well-being)
• Sense of place (Sirgy *et al*.'s neighbourhood well-being)
• Access to nature (Sirgy *et al*.'s environmental well-being)
Capabilities
(Factors enabling individuals and communities to act meaningfully to pursue their goals)
• Knowledge systems (Sirgy *et al*.'s education well-being)
• Political participation (Sirgy *et al*.'s political well-being)
• Governance (Sirgy *et al*.'s political well-being)
• Activities (Sirgy *et al*.'s leisure well-being, work well-being)
Cross-cutting domains
• Equity and justice (Sirgy *et al*.'s social well-being)
• Security (Sirgy *et al*.'s safety well-being)
• Resilience and sustainability (Sirgy *et al*.'s environmental well-being)

Within a CBC context, environmental well-being is closely linked to conservation practice, as perceptions of environmental quality, access to natural resources, and environmental safety influence both individual life satisfaction and willingness to engage in conservation-related behaviours.

Our model also draws on Nelson and Prilleltensky's (2001) conceptualisation of community wellness. Central to the model of community wellness is the recognition of the interconnectedness of personal, relational and collective well-being. Achieving overall well-being within communities, they argue, will arise from the balanced satisfaction of this 'tripartite' of needs, provided that these needs are supported by appropriate values and resources. Neglecting any one of these dimensions, they cautioned, can undermine community well-being. Within CBC initiatives, this interconnectedness is particularly relevant where collective well-being is shaped by shared relationships with local natural resources and conservation-related institutions.

Based on this tripartite conceptualisation of wellness, it becomes clear that well-being encompasses more than just emotional and cognitive aspects. It is also deeply influenced by socio-political and institutional contexts, which therefore need to be taken into consideration in the context of CBC initiatives. These socio-political and institutional contexts often intersect with conservation governance structures, influencing how communities experience and respond to conservation interventions. **Table 2** illustrates Nelson and Prilleltensky's (2001) tripartite conceptualisation of wellness.

Notably, the inclusion of a healthier environment within the collective dimension reinforces the relevance of environmental conditions to broader community wellness, without positioning environmental outcomes as the primary analytical focus. Based on the tripartite conceptualisation of well-being, Prilleltensky and Nelson (2002) advocate that relying solely on individual-focused interventions is not sufficient to overcome systemic disadvantages present at the collective level. For instance, challenges related to employment in the community require interventions that not only focus on building individual skills and capacity but also target broader societal structures and institutions that affect the employability of the individual.

Table 2: Community wellness model.

**Adapted from Nelson and Prilleltensky (2005), Prilleltensky and Nelson (2002) and Prilleltensky et al. (2001)*

	Personal	**Relational**	**Collective**
Community wellness model	Sense of control over one's life, physical health, love, competence, optimism and self-esteem	Social support, affection, belonging, cohesion, collaboration, respect for diversity, and democratic participation	Economic security, social justice, adequate health and social services, low crime, safety, adequate housing and social structures (e.g. educational, recreational and transportation facilities) and a clean environment

A similar logic applies to conservation-related challenges, where individual behaviour change is unlikely to be sustained without supportive institutional, social and infrastructural conditions.

In developing our model, we therefore opted to integrate the perspectives of both the socio-psychological theories (i.e. those that mainly focus on individual-level determinants of behaviour) and the institutional theories (i.e. those that mainly focus on the role of broader societal structures and norms and their role in shaping individual and collective behaviour). The following sections are devoted to a brief overview of these two perspectives, both of which are relevant for understanding behaviour in socially embedded conservation settings.

Psychological and social theories relating to human behaviour

Comprehending human behaviour is crucial for fostering positive change, especially when promoting sustainability and encouraging behavioural change. Socio-psychological theories provide significant insights into the intricate dynamics of personal motivations, social interactions, and environmental factors that drive human actions and choices, which is instrumental in crafting effective strategies to tackle specific behavioural issues. This is particularly pertinent where behavioural change is linked to sustainability and conservation-related practices. Within the vast array of socio-psychological theories, those outlined in **Table 3** are particularly pertinent to our model.

While socio-psychological theories explain individual motivations and behaviour, institutions, defined as "the rules of the game in a society" (North, 1990), are often seen as the foundation of societies, influencing the attitudes, actions and decisions of the individuals within them. This includes not only formalised rules and regulations such as constitutions, laws, policies and property rights, but also informal rules like conventions, traditions and norms that direct behaviour (Vatn, 2015; North, 1990). These institutional structures establish expectations and meanings, thus moulding individuals' interpretations of situations and steering their choices. Vatn (2015) suggests that institutions shape attitudes and actions by defining what is perceived as the "natural" way to act (conventions), the "correct" way to act (norms) and the "sanctioned" way to act (policies and regulations).

Institutional contexts differ across various domains of human interaction, such as the market, community or family, each highlighting unique forms of rationality (Vatn, 2009). Within these settings, individuals assume roles that enforce certain norms and conventions, thereby moulding their rational behaviour (Soma & Vatn, 2014, 2010). Institutions have the power to modify beliefs and attitudes about issues, thereby influencing group actions (Stern & Dietz, 2002). Moreover, institutions do not only dictate appropriate actions but also shape what individuals can envision doing in each scenario (Hall & Taylor, 1996). Institutions comprise symbolic systems, cognitive scripts and classifications that assign meaning to circumstances and form mental models that lead to actions (Vatn, 2015; North, 1990).

Similarly, individuals' worldviews are shaped by their cultural background, personal experiences and prevailing social norms, which in turn affect their beliefs and behaviours (Berger & Luckmann, 1967). Therefore, it is evident that institutions significantly influence human motivations, preferences and values (Vatn, 2015; Hukkinen, 1999). Different institutional contexts result in varied behavioural patterns and outcomes, underscoring the

need to comprehend institutional dynamics to tackle societal and conservational issues (Vatn, 2017).

In summary, both institutional and socio-psychological theories are vital for understanding the formation of human behaviours which help to strengthen the social outcomes of CBC and, by extension, supports the social conditions necessary for effective conservation practice. To add more rigour, we applied Thøgersen et al.'s (2021) framework to merge individual-level behaviour determinants from socio-psychological theories with the collective impacts of institutional theory in our change-praxis model (**Table 4**).

Table 3: Social and psychological theories relevant to our praxis-of-change model.

Theories	Founder(s)	Date	Relevance to behavioural change
Social cognitive theory	Bandura	1977	Highlights the role of observational learning and social reinforcement in shaping behaviour.
Ecological systems theory	Bronfenbrenner	1979	Emphasises the interconnectedness between individuals and their social environments by taking various systems' influences on behaviour into account.
Theory of planned behaviour	Ajzen	1991	Focuses on attitudes, subjective norms and perceived behavioural control, thereby influencing individuals' intentions and subsequent behaviour.
Socio-ecological theory	Stokols	1996	Examines multi-level influences on behaviour, encompassing individual, interpersonal, community and societal factors.
Self-determination theory	Deci & Ryan	2000	Emphasises intrinsic motivations for autonomy, competence and relatedness, thereby fostering well-being and optimal functioning.

Table 4: An integrated institutionalist socio-psychological framework based on Thøgersen *et al*. (2021).

	Level 1	**Level 2**	**Level 3**
	General broad, stable characteristics	*Issue-specific mental model*	*Behaviour-specific mental model*
Factors that influence choices and actions	The characteristics that are common to and stable in most individuals.	Mental models directly related to the specific issue being addressed	This level comprises factors that directly influence specific actions or behaviours related to the issue being addressed
Physical context	The broader physical contexts that individual values and traits inhabit		The behaviour-specific physical context encompasses the options and resources that are available to address the issue To address a litter problem, for example, these would encompass infrastructure such as waste-disposal facilities as well as community cleanliness initiatives

Individual beliefs and personal norms	General beliefs	Issue-specific individual beliefs and attitudes with regards to, for example, littering that can range from acceptance to scepticism or concern Issue-specific personal norms that relate to an individual's general sense of responsibility or obligation to contribute towards addressing a specific issue (e.g. littering)	Behaviour-specific beliefs refer to individual perceptions and considerations with regards to behaviour and practices (e.g. the perceived benefits, challenges or opportunities associated with those behaviours or practices)
Social context	The broader social contexts individual values and traits inhabit	The issue-specific social context pertains to the significance of a specific issue within an individual's social circles, including the importance friends and family attach to that issue	Behaviour-specific social norms involve perceptions about what is socially expected or acceptable in terms of behaviours related to a specific issue (e.g. how people act to address the issue as influenced by social pressures and cultural norms as well as observations of others' behaviours)
Influence[1]	Mostly indirect	Mostly direct	Mostly direct

Illustrated in **Figure 1** is our theoretical framework that integrates insights from diverse theoretical and philosophical perspectives on the elements that drive behavioural change in communities. It outlines a methodical process for pinpointing intervention opportunities that enhance community well-being (in the context of CBC) and aids in creating metrics to assess the success of these interventions. In a CBC setting, these behavioural change processes are also relevant for shaping stewardship-oriented practices that support conservation objectives.

This model was implemented in the context of CBC initiatives in two rural communities in the Western Cape - Kurland and Covie - where everyday livelihoods are closely intertwined with surrounding protected and conservation-managed landscapes, as will be outlined next.



[1] While these factors may exert influence on all individual choices and activities, their impact is typically indirect and mediated through more specific factors in subsequent levels.

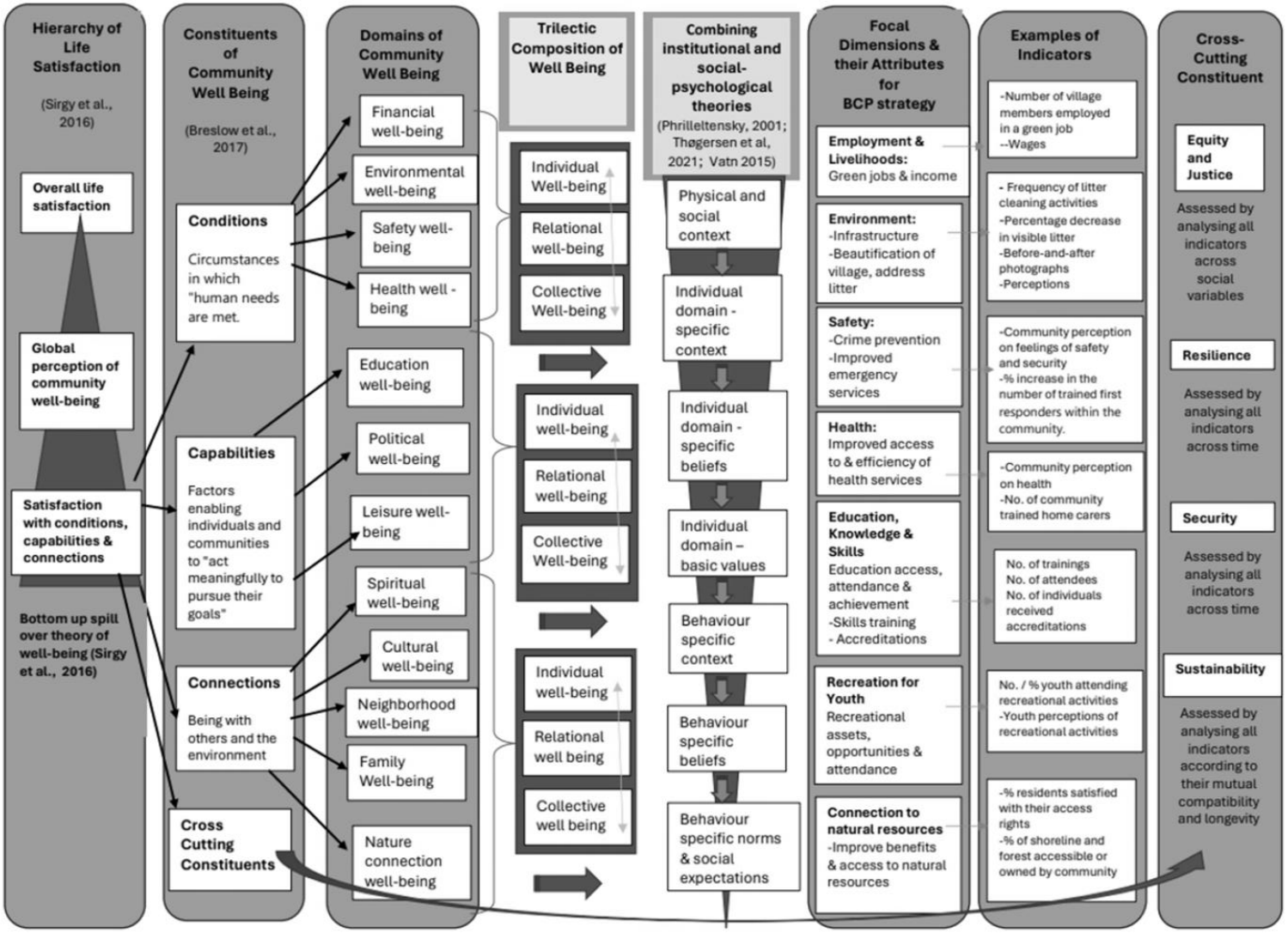


Figure 1: Conceptual framework that underpins the community-based cycle-of-praxis model.

Community context

The Kurland and Covie communities, which contributed to the model's advancement, are located in the Western Cape Province of South Africa, flanking Nature's Valley village to the west and east. These communities are under the jurisdiction of the Bitou Local Municipality (with Plettenberg Bay as the largest town in the municipality), within the Garden Route District Municipality and are encircled by the Tsitsikamma Section of the Garden Route National Park.

Around 4,030 individuals (1,500 households) live in Kurland, whereas Covie is home to roughly 100 people (32 households). In both communities, the majority self-identify as Coloured and predominantly speak Afrikaans (Stats-SA, 2022). Covie presents a more uniform community, while Kurland is more diverse, with IsiXhosa-speaking residents and immigrants from other African countries, notably Malawi and Somalia. The former group is typically employed in farming and domestic roles, whereas the latter often run local shops. Kurland has basic infrastructure and limited services, in contrast to Covie, which is almost entirely undeveloped. Residents of both communities usually have to travel 30 to 40 kilometres to Plettenberg Bay for more specialised goods and services.

The social study which informed the model was approved by the North-West University's Basic Sciences Research Ethics Committee (BASSREC) (NWU-01023-23-A7). All prescribed ethical guidelines were followed before, during and after the project.

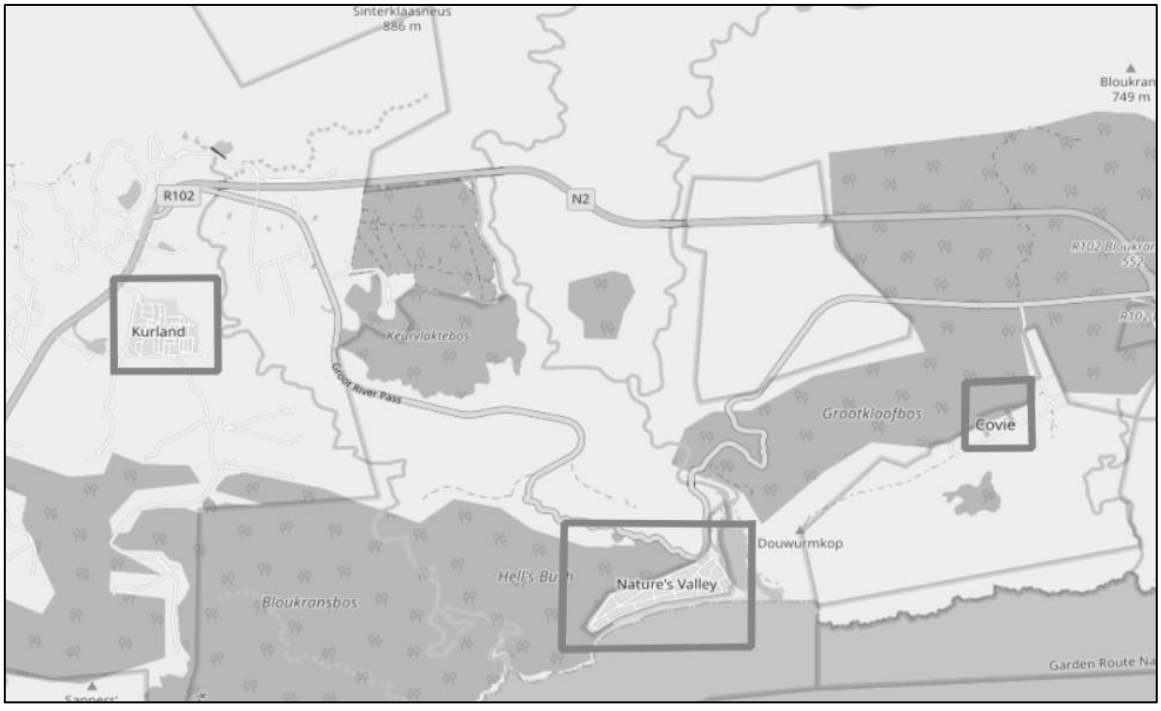


Figure 2: Map of study area showing the Kurland and Covie communities in relation to Nature's Valley, Western Cape, South Africa.

Applying the community conservation cycle-of-praxis model in Kurland and Covie

As depicted in **Figure 3**, the community-based cycle-of-praxis model consists of seven praxis cycle steps. Steps 1, 2 (except the identification of barriers), 3, and 6 correspond with the community wellness model and the community wellness cycle-of-praxis while steps 4, 5, and 7 are additions that are based on our new conceptual framework.

The following sections outline the implementation of these steps in Kurland and Covie.

Praxis cycle step 1: Identifying community ideals

Step 1 involved the facilitation of participatory situational analysis workshops (Du Plessis, 2024) to determine Kurland's and Covie's community's ideals (i.e. their values and visions for the future). Their responses were compiled and categorised in accordance with the community wellness model of personal, relational, and collective well-being (refer to **Tables 5** and **Table 6**). In so doing, we not only gained a better understanding of where the respective communities' ideals fit within that community's level of well-being, but also highlighted those areas where cooperation and collective efforts will be required if the communities' ideals are to be realised.

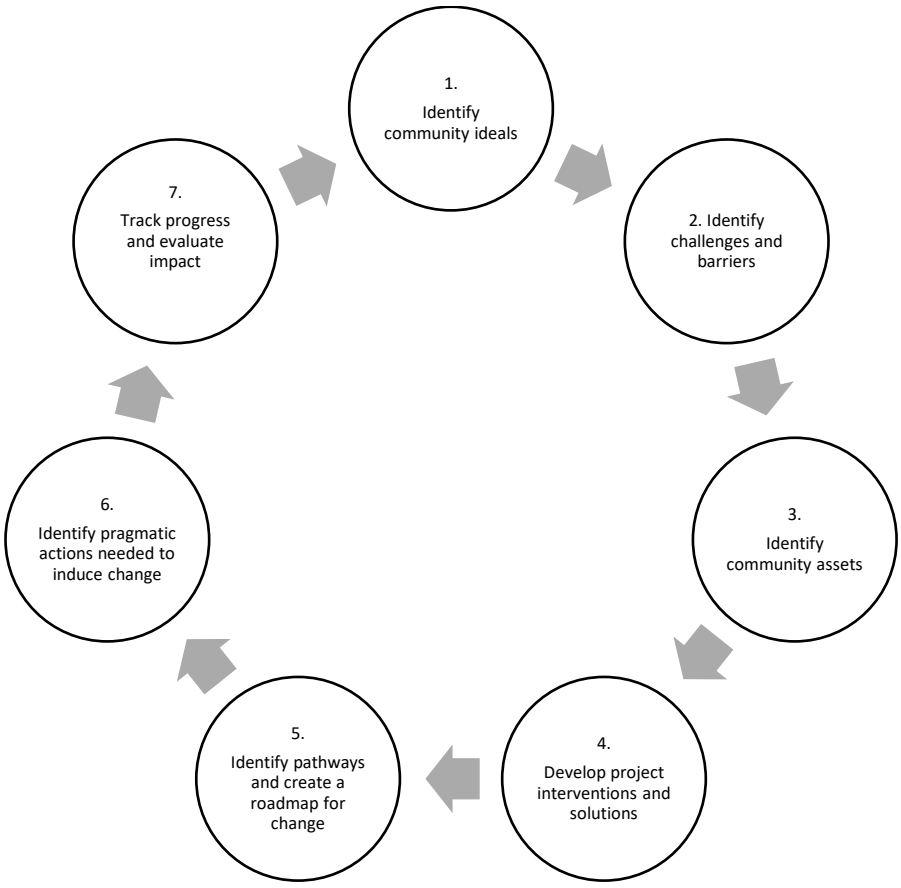


Figure 3: Community-based praxis-cycle-steps.

Table 5: Kurland community's ideals.

Personal	Relational	Collective
Neighbourhood and social well-being ideals		
Securing land from the municipality for food gardens, aiming to sell the harvest locally, thus reducing dependence on external sources.		
Securing a permanent working space for Kurland ladies' sewing team.		
Social well-being ideals		
	Forming a team of local volunteers to connect community needs with external support, acting as liaisons for organisations wanting to assist the Kurland community, ensuring alignment with the community's actual needs for greater impact.	
Education well-being ideals		
Creating a learning and skills development centre focusing on carpentry, electrical work, computer literacy and educational support. The centre will include a dedicated area for learning and reading, complete with computers, functioning as a holistic hub for educational and vocational advancement. It will provide a conductive environment for the community's youth to develop skills and pursue learning. The centre will also offer accreditation for the skills acquired, aiding in employment opportunities.		
Economic well-being ideals		
Developing a sustainable marketplace for local products by forming partnerships with hospitality industry entities like restaurants and guesthouses.		

Table 6: Covie community's ideals.

Personal	Relational	Collective
Safety, neighbourhood and social well-being ideals		
Ensuring a tranquil, safe environment for raising children.		
	Fostering a sense of safety and security within neighbourhoods.	
	Reducing drug and alcohol abuse.	
Health well-being ideals		
		Establishing a permanent healthcare clinic in Covie village.
Spiritual well-being ideals		
	Increasing attendance at church services and extending their community outreach.	
Educational well-being ideals		
		Providing support to re-establish a school within the community.
Environmental well-being ideals		
		Enhancing infrastructure by properly tarring roads and providing additional housing.
		Living in harmony with nature.

	Creating community food gardens/nurseries.
Economic well-being ideals	
Developing a small industry park for vocational training and skills development.	
Opening restaurants, eateries and stores to create job opportunities for youth and homemakers seeking work outside the home.	
Encouraging Covie community members to sustainably cultivate and harvest produce, selling any surplus to augment household income.	
Ensuring Covie fishermen can catch enough fish to support their families and sell any excess.	
Guaranteeing the ability to provide food for the table.	
	Advocating for the lifting of zone restrictions along Covie shores, granting Covie fishermen the right to fish in their own boats unimpeded.

Praxis cycle step 2: Identifying challenges and barriers

Step 2 entailed a two-step process of identifying community challenges and barriers by establishing what their needs are, as well as what might be preventing them from achieving their desired ideals.

Nell and Coetzee (2024) conducted a mixed-methods study to understand the challenges of the Kurland and Covie communities, which revealed that both communities face significant challenges that can be ascribed to a lack of, or the inaccessibility of basic needs such as housing, healthcare, education, transportation, etc. Consequently, Nell and Coetzee concluded that residents find themselves in an "undesirable, underdeveloped state" where they must struggle day after day to merely survive and have little to no prospect of ever being able to improve their well-being by, for example, gaining access to employment opportunities. This situation was found to be associated with lowered levels of connectedness to nature and limited or absent pro-environmental behaviour. They concluded the study by outlining the most significant challenges that ought to be addressed in all projects intended to improve the well-being of the Kurland and Covie communities:

- Limited economic and employment opportunities resulting in persistent financial insecurity.
- Restricted access to education and skills development, limiting residents' ability to improve employment prospects.
- High levels of insecurity and crime, negatively affecting personal and community safety.
- Inadequate access to healthcare services, constraining physical well-being and timely medical care.

In this step, we incorporate the Logical Framework Approach (LFA) problem tree into our model to chart the barriers and understand the cause-and-effect relationships. While the community wellness model and the community wellness cycle-of-praxis do not encompass this step, we have included this to distinguish between the barriers that impede well-being and the essential elements required for achieving well-being. We hold that it is crucial for our praxis-of-change model to not only pinpoint what is lacking for the community's well-being, but also to identify the barriers preventing

the community from attaining well-being. This fosters a more effective strategy, allowing the community to comprehend the fundamental causes of the deficiencies and to either eliminate these barriers or surmount them, thereby more adequately addressing their needs and realising their ideal well-being.

By synthesising insights from the participatory situational analysis workshops (Du Plessis, 2024) with the social study conducted by Nell and Coetzee (2024), we pinpointed the barriers that have obstructed community well-being and impeded the two communities from attaining their desired state (ideal). The "problem trees" depicted in **Figure 4** and **Figure 5** visually represent the cause-and-effect relationships that define the barrier to well-being in Kurland (**Table 7**) and Covie (**Table 8**).

The problem tree in **Figure 4** visually represents the interconnected causes and effects of the current conditions and social challenges identified in step 2.

Table 7: Barriers obstructing well-being in Kurland.

Community services	Previously, the absence of a fire station compromised community safety, particularly in responding to shack fires. This issue has been addressed with the establishment of a local fire station. However, the lack of nearby ambulance services and their delayed response times from town (Plettenberg Bay) remain a concern during emergencies. Additionally, the absence of a police station in the village leads to significant concerns over accessible law enforcement.
Accessibility of healthcare services	The Kurland community faces healthcare challenges as the small clinic is unable to meet all the residents' needs. For instance, the absence of nearby dental services forces residents to undertake long trips for essential care. The healthcare system's strain is further aggravated by the clinic's long waiting times, attributed to being understaffed and too small, with staff often taking extended tea or smoke breaks, disregarding the waiting patients.
Housing and infrastructure	A severe housing shortage significantly affects the well-being of the Kurland community. Moreover, authorities appear to neglect structural issues with existing housing. In terms of infrastructure, residents report that public toilets are non-functional, street lights are inoperative, and the village's dirt roads are poorly maintained. Additionally, cultural groups like the "Elders' Group" lack a dedicated space for their activities.
Unemployment	Unemployment presents a major barrier to the well-being of the Kurland community. Many residents are not aware of job market opportunities and those who seek employment often find themselves either lacking necessary skills or possessing unaccredited skills, further complicating their employment prospects.
Safety	High crime rates significantly impact the sense of safety and overall well-being of Kurland residents. The presence of drug houses and shebeens selling substances to minors, along with pit bulls roaming the streets and an unprotected dam near the Sout River, where many non-swimming children have drowned, are major concerns.
Education	The lack of a local secondary school and scarce educational resources compel children and youth desiring further education to travel great distances. This barrier to academic progression often leads to increased drug and alcohol abuse among young people.
Recreation	Limited healthy recreational options for the youth lead to boredom, which in turn increases their vulnerability to substance abuse.
Connectedness with nature	A minority of Kurland's community members place special value on nature, as reported by Nell & Coetzee (2024). However, they also observed that the community does not view nature as an exploitable resource, and anti-environmental behaviours are uncommon. This

	comparison of low nature connectedness with moderately positive environmental attitudes indicates a possible gap between environmental consciousness and action, underscoring the challenge of advancing conservation in Kurland (Nell & Coetzee, 2024).

Table 8: Barriers obstructing well-being in Covie.

Fishing rights	Covie, a community with a rich history in fishing, was forcibly removed from their land in the 1970s, which disrupted their lifestyle. Although they have reclaimed their land in recent years, residents are disheartened to see tourists on the Otter Trail accessing the coastline, a place that belongs to them, while they are denied access. The loss of traditional fishing rights, crucial to Covie's identity and sustenance, has deepened the residents' sense of disillusionment and inequality.
Infrastructure and transport	Covie village is grappling with significant infrastructure challenges, including inadequate housing and sanitation facilities. The lack of playgrounds deprives children of recreational areas, and the scarcity of shops makes accessing basic necessities difficult. The absence of permanent medical facilities further aggravates health care concerns. Additionally, the village's remote location means that public transportation is both limited and expensive.
Drug and alcohol abuse	Drug and alcohol abuse in Covie significantly affects the well-being of families and the broader community. This issue is particularly detrimental to the youth, exposing them to a range of social and health-related challenges. It also disrupts their educational pursuits, impeding their ability to finish school and limiting their future prospects. Furthermore, substance abuse contributes to a lack of safety in the neighbourhood, fostering a climate of insecurity and instability.
Economic challenges	The economic challenges in Covie are complex, characterised by a lack of employment opportunities and a shortage of well-paying jobs, leading to widespread unemployment. The erosion of traditional fishing rights has severely affected the community's conventional means of subsistence, undermined established income sources and intensified financial difficulties. Moreover, the village has seen little development over time, with scant government support for development projects. These issues continue to hinder the community's efforts to achieve sustainable livelihoods and economic stability.
Education and recreation	Education and recreation are crucial for a community's well-being. The shutdown of Covie's local school presents substantial difficulties, necessitating the transportation of children to Kurland and imposing extra costs on economically strained families. Additionally, the absence of after-school programs in the village limits constructive opportunities for the youth, leading many to resort to drugs and alcohol, thus perpetuating the cycle of substance abuse in the community.

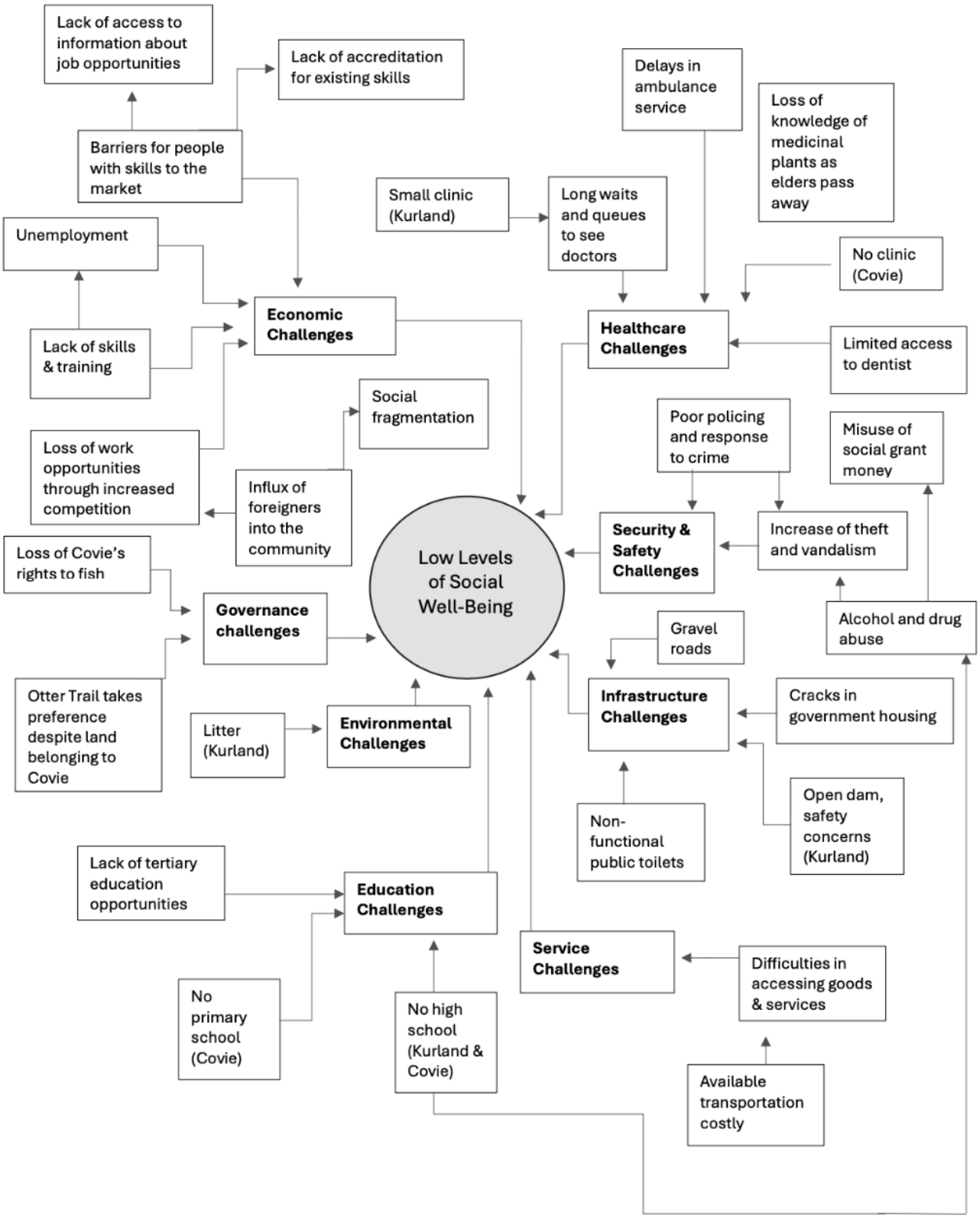


Figure 4: Problem tree.

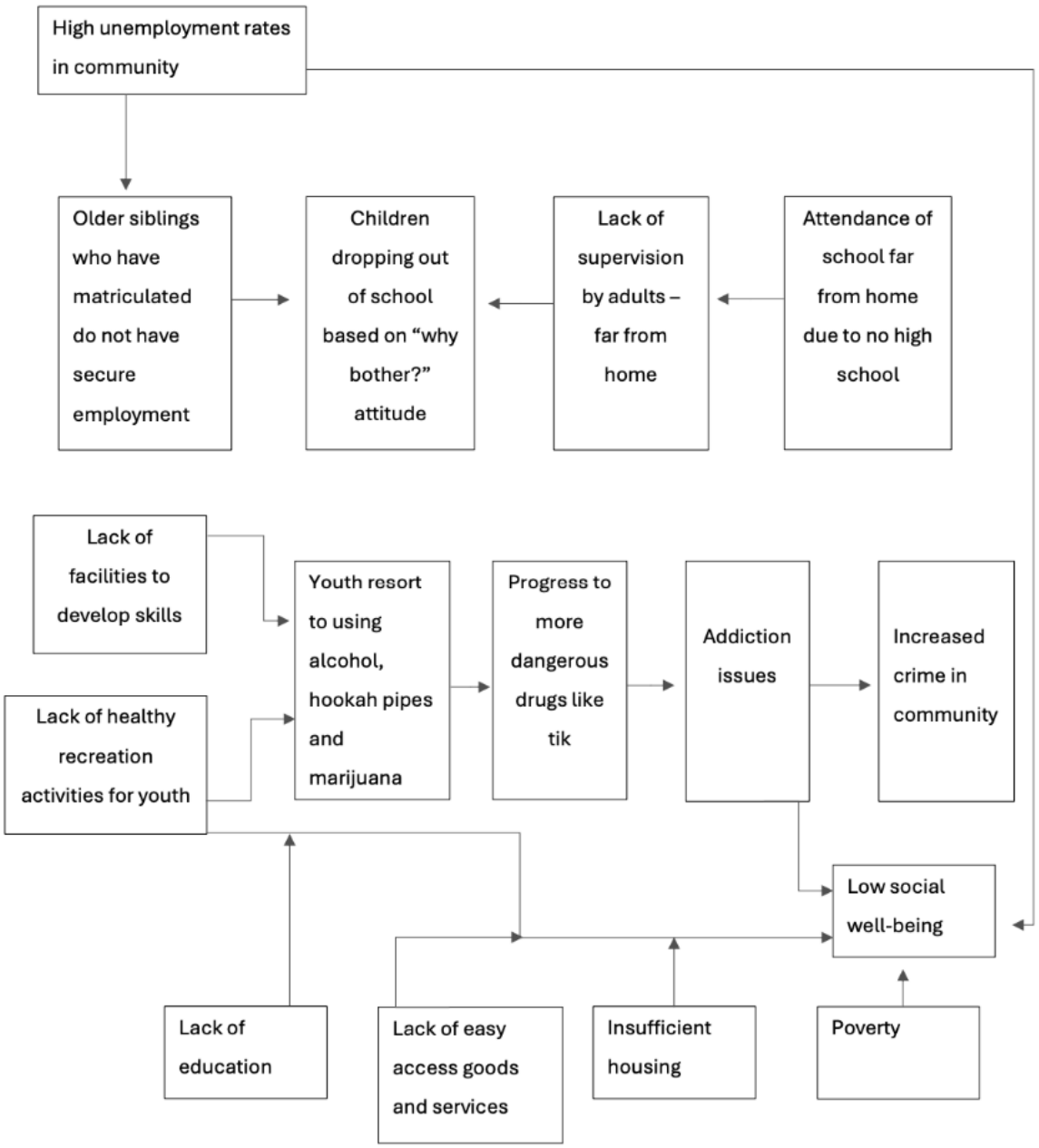


Figure 5: Cause-and-effect dynamics: Impact of education and skills development issues in Covie and Kurland villages.

Figure 5 provides a closer look at one branch of the problem tree, illustrating the causes and effects of selected educational and economic challenges prevailing in Kurland and Covie.

Figure 5 demonstrates the significant barriers faced by the youth in Kurland and Covie due to the absence of a nearby high school, limited opportunities for skills development, and a lack of healthy recreational activities. This scarcity of opportunities has resulted in an increased prevalence of harmful behaviours, particularly substance addiction issues among the youth, which adversely impacts individual well-being and the overall health and social dynamics of the community.

In Covie and Kurland, there is a pressing need for effective interventions to target the challenges associated with youth involvement

in criminal activities and their struggle to acquire skills and education (Nell & Coetzee, 2024). Such interventions should promote positive social identities and move away from stigmatising labels. For instance, reframing the perception of struggling learners from being labelled as "stupid" to recognising their diverse talents and potential is crucial (Nell & Coetzee, 2024). The authors suggest that developing interventions aimed at bolstering residents' self-efficacy through small, visible actions can lead to significant outcomes.

Table 9 provides an illustrative example of how institutional and socio-psychological theories have been integrated by applying Thøgersen *et al.*'s (2021) framework to arrive at an understanding of the impact values, beliefs and social norms have on behavioural choices in the specific contexts depicted in **Figure 5**.

Table 9: Impact of values, beliefs and social norms on behavioural choices.

	Level 1 - General characteristics
Stable individual traits	The youth in Kurland often display enduring traits like resilience, self-efficacy and motivation. These characteristics are formed through their upbringing and environmental experiences and prompt resourceful behaviour in earning an income or making the most of limited resources such as money. Likewise, environmental attitudes remain fairly stable over time and greatly impact environmentally-oriented behaviour. Nonetheless, the broader physical context of their lives may either test or strengthen these traits.
Fundamental values	The core values that the youth hold are shaped by their interaction with their environment. In the context of CBC, this particularly includes values related to nature and environmental conservation. In turn, such values are expressed behaviourally as either pro or anti-environmental actions (such as littering). Interaction with teachers in local schools and educational outcomes might shape values towards either pro or anti-educational orientations, which in turn greatly impacts consequent behaviour (e.g., staying in school versus dropping out)..
Broader physical context	The youth's well-being and prospects are significantly influenced by their physical surroundings, including housing quality, availability of basic amenities, and social challenges. Those from disrupted homes or communities afflicted by substance abuse confront extra hurdles in education and employment. A lack of access to vital services like healthcare, social support, and tertiary education can intensify feelings of despair and powerlessness and result in social exclusion or marginalization. The availability and accessibility of nature-spaces also influence well-being, a sense of connectedness to nature, and consequent environmental attitudes and behaviours.
	Level 2 - Issue-specific factors (youth unemployment and education)
Beliefs about education and employment	Youths' beliefs regarding education and employment are moulded by their broader physical context experiences. If they view their environment as deficient in opportunities or support systems, this may lead to negative attitudes toward education and the likelihood of securing meaningful employment, and an increased likelihood of dropping out of school.
Formal and informal institutions	Community social norms are shaped by the dominant physical conditions. If substance abuse or school dropout rates are high, youths may adopt these behaviours which are regarded as normative by their peers, engaging in

	behaviours that continue the cycle of poverty and disenfranchisement. Likewise, formal and informal norms, customs and regulations influence youths' attitudes and actions toward education and employment. For instance, a lack of effective policies for addressing youth unemployment or ensuring access to quality education can reinforce adverse perceptions and obstruct advancement opportunities. In villages where effective policing and formal regulations are absent, youths face increased risks of drug and alcohol abuse, which can negatively affect educational attainment and result in increased school dropout rates.
	Level 3 - Behaviour-specific factors
Education	Behaviour-specific constraints can have a direct impact on young people's ability to engage in academic activities. Factors such as low perceived control, limited confidence in academic ability, and repeated experiences of failure can impede their motivation, focus, and sustained participation in school and learning-related activities.
Skills development	In disadvantaged communities, access to skills development opportunities can be constrained by behaviour-specific barriers rather than infrastructure alone. These include limited awareness of available programmes, transport costs, and competing household responsibilities, which can hinder participation in vocational training or job-readiness programmes and contribute to ongoing unemployment and economic marginalisation.

Praxis cycle step 3: Identifying community assets

Achieving the community ideal through strength-based approaches involves actively engaging community members in a participatory process to identify local assets, such as resources, strengths, knowledge, skills, and networks (Nell & Coetzee, 2024). This process strengthens participation by recognising community members as knowledge holders and contributors, rather than passive beneficiaries. Identifying these assets enables the development of more contextually aligned, inclusive, and sustainable interventions that build on existing capacities, thereby enhancing local ownership and increasing the likelihood of long-term success. Mechanisms such as the NatureCredit programme build on these assets by recognising and rewarding community contributions to environmental stewardship.

Table 10: Assets contributing to well-being in Kurland.

Infrastructure	The Kurland Community Hall is recognised as an asset for providing temporary shelter during emergencies, such as fires and for hosting social events. The community library is also a significant asset, offering essential internet access for online registration and educational resources for the youth. The sports and rugby fields are seen as moderate assets for recreation. However, these contrast with the broader infrastructure challenges, which are viewed as urgent needs.
Environment	Covie village residents regard the natural environment as a key asset, while in Kurland, it is of minor importance. Agriculture and farming have a moderate value as community assets. Kurland's community gardens are appreciated for supplying local food and reducing the necessity for long-distance travel to stores.
Schools	The quality of Kurland's schools and teachers are seen as community strengths.

Local businesses and economic situation	In Kurland, local economic development is seen as moderately valuable, with industries, wine farms, and restaurants being potential employment sources. The introduction of Steers, a fast-food chain, and a new fuel station has not only provided quick meal options but also added convenience with accessible ATMs. Spaza shops are essential to the community, offering a local alternative for necessary food items. The sawmill has been crucial in providing jobs, aiding in housing construction and making significant donations to the community. Kurland Bricks has also contributed to employment and housing construction, positively affecting the community's economic fabric. While these businesses are viewed as moderate assets by Kurland's residents, there is an acknowledgment that these opportunities are inadequate to solve the village's extensive unemployment issue.
Skills	The skills of community members, particularly in construction and carpentry, are deemed valuable assets to the community.
Healthcare	Community members value the healthcare services within the village, despite the system's efficiency issues and the recognised need for improvement, considering it a moderate to significant asset.
Welfare	Recognised as both a community need and an asset, welfare services are crucial. Organisations and NGOs like Born in Africa and Kurland Bricks are pivotal in providing this support. Born in Africa aids about 450 children with educational assistance yearly, significantly aiding community development. Kurland Bricks also supports welfare by creating a community crèche, reflecting its commitment to corporate social responsibility. Additionally, NVT's engagement in community greening, clean-ups, gardening clubs and environmental education at Crags Primary School enhances community welfare. Moreover, rehabilitation for youths battling drug addiction and local soup kitchens offering free meals to those at the clinic exemplify the community's welfare resources.
Cultural and recreational assets	The soccer and rugby fields are vital to Kurland's community spirit. They are instrumental in promoting community engagement, social cohesion and active lifestyles. Despite being identified as a strength, the social study revealed a need for more recreational activities for youth to alleviate boredom and deter drug and alcohol abuse, as noted in participatory workshops.
Local economic development	Service delivery is deemed a moderate asset. Although it ranked low in the social study, local economic development is still considered a moderate asset.
Local leadership	Like local economic development, local leadership received a low ranking in the social study but is viewed as a marginal asset.
Religion	Religion holds moderate value as asset within the community. Although many residents rarely participate in local church events, certain smaller groups consider them to be of great importance, particularly in addressing drug-related issues. The community exhibits varied degrees of involvement in church activities, with some congregations becoming more proactive in addressing drug concerns.

Table 11: Assets contributing to well-being in Covie.

Cultural identity	The historical discovery, establishment, and development of Covie hold significant importance for its members. It shapes their sense of place and cultural identity, and fosters community cohesion and pride.
Infrastructure	The Covie community's well-being has been greatly enhanced by several key developments. These include the provision of running water and electricity, road improvements for better transportation, the creation of a library filled with educational resources, a central church/school building for communal gatherings and education, a mobile clinic to improve healthcare access and transportation services that allow children to attend school in Kurland.
Economic	The traditional harvesting of honeybush tea, which is no longer practiced due to its illegality, once contributed to Covie's economic health. Nowadays, income from backpackers and job opportunities provided by local businesses like Platbos Hotel, Bloukrans Bungy, the tollgate and King Fisher B&B play a role in the community's economy. Like Kurland, these businesses are seen as moderate assets, and although they provide some employment, they are not enough to overcome the extensive unemployment in the village. The level of service delivery in the village is also viewed as a moderate asset.
Skills	In the Covie community, skills in construction and carpentry are highly valued as community assets.
Educational	The well-being of the Covie community is enhanced by educational contributions, such as the provision of a kindergarten and transportation for children to attend school in Kurland.
Environmental	Located in a rural setting, Covie is encircled by forests and fynbos. The natural environment and resources are crucial assets, providing medicinal plants and firewood. Additionally, the beauty of the natural surroundings is appreciated for its aesthetic value. While agriculture and farming are seen as moderate assets, community gardens are treasured for their role in feeding the community. Initiatives like the Covie Fishing Club and a hiking trail through the local fynbos have positively impacted community well-being.
Safety	Although issues with drug and alcohol abuse leading to crime pose safety concerns, there is a sentiment within the community that the current level of community safety is a valuable asset.

Praxis cycle step 4: Developing project interventions and solutions

Step 4 focuses on bridging the gap between the current situation - defined by identified needs, barriers, and existing community assets - and the community's desired ideals. Drawing on the integrated socio-psychological and institutional frameworks outlined above, this step translates insights on values, norms, institutional constraints, and capabilities into practical interventions that target personal, relational, and collective dimensions of community well-being in CBC settings.

The development of project interventions occurs in three phases. First, the problem tree developed in Step 2 of the praxis cycle is transformed into a solution tree (see Figure 6). This process enables the identification of context-appropriate solutions that respond to both individual-level behavioural determinants and broader institutional conditions, forming the basis for targeted project interventions.

In Kurland, one such intervention was the NatureCredits programme, designed to link community participation in environmental stewardship with incentives that support skills development and livelihood pathways; environmental outcomes associated with this intervention will be examined in version 2.0 of this model.



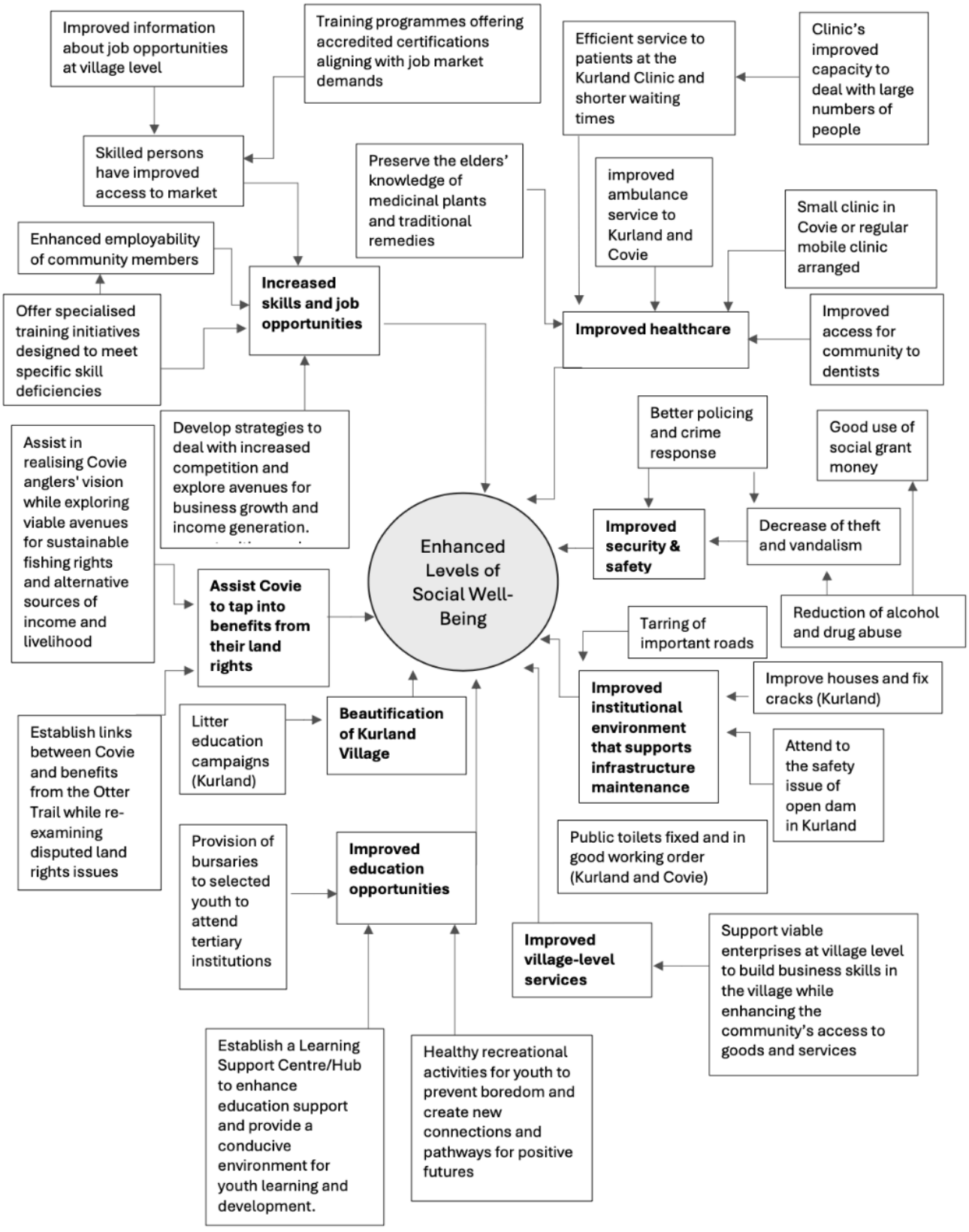


Figure 6: Solution tree showing cause-and-effect pathways.

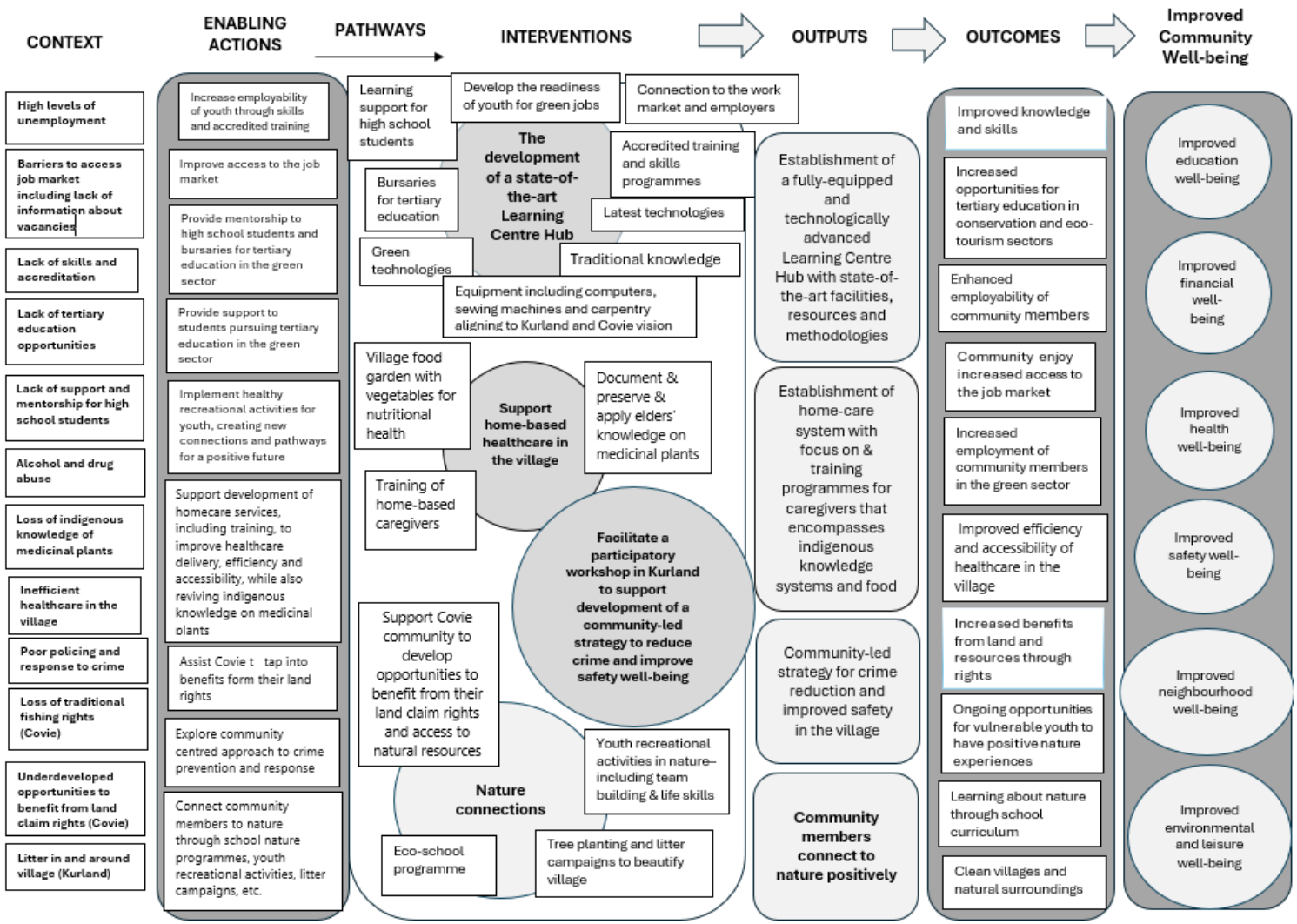


Figure 7: The praxis-of-change model: A strength-based approach to positively impact community well-being in the Covie and Kurland villages.

Praxis cycle step 5: Identifying pathways and creating a roadmap for change

Step 5 entails the creation of a roadmap for change, consisting of a series of concrete action-steps. Prilleltensky (2001) characterises the final step of his value-based praxis model as the translation of research findings into community action through various activities, while Totikidis (2003) specifies actions to improve community well-being in the final step of her model (refer to the next step). Before moving to action, we introduce an additional step: "creating a roadmap for action." This roadmap is crucial to ensure that the actions are in line with the underlying theory and context, namely the ideals, barriers, strengths, assets and needs of the community within CBC initiatives.

Our praxis-of-change model, which is a result of steps 1 to 5, is depicted in **Figure 7**. This Figure visually illustrates the pathways to affect community well-being in Kurland and Covie. It includes the interventions (inputs), outputs, outcomes, and context, along with a series of enabling actions for each pathway.

Praxis cycle step 6: Identifying the pragmatic actions needed to induce change

Step 6 entails a transition from conceptualisation to practical implementation. This step, which is absent in the two foundational models we used, has been incorporated into our praxis cycle to ensure that the first stages of the model get translated into practical action. To drive real-world change, this involves converting our concepts, ideas, theories, visions and plans into meaningful outcomes. In short, this step brings our theory of change to life. In the context of our intervention in Kurland and Covie, the notion of space was identified as a useful lens through which to translate the identified outcomes into concrete, practical action, as will be discussed next.

Interconnecting the mental, physical, and social space

The concept of "space" has experienced a significant paradigm shift in sociological theory since the mid-nineteenth century. Henri Lefebvre, a notable social science scholar, redefined traditional notions of space in his seminal work, *The Production of Space* (1991). Lefebvre's theoretical framework recognises the interplay between physical, mental, and social spaces, underscoring their collective impact on human environments and behaviours. Contrary to the traditional view of space as an "empty abstraction," he posited that space is socially constructed and deeply entwined with human experiences and interactions. Subsequent scholars like Gruenewald (2003), Warf and Arias (2009), and Rivera-Yevenes (2023) have further highlighted the significance of space in shaping and transforming social life, especially within educational contexts such as learning centres. This perspective is particularly relevant in community-based initiatives where social change, learning, and relationships with place are closely intertwined.

Creating a platform for pragmatic action

Drawing from these insights into the transformative role of space in social life, and anchored by the collective ideals of the communities, we propose that a promising option to create a platform for pragmatic action is the establishment of a physical space such as a resource/learning centre. This would be a place where fundamental values, beliefs, social norms, and behavioural choices can be influenced. However, the space will have to address challenges (e.g. needs and barriers) and take

into consideration the community assets that are available. Based on the Kurland and Covie contexts, we propose that the space consist of three interconnected focus areas/hubs: a learning hub, community care hub, and a nature connect hub (**Table 12**). Each hub should then be further planned to address specific issues in the community.

Table 12: Interconnected hubs forming the basis of a space that can be used to influence values, beliefs, social norms and behavioural choices.

Learning hub (Committed to fostering lifelong learning, skill enhancement, and personal development.) • Resources for adult education in personal growth, skill advancement, and the sharing of traditional knowledge. • Supportive educational services for children and youth, such as tutoring, mentoring, and after-school homework assistance. • Financial literacy training, including budgeting and financial planning resources, alongside support for entrepreneurial and small business growth. • Enhancing the preparedness and employability of youth in green jobs and broadening their job market access.
Community care hub (Aimed at advancing physical and mental well-being and community safety.) • Advancement of physical and mental health via medical services, wellness projects, preventive care, home-based care, traditional knowledge, and health education initiatives. • Projects and services to fortify family ties, encourage healthy relationships, and provide support for familial issues and challenges. • Healthy recreational activities that impart valuable life lessons and address drug and alcohol prevention, targeting vulnerable youth to foster well-being. • Tools for crime prevention, emergency response training and assistance for victims of violence or abuse.
Nature connection hub (Focused on enhancing environmental sustainability, conservation efforts, eco-friendly living, and aiding Covie and Kurland in utilising their rights to natural resource benefits.) • Educational projects on environmental sustainability and conservation within nearby protected areas and spaces. • Community-based educational initiatives to encourage eco-conscious living, recycling, and community gardening projects. • Opportunities for community engagement in environmental stewardship and sustainable practices. • Utilisation of communal rights to land and natural resources for the benefit of the community.

It should be noted that the resource or learning centres housing the hubs need not be expansive, state-of-the-art spaces. Rather, existing or underutilised buildings within a community could serve this purpose. In Nature's Valley, as in other regions globally, there are often available municipal or government buildings that can be repurposed. Informal structures may also be considered. Furthermore, the three-hub concept can be decentralised, meaning it can be distributed across various locations within the community.

Praxis cycle step 7: Tracking progress and evaluating impact

In this final step of our praxis cycle, we propose tracking progress and evaluating impact in the Kurland and Covie contexts to ensure that planned activities address both social and environmental priorities and remain aligned with community challenges and assets. Adding this step also addresses a recognised shortcoming in CBC, namely the limited measurement of social outcomes (Kaplan-Hallam & Bennett, 2017), while incorporating environmental indicators linked to the piloted NatureCredits programme, such as trees planted, alien vegetation cleared, and litter removed from villages and beaches. We therefore recommend the use of a Monitoring, Evaluation and Learning (MEL) approach to track progress, learn from implementation, and ensure that interventions are both impactful and efficiently executed (**Figure 8**).



As depicted in **Figure 8**, the proposed MEL approach can guide the assessment of the overall performance of planned activities. In this way, NGO-led organisations can constantly refine the strategy to enhance the overall success of their interventions. A logical framework can be used to provide clarity by tracking the progress made towards achieving the objectives. This also serves as a crucial link between a theory of change and the actions that can be taken to implement it effectively. Essential elements such as a vision, goals, objectives, outputs, activities (inputs), quantitative and qualitative indicators, verification and underlying assumptions should be included.

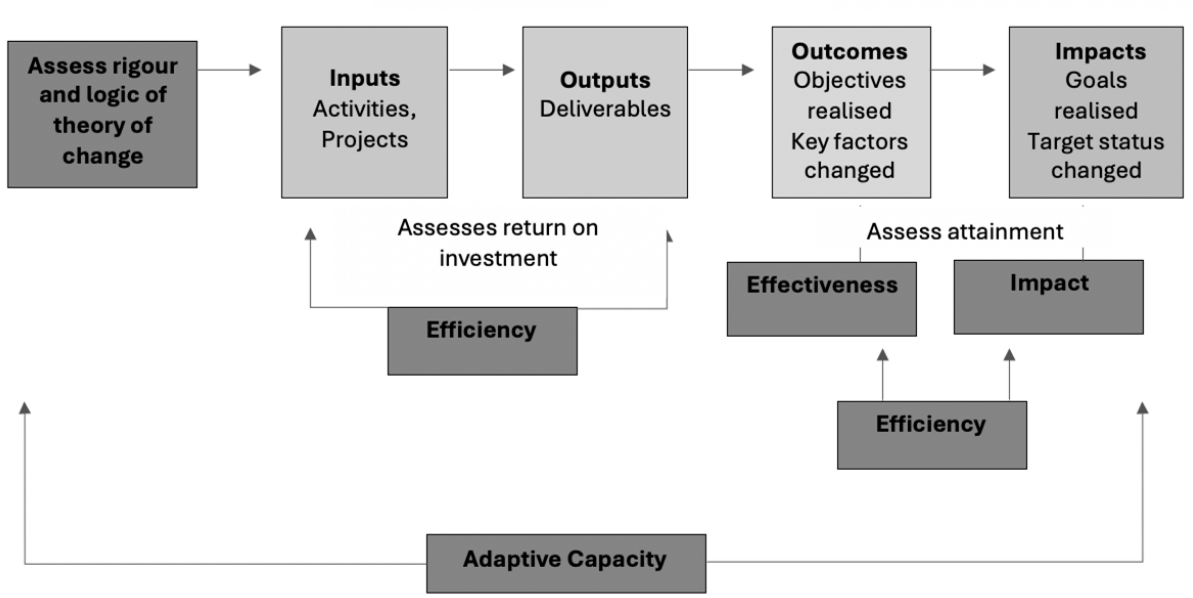


Figure 8: Monitoring, Evaluation and Learning approach.

Partnerships and collaboration

Considering that the primary mandate of most NGO-led organisations does not include the provision of community facilities and services, the success of our proposed model depends on collaboration and partnerships with relevant organisations and service providers operating within existing institutional frameworks. For example, by partnering with education providers and social support organisations that can supply the necessary infrastructure and expertise for skills development, education, and healthcare initiatives, NGO-led CBC programmes can strengthen the social dimensions of CBC without needing to hold this expertise internally.

Conclusion

In this article we outlined our community-based cycle-of-praxis model, which has been designed for use as a practical tool in CBC contexts to systematically develop contextually sensitive and practically actionable interventions aimed at contributing to conservation practice by supporting the social conditions necessary for stewardship, compliance, and long-term conservation effectiveness. In particular, the model enables the identification of clear actions and pathways that enhance the social dimensions of CBC, which have been identified as a shortcoming in most other models. Viewing these dimensions through the lens of community well-being, socio-psychological and behavioural change can significantly improve social outcomes and lead to more effective CBC initiatives by addressing complex challenges throughout the social-ecological system that serve as barriers to pro-environmental behaviour. Such efforts are poised to make a greater impact on mitigating complex social and environmental challenges, including biodiversity loss, poverty, and job creation. By prioritising community needs and capitalising on local assets, our model can create strong links between conservation objectives and community well-being. However, achieving

this requires a strategic shift towards impactful interventions that address the underlying causes of societal issues, thereby fostering healthier and more sustainable communities, which in turn, support the aim of creating a healthier natural environment.

References

Ajzen, I. 1991. The theory of planned behavior. Organizational Behavior and Human Decision Processes, 50(2), 179-211.

Bandura, A. 1977. Social learning theory. Prentice Hall.

Berger, P. L. & Luckmann, T. 1967. The social construction of reality treatise in the sociology of knowledge. Anchor Books.

Biedenweg, K., Stiles, K.A. & Wellman, K. 2016. A holistic framework for identifying human wellbeing indicators for marine policy. Marine Policy, 31-37.

Breslow, S. J., Allen, M., Holstein, D., Sojka, B., Barnea, R. & Basurto, X. 2018. Evaluating indicators of human well-being for ecosystem-based management. Ecosystem Health and Sustainability, 3(12), 1-18.

Britton, E. & Coulthard, S. 2013. Assessing the social wellbeing of Northern Ireland's fishing society using a three dimensional approach. Marine Policy, (37), 28-36.

Bronfenbrenner, U. 1979. The ecology of human development: Experiments by nature and design. Harvard University Press.

Deci, E. L. & Ryan, R. M. 2000. Self-determination theory: When mind mediates behavior. The Journal of Mind and Behavior, 21(4), 377-400.

Du Plessis, S. 2024. Social baseline assessment for the NVT Salt River project. NVT internal report.

Esmail, N., McPherson, J.M., Abulu, L., Amend, T., Amit, R., … Tawake, A. & Wintle, B. 2023. What's on the horizon for community-based conservation? Emerging threats and opportunities. Trends Ecol Evol., 38(7), 666-680.

Galvin, K. A., Beeton, T. A. & Luizza, M. W. 2018. African community-based conservation: a systematic review of social and ecological outcomes. Ecology and Society, 23(3), 39.

Gruenewald, D. A. (2003). Foundations of place: A multidisciplinary framework for place conscious education. American Educational Research Journal, 40(3), 619-654.

Hall, P. A. & Taylor, R.C.R. 1996. Political science and the three new institutionalisms. Political Studies, 44(5), 936-957.

Hukkinen, J. 1999. Institutions of environmental management. Constructing mental models and sustainability. Routledge.

Kaplan-Hallam, M. & Bennett, N.J. 2017. Adaptive social impact management for conservation and environmental management. Conserv Biol., 32(2), 304-314.

Lefebvre, H. (1991). The production of space. Blackwell.

Mahajan, S.L., Glew, L. Rieder, E., et al. 2019. Systems thinking for planning and evaluating conservation interventions. Conservation Science and Practice, 1, e44.

Nell, W. & Coetzee, H.C. 2024. Community needs, assets, psycho-social well-being and environmental attitudes in Kurland and Covie, Western Cape, South Africa. Implications for interventions and community-based conservation. NVT internal report.

Nelson, G. & Prilleltensky, I. (Eds.). 2005. Community psychology: In pursuit of liberation and well-being. Palgrave Macmillan.

North, D. C. 1990. Institutions, institutional change, and economic performance. Cambridge University Press.

Prilleltensky, I. & Nelson, G. 2002. Doing psychology critically: Making a difference in diverse settings. Palgrave Macmillan.

Prilleltensky, I. 2001. Value-based praxis in community psychology: Moving toward social justice and social action. American Journal of Community Psychology, 29(5), 747-778.

Rice, W., Sowman, M. & Bavinck, M. 2021. Informing a conservation policy-praxis disjuncture: A 'commons' perspective to tackling coastal-marine community conserved area implementation in South Africa. Biological Conservation, 261, 109296.

Rivera-Yevenes, C. 2023. Theorising the development, implementation and sustainability of schools as community hubs. In B. Cleveland, S. Backhouse, P. Chandler, I. McShane, J.M. Clinton, & C. Newton (Eds.). Schools as community hubs. Building 'more than a school' for community benefit (pp. 279-292). Springer.

Salerno, J., Romulo, C., Galvin, K.A., Brooks, J., Mupeta-Muyamwa, P. & Glew, L. 2021. Adaptation and

evolution of institutions and governance in community-based conservation. Conservation Science and Practice, 3, e355.

Sirgy, M. J. & Cornwell, T. 2002. Further validation of the Sirgy et al.'s measure of community quality of life. Social Indicators Research, 56, 125-143.

Sirgy, M. J., Rahtz, D., Cicic, M. & Underwood, R. 2000. A method for assessing residents' satisfaction with community-based services: A quality-of-life perspective. Social Indicators Research, 49(3), 279-316.

Sirgy, M. J., Widgery, R. N., Lee, D. & Yu, G. B. 2010. Developing a measure of community well-being based on perceptions of impact in various life domains. Social Indicators Research, 96(2), 295-311.

Soma, K. & Vatn, A. 2010. Is there anything like a citizen? A descriptive analysis of instituting a citizen's role to represent social values at the municipal level. Environmental Policy and Governance, 20(1), 30-43.

Soma, K. & Vatn, A. 2014. Land use policy representing the common goods – stakeholders vs. citizens. Land Use Policy, 41, 325-333.

Stern, P. & Dietz, T. (Eds.). 2002. New tools for environmental protection: Education, information, and voluntary measures. National Academic Press.

Stokols, D. 1996. Translating social ecological theory into guidelines for community health promotion. American Journal of Health Promotion, 10(4), 282-298.

Thøgersen, J., Vatn, A., Aasen, M., Dunlap, R. E., Fisher, D. R., Hellevik, O. & Stern, P. 2021. Why do people continue driving conventional cars despite climate change? Social psychological and institutional insights from a survey of Norwegian commuters. Energy Research and Social Science, 79, 1-14.

Totikidis, V. & Prilleltensky, I. 2006. Engaging community in a cycle of praxis: Multicultural perspectives on personal relational and collective wellness. Community ,Work and Family, 9(1), 47-67.

Totikidis, V. 2003. Understanding community wellness from a multicultural perspective. (Unpublished master's dissertation). Victoria University, St Albans, Melbourne, Australia.

Vatn, A. 2015. Environmental governance: Institutions, policies and actions. Edward Elgar.

Wachira W., J, Atela, J, Stacey, P.A. & Outa, G. 2024. NGO-Led Community-Based Conservation: A New Frontier of Territorialization with Implications for Pastoralists. Land Tenure and Climate Change Adaptation, 13(6), 740.

Warf, B. & Arias, S. 2009. Introduction: The reinsertion of space into the social sciences and humanities. In B. Warf, & S. Arias (Eds.). The spatial turn: Interdisciplinary perspectives (pp. 1-10). Routledge.



Dung beetle assemblages across contrasting fynbos age classes and forest habitats in the southern Cape, South Africa: A pilot study

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How to cite this article:

Van Hasselt, C., Roets, F. & Swart, R.C. 2026. Dung beetle assemblages across contrasting fynbos age classes and forest habitats in the southern Cape, South Africa: A pilot study. *Nature's Letters* **3**:89-99. URL: https://nvt.org.za/natures-letters/

Abstract

Dung beetles are key ecosystem engineers, yet their responses to fire-driven fynbos dynamics remain poorly understood. We assessed dung beetle assemblages across burnt fynbos (≈2 years post-fire), dense fynbos (≈26 years), and southern Afrotemperate forest in the Nature's Valley region, South Africa, using baited pitfall traps. A total of 9 species and 640 individuals were recorded. Total dung beetle abundance differed significantly among habitats (Kruskal–Wallis: $\chi^2 = 8.92$, df = 2, $p = 0.0116$), with dense fynbos supporting higher abundances than burnt fynbos and forest (pairwise Wilcoxon tests, $p = 0.036$). Species richness and Shannon diversity did not differ significantly among habitats. Community composition varied significantly (PERMANOVA: $F_{2,12} = 3.66$, $R^2 = 0.38$, $p = 0.002$), with forest assemblages distinct from both fynbos sites. To our knowledge, this study provides the first pilot assessment of dung beetle diversity and abundance in the wider Nature's Valley region and the first direct comparison of two fynbos regeneration stages with southern Afrotemperate forest.

Introduction

Dung beetles (Coleoptera: Scarabaeinae) are widely recognised as key ecosystem engineers that contribute to nutrient cycling, secondary seed dispersal, soil bioturbation, and parasite suppression (Davis, 1994a; Halffter & Favila, 1993; Davis & Scholtz, 2020a). Through dung burial and manipulation, dung beetles accelerate the incorporation of organic matter into soils, influence microbial activity, and affect plant recruitment, thereby playing a disproportionate role in ecosystem functioning relative to their biomass (Doube, 1990; Davis, 1996).

Dung beetle assemblages are structured by a combination of trophic resources, habitat characteristics, and landscape contexts, operating at both microhabitat (the dung pat and immediate surroundings) and macrohabitat (vegetation structure, soil type, climate, and mammal assemblages) scales (Davis, 1997; Davis *et al*., 1994; Pryke *et al*., 2022). Species differ markedly in dung exploitation strategies, body size, and nesting behaviour, leading to the functional group framework proposed by Doube (1990), which classifies dung beetles into telecoprids (rollers), paracoprids (tunnelers), kleptocoprids (stealers), and endocoprids (dwellers). These functional differences influence species' sensitivity to disturbance, resource scarcity, and habitat change (Davis, 1996; Jankielsohn *et al*., 2001).

Although dung beetle ecology has been extensively studied in African savanna systems, comparatively little is known about dung beetle assemblages in fynbos and Afrotemperate forest ecosystems of the Cape Floristic Region, a global biodiversity hotspot which saw the extirpation of megaherbivores, like the Knysna elephant, with the start of European settlement (Moolman *et al*., 2019). This represents a notable knowledge gap, given the ecological uniqueness of fynbos and its globally recognised biodiversity value.

Fynbos is a sclerophyllous shrubland vegetation type endemic to the south-western Cape of South Africa, occurring within a Mediterranean-type climate characterised by winter rainfall and summer drought (van Wilgen *et al*., 1985; Smit *et al*., 2024). Fynbos ecosystems are inherently fire-driven, with natural fire return intervals typically ranging from 6 to 40 years (Bond, 1980; Kruger & Bigalke, 1984; Kraaij & van Wilgen, 2014). Fire is essential for maintaining plant diversity, driving nutrient release in nutrient-poor soils, and promoting post-fire regeneration through seed germination and resprouting mechanisms. Post-fire succession in fynbos leads to pronounced changes in vegetation structure, ground cover, microclimate, and soil properties. Early post-fire habitats are structurally open and dominated by pioneer vegetation, offering fewer micro-niches for ground-dwelling arthropods compared to mature fynbos with accumulated litter and complex vegetation layers (Kerdoncuff *et al*., 2023). Arthropods, including dung beetles, are therefore expected to respond strongly to post-fire habitat changes, making them valuable indicators of habitat recovery and land-management impacts (Pryke & Samways, 2012).

Previous studies have shown that dung beetle assemblages can be sensitive to habitat transformation, fragmentation, and edge effects, particularly at forest–open habitat ecotones (Davis, 1994b; Martínez-Falcón *et al*., 2018; Swart *et al*., 2018; van Bergen *et al*., 2025). However, dung beetle responses to fynbos fire regimes, post-burn vegetation age, and fynbos–forest contrasts remain poorly documented, especially in the Cape region.

Study rationale and objectives

Given the ecological importance of dung beetles and the dynamic, fire-driven nature of fynbos ecosystems, understanding how dung beetle communities vary across habitats of differing post-fire age is critical for conservation planning and biodiversity monitoring. This is particularly relevant in the context of prescribed burning, which is increasingly used to manage fynbos fuel loads and maintain habitat heterogeneity (Smit *et al*., 2024). This study aimed to investigate dung beetle abundance, diversity, and community composition across recently burnt fynbos, dense (mature) fynbos, and Afrotemperate forest habitats around Nature's Valley, South Africa. Specifically, the study tested whether dung beetle assemblages differ between forest and fynbos habitats, and whether post-fire vegetation age influences dung beetle diversity and abundance within fynbos.

By providing baseline data for dung beetle communities in this under-studied region, this research contributes to the NVT's broader objectives of biodiversity monitoring and habitat restoration, and offers insights into how fire-driven landscapes influence insect-mediated ecosystem functions.

Materials and methods

Study area

Fieldwork was conducted in the De Vasselot section of the Tsitsikamma National Park within the Garden Route region of the Western Cape, South Africa (**Figure 1**). Three study sites were selected: burnt fynbos – a post-fire fynbos site with a crude vegetation age of approximately 2 years since last burn; dense fynbos – a mature fynbos site with a crude vegetation age of approximately 26 years; southern Afrotemperate forest – a forest control site located within Hebron Private Nature Reserve. Vegetation age and habitat boundaries were derived from SANParks vegetation layers, and site locations were mapped using QGIS version 3.40.1 (**Figure 1**).

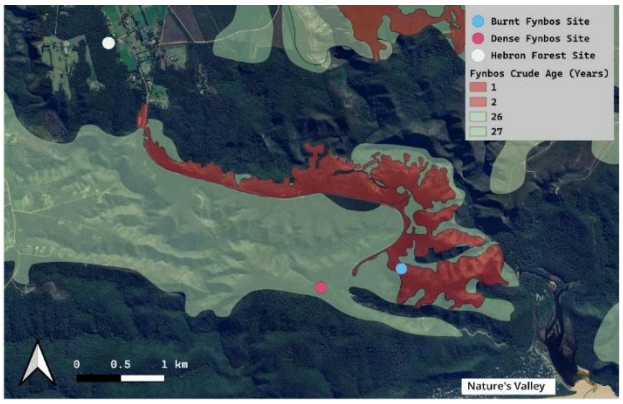


Figure 1: Map of the study area in and around Nature's Valley, South Africa, showing the locations of dung beetle sampling sites across burnt fynbos, dense fynbos, and Afrotemperate forest habitats. Coloured polygons represent fynbos vegetation age (years since last fire) derived from SANParks vegetation layers, while points indicate the locations of the three sites.

Study design and aims

This study employed a comparative field design to investigate differences in dung beetle (Scarabaeinae) abundance, diversity, and community composition across habitats differing in vegetation structure and regeneration stage. Specifically, dung beetle assemblages were compared among burnt fynbos, dense (mature) fynbos, and Afrotemperate forest habitats in and around Nature's Valley, South Africa. The primary aim was to assess whether dung beetle species assemblages differ between forest and fynbos habitats, and whether differences occur between fynbos sites of contrasting post-fire age. The following hypotheses were tested: H0: There is no significant difference in dung beetle diversity (species richness and abundance) between post-burn, mature fynbos and forest sites; H1: There is a significant difference in dung beetle diversity between post-burn, mature fynbos and forest sites.

Trap design and deployment

Dung beetles were sampled using randomised baited pitfall traps (RBAs) following the CSS modified model (Krell, 2007). In this design, bait is suspended at the soil surface to reduce rapid degradation and flooding while maintaining attractiveness to dung beetle functional groups. At each site, 10 pitfall traps were deployed, arranged as five paired sampling points, with one large and one small pitfall trap placed at least 5 m apart at each sampling point. Pairing traps in this manner allowed comparison of trap sizes while maintaining spatial independence at the sampling-point level. Trap design and deployment were standardised across all sites to minimise sampling bias. Traps were baited with a mixed bait 50:50 ratio of fresh pig dung and carrion (chicken liver) to maximise attraction across dung beetle guilds, following recommendations by van Bergen *et al*. (2025). Traps were deployed for seven consecutive days in mid-April 2025, with periodic site visits during the sampling period to check trap integrity and minimise bycatch.

Specimen collection and identification

Following retrieval, dung beetle specimens were preserved in 96% ethanol and transported to the laboratory for processing. Individuals were sorted and identified to species or morphospecies level using external morphological characteristics, including body shape, pronotal structure, elytral sculpturing, and horn development where applicable. Reference images of representative individuals were taken using a Dino-Lite digital handheld microscope, with images captured using DinoCapture 3.0 software. Macroscopic images illustrating key diagnostic features for each species are shown in **Figure 2**. Species identifications were cross-checked using regional taxonomic keys and reference material; then a specimen reference collection was sent for further identification confirmation through Stellenbosch University.

Statistical analyses

Abundance data were recorded as counts of individuals per species per trap. For statistical analyses, paired traps (large and small) were pooled within each sampling point, resulting in five independent sampling points per habitat. Pooling traps at this level avoided pseudoreplication while preserving spatial replication across sites.

All statistical analyses were conducted in R (version 4.2.1). Because abundance and diversity data did not meet assumptions of normality and sample sizes were small (n = 5 per habitat), non-parametric tests were used throughout. Differences in total abundance, species richness, and Shannon diversity among habitats were tested using Kruskal–Wallis rank-sum tests. Where significant differences were detected, pairwise Wilcoxon rank-sum tests with Holm-adjusted p-values were used for post-hoc comparisons between habitat pairs. To assess differences in community composition among habitats, a permutational multivariate analysis of variance (PERMANOVA) was performed using Bray–Curtis dissimilarities based on species abundance data. The analysis was run with 999 permutations. Homogeneity of multivariate dispersion among habitats was assessed using betadisper to ensure that significant PERMANOVA results were not driven by differences in within-group variability. Non-metric multidimensional scaling (NMDS) ordination was used to visualise patterns in dung beetle community composition among habitats. NMDS was conducted using Bray–Curtis dissimilarities with two dimensions (k = 2), and ordination quality was assessed using stress values.

Ethical approval and permitting

All fieldwork was conducted under a SANParks research permit approved by the Garden Route National Park Scientific Services (Permit Number SS1516). Sampling methods were designed to minimise environmental disturbance and non-target bycatch.



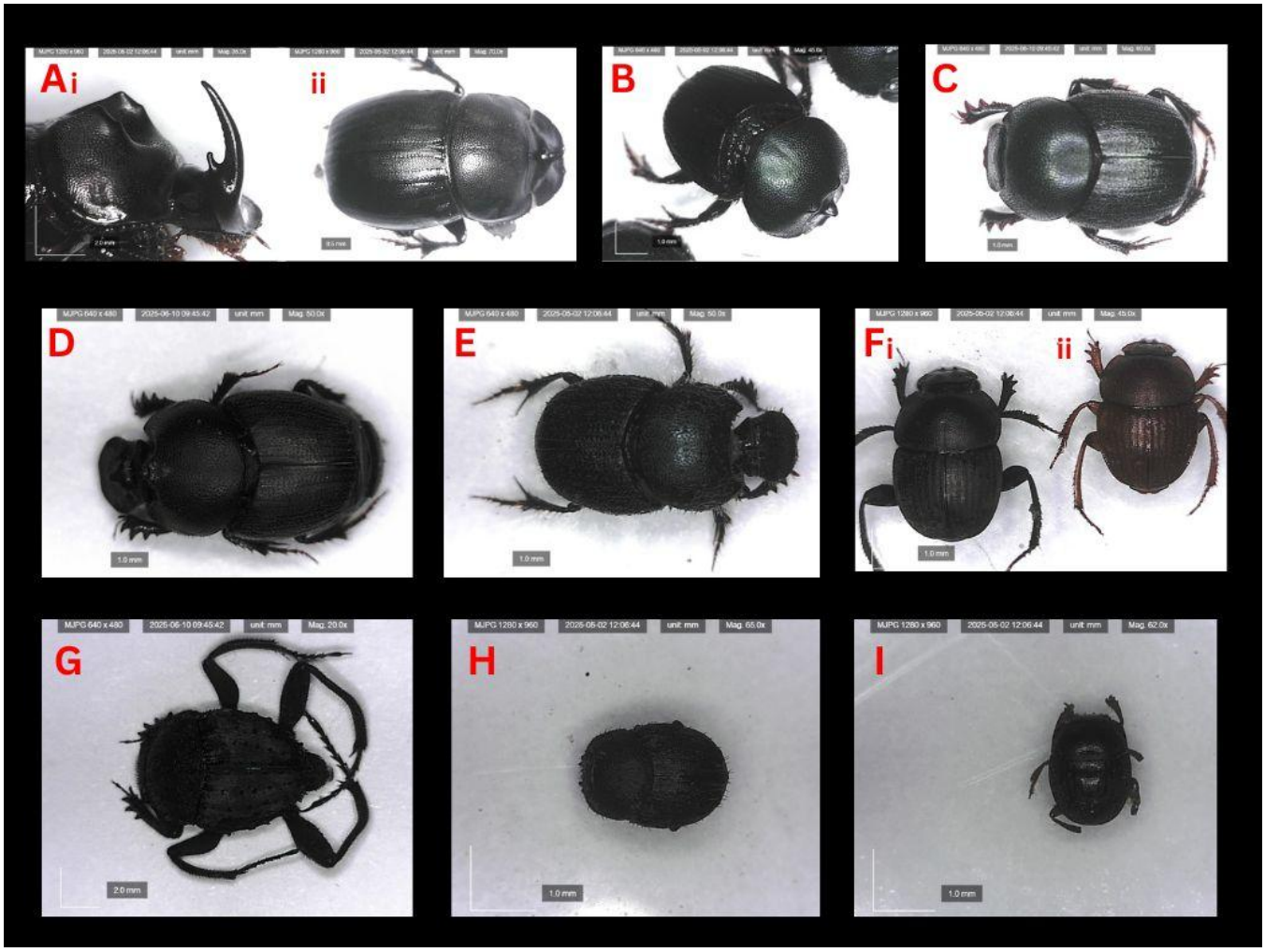


Figure 2: Macroscopic images of dung beetle species recorded during the study, captured using DinoCapture 3.0. Panels show representative individuals of: (**A**) *Copris fidius*; (**B**) *Onthophagus cf. fimetarius*; (**C**) *Onthophagus deterrens*; (**D**) *Onthophagus hyaena*; (**E**) *Onthophagus pilosus*; (**F**) *Epirinus hilaris* (**i**) and *Epirinus silvestris* (**ii**); (**G**) *Sisyphus muricatus*; (**H**) *Odontoloma pygidiale*; and (**I**) *Bohepilissus subtilis*. Images illustrate dorsal and lateral morphological features used for species identification; scale bars are shown in each panel.

Results

Across all habitats, a total of 9 dung beetle species were recorded, comprising 640 individual specimens. Species composition and total abundance varied among habitat types (**Table 1**). Burnt fynbos supported the highest overall species richness, with 7 species recorded and a total of 111 individuals. The most abundant species in burnt fynbos was *Epirinus* spp. (n = 69), followed by *Onthophagus cf. fimetarius* (n = 22) and *Copris fidius* (n = 13). Additional species recorded exclusively in burnt fynbos included *Odontoloma pygidiale* (n = 1) and *Onthophagus hyaena* (n = 2). Dense fynbos supported fewer species, but exhibited the highest overall abundance, with 449 individuals recorded. Assemblages were strongly dominated by *Epirinus* spp. (n = 405), followed by *Onthophagus cf. fimetarius* (n = 28) and *Copris fidius* (n = 12). Two species were recorded only in dense fynbos: *Onthophagus deterrens* (n = 1) and *Sisyphus muricatus* (n = 2). Forest sites supported three species and a total of 80 individuals, with assemblages dominated by *Epirinus* spp. (n = 66), followed by *Copris fidius* (n = 13). *Bohepilissus subtilis* was

recorded exclusively in forest habitat, represented by a single individual (n = 1). Several species were restricted to fynbos habitats and occurred across both burnt and dense fynbos sites, including *Onthophagus cf. fimetarius* and *Onthophagus pilosus*. In contrast, no species were shared exclusively between forest and fynbos habitats without also occurring in both fynbos types (**Table 1**).

Total dung beetle abundance per sampling point differed significantly among habitat types (Kruskal–Wallis rank sum test: χ^2 = 8.9168, df = 2, p = 0.0116). Post-hoc pairwise comparisons using Wilcoxon rank-sum tests with Holm correction indicated that dung beetle abundance in dense fynbos differed significantly from both burnt fynbos (p = 0.036) and forest habitats (p = 0.036). In contrast, no significant difference in abundance was detected between burnt fynbos and forest sites (p = 0.599). Median dung beetle abundance per sampling point was highest in dense fynbos (median = 75 individuals, IQR = 54), compared to burnt fynbos (median = 19, IQR = 20) and forest habitats (median = 16, IQR = 2) (**Figure 3**).

Species richness per sampling point did not differ significantly among habitats (Kruskal–Wallis rank sum test: χ^2 = 3.2732, df = 2, p = 0.1946). Pairwise Wilcoxon rank-sum tests with Holm correction similarly showed no significant differences in species richness between any habitat pairs (burnt vs. dense fynbos: p = 0.76; burnt fynbos vs. forest: p = 0.22; dense fynbos vs. forest: p = 0.76). Median species richness was similar across habitats, with burnt fynbos supporting a median of 3 species per sampling point (IQR = 0), dense fynbos 2 species (IQR = 1), and forest 2 species (IQR = 0), based on pooled paired pitfall traps (n = 5 sampling points per habitat).

Shannon diversity (H′) also showed no significant differences among habitats (Kruskal–Wallis rank sum test: χ^2 = 3.9721, df = 2, p = 0.1372). Pairwise Wilcoxon rank-sum tests with Holm correction revealed no significant differences between burnt and dense fynbos (p = 0.30), burnt fynbos and forest (p = 0.29), or dense fynbos and forest (p = 0.60). Median Shannon diversity values were broadly similar across habitats, indicating comparable patterns of species evenness despite differences in total abundance (**Figure 4**).

Multivariate community composition

Dung beetle community composition differed significantly among habitats (PERMANOVA based on Bray–Curtis dissimilarities: $F_{2,12}$ = 3.66, p = 0.002), with habitat type explaining approximately 38% of the variation in species composition. Tests for homogeneity of multivariate dispersion were not significant (p = 0.36), indicating that observed differences were due to shifts in species composition rather than differences in within-habitat variability.

NMDS ordination provided a good representation of community structure and revealed distinct clustering of forest sites relative to fynbos habitats (**Figure 5**). Burnt and dense fynbos assemblages showed partial overlap in ordination space, indicating similarities in species presence but differences in relative abundances among sites. These patterns visually support the PERMANOVA results and highlight compositional differences among habitats despite comparable levels of species richness and Shannon diversity.

Table 1: Total number of dung beetle individuals per species recorded in fynbos and forest habitats.

Species	Dense Fynbos	Burnt Fynbos	Forest
Copris fidius	12	13	13
Onthophagus cf fimetarius	28	22	0
Sisyphus muricatus	2	1	0
Onthophagus pilosus	1	3	0
Epirinus spp.	405	69	66
Onthophagus deterrens	1	0	0
Odontoloma pygidiale	0	1	0
Onthophagus hyaena	0	2	0
Bohepilissus subtilis	0	0	1

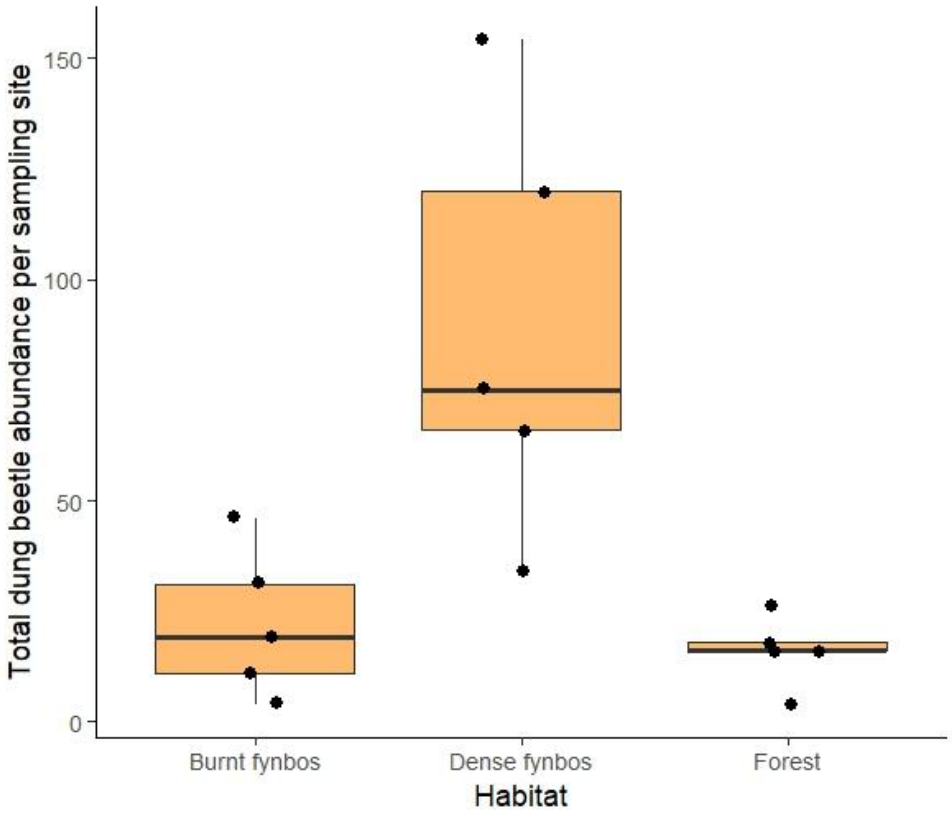


Figure 3: Dung beetle species richness per sampling point across burnt fynbos, dense fynbos, and forest habitats. Median species richness was similar across habitats, with burnt fynbos (median = 3 species, IQR = 0), dense fynbos (median = 2, IQR = 1), and forest (median = 2, IQR = 0 calculated from pooled paired pitfall traps (n = 5 per habitat).

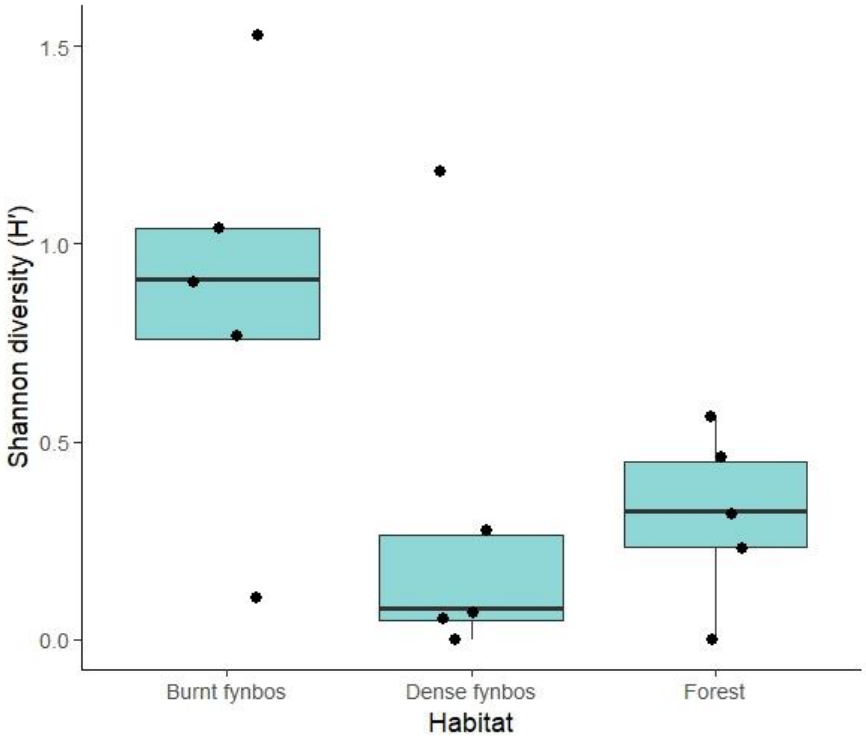


Figure 4: Shannon diversity (H′) of dung beetle assemblages per sampling point across burnt fynbos, dense fynbos, and forest habitats. Diversity values were calculated from pooled paired pitfall traps at each sampling point. Jittered points represent individual sampling points (n = 5 per habitat).

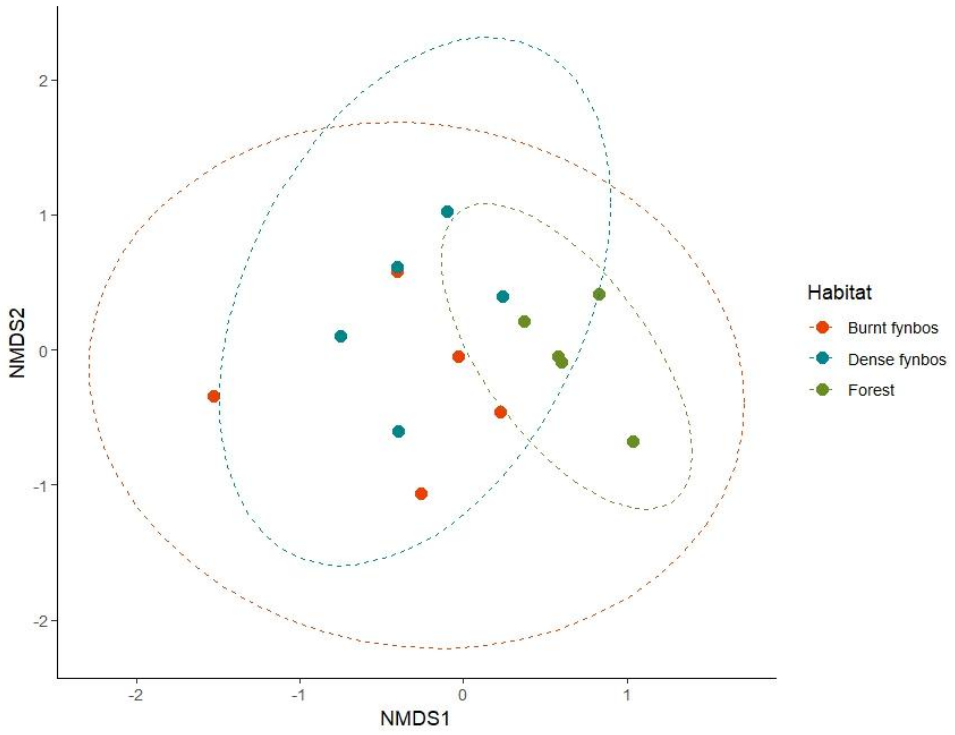


Figure 5: NMDS ordination of dung beetle assemblages across forest, burnt fynbos, and dense fynbos habitats based on Bray–Curtis dissimilarities. Points represent pooled sampling points, with dashed ellipses indicating habitat groupings. Community composition differed significantly among habitats (PERMANOVA, p = 0.002; stress = 0.12).

Discussion

Dung beetle abundance differed significantly among habitats, with dense fynbos supporting substantially higher abundances than both burnt fynbos and forest sites. This pattern suggests that mature fynbos provides more favourable conditions for dung beetle activity, potentially due to greater habitat complexity, litter accumulation, and microclimatic buffering.

In dense fynbos, accumulated plant material and increased structural heterogeneity may enhance moisture retention and provide more stable microhabitats for ground-dwelling beetles, consistent with findings from other arthropod studies in heathland and post-fire systems (Kerdoncuff *et al*., 2023). In contrast, burnt fynbos sites, characterised by open vegetation and reduced litter cover, may impose harsher thermal and desiccation stress on dung beetles, limiting overall abundance despite the availability of dung resources. Interestingly, forest sites supported similarly low dung beetle abundances to burnt fynbos, despite their structural complexity. This may reflect differences in mammal dung availability, soil conditions, or microhabitat suitability within southern Afrotemperate forest, where dense canopy cover and cooler conditions may favour a smaller subset of dung beetle species (Davis, 1996; Martínez-Falcón *et al*., 2018).

In contrast to abundance, species richness and Shannon diversity did not differ significantly among habitats, indicating that similar numbers of species were present across burnt fynbos, dense fynbos, and forest sites. This pattern suggests that habitat differences influenced the number of individuals and relative dominance of species, rather than the presence or absence of species *per se*. Such decoupling of abundance and richness has been reported in other dung beetle studies, particularly in resource-limited or disturbed systems where generalist species persist across habitats but vary in dominance (Davis, 1997; Jankielsohn & Daniel, 2024). The dominance of *Epirinus* spp. across all habitats, especially in dense fynbos, supports the idea that generalist or disturbance-tolerant species can maintain broad distributions even in structurally contrasting environments. The relatively low overall species richness recorded in this study is consistent with expectations for fynbos systems, which support lower large-mammal diversity compared to savanna ecosystems, thereby limiting dung resource diversity (Davis & Scholtz, 2020a).

Community composition and habitat specificity

Despite similarities in species richness, multivariate analyses revealed significant differences in dung beetle community composition among habitats, with forest assemblages clustering distinctly from fynbos sites. This indicates that habitats supported different species assemblages and dominance structures, rather than simply varying numbers of the same species. Several species showed clear habitat associations. Species such as *Onthophagus deterrens* and *Sisyphus muricatus* were restricted to dense fynbos, while *Odontoloma pygidiale* and *Onthophagus hyaena* occurred only in burnt fynbos. In contrast, *Bohepilissus subtilis* was recorded exclusively in forest habitat. These patterns suggest that certain species may be sensitive to vegetation structure, microclimate, or post-fire conditions, consistent with functional group-based responses reported in previous studies (Doube, 1990; Davis, 1996). The partial overlap between burnt and dense fynbos assemblages observed in the NMDS ordination suggests shared species pools but differing relative abundances, potentially reflecting post-fire recolonisation dynamics. Early-

successional fynbos may initially support a subset of disturbance-tolerant species, with assemblages gradually shifting as vegetation structure and soil conditions recover.

Conclusion

This study demonstrates that dung beetle communities in the Nature's Valley region vary markedly across burnt fynbos, dense fynbos, and Afrotemperate forest habitats. While species richness and Shannon diversity were broadly similar among habitats, dung beetle abundance and community composition differed significantly, with dense fynbos supporting higher abundances and forest sites forming distinct assemblages. These patterns indicate that habitat type and post-fire vegetation structure influence dung beetle assemblages primarily through changes in abundance and species dominance rather than through species turnover alone.

The findings highlight the importance of vegetation structure and post-fire habitat age in shaping dung beetle communities within fynbos ecosystems. In particular, mature fynbos appears to provide more favourable conditions for dung beetle activity, while forest habitats support compositionally distinct assemblages despite lower overall abundance. Together, these results support the use of dung beetles as indicators of habitat condition and post-disturbance recovery and provide valuable baseline data to inform biodiversity monitoring and fire-management strategies within the Nature's Valley Trust and broader conservation initiatives.

Study limitations

While this study provides important baseline insights, it is subject to several limitations. Sampling was conducted over a single week within one season, and dung beetle assemblages are known to exhibit strong seasonal variation in activity and composition (Davis, 1997). In addition, mammal dung availability was not quantified, limiting the ability to directly assess resource–beetle relationships across habitats. Other environmental variables that were not recorded due to time and resource constraints – including soil properties, vegetation structure (e.g. vegetation height classes), and rainfall patterns – are also likely to influence dung beetle distribution and activity and should be considered in future work. Future research could build on these findings by incorporating seasonal sampling, functional group–based analyses, and direct measures of dung availability, alongside more detailed measurements of soil characteristics, vegetation structure, distance to habitat patch edge and local rainfall. Expanding sampling across additional sites and a broader range of post-fire ages would further strengthen inference regarding post-fire succession and recolonisation processes in fynbos landscapes.

By linking fire ecology, vegetation structure, and insect community dynamics, this study contributes to a growing understanding of how fire-driven landscapes influence dung beetle assemblages and the ecosystem services they provide in this region.

Acknowledgements

We thank the NVT interns for their assistance with fieldwork, often under challenging weather conditions. We are grateful to SANParks for providing access to their property. We also thank John Wilson for granting access to the forest study site within Hebron Private Nature Reserve.

References

Bond, W. 1980. Fire and Senescent Fynbos in the Swartberg, Southern Cape. South African Forestry Journal, 114(1), 68–71.

Davis, A.L.V. 1997. Climatic and biogeographical associations of southern African dung beetles (Coleoptera: Scarabaeidae). African Journal of Ecology, 35(1), 10–38.

Davis, A.L.V. 1994a. Associations of Afrotropical Coleoptera (Scarabaeidae: Aphodiidae: Staphylinidae: Hydrophilidae: Histeridae) with dung and decaying matter: implications for selection of fly-control agents for Australia. Journal of Natural History, 28(2), 383-399.

Davis, A. L.V. 1994b. Habitat fragmentation in southern Africa and distributional response patterns in five specialist or generalist dung beetle families (Coleoptera). African Journal of Ecology, 32(3), 192–207.

Davis, A.L.V. 1996. Community organization of dung beetles (Coleoptera: Scarabaeidae): differences in body size and functional group structure between habitats. African Journal of Ecology, 34(3), 258–275.

Davis, A.L.V. & Scholtz, C.H. 2020. Dung beetle conservation biogeography in southern Africa: current challenges and potential effects of climatic change. Biodiversity and Conservation, 29(3), 667–693.

Doube, B.M. 1990. A functional classification for analysis of the structure of dung beetle assemblages. Ecological Entomology, 15(4), 371–383.

Halffter, G. & Favila, M.E. 1993. The Scarabaeinae (Insecta: Coleoptera), an animal group for analysing, inventorying and monitoring biodiversity in tropical rainforest and modified landscapes. Biology International, 27, 15–21.

Jankielsohn, A. & Daniel, G. 2024. Protection and Restoration of Natural Habitats as a Nature-Based Solution for Sustaining Insect Biodiversity and Associated Ecosystem Functions to Ensure Sustainable Food Production. Rochester, NY.

Jankielsohn, A., Scholtz, C.H. & Louw, S. 2001. Effect of Habitat Transformation on Dung Beetle Assemblages – A Comparison Between a South African Nature Reserve and Neighboring Farms. Environmental Entomology, 30(3), 474–483.

Kerdoncuff, M., Måren, I.E. & Eycott, A.E. 2023. Traditional prescribed burning of coastal heathland provides niches for xerophilous and sun-loving beetles. Biodiversity and Conservation, 32(12), 4083–4109.

Kraaij, T. & van Wilgen, B.W. 2014. Drivers, ecology, and management of fire in fynbos', in N. Allsopp, J.F. Colville, and G.A. Verboom (eds) Fynbos: Ecology, Evolution, and Conservation of a Megadiverse Region. Oxford University Press.

Krell, F.-T. 2007. Dung Beetle Sampling Protocols. DMNS Technical Report.

Kruger, F.J. & Bigalke, R.C. 1984. Fire in Fynbos', in P.V. de Booysen and N.M. Tainton (eds) Ecological Effects of Fire in South African Ecosystems. Berlin, Heidelberg: Springer, pp. 67–114.

Martínez-Falcón, A.P., Verdú, J.R. & Lumaret, J.P. 2018. Populations and assemblages living on the edge: Dung beetles responses to forests–pasture ecotones. PeerJ, 6, e6148.

Moolman, L., Ferreira, S.M. & Gaylard, A. 2019. The decline of the Knysna elephants: Pattern and hypotheses. South African Journal of Science, 115(3–4), 1–7.

Pryke, J.S., Roets, F. & Samways, M.J. 2022. Large African herbivore diversity is essential in transformed landscapes for conserving dung beetle diversity. Journal of Applied Ecology, 59(5), 1372–1382.

Pryke, J.S. & Samways, M.J. 2012. Importance of using many taxa and having adequate controls for monitoring impacts of fire for arthropod conservation. Journal of Insect Conservation, 16(2), 177–185.

Smit, I.P.J., Baard, J.A. & van Wilgen, B.W. 2024. Fire regimes and management options in mixed grassland-fynbos vegetation, South Africa. Fire Ecology, 20(1), 29.

Swart, R.C., Pryke, J.S. & Roets, F. 2018. Arthropod assemblages deep in natural forests show different responses to surrounding land use. Biodiversity and Conservation, 27(3), 583-606.

van Bergen, G., Coetzee, A., Venter, J.A., Roets, F. & Swart, R.C. 2025. Dung and carrion beetle diversity in small forest patches versus large continuous forest in the southern Cape, South Africa. African Zoology, 60(3), 199–207.

van Wilgen, B.W. 2013. Fire management in species-rich Cape fynbos shrublands. Frontiers in Ecology and the Environment, 11(s1), 35-44.

van Wilgen, B.W., Le Maitre, D.C. & Kruger, F.J. 1985. Fire Behaviour in South African Fynbos (Macchia) Vegetation and Predictions from Rothermel's Fire Model. Journal of Applied Ecology, 22(1), 207–216.

Leopard (*Panthera pardus*) occurrence records from camera traps and citizen science in Nature's Valley, Tsitsikamma

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How to cite this article:

Talbot-Ponsonby, N.& Swart, R.C. 2026. Leopard (*Panthera pardus*) occurrence records from camera traps and citizen science in Nature's Valley, Tsitsikamma. *Nature's Letters* **3**:100-108. URL: https://nvt.org.za/natures-letters/

Abstract

Leopards (*Panthera pardus*) persist across heterogeneous mosaic landscapes in the Eastern and Western Cape, yet its spatial ecology remains poorly understood in many regions. This study combined a camera trap survey and citizen science data to compile a preliminary assessment of potential leopard presence and distribution in Nature's Valley, Tsitsikamma. Camera traps recorded one leopard photograph during the study period, while citizen-reported and verified sightings, tracks, and scat extended spatial coverage. A nearest-neighbour analysis of leopard records within the Nature's Valley landscape indicated no significant departure from a random spatial distribution (NNI = 1.07, z = 0.69, p = 0.49). Although kernel density heat mapping revealed a clear core area of activity around Kalanderkloof, the presence of several more dispersed peripheral records across the wider landscape increased mean inter-point distances, resulting in a spatial pattern that was not globally clustered. This suggests that leopard distribution is characterised by a central high-use area embedded within a broader matrix of more scattered movements. Leopard signs were found near human-use areas, highlighting potential biases in citizen science data towards areas that humans frequent. However, it confirms leopard persistence in the landscape and even within the perimeters of the Village. These findings emphasize the importance of maintaining habitat connectivity, conserving prey populations, and integrating systematic surveys with community involvement to inform effective leopard conservation strategies.

Introduction

The leopard (*Panthera pardus*) is the most widespread felid, with global distributions stretching from the southernmost tip of Africa to the tropical sub-regions of Asia (Stein *et al*., 2025). Its widespread distribution and ecological success mainly result from its opportunistic hunting and feeding behaviours, its ability to evade detection by humans, as well as its adaptability to a wide range of different terrains, from arid savannah to forests and mountainous ecoregions (Jacobson *et al*., 2016). However, despite its adaptability, leopards across Africa are still facing anthropogenic pressures limiting their survival, through habitat fragmentation, illegal hunting, depletion of natural prey sources and direct conflict with humans (Loveridge *et al*., 2022). In 2016, the African leopard was described as "vulnerable" under the IUCN Red List assessment (Stein *et al*., 2025).

Although not taxonomically different from other African leopards, leopards in the Cape do exhibit some morphological and ecological differences, including a smaller body size and much larger home ranges in comparison to other sub-populations (Martins & Martins, 2006). For example, whereas male African leopards usually weigh between 50-70kg, Cape leopard males typically only weigh between 30-35kg (Hayward *et al*., 2006). Whereas savannah leopards typically target medium to large (10-40kg) ungulates such as impala, nyala and duiker (Hayward & Kerley, 2008), leopards of the Cape mainly rely on smaller prey species like rock hyrax, klipspringer, Lagomorpha, rats, and bushbuck (Jansen *et al*., 2023; Braczkowski *et al*., 2012). This difference in prey composition is closely linked to their smaller body size and the limited abundance of large herbivores within their habitat.

Several camera trap studies in the Western and Eastern Cape have been conducted over the last two decades, mainly to document population density. For example, Müller *et al*. (2022), carried out a camera trap survey between 2017 and 2018 in the Cederberg Mountains (a dry, mountainous region of the Western Cape) to assess population density and size, activity hotspots and movement behaviours in a particular population of Cape leopard. They reported frequent low densities of leopard, approximately 1.5–1.6 individuals per 100 km² (e.g., home range of 62.5-66.6 km² - Müller *et al*., 2022). These are relatively low densities when compared to savannah leopards, which have been found to have female home ranges ranging from 5.65 – 61.81 km² and male home ranges ranging from 16.28 – 89.88 km² (Le Roex *et al*., 2022). However, other studies reported home ranges of 35–50 km² for females and 100–150 km² for males in well-resourced savannah ecosystems (Bailey, 1993).

Cape leopards inhabit a diverse range of environments, including fynbos, indigenous forest, pine plantations, mountainous regions, and farmland (Martins & Martins, 2006). The persistence of leopards across such fragmented and human-modified landscapes depends heavily on the maintenance of wildlife corridors that allow for dispersal between isolated ranges (Devens *et al*., 2021). However, these pathways are increasingly threatened by expanding human development (Devens *et al*., 2021). Despite these pressures, the species remains a flagship predator in the Western Cape and the last surviving terrestrial large carnivore in the region. The ability to conserve and protect large carnivores, such as leopards, is reliant on a detailed understanding of their local ecology and activity, including habitat use and spatial distribution. This information is vital for informing conservation strategies, identifying potential corridors and reducing human-wildlife conflict.

The southern Cape area supports the largest indigenous forests in South Africa, which are interspersed with fynbos, a Mediterranean-type shrubland (Mucina *et al*., 2006). Within the southern Cape, the Garden Route National Park (GRNP) is the largest conservation authority. A unique village can be found completely surrounded by the GRNP, named Nature's Valley. This village can only be accessed by two mountain passes, traversing steep mountains and dense forest vegetation. The Village itself lies at sea level. Development is restricted to ca. 400 private houses, one retailer and one restaurant, and the Village retains a semi-natural state. Research on leopards in the broader southern Cape area is scant (Braczkowski *et al*., 2012; McManus *et al*., 2022); none has focussed solely on Nature's Valley. Furthermore, few studies have

integrated local knowledge, i.e., citizen science, and sightings of the species to identify their spatial distribution. Here, we aim to identify leopard occurrence records in and around Nature's Valley to better understand their distribution over the landscape, and inform stakeholders accordingly.

Study area

This study took place in the Tsitsikamma region of the Garden Route, specifically around Nature's Valley (**Figure 1**). Nature's Valley and the surrounding areas provide a unique mosaic of indigenous forest, pine plantations, fynbos and agricultural land. This unique composition of natural and transformed habitats presents both opportunities and challenges for leopard conservation, making it an ideal location to investigate spatial distribution patterns.

Methods

Camera trap deployment

This study focused primarily on monitoring leopard occurrence patterns through the use of camera traps. Camera traps were placed out on the 14th July 2025 and taken down on 31st August 2025. Initially, three camera traps were strategically deployed along key locations based on previous sightings and evidence of occurrence: the Salt River hiking trail in an indigenous forest (camera 2), the West Pass in natural fynbos (camera 3), and the East Pass adjacent to the pine plantation, in thicket vegetation (camera 1) (**Figure 1**). Unfortunately, camera 3 was stolen, resulting in a loss of data from that site. Camera 2 was moved from an indigenous forest site along the Salt River hiking trail to an indigenous forest site at the bottom of Kalanderkloof, after security concerns, and at a location where a leopard was sighted only two months prior (**Figure 1**).

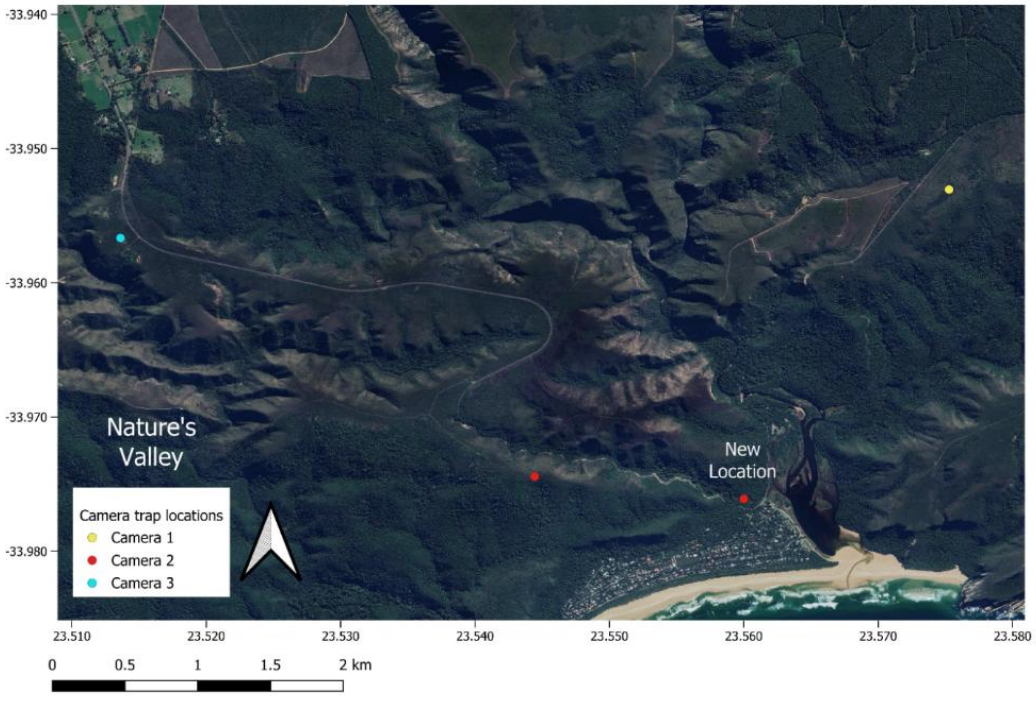


Figure 1: Camera trap locations (before and after relocation) used in the current study. Camera 3 was stolen after one week.

We used NUMaxes infrared cameras (12-megapixel resolution) with a trigger distance of approximately 6m. Each unit was set to light mode on and programmed to capture three consecutive photographs per detection event. Cameras were mounted at a height of 50cm above ground level. Camera 1 was oriented west and operated for four weeks, camera 3 was oriented south and deployed for one week before being stolen, and camera 2 was oriented south and remained active for a total of five weeks, of which three weeks were along the Salt River hiking trail and two weeks at the bottom of Kalanderkloof.

Community reported sightings

Citizen science data was collected to supplement camera trap detection photographs by compiling community-reported leopard signs (live sightings, tracks, scat, and suspected kill-sites) over a two-month period coinciding with and immediately following camera trap deployment. Reports were submitted via community communication channels facilitated by the NVT. Given the high potential for misidentification of leopard signs, particularly confusion with caracal, domestic dog, or baboon spoor and scat, photos had to be submitted of reported signs. Records from pre-2020 were discarded. Records were then independently verified using submitted photographs and making use of field guides (Liebenberg, 2008; Stuart & Stuart, 2019).

Data analysis

To assess whether leopard records were randomly distributed, clustered, or spatially dispersed across the study area, a Nearest Neighbour Index (NNI) was calculated using the geographic coordinates of all recorded and verified signs, including sightings, tracks, scats, and kill sites. Spatial data were standardised and processed prior to statistical analysis in R, through which the NNI was computed. This index compares the observed mean nearest-neighbour distance between points to the expected distance under a spatially random (Poisson) distribution, with values below one indicating clustering, values close to one indicating randomness, and values above one indicating spatial dispersion. Spatial patterns were further visualised in QGIS to support interpretation of leopard space use across the landscape using the kernel density heat map function. To characterise the relative frequency of different types of leopard evidence reported by community members, records were also classified by sign type.

Results

We verified 29 records of leopards in the Nature's Valley area, starting in April 2020 to December 2025. Leopard signs were mostly found near roads, cycling tracks or along hiking trails (**Figure 2**). The northernmost sign was next to the Bobbejaansriver and the southernmost sign was on the beach of Nature's Valley by beach entrance 1, near Blue Rocks. Leopard signs were found in close proximity to the Village. For example, three signs were within village boundaries; the latest record was a live sighting in St Andrews drive during December 2025. The north-easternmost sign was a sighting during the survey period; a leopard captured on camera trap 1 (**Figure 3**).

The kernel density heat map showed areas of higher reporting frequency of leopard records within the study area (**Figure 4**). The highest concentration of reported records occurred immediately north of Kalanderkloof, where seven reported signs were located in close proximity. Additional areas of elevated reporting density were observed further north

and east. These patterns reflect the spatial distribution of reported leopard records within the dataset rather than confirmed areas of activity or movement. The most common evidence of leopard activity was tracks, followed by sightings, scat, and kill sites (**Figure 5**).

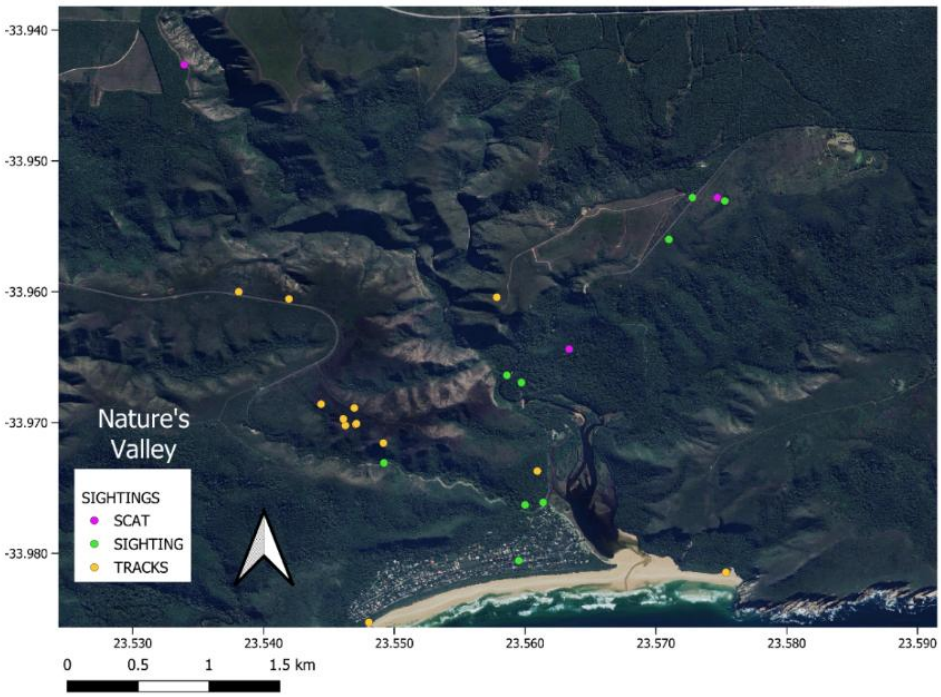


Figure 2: Locations of verified leopard records (sightings, tracks and scat) within and around Nature's Valley from the period 2020-2025.



Figure 3: During the survey period, we photographed one leopard through camera trap 1, on the 28th of July 2025 at 23:36.

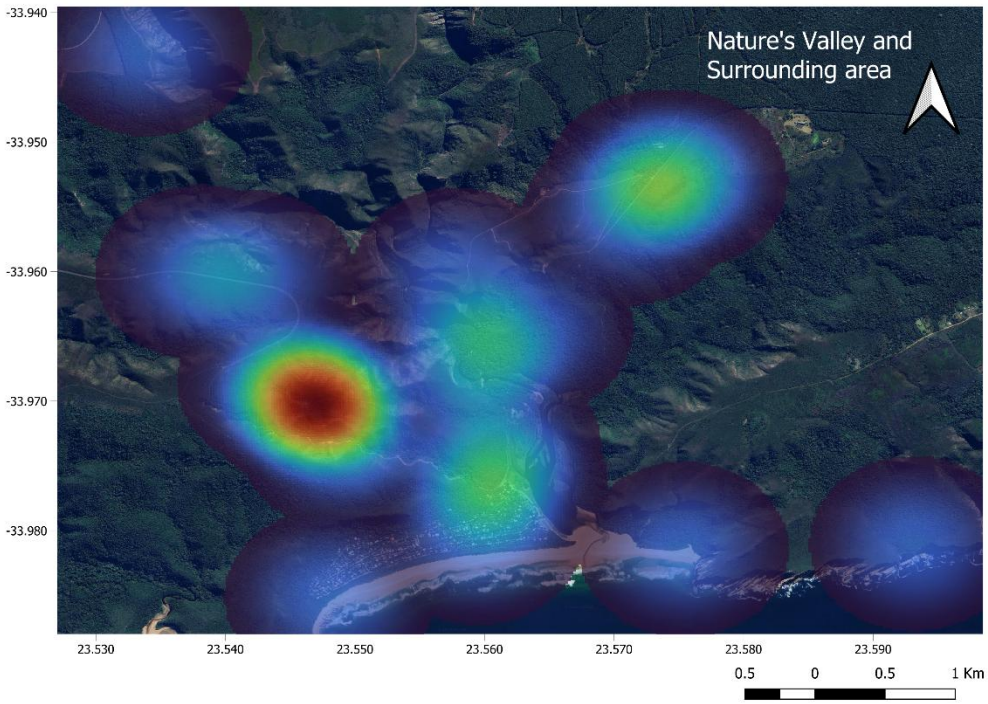


Figure 4: Kernel density heat map showing densities of location sightings based on community-reported records and a camera trap survey.

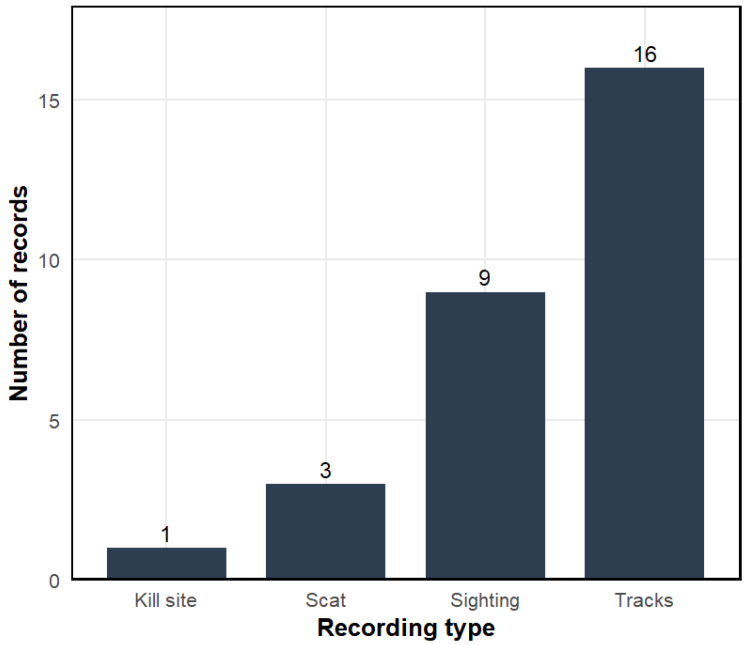


Figure 5: Bar chart showing the frequency of leopard records (n = 29) in the Nature's Valley area, separated by type of evidence (tracks, scat, sightings, and kill site). Tracks were the most commonly reported signs, followed by sightings and scat, while kill sites were rare.

A nearest-neighbour analysis of leopard records within the Nature's Valley landscape indicated no significant departure from a random spatial distribution (NNI = 1.07, z = 0.69, p = 0.49). Although kernel density mapping revealed a clear core area of activity around Kalanderkloof, the presence of several more dispersed peripheral records across the wider landscape increased mean inter-point distances, resulting in a spatial pattern that was not globally clustered. This suggests that leopard space use is characterised by a central high-use area embedded within a broader matrix of more scattered movements.

Discussion

Although only one leopard was recorded on the camera traps during a very short camera deployment period, the integration of community-sourced records (tracks, scat, sightings, and one kill site) provided a snapshot of leopard presence across the Nature's Valley area. Despite the clear limitation of our study, being limited time and resources in the form of cameras, the community reported records extended the spatial coverage of the study and highlight the importance of incorporating citizen science into monitoring efforts. Citizen observations revealed at least eight separate sighting locations, clustered mainly around the forest–fynbos and plantation–farmland interfaces, suggesting that leopards are moving between multiple habitat types and not restricted to a single habitat type, nor to natural vegetation types only. Surprisingly, the most recent live sighting was in December 2025, a peak tourist period, within the Village of Nature's Valley itself, at ca. 23:00 crossing St Andrews drive.

Kernel density heat mapping showed spatial variation in the reporting frequency of leopard records across the study area; however, the Nearest Neighbour Index indicated no significant departure from a random spatial distribution at the landscape scale (NNI = 1.07, p = 0.49). This suggests that while leopards concentrate their activity within a central high-use zone, records are also distributed across a broader landscape matrix, resulting in an overall pattern that is not globally clustered. Our results suggest that leopard space use is locally concentrated around Kalanderkloof, but globally diffuse across the wider mosaic landscape. One drawback of citizen science in this context is that leopard signs are largely restricted to areas of human use, such as hiking trails, cycling tracks, or near recreational areas like picnic spots. Consequently, large portions of the landscape, particularly remote or less-accessible habitats, may be undersampled or completely missed. This spatial bias can result in an incomplete understanding of leopard distribution and habitat use, as areas with low human presence may be underrepresented in the data. Supplementing citizen science with more systematic survey approaches could help capture a more accurate picture of leopard activity across the landscape. For such an all-encompassing survey, more camera traps will need to be deployed. If camera traps are to be deployed in the future, it is advised that they be placed in pairs, to assist in identifying individual leopards.

The combination of camera trap and citizen science data confirms that leopards continue to persist across this heterogeneous, mosaic landscape. However, the relatively low frequency of direct detections, along with the predominance of indirect signs such as tracks and scat, could align with previous research showing that Cape leopards occur at very low densities across the Eastern and Western Cape (Müller *et al*., 2022). This is consistent with previous studies showing that leopards in the Cape and other human-modified landscapes can occupy large home ranges, particularly

where prey availability is low and human disturbance is high. We were also not able to identify any clear movement corridors via our methodology, and propose further work on this important aspect.

The integration of community-reported data also underscores the value of citizen involvement in conservation science. These contributions not only expanded our dataset, but could also promote local stewardship of wildlife by making citizens involved in active research. However, caution is needed when interpreting such data, as reports vary in reliability and may be influenced by misidentification. In this study, records were cross-referenced with habitat suitability and photographs, and previous confirmed activity hotspots, reducing the likelihood of error.

Overall, the findings reinforce that although leopards in the Tsitsikamma region face considerable pressure from habitat transformation, snares, and high levels of human activity, they still maintain a presence in close proximity to human settlements and developments. However, the limited scope of the dataset precludes inferences about movement corridors or prey availability. Future research should prioritise long-term, systematic monitoring projects utilising larger camera-trap networks, alongside well-documented citizen science initiatives, to support evidence-based conservation planning and human–wildlife coexistence (McManus *et al*., 2022).

Acknowledgements

We thank John Wilson of Hebron Private Nature Reserve for donating three cameras for use in this study. We also want to thank the local residents who generously shared their knowledge of leopard signs, your contributions have been invaluable and enriched the project. Special appreciation goes to the volunteers and field assistants whose dedication and hard work in the field ensured the success of this study. We are also thankful to Giorgio, Hendri, and Elias for kindly sharing previous reports, which provided essential background and context for this work.

References

Braczkowski, A., Watson, L., Coulson, D. & Randall, R. 2012. Diet of leopards in the southern Cape, South Africa. African Journal of Ecology, 50(3), 377.

Devens, C. H., Hayward, M. W., Tshabalala, T., Dickman, A., McManus, J. S., Smuts, B. & Somers, M. J. 2021. Estimating leopard density across the highly modified human dominated landscape of the Western Cape, South Africa. Oryx, 55(1), 34-45.

Bailey, T.N. 1993. The African Leopard: Ecology and Behavior of a Solitary Felid. The Journal of Wildlife Management, 59(1), p.190.

Hayward, M. W. & Kerley, G. I. (2008). Prey preferences and dietary overlap amongst Africa's large predators. South African Journal of Wildlife Research-24-month delayed open access, 38(2), 93-108.

Hayward, M.W., Henschel, P., O'Brien, J., Hofmeyr, M., Balme, G. & Kerley, G.I.H. 2006. Prey preferences of the leopard (Panthera pardus). Journal of Zoology, 270(2), 298-313.

Jacobson, A. P., Gerngross, P., Lemeris Jr, J. R., Schoonover, R. F., Anco, C., Breitenmoser Würsten, C. & Dollar, L. 2016. Leopard (Panthera pardus) status, distribution, and the research efforts across its range. PeerJ, 4, e1974.

Jansen, C., Leslie, A. J., Cristescu, B., Teichman, K. J. & Martins, Q. E. 2023. Leopards living at the farmland protected area interface prefer wild prey but consume high biomass of livestock. African Journal of Ecology, 61(4), 1023-1029.

Le Roex, N., Mann, G. K. H., Hunter, L. T. B. & Balme, G. A. 2022. Big competition for small spots? Conspecific density drives home range size in male and female leopards. Journal of Zoology, 316(3), 178 187.

Liebenberg, L. 2008. A field guide to the animal tracks of southern Africa. David Philip Publishers.

Loveridge, A.J., Sousa, L.L., Seymour-Smith, J.L., Mandisodza-Chikerema, R. & Macdonald, D.W. 2022. Environmental and anthropogenic drivers of African leopard Panthera pardus population

density. Biological Conservation, 272, 109641.

Martins, Q. & Martins, N. 2006. Leopards of the Cape: conservation and conservation concerns, International Journal of Environmental Studies, 63(5), 579–585.

McManus, J., Schurch, M.P.E., Goets, S., Faraut, L., Couldridge, V. & Smuts, B. 2022. Delineating Functional Corridors Linking Leopard Habitat in the Eastern and Western Cape, South Africa. Conservation, 2(1), 99 122.

Mucina, L., Geldenhuys, C. J., Rutherford, M. C., Powrie, L. W., Lötter, M. C., & Von Maltitz, G. P. 2006. Afrotemperate, subtropical and azonal forests. The vegetation of South Africa, Lesotho and Swaziland. Strelitzia, 19, 584-614.

Müller, L., Briers-Louw, W.D., Seele, B.C., Stefanus Lochner, C. & Amin, R. 2022. Population size, density, and ranging behaviour in a key leopard population in the Western Cape, South Africa. PLOS ONE, 17(5), 0254507.

Stein, A.B., Athreya, V., Gerngross, P., Balme, G., Henschel, P., Karanth, U., Miquelle, D., Rostro Garcia, S., Kamler, J.F., Laguardia, A., Khorozyan, I. & Ghoddousi, A. 2025. Panthera pardus. The IUCN Red List of Threatened Species 2025: e.T15954A102421779.

Stuart, C. & Stuart, M. 2019. Stuarts' Field Guide to Tracks & Signs of Southern, Central & East African Wildlife. Penguin Random House South Africa.

An exploration of marine and coastal Drivers, Pressures, State, Impacts and Responses (DPSIR) in Plettenberg Bay, South Africa

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How to cite this article:

Du Plessis, S., Vargas-Fonseca, A. & Coetzee, H. 2026. An exploration of marine and coastal Drivers, Pressures, State, Impacts and Responses (DPSIR) in Plettenberg Bay, South Africa. *Nature's Letters* **3**:109-134. URL: https://nvt.org.za/natures-letters/

Abstract

Coastal and marine ecosystems are vital for biodiversity, food security and human well-being, yet face escalating pressures from human activity and climate change. This study investigates the socio-ecological dynamics of Plettenberg Bay, South Africa, spanning from the Tsitsikamma Marine Protected Area to Kranshoek, using the Drivers, Pressures, State, Impacts and Responses (DPSIR) framework. User groups were categorised into extractive (fishers, bait collectors, marine harvesters) and non-extractive (beach and water users, land-based nature enthusiasts, and commercial boat operators). Findings reveal diverse drivers — including urbanisation, tourism expansion, and livelihood dependence on marine resources — alongside broader structural conditions such as socio-economic inequality and shifting livelihood dynamics, which generate pressures including resource depletion, pollution, habitat disturbance and noise impacts. Estuarine systems (Piesang, Keurbooms, Salt, Groot) exhibit varying states of degradation, from urbanisation-driven eutrophication to relatively pristine conditions threatened by runoff. Climate change further exacerbates ecosystem vulnerability through rising temperatures, erosion, and species shifts. Responses identified include stricter regulations, habitat restoration, adaptive management, community co-management, and innovative tourism models. The study underscores the interconnectedness of ecological and socio-economic systems and calls for integrated governance strategies that balance conservation, livelihoods, and sustainable development to secure the long-term resilience of Plettenberg Bay's marine and coastal environments.

Introduction

Coastal areas are highly dynamic, interconnected systems that link terrestrial and marine environments, supporting exceptional biodiversity and critical ecosystem services that underpin human well-being. Plettenberg Bay, on the southern Cape coast of South Africa, is a region of notable ecological importance. Its diverse marine and coastal habitats support a wide range of species. However, increasing pressures from fishing, tourism, and coastal development are placing these ecosystems at risk, highlighting the need for integrated and adaptive management approaches (Harris, 2025).

To address these challenges, this study applies the Drivers, Pressures, State, Impacts and Responses (DPSIR) framework as a structured tool to analyse the interactions between human activities and environmental change. Widely used in coastal and marine management, the DPSIR framework links socio-economic drivers to environmental pressures, evaluates resulting changes in ecosystem

state, and considers their ecological, social, and economic impacts, thereby informing appropriate management responses (Elliott *et al*., 2017; Obubu *et al*., 2022; Sun *et al*., 2016; Timmerman *et al*., 2011).

Applied to Plettenberg Bay, this framework enables a holistic assessment of environmental conditions and emerging threats. Key pressures, including urban expansion, intensifying tourism, and unsustainable resource use, are contributing to habitat degradation, pollution, and biodiversity loss, particularly within estuarine and nearshore systems. These trends underscore the urgency of proactive and evidence-based conservation strategies. Importantly, the DPSIR framework also helps bridge science and management by integrating ecological dynamics with socio-economic dependencies, such as fisheries, ecotourism, and recreation. By identifying priority areas, potential trade-offs, and pathways for intervention, it supports more balanced decision-making that aligns conservation goals with sustainable development, ultimately enhancing the long-term resilience of Plettenberg Bay's ecosystems and the communities that depend on them.

The aim of this study is to apply and adapt the DPSIR framework to systematically analyse the socio-ecological dynamics of Plettenberg Bay, with specific reference to four primary coastal user groups and four key estuarine systems. The study seeks to (1) identify and categorise major drivers and pressures shaping marine and coastal conditions, (2) assess their ecological and socio-economic states and impacts, and (3) evaluate existing and potential response pathways that support integrated and adaptive governance. Through this analysis, the study aims to provide an evidence-based foundation for strengthening marine and coastal management in the Bay.

Study area

Ecological context of Plettenberg Bay

Regional and terrestrial context

Plettenberg Bay area lies within the Cape Floristic Region, a globally recognised biodiversity hotspot, characterised by high levels of endemism and floristic diversity (Myers *et al*., 2000; SANBI, 2019). The Tsitsikamma region, including Plettenberg Bay, it is also recognised as one of South Africa's eleven Strategic Water Source Areas (SWSAs), reflecting its national importance for freshwater provision. This designation reflects the integration of terrestrial and marine systems within the region, reinforcing its importance as a coupled socio-ecological landscape.

The area forms part of the Garden Route Biosphere Reserve, a UNESCO-designated landscape spanning approximately 698,363 ha. This biosphere reserve integrates multiple protected areas, including Goukamma, Robberg, and Tsitsikamma, and encompasses the wider Garden Route Protected Area (88,042 ha), which includes the Garden Route National Park and sections of the Outeniqua Mountains (**Figure 1**).

The surrounding landscape supports Afro temperate forests, which form part of the Southern Cape Forest complex, the largest remaining indigenous forest complex in South Africa (Mucina & Rutherford, 2006). Nationally, the forest biome occupies less than 0.5% of the country's land surface and is considered one of the most spatially restricted and threatened biomes (SANBI, 2019). This combination of high biodiversity value and ecological connectivity highlights the importance of integrated conservation strategies across the region.

Marine and coastal ecosystems

Plettenberg Bay lies within the Eastern Agulhas Bank (Agulhas ecoregion) and was identified in the 2018 National Biodiversity Assessment as one of South Africa's major bays (SANBI, 2019). With average depths of 40–60 m, this marked the first time bays were recognised as distinct ecosystem types, underscoring their ecological importance. Plettenberg Bay comprises a range of coastal habitats, including dunes, sandy beaches, rocky shores, inshore and offshore reefs, and estuarine systems. The bay's inshore reef systems are particularly ecologically important, especially within and adjacent to the Tsitsikamma Marine Protected Area (MPA), the oldest marine reserve in Africa. These habitats support high biodiversity, including endemic fish species, marine mammals, seabirds and benthic communities. Due to this ecological richness, Plettenberg Bay is recognised as one of South Africa's eighteen Ecologically or Biologically Significant Area (EBSA) (Sink *et al*., 2012, 2019). Within the Tsitsikamma–Robberg EBSA, several ecosystem types are classified as endangered or vulnerable (**Figure 2**).

Bays function as semi-enclosed coastal systems that provide sheltered conditions for marine species during critical life stages. Their physical characteristics promote longer water residence times, larval retention, thermal stratification and enhanced phytoplankton productivity. However, these same characteristics also increase the potential for pollutant accumulation (Pfaff *et al*., 2019; SANBI, 2019). The enclosed nature of Plettenberg Bay supports high productivity that is essential for the life stages of many marine species, including commercially important fish. Despite this ecological value, cumulative anthropogenic pressures have degraded ecosystem condition. Current assessments indicate that approximately 28% of the EBSA is in poor ecological condition, while only 37% remains in good condition. **Figure 3** shows the cumulative pressures in the area. Across the broader EBSA, 19 ecosystem types have been identified, of which ten are classified as threatened, further emphasising the need for strengthened conservation and management interventions (Harris *et al*., 2019; Sink *et al*., 2019).



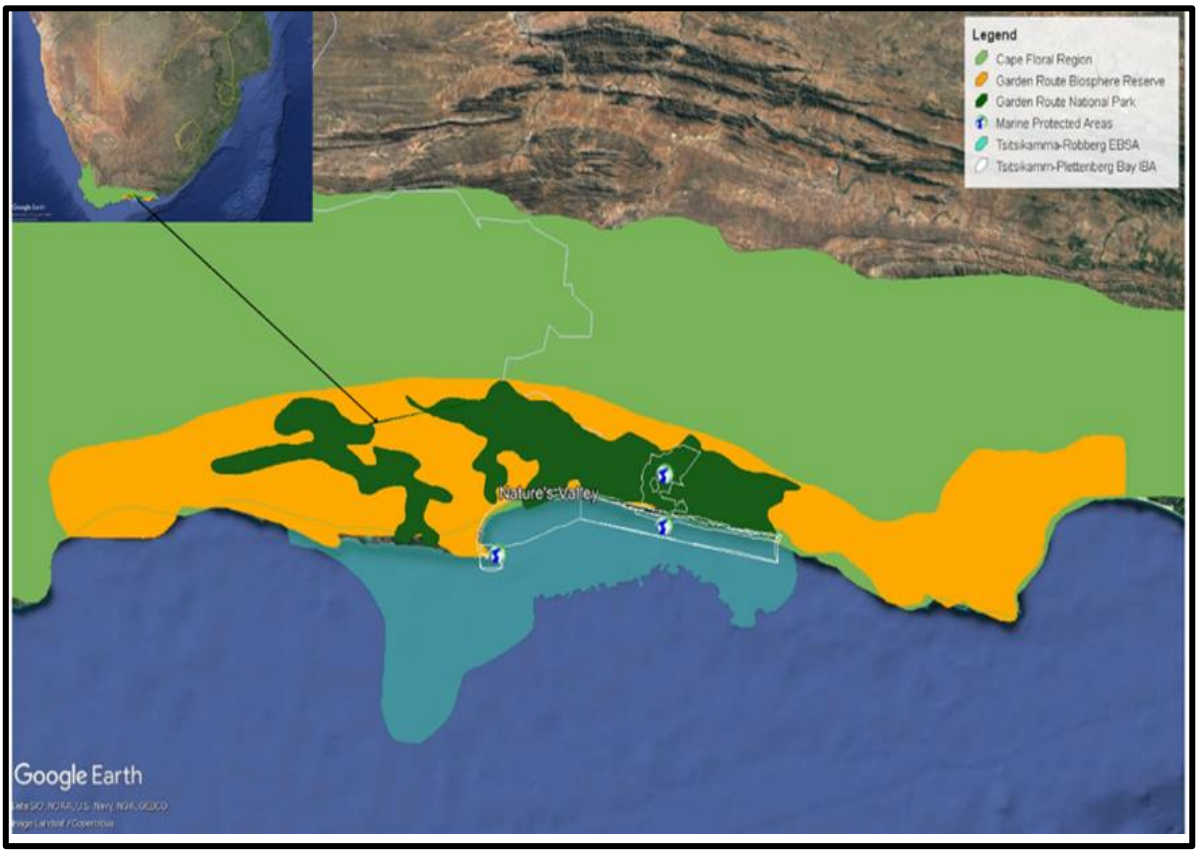


Figure 1: Broader conservation landscape in the Plettenberg Bay area.

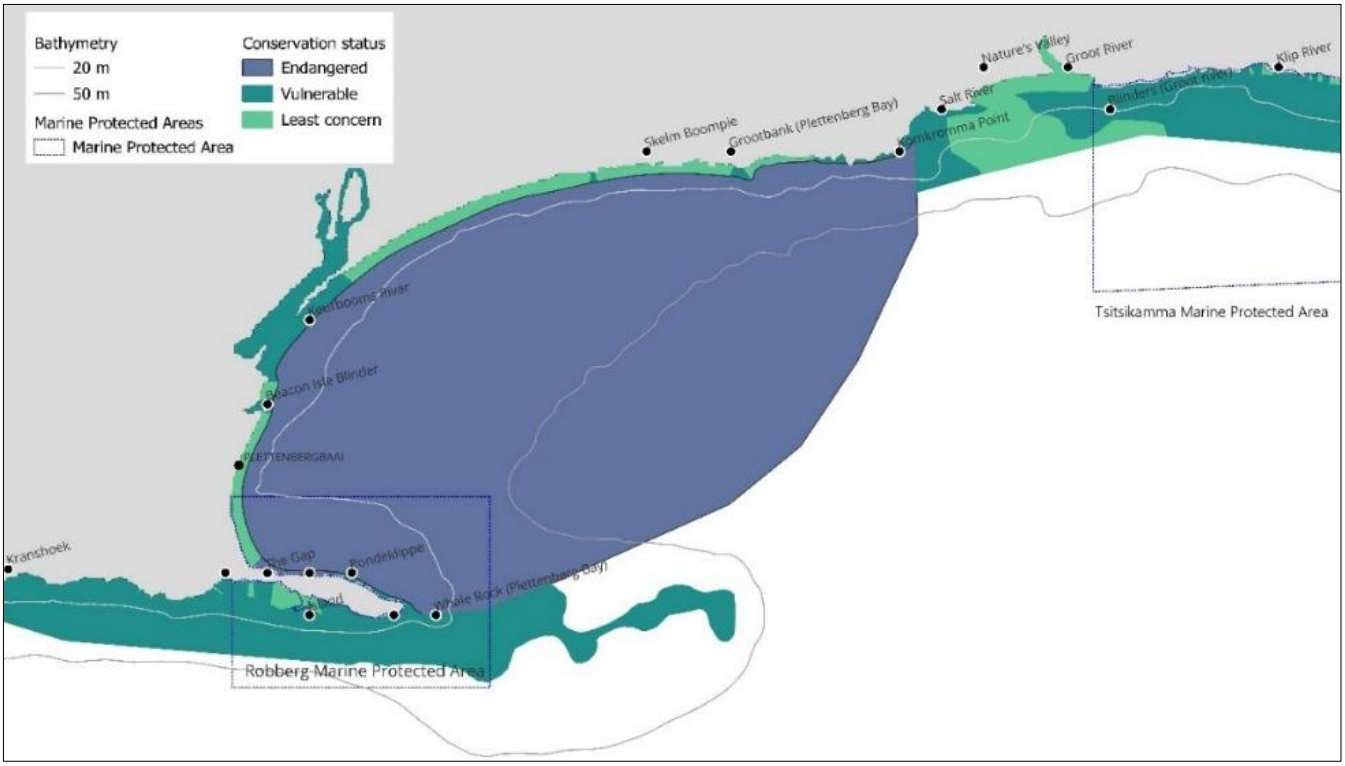


Figure 2: Map indicating conservation status of the Bay.

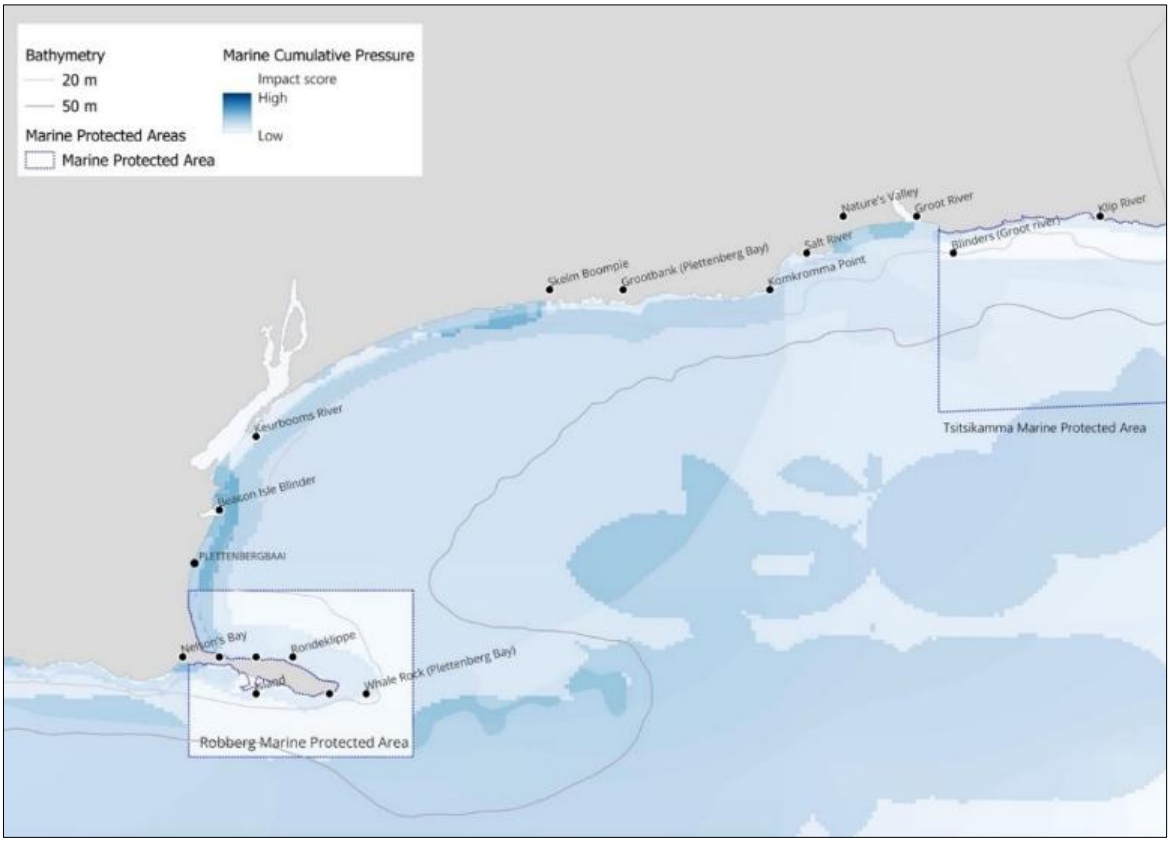


Figure 3: Map of the cumulative pressures in the study area.

Rivers and estuaries

Several ecologically important rivers and estuaries discharge into Plettenberg Bay and play a key role in maintaining regional ecological processes. The Piesang Estuary is located within the urban centre of Plettenberg Bay. The Keurbooms Estuary is of high ecological importance, supporting the endangered Knysna seahorse (*Hippocampus capensis*) and providing habitat for commercially valuable fish species.

The Salt River Estuary, which is less affected by urban development, supports unique assemblages of mayflies (Ephemeroptera) and caddisflies (Trichoptera), including previously undescribed insect genera and species (de Moor & Bellingan, 2010). The Groot River Estuary, located near Nature's Valley, connects directly to the Tsitsikamma MPA. Collectively, these estuarine systems provide critical nursery habitats for juvenile fish, support diverse bird communities, sustain a wide range of plant and animal species, and contribute to carbon sequestration.

Biodiversity designations

The Plettenberg Bay area is a globally significant conservation site, supporting over 300 bird species and designated as an Important Bird and Biodiversity Area (IBA), one of 112 in South Africa (BirdLife South Africa, 2014; Kingwell *et al*., 2020). It is also recognised as a Mission Blue Hope Spot, one of only 165 worldwide (Mission Blue, 2025), an international initiative aimed at conserving oceans and their biodiversity.

The bay forms part of the broader Southern Coastal and Shelf Waters Important Marine Mammal Area (IMMA), identified by the IUCN Marine Mammal Protected Areas Task Force for its importance to multiple whale and dolphin species. Plettenberg Bay is internationally recognised as a Whale Heritage Site, one of only ten worldwide (World Cetacean Alliance, 2019) providing critical habitat for the vulnerable and resident Bryde's whales (*Balaenoptera edeni*; Penry *et al*., 2016) and the Indian Ocean humpback dolphin (*Sousa plumbea*; Vargas-Fonseca *et al*., 2025). In addition, the Bay hosts an annual Ocean Festival, which features a Marine Science Symposium that attracts local and regional scientists to share the latest findings in marine science and conservation.

Social context in Plettenberg Bay

The Bay is also home to a diverse population shaped by its historical roots and migration patterns. The main ethnic groups include the Khoikhoi and San (Bushmen), the "First People" of the Bay, with a long history in the region. The Xhosa arrived in the 16th century, blending agricultural and fishing practices with the Khoikhoi. Afrikaans-speaking South Africans, descendants of later European settlement linked to Dutch colonial administration from the late 18th century onwards, form a significant portion of the population. Later settlement and migration have included Indian communities and migrants from neighbouring countries such as Zimbabwe and Malawi, alongside internal migration within South Africa, contributing to the region's rich and evolving cultural fabric.

The modern-day Plettenberg Bay functions as a dynamic socio-ecological system shaped by rapid population growth and changing patterns of settlement and land use. Over the past two decades, Plettenberg Bay's population has tripled, with a 5.54% annual growth rate driven by remote work, tourism and lifestyle migration (World Population Review, 2024). The most recent growth was driven by the COVID-19 pandemic. People from major urban centres such as Pretoria, Johannesburg, Cape Town and Gqeberha have flocked to Plettenberg Bay, diversifying the population and increasing pressure on natural resources. As a result, urbanisation has become the greatest threat to the Bay's ecological integrity, with rapid population growth surpassing regional and provincial averages. The town accounts for 23.4% of the regional population, with significant increases seen in urban areas such as Plettenberg Bay and Kurland, while rural areas have declined (Western Cape Government, 2023).

Despite its reputation as a tourist hub, Plettenberg Bay's economy is not solely driven by tourism. Key contributors to the region's GDP include finance (31.2%), trade (18.0%), and government (11.0%) (Western Cape Government, 2023). The finance sector, encompassing activities such as accounting, property leasing and insurance, plays a crucial role, with the Bitou Municipality serving as a major employer.

The region is also marked by a complex socio-political landscape, shaped by its apartheid legacy. Historical spatial planning produced enduring inequalities in access to land, infrastructure and marine livelihoods, influencing patterns of resource dependence. Underlying tensions exist between commercial operators and small-scale fishers over access rights, between conservation authorities and resource users over regulatory enforcement, and between coastal development interests and traditional access routes. Ongoing migration from the Eastern Cape increases demand for housing, services and employment,

intensifying pressure on local infrastructure and natural resources. Nature's Valley, as an eco-tourism hub and endpoint of the Otter Trail, and Covie, a small settlement situated within the Tsitsikamma National Park, further illustrate the intersection of conservation, tourism and local livelihoods. These dynamics are ecologically relevant, as unequal access and limited participation in governance processes may influence compliance, informal harvesting practices and the effectiveness of co-management initiatives. Recognising these structural and relational dynamics is therefore essential for interpreting drivers and pressures within the DPSIR framework.

Methodological approach

Our methodological approach consisted of three key components designed to assess the complex interactions between human activities and environmental and social changes in the Bay. These components included (a) identifying activities in the Bay and categorising user groups, (b) applying the DPSIR framework to five purposively selected case studies representing key socio-ecological interaction zones across extractive, non-extractive, marine and estuarine systems, and (c) synthesising key findings. The case studies were selected based on ecological significance, intensity of human use, management relevance and data availability, enabling comparative analysis while maintaining analytical depth.

The first component involved classifying the diverse activities in the Bay into distinct user groups to address the complexities of the identified overlapping activities and competing interests. This categorisation enabled a more precise analysis of the specific pressures exerted by each group and their impacts on environmental, social and economic conditions.

The second methodological component employed an expanded DPSIR framework (**Table 1**) to assess the four user groups of the Bay and the estuaries. This framework, adapted from Allen (2022), incorporates three categories of response strategies: adaptation, restoration and mitigation. Each of these categories addresses specific system dynamics (**Figure 4**).

Table 1: Adapted DPSIR framework (Allen, 2022).

Framework component	Description
Drivers	Underlying socio-economic forces driving change. Analysed to identify the underlying anthropogenic forces influencing environmental and social change. These included urbanisation, tourism and fisheries demand, which shape production, consumption and human activity patterns in the Bay.
Pressures	Encompassed the stressors imposed by human activities on the environment and communities. The distinction between manageable (endogenic) and unmanageable (exogenic) pressures will help determining appropriate intervention strategies.
State	Assessments examined the current ecological and socio-economic conditions of the Bay, including biodiversity health, habitat integrity and economic dependence on marine resources. These assessments highlighted vulnerabilities in estuarine ecosystems and the fragility of fishing-dependent communities.
Impacts	Focused on the consequences of environmental changes for both ecosystems and human livelihoods. The loss of marine biodiversity, declining fish stocks and tourism pressures were identified as primary concerns.

Responses	Policies, tools and economic instruments to address environmental and socio-economic issues were categorised into adaptation (e.g., rotational fisheries closures), restoration (e.g., estuarine habitat recovery) and mitigation (e.g., stricter pollution controls and enforcement of fishing regulations). The integration of ecological, socio-economic and governance facilitated a holistic management approach.
Recovery	The improvement in environmental and socio-economic conditions following responses.

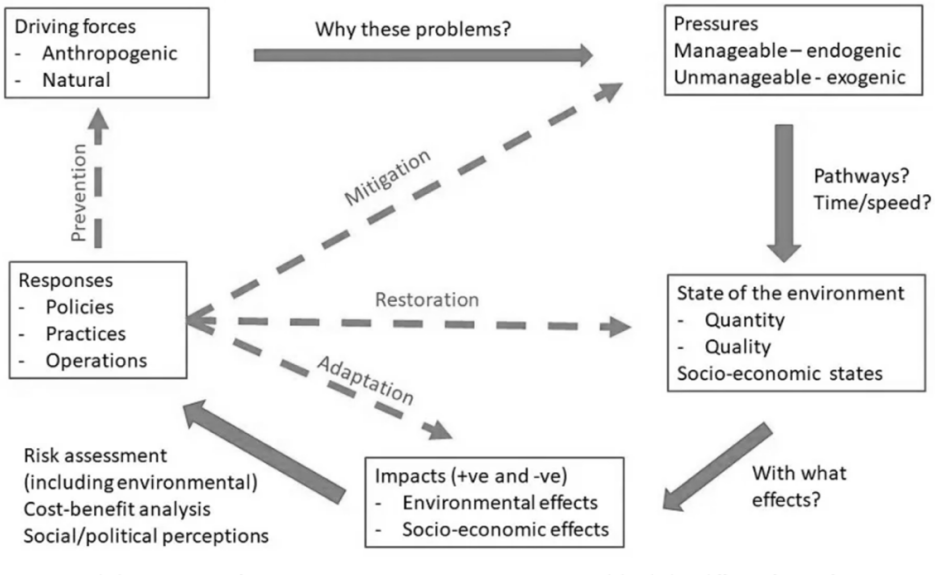


Figure 4: An extended DPSIR framework showing the different forms of "Responses" (Based on Allen, 2022).

The final methodological component synthesised the findings into two overarching summaries: one overarching framework for the Bay and one for the estuaries. These summaries provided a comprehensive overview of human-environment interactions in the Bay and the estuaries, outlining key challenges and potential management interventions. The first synthesis focused on ecological and socio-economic linkages, emphasising the feedback loops between environmental degradation and community vulnerabilities. The second synthesis highlighted priority areas for conservation and governance, recommending stakeholder engagement, sustainable resource management and policy enforcement improvements.

Data collection and analysis

The data collection methods included a mixed-methods approach to assess human-environment interactions, integrating direct observation, document analysis, focus group discussions (FGDs) and semi-structured interviews. Structured observations were conducted at selected beaches and estuaries to record activity types, user intensity and visible environmental impacts, documented through systematic field notes to identify recurring interaction zones and pressure points.

Secondary data were drawn from municipal reports, estuarine management plans, biodiversity assessments, fisheries documentation and NGO publications. These materials were analysed using structured content analysis aligned with DPSIR categories (drivers,

pressures, state, impacts and responses), enabling triangulation between documentary and field-based findings.

The FGDs were conducted with purposively selected local resource users, including subsistence-oriented fishers, recreational users and community members, using a semi-structured guide organised around DPSIR themes. Participants were selected through purposive sampling to ensure representation of key livelihood groups and stakeholder categories. Focus group discussions and interviews were conducted in accessible community locations and coastal settings familiar to participants. Discussions were audio-recorded with consent and supplemented by detailed field notes. Semi-structured interviews with conservation managers, municipal officials, tourism operators and community representatives explored perceptions of environmental change, regulatory effectiveness and socio-economic dependencies. The study adhered to standard ethical principles for research involving human participants; participation was voluntary, informed consent was obtained, and anonymity was maintained where requested.

Qualitative data were analysed using thematic content analysis. Coding was conducted deductively using DPSIR categories, followed by inductive identification of emergent sub-themes. Cross-case comparison was undertaken across user groups and estuarine systems, and triangulation of observational, documentary and interview data enabled validation of recurring patterns across data sources.

Results and discussion

Activities and user groups identified in Plettenberg Bay

The Bay supports a range of stakeholders engaged in both non-commercial and commercial activities, each contributing to the local economy and community well-being (**Table 2).** These activities, spanning from recreation to economic ventures, reflect diverse interests and interactions. While they provide social and financial benefits, they also impose varying pressures on the Bay's socio-ecological systems, shaping conservation and management challenges.

Table 2 presents the range of activities observed in the Bay, irrespective of whether they are undertaken commercially or non-commercially. Many activities occur across multiple modes of engagement (e.g. self-directed and guided) and are therefore not mutually exclusive.

Users were categorised into four functional groups: fishers and bait collectors (extractive users), recreational water users (non-commercial), land-based nature users, and commercial marine tourism operators. Commercial operators were distinguished from general water-based users due to the structured, economically driven and repeated nature of their operations, which often involve direct interaction with marine species and generate distinct types and intensities of socio-ecological pressures. The categorisation distinguishes user groups based on the nature and intensity of pressures generated, rather than the activities themselves, which may occur across both commercial and non-commercial contexts.

The functional categorisation presented above (**Figure 5**) is applied to interpret how these activities translate into different types and intensities of socio-ecological pressures. The categorisation of user groups and key stakeholders is presented in **Figures 5** and **6**, which illustrate the diversity of actors, activities and affiliations shaping the socio-ecological dynamics of the Bay.

Table 2: Non-commercial and commercial activities in Plettenberg Bay.

Activity	Description	Location
Picnicking, braaing and social gatherings	Recreational use of designated braai and picnic facilities along beaches and estuaries.	Robberg Nature Reserve and Nature's Valley Beach Estuaries: Keurbooms and Piesang
Swimming	Recreational swimming in estuaries or at beaches, including during peak seasonal periods when user intensity increases.	Beaches: Central, Robberg, Lookout, Keurbooms and Nature's Valley Estuaries: Piesang, Keurbooms, Salt River and Groot River
Boating	Popular activity, own boats, rentals or tours.	Keurbooms River and the Bay
Kayaking / canoeing / surf skiing	Popular activity can be used for wildlife viewing and sports. Own equipment or rental services (guided tours).	Beaches: Central, Lookout, Robberg and Nature's Valley. Estuaries: Piesang, Keurbooms and Groot River
Fishing	Subsistence, recreational or commercial fishing from the shore[1], estuaries or along the coast.	Keurbooms River, Salt River Estuary, Robberg Beach and Nature's Valley
Bird watching	Popular bird-watching opportunities.	Nature's Valley, Robberg Nature Reserve, Keurbooms and Salt River
Hiking	Popular activity, scenic coastal and forest hikes.	Robberg Nature Reserve, Keurbooms River, Nature's Valley, Kalander Kloof, Salt River, Groot River and Otter Trail
Body boarding	Annual body boarding competitions are common on Wedge Beach.	Beaches: Wedge, Lookout, Robberg 5 and Nature's Valley
Surfing	Popular activity self-guided, rentals or surf schools provide beginner lessons in safe conditions.	Beaches: Lookout, Robberg 5 and Sanctuary
SUP	Stand-up paddle boarding in calm estuary waters.	Estuaries: Piesang, Keurbooms and Groot River
Windsurfing/ Kitesurfing	Popular in windy conditions.	Bay beaches and Keurbooms estuary.
Whale and dolphin watching	Boat-based tours to view marine wildlife, including dolphins and whales.	Plettenberg Bay, Keurbooms Beach, Nature's Valley and Robberg Beach
Eco-tourism tours	Focused on conservation and biodiversity, especially around marine life and estuaries.	Keurbooms River, Plettenberg Bay and Nature's Valley
Seal swimming	Tour operators offer swimming with Cape Fur seals.	Robberg Nature Reserve

Scuba / snorkelling	Tours offer scuba diving and snorkelling along the coastline.	Robberg Nature Reserve and the Bay
Deep sea and bottom fishing	Fishing tours targeting species like hake and gurnard (*Dichistius capensis*).	Coastal waters in the Bay
Sailing charters	Tours for viewing marine life, such as Cape fur seals.	Coastal waters in the Bay

[1]*Fishing operators in the Bay: Enrico's Fishing Safaris, Garden Route Fishing, Plett Deep Sea Safari, Plett Fishing Charters & Plettenberg Bay Rock and Surf -Angling Adventures.*

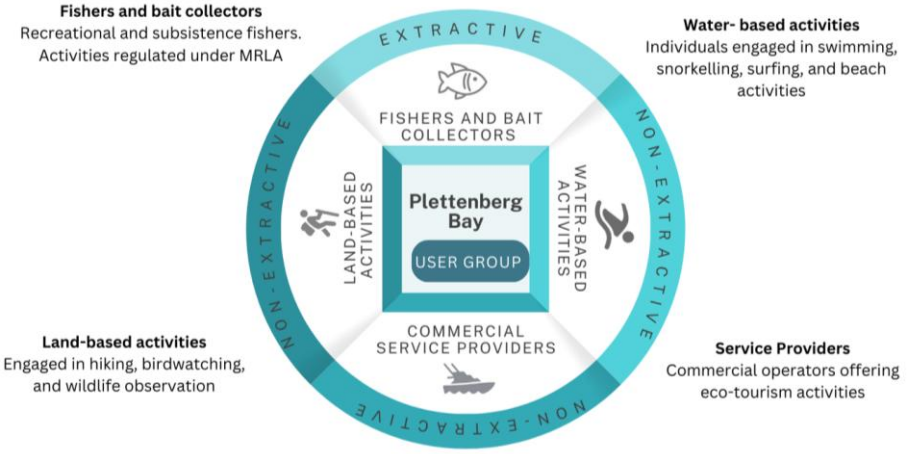


Figure 5: Four categories of user groups of the Bay.

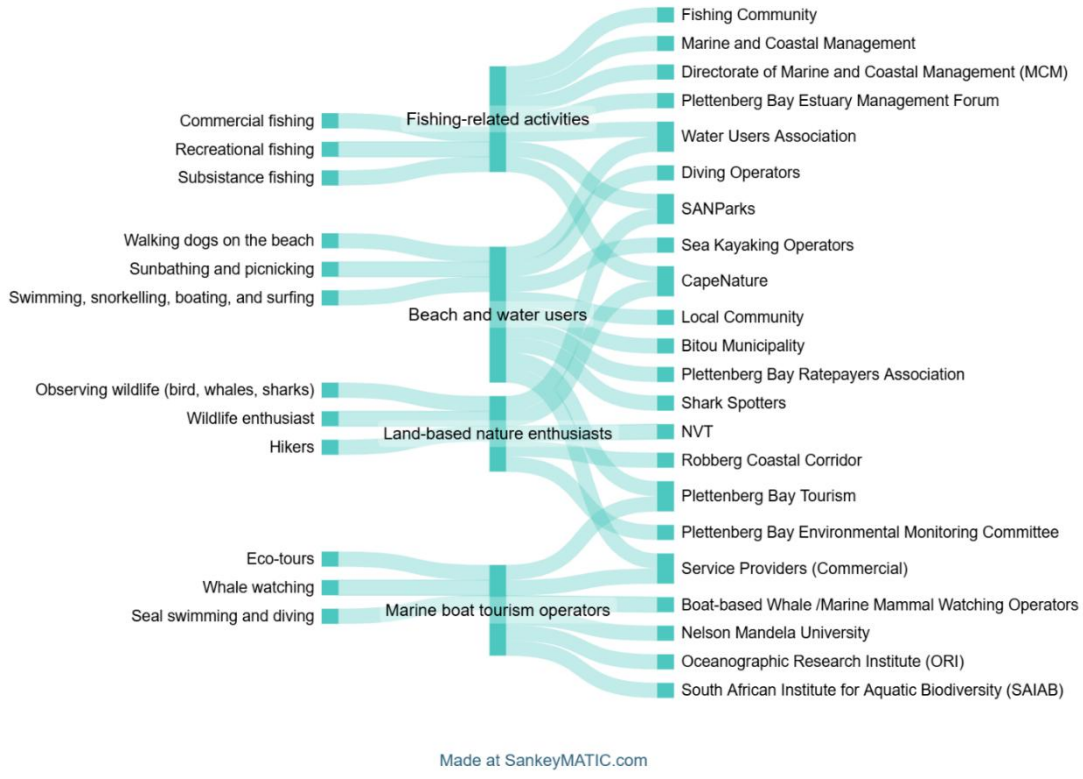


Figure 6: Grouping of users in the Bay and key stakeholders.

The increasing human presence in the Bay brings diverse user groups into closer interaction, heightening the potential for conflicts of interest. These conflicts often stem from differing priorities and perceptions regarding resource use, access and environmental management. In high ecological and recreational value areas like the Bay, the coexistence of extractive and non-extractive activities (alongside a mix of commercial, recreational and conservation-focused interests) creates a complex dynamic that requires careful management.

User groups frequently hold distinct and sometimes competing values and objectives. For instance, some may prioritise resource extraction for livelihood or recreation, while others emphasise the conservation of habitats and species. Similarly, tourism activities that depend on undisturbed natural settings may clash with infrastructure developments or other high-impact activities. These differences can lead to tensions, especially in environments where space and resources are limited and where seasonal peaks in activity intensify competition.

Such conflicts are not inherently problematic. They are a natural outcome of the diversity of human interests in shared environments. Recognising and understanding these potential areas of divergence is crucial for designing effective management strategies that balance ecological, social and economic goals. By anticipating conflicts, stakeholders can implement proactive measures, such as fostering dialogue, promoting shared governance frameworks and integrating diverse perspectives into decision-making processes.

Overarching DPSIR framework for Plettenberg Bay

The literature review findings on past DPSIR elements, even those not explicitly structured within the framework, are presented in **Annex 1**. These insights have informed and shaped the current DPSIR framework, which is outlined in **Table 3**. **Figure 7** provides an overview (summary) of the drivers, pressures, state, impacts and responses for each of the four user groups in the Bay. The analysis identifies a total of 12 drivers, 12 endogenic (manageable) pressures and five exogenic (unmanageable) pressures, along with eight states and nine impacts. While the preceding analysis considered the Bay as a connected marine system shaped by user groups, estuaries represent ecologically distinct but functionally integrated subsystems. Given their nursery function, hydrological sensitivity and land-sea interface dynamics, a focused DPSIR analysis of the four principal estuaries provides deeper insight into localised pressures and management needs. The DPSIR framework for the estuaries is summarised in **Table 4**.



Table 3: DPSIR summary for the four user groups in Plettenberg Bay.

Framework / User groups	Fishers and bait collectors	Water-based activities	Land-based nature activities	Service providers
Drivers	Economic necessity and market demand for marine resources. Cultural traditions of fishing. Commercial profitability of target species.	Growth in domestic and international tourism. Improved infrastructure and promotion of coastal recreation. Post-pandemic interest in outdoor, family friendly activities.	High demand for access to pristine natural areas. Global popularity of destinations like Robberg and Tsitsikamma MPAs. Increased domestic tourism driven by accessible MPAs.	Expansion of marine-based tourism markets. Economic potential of wildlife tours (e.g. whale, dolphin and seal tourism). Marketing of the Bay as a marine tourism destination.
Pressures (Manageable - Endogenic)	Overfishing of key species such as galjoen (*Dichistius capensis*) and dusky kob (*Argyrosomus japonicus*). Habitat damage from ghost gear. Competition for resources among fishers.	Habitat degradation from trampling and recreational activities. Noise and water pollution from motorised water sports. Littering from beach activities.	Trail erosion and habitat fragmentation in MPAs. Disturbance to nesting seabirds and sensitive wildlife. Off trail activities exacerbating habitat degradation.	Noise pollution from frequent boat tours. Damage to seafloor habitats from anchoring. Overcrowding in marine areas during peak seasons (e.g. habitat degradation risks).
Pressures (Unmanageable - Exogenic)	Reduce of fish stocks from climate driven shifts in species distribution.	Coastal erosion and habitat degradation from climate induced sea level rise and storm surges.	Habitat stress from prolonged droughts and vegetation loss due to climate impacts.	Rising ocean temperatures and prey shifts affect marine biodiversity.

State	Declining fish stocks and reduced reproductive success in key species. Ecosystem changes driven by overfishing	Degraded coastal and intertidal ecosystems. Reduced shorebird populations from disturbances.	Trail degradation and fragmented habitats in MPAs. Localised damage to sensitive habitats like nesting sites.	Marine biodiversity stress from frequent human interactions. Risks to sustainability of marine tourism without stricter management.
Impacts	Biodiversity loss and declining fish stock resilience. Food insecurity for subsistence fishers. Social conflicts over resource allocation.	Loss of ecosystem resilience in dunes and intertidal zones. Reputation as a pristine destination at risk, reducing tourism revenue.	Degraded visitor experiences due to overcrowding and environmental damage. Reduced ecological integrity of MPAs.	Behavioural disruptions to marine species. Longterm risks of overcommercialisation and declining ecosystem quality.
Responses (Mitigation)	Sustainable fishing education programmes. Ghost gear retrieval and biodegradable gear trials. Inclusive governance for marginalised groups. Monitoring socioeconomic metrics to guide adaptive management and understand resource dependencies.	Enhanced litter management and regular cleanups. Seasonal restrictions during critical wildlife activity periods. Non-motorised and dog free zones in critical habitats.	Visitor quotas during peak seasons. Smart trail design to avoid sensitive habitats. Promoting conservation awareness through nature-based tourism.	Stricter ecotourism guidelines, including limits on the number of boat tours and maintaining strict guidelines for wildlife interactions. Collaborative initiatives with local communities. Monitoring socioeconomic indicators like employment and income.

	Establish systems to monitor key indicators such as employment, fishing activities and income from fisheries.			
Responses (Adaptation)	Best practices for catch and release fishing. Economic alternatives for coastal communities (e.g., ecotourism).	Education and outreach on ecological importance of coasts. Citizen science and biodiversity monitoring.	Conservation education through guided tours and apps.	Collaboration with local communities to enhance ecotourism benefits.
Responses (Restoration)	Habitat restoration to counteract damage from ghost gear. Community based conservation to restore fish stocks.	Restoration of sensitive marine habitats, including estuaries, impacted by boats and anchoring. Monitoring high-use areas to better understand user impacts and inform management. Monitoring beach microfauna to guide strategies for biodiversity and ecosystem resilience.	Trail maintenance and rotational closures in sensitive areas. Incentives for sustainable tourism practices.	Licensing credits for marine operators supporting conservation. Data collection to support research that enhances marine conservation in the Bay.

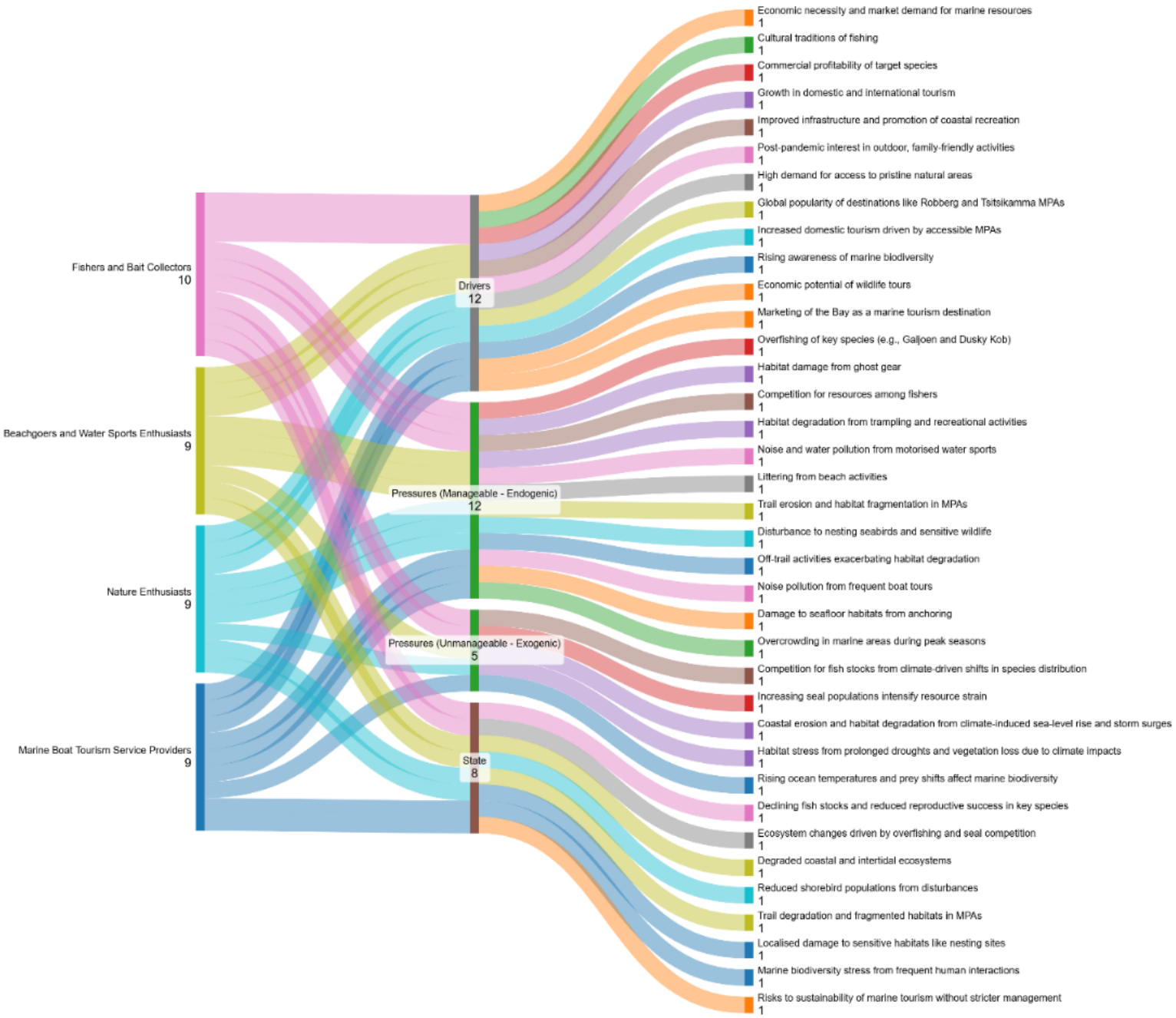


Figure 7: Combined summary of DPSIR for Plettenberg Bay (Made at SankeyMATIC.com).

Table 4: Summary of DPSIR framework for the estuaries in Plettenberg Bay.

Framework	Piesang River	Keurbooms River	Salt River	Groot River
Drivers	Economic growth (population, urbanisation, tourism increase) and land use changes.			
Pressures	Infrastructure development, altered hydrology, freshwater abstraction,	Infrastructure development, boating activities, sedimentation from agriculture and urban runoff, pollution,	Agricultural runoff and sedimentation.	Agricultural and urban runoff and

	sedimentation, pollution and recreational activities.	dogs' wildlife disturbance and trampling eelgrass meadows.	Limited tourism pressures.	sedimentation. Tourism trampling and dogs' wildlife disturbance.
State	Moderately degraded, affected by urbanisation, salinity changes, sediment shifts, habitat loss and eutrophication.	Relatively good condition, with fluctuating water quality and intact eelgrass meadows.	Pristine condition, limited direct development, with high water quality, minimal sedimentation and intact biodiversity.	Good ecological health, with limited development impact and high biodiversity.
State (threat status)	Vulnerable	Vulnerable	Vulnerable	Least concern
Impact	Loss of biodiversity due to pollution, habitat changes and fishing. Seagrass bed deterioration, affecting ecosystem structure. Degraded water quality and altered salinity, leading to ecological shifts and eutrophication.	Loss of biodiversity due to pollution, habitat changes and fishing. Eelgrass bed deterioration, affecting ecosystem structure.	Low impact (minimal upstream urban development and industry).	Limited
Responses	Regulation and policy measures: Stricter fishing regulations, zoning regulations and stricter pollution control. Conservation and restoration efforts: Habitat restoration and maintaining biodiversity. Monitoring and research: Continuous monitoring (water quality, sedimentation, runoff impacts) and research on sedimentation and pollution impacts, monitoring agricultural runoff, etc. Community engagement and education: Community-led initiatives, public awareness campaigns and citizen science engagements.			

Table 5: Conflicts of interest in Plettenberg Bay.

Category	Conflict of interest	Details
Fishermen and livelihoods	Access to fishing spots.	Development projects denying access to local subsistence fishing areas, impacting fishermen's livelihoods.
Fishermen and conservation	Tensions between conservation efforts and fishing practices.	Conservation measures restricting fishing areas, leading to conflicts with local subsistence fishers. Examples include conflicts arise as pyjama sharks (*Poroderma africanum*) consume small fish, crustaceans, and cephalopods (organisms that fishermen rely on for bait) creating tensions that lead to sharks being killed by the fishers.
Fishermen and resource management	Lack of opportunities for self-monitoring of resources.	Limited opportunities for local subsistence fishers to engage in resource monitoring, leading to reduced participation in sustainable management.
Fishermen and regulations	Regulatory restrictions.	Stringent regulations and zoning affecting subsistence fishing practices and access to resources.
Regulatory compliance and enforcement	Poor enforcement of fishing regulations.	Inadequate monitoring and enforcement leading to illegal fishing practices and further competition for resources.
Development and recreation	Competition between recreational activities and fishing.	Increased recreational use of coastal areas causing conflicts with local subsistence fishing activities.
Development and environment	Coastal development projects.	Construction and land use changes affecting marine ecosystems and access to local subsistence fishing areas.
Development and community	Displacement of local communities.	New developments leading to displacement of communities who rely on subsistence fishing for their livelihoods.
Commercial interests and local communities	Commercial activities impacting subsistence fishing rights.	Large-scale commercial operations dominate market access affecting the ability of local fishermen to develop their own commercial operations and limiting opportunities for them to compete in formal markets.

Conflicts of interest

Recognising the increasing diversity of users and their competing interests is essential for developing informed, equitable and sustainable management strategies. Conflicts of interest (**Table 5**) among user groups further complicate the social landscape. The diverse stakeholder community includes subsistence fishers, tourism operators, commercial entities and conservation authorities - all with competing priorities. These conflicts reveal deeper structural issues, such as imbalanced power dynamics, limited resource access for marginalised groups and tensions between economic

growth and environmental stewardship. Such complexities highlight the inherent difficulty of managing shared ecosystems where diverse user groups with differing needs and expectations interact.

Within the DPSIR framework, conflicts of interest can be interpreted as manifestations of underlying drivers (economic inequality, livelihood dependence) interacting with regulatory pressures and governance responses. Where responses are perceived as exclusionary or inequitable, compliance declines, reinforcing pressures and potentially degrading system state. Understanding conflict therefore becomes central to adaptive response design. Recognising the interconnectedness of these conflicts is essential to developing governance approaches that address root causes rather than just symptoms (Pomeroy & Douvere, 2008). Rather than discrete actions, effective responses must be part of an integrated strategy that acknowledges the cumulative and dynamic nature of pressures within and beyond the Bay. For example, Marine Spatial Planning (MSP) and zoning frameworks can help mediate resource competition by balancing ecological protection with user needs, but their success also depends on participatory processes that also include historically marginalised groups, such as small-scale fishers. The integration of community-based resource management into MSP can strengthen trust and ownership among stakeholders, creating a platform for long-term resilience (Stephenson *et al*., 2019a).

Effective governance of the Bay requires integrating ecological sustainability with socio-economic realities. Structural inequities that limit access to resources and decision-making must be addressed to improve regulatory compliance. Educational initiatives and capacity building align stakeholder interests with sustainability goals while reducing dependency on vulnerable resources. Conflict resolution should extend beyond immediate disputes to address broader challenges such as climate change and external fishing pressures, which shape the Bay's socio-ecological dynamics.

Key responses to improve outcomes in the Bay

The DPSIR analysis highlights that responses in the Bay cannot be addressed through isolated interventions, but require integrated, system-level approaches that link drivers, pressures, ecological state and socio-economic outcomes. The response pathways identified across estuarine and coastal systems converge around three core domains: regulation and enforcement, ecological restoration, and monitoring and knowledge generation, all of which must be underpinned by meaningful stakeholder engagement. In particular, cumulative pressures in high-use and ecologically sensitive areas, such as estuaries and heavily utilised coastal zones, require targeted, place-based responses that directly address underlying drivers of change, including both ecological degradation and socio-economic dependence on coastal resources.

Regulatory responses remain essential, particularly in addressing pressures associated with resource extraction, habitat disturbance and pollution. Measures such as spatial zoning, stricter fishing regulations, visitor management, and controls on tourism and boating activities are necessary to reduce direct ecological pressures. This includes limits on wildlife interactions, controls on access in sensitive habitats, and the management of recreational activities in high-use zones. Targeted measures such as seasonal restrictions, non-motorised access in sensitive areas, and limits on visitor intensity where ecological thresholds are exceeded are particularly important. However,

regulatory approaches alone are insufficient in a context of diverse user groups and socio-economic inequalities. Conflicts between fishers, tourism operators and conservation authorities indicate that perceptions of fairness, inclusion and legitimacy strongly influence compliance. In contexts where access to resources underpins livelihoods, enforcement without participation may exacerbate conflict and undermine long-term sustainability, limiting the ability of regulation to address root drivers.

Ecological restoration emerges as a critical pathway in systems already experiencing degradation, particularly within estuaries affected by sedimentation, altered hydrology and habitat loss. The restoration of eelgrass beds, rehabilitation of riparian zones, and protection of sensitive habitats are essential for maintaining ecosystem structure and function. In addition, targeted restoration linked to specific pressures, such as damage from lost fishing gear, marine litter accumulation, or intensive recreational use, supported by targeted clean-up and recovery initiatives, alongside improved management of high-use areas, can support recovery processes. Importantly, restoration is not only an ecological intervention but also a socio-economic one, as the recovery of ecosystem services directly supports fisheries, tourism and local well-being, especially in areas where cumulative impacts have altered ecosystem dynamics.

Monitoring and research form the backbone of adaptive management. Continuous tracking of ecological indicators such as water quality, sedimentation and biodiversity, alongside socio-economic indicators including employment, income and resource dependence, enables a more responsive and evidence-based governance approach. Expanding monitoring systems to include user behaviour, fishing practices, tourism intensity and ecosystem responses, as well as tracking the effectiveness of management interventions and incentive-based approaches, allows for more targeted interventions and adaptive learning. The integration of scientific monitoring with local ecological knowledge, including participatory approaches such as citizen science, enhances both the relevance and accuracy of data, while fostering stakeholder ownership and enabling management responses to evolve over time. These findings are consistent with broader work highlighting the need to integrate ecological condition with social and governance dimensions in coastal systems, including conservation planning gaps (Harris *et al*., 2014), ecosystem service frameworks (Harris & Defeo, 2022), and the role of governance and social outcomes in resource management (Mascia *et al*., 2010; Isaacs, 2013, Stephenson *et al*., 2019b).

Across these response domains, fragmented governance remains a key constraint. The diversity of user groups, overlapping institutional mandates, and socio-economic inequalities create conditions in which single-sector interventions are insufficient. In this context, co-management emerges as a key pathway to improve outcomes in the Bay.

Co-management provides a framework through which state authorities, local communities and resource users collaboratively participate in decision-making, implementation and monitoring. By actively engaging stakeholders, particularly small-scale fishers and local communities, in planning, monitoring and enforcement, while recognising secure and equitable access to marine and coastal resources and safeguarding these groups from exclusion in decision-making and resource use, co-management bridges the gap between policy and practice and strengthens stewardship of marine resources. It also enables the integration of local ecological knowledge with formal scientific systems, enhancing adaptive capacity in

the face of environmental and socio-economic change.

Beyond institutional coordination, co-management addresses the social conditions that underpin effective governance. By improving perceptions of fairness, inclusion and legitimacy, it strengthens compliance and reduces conflict between user groups. This is particularly important in the Bay, where livelihood dependence on marine and coastal resources intersects with conservation objectives. Through more equitable distribution of costs and benefits, and through mechanisms such as incentive-based schemes, stewardship-linked benefits, and collaborative initiatives, co-management supports more inclusive and durable management outcomes.

Ultimately, improving outcomes in the Bay requires a shift from fragmented, reactive management towards collaborative and adaptive governance that explicitly recognises cumulative pressures and addresses their underlying drivers. Co-management provides the mechanism through which regulatory, restoration and monitoring responses can be integrated, strengthening shared responsibility, reducing conflict, and aligning conservation objectives with livelihood needs.

Conclusion

This study demonstrates that Plettenberg Bay is shaped by a tightly coupled set of socio-economic drivers and ecological responses, where tourism growth, livelihood dependence on marine resources, and rapid urbanisation collectively generate concentrated pressures on coastal and estuarine systems. Across user groups, the most consistent pressures identified include overexploitation of key species, habitat disturbance from recreational and tourism activities, pollution and runoff affecting estuarine water quality, and noise and physical impacts from boat-based operations. These pressures are most evident in high-use areas and urbanised estuaries such as the Piesang and Keurbooms systems, where ecological condition is already degraded or vulnerable.

These pressures translate into measurable ecological and socio-economic impacts. Declining fish stocks, habitat degradation and reduced water quality undermine ecosystem functioning and resilience, while also affecting livelihoods, food security and the sustainability of tourism. At the same time, increasing competition for space and resources intensifies conflicts among user groups, particularly where access, regulation and benefits are unevenly distributed. Importantly, relatively intact systems such as the Salt and Groot estuaries highlight the value of lower-impact conditions, but also their vulnerability to emerging pressures.

The findings point to a set of priority, actionable responses. First, spatially targeted management is needed in high-impact zones, including stricter regulation of boating, anchoring and recreational use, visitor management measures and limits in ecologically sensitive areas, alongside enforcement of fishing controls. Second, restoration efforts should focus on sensitive habitats, particularly estuarine and nearshore systems already showing signs of degradation, including areas affected by marine litter and fishing-related debris. Third, sustained ecological and socio-economic monitoring is essential to track changes, improve understanding of user impacts, and support adaptive management, including participatory and citizen-based monitoring approaches. Finally, strengthening participatory and inclusive governance, particularly through equitable participation of small-scale fishers and local communities in decision-making, is critical to improving compliance, reducing conflict and aligning conservation with livelihood needs.

This should be supported by approaches that ensure fair access to marine and coastal resources, alongside education, awareness and incentive-based mechanisms that promote stewardship behaviours.

Overall, the study highlights that pressures in Plettenberg Bay are cumulative, spatially uneven and closely linked to human use patterns. Addressing them requires integrated, place-based management that combines regulation, restoration and co-management. By linking specific drivers, pressures, impacts and responses, the DPSIR framework provides a practical foundation for prioritising interventions and supporting more adaptive, equitable and effective coastal governance in the Bay.

References

Allen, W. 2022. Using a DPSIR framework to support natural resource management and policy. Learning for Sustainability. Retrieved from https://learningforsustainability.net

Atkins, J.P., Burdon, D., Elliott, M. & Gregory, A.J. 2011. Management of the marine environment: Integrating ecosystem services and societal benefits with the DPSIR framework in a systems approach. Marine Pollution Bulletin, 62(2), 215-226.

Berkes, F. 2009. Evolution of co-management: Role of knowledge generation, bridging organizations and social learning. Journal of Environmental Management, 90(5), 1692-1702.

Berkes, F. & Folke, C. 1998. Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience. Cambridge: Cambridge University Press.

BirdLife South Africa 2014. Tsitsikamma-Plettenberg Bay Important Bird and Biodiversity Area. Important Bird and Biodiversity Areas.

Bornman, T.G. & Adams, J.B. 2005. Keurbooms Reserve Determination Study: Scoping Phase Estuaries. Institute for Environmental and Coastal Management, Nelson Mandela Metropolitan University. IECM Research Report No. 44.

CapeNature 2023. Keurbooms Estuary Management Plan. Cape Town: Department of Environmental Affairs and Development Planning.

CapeNature and Bitou Municipality. 2011. Keurbooms Estuary Draft Estuarine Management Plan.

Chadwick, P., Lombard, A.T., Attwood, C.G., Harris, J.M. & Mann, B.Q. 2014. Assessing the effectiveness of a marine protected area network in South Africa. African Journal of Marine Science, 36(1), 45-56.

Cinner, J.E., McClanahan, T.R., MacNeil, M.A., Graham, N.A.J., Daw, T.M., Mukminin, A., Feary, D.A., Rabearisoa, A.L., Wamukota, A., Jiddawi, N., Campbell, S.J., Baird, A.H., Januchowski-Hartley, F.A., Hamed, S., Lahari, R., Morove, T. & Kuange, J. 2012. Co-management of coral reef social ecological systems.Proceedings of the National Academy of Sciences, 109(14), 5219-5222.

Crutzen, P.J. & Stoermer, E.F. 2000. The "Anthropocene". Global Change Newsletter, 41, 17-18.

De Moor, F.C. & Bellingan, T.A. 2010. Evaluation of the conservation requirements of Trichoptera from the Tsitsikamma mountain streams in South Africa. Water SA, 36(1), 1–8.

Department of Environmental Affairs 2012. South African Water Quality Guidelines for Coastal Marine Waters. Volume 2: Guideline for Recreational Use. Pretoria: Department of Environmental Affairs.

Department of Environmental Affairs and Tourism 2003. National Environmental Management: Biodiversity Act 10 of 2004. Pretoria: DEAT.

Duarte, C.M., Agusti, S., Barbier, E., Britten, G.L., Castilla, J.C., Gattuso, J.P., Fulweiler, R.W., Hughes, T.P., Knowlton, N., Lovelock, C.E., Lotze, H.K., Predragovic, M., Silliman, B.R., Thrush, S.F. & Worm, B. 2020. Rebuilding marine life. Nature, 580, 39-51.

Elliott, M., Burdon, D., Atkins, J.P., Borja, A., Cormier, R., de Jonge, V.N. & Turner, R.K. 2017. A unifying framework for marine environmental management. Marine Pollution Bulletin, 118(1–2), 27–40.

Elwen, S.H., Findlay, K.P., Kiszka, J. & Weir, C.R. 2011. Cetacean research in the southern African subregion: a review of previous studies and current knowledge. African Journal of Marine Science, 33(3), 469-493.

Harris, L. R. 2025. Biodiversity survey of sandy beach invertebrates between Plettenberg Bay and Nature's Valley: Changes since the 1980s. Nature's Letters, 2, 64-75.

Harris, L. R. & Defeo, O. 2022. Sandy shore ecosystem services, ecological infrastructure, and bundles: New insights and perspectives. Ecosystem Services, 57, 101477.

Harris, L., Campbell, E.E., Nel, R. & Schoeman, D. 2014. Rich diversity, strong endemism, but poor protection:

addressing the neglect of sandy beach ecosystems in coastal conservation planning. Diversity and Distributions, 20(10), 1120–1135.

Harris, L.R., Sink, K.J., Skowno, A.L. & Van Niekerk, L. (eds.) 2019. South African National Biodiversity Assessment 2018, Volume 5: Coast. South African National Biodiversity Institute.

Holling, C.S. 1990. The resilience of terrestrial ecosystems: Local surprise and global change, in Clark, W.C. and Munn, R.E. (eds.) Sustainable Development of the Biosphere. Cambridge: Cambridge University Press, pp. 292-317.

Isaacs, M. 2013. Small-scale fisheries governance and understanding the snoek (Thyrsites atun) supply chain in the Ocean View fishing community, Western Cape, South Africa. Ecology and Society, 18(4), Art. 17.

Kingwell, R. 2009. Economic trends and drivers affecting the South African fishing industry. South African Journal of Marine Science, 31(3), 345-358.

Kingwell, J., Crookes, D., Hauvette, C., Brown, M., Mander, M., Lombard, A.T. & Penry, G.S. 2020. An economic valuation of marine boat-based tourism in Plettenberg Bay, South Africa. Technical Report. WWF South Africa, Sustainable Marine Tourism Programme.

Martins, F., Falcão, M., Santos, M.N., Ferreira, J.G. & Cabral, H.N. 2012. Coastal marine biodiversity threats in the Cape Floristic Region. Ocean & Coastal Management, 69, 58-68.

Mascia, M.B., Claus, C.A. & Naidoo, R. 2010. Impacts of marine protected areas on fishing communities', Conservation Biology, 24(5), 1424–1429.

McGwynne, L., & Adams, J.B. 2004. Protocols contributing to the management of estuaries in South Africa, with a particular emphasis on the Eastern Cape province. Volume II, Report E: A monitoring protocol for South African estuaries. Water Research Commission Report No. TT 237/04. Pretoria: Water Research Commission.

Mission Blue. 2025. Hope Spots. Available at: https://www.mission-blue.org/hope-spots/ (Accessed 1 March 2025).

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. & Kent, J. 2000. Biodiversity hotspots for conservation priorities. Nature, 403, 853–858.

Mucina, L. & Rutherford, M.C. (eds.) 2006. The Vegetation of South Africa, Lesotho and Swaziland. Strelitzia 19. South African National Biodiversity Institute, Pretoria.

Obubu, J.P., Odong, R., Alamerew, T., Fetahi, T. & Mengistou, S. 2022. Application of DPSIR model to identify the drivers and impacts of land use and land cover changes and climate change on land, water, and livelihoods in the L. Kyoga basin: implications for sustainable management. Environmental Systems Research, 11(1), 1–21.

Ostrom, E. 1990. Governing the Commons: The Evolution of Institutions for Collective Action. Cambridge: Cambridge University Press.

Pauly, D. & Zeller, D. 2016. Catch reconstructions reveal that global marine fisheries catches are higher than reported and declining. Nature Communications, 7, 10244.

Penry G, Findlay K, Best P. 2016. A conservation assessment of Balaenoptera edeni. In Child MF, Roxburgh L, Do Linh San E, Raimondo D, Davies-Mostert HT, editors. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

Pfaff, M.C., Logston, R.C., Raemaekers, S.J.P.N., Hermes, J.C., Blamey, L.K., Cawthra, H.C., Cockcroft, A.C., Daniels, S.R., Griffiths, C.L., Götz, A., Paterson, A.W., Penry, G.S., Ras, C., Strydom, N.A., Turpie, J.K., Winker, H. & Sink, K.J. 2019. A synthesis of three decades of socio-ecological change in False Bay, South Africa: setting the scene for multidisciplinary research and management. Elementa: Science of the Anthropocene, 7(32).

Pomeroy, R. & Douvere, F. 2008. The engagement of stakeholders in the marine spatial planning process. MarinebPolicy, 32(5), 816–822.

SANBI 2019. National Biodiversity Assessment 2018: The status of South Africa's ecosystems and biodiversity. Synthesis Report. South African National Biodiversity Institute, Pretoria.

Sink, K.J., Holness, S., Harris, L., Majiedt, P.A., Atkinson, L.J., Kirkman, S.P. & Karenyi, N. 2012. National Biodiversity Assessment 2011: Technical Report. Volume 4: Marine and Coastal Component. South African National Biodiversity Institute, Pretoria.

Sink, K.J., Van der Bank, M.G., Majiedt, P.A., Harris, L.R., Atkinson, L.J., Kirkman, S.P.,Karenyi, N. (eds). 2019. South African National Biodiversity Assessment 2018 Technical Report Volume 4: Marine Realm. South African National Biodiversity Institute, Pretoria.

Smith, J., Huizinga, P. & Van Niekerk, L. 2019. Mouth Management Plan for the Piesang River Estuary. Department of Environmental Affairs and Development Planning.

Steffen, W., Grinevald, J., Crutzen, P. & McNeill, J. 2011. The

Anthropocene: From Global Change to Planetary Stewardship. Ambio, 40, 739-761.

Stephenson, R.L., Hobday, A.J., Allison, E.H., Armitage, D., Brooks, K., Bundy, A., Cvitanovic, C., Dickey-Collas, M., de Miranda Grilli, N., Gomez, C., Jarre, A., Kaikkonen, L., Kelly, R., Lopez, R., Muhl, E.-K., Pennino, M.G., Tam, J., Teixeira, C., van Putten, I.E. & Wiber, M. 2019b. The Quilt of Sustainable Ocean Governance: Patterns for Practitioners. Frontiers in Marine Science, 6(610).

Stephenson, R.L., Paul, S., Pastoors, M.A., Kraan, M., Holm, P., Wiber, M., Mackinson, S., Dankel, D.J., Brooks, K., and Benson, A. 2019a. Integrating fishers' knowledge research in science and management. ICES Journal of Marine Science, 76(2), 467–476.

Sun, S., Wang, Y., Liu, J., Cai, H., Wu, P. & Geng, Q. 2016. Sustainability assessment of regional water resources under the DPSIR framework. Journal of Hydrology, 532, 140–148.

Tallis, H. & Lubchenco, J. 2014. Working together: A call for inclusive conservation. Nature, 515(7525), 27-28.

Timmerman, J. G., Pahl-Wostl, C. & Moltgen, J. 2011. Developing transboundary river basin monitoring programmes using the DPSIR framework. Journal of Environmental Monitoring, 13(10), 2808–2818.

Van Niekerk, L. & Taljaard, S. 2007. Proposed generic framework for Estuarine Management Plans. C.A.P.E. Estuaries Programme.

Vargas-Fonseca OA, Atkins S, Vermeulen E, Dines S & da Silva JM. 2025. A conservation assessment of Sousa plumbea. In Patel T, Smith C, Roxburgh L, da Silva

JM & Raimondo D, editors. The Red List of Mammals of South Africa, Eswatini and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

Western Cape Government 2023. Bitou Municipality Socio-economic Profile 2023. https://www.westerncape.gov.za/provincial-treasury/sites/provincial treasury.westerncape.gov.za/files/atoms/files/Bitou%20Municipality%20SEPLG%202023.pdf

World Population Review. (2024). Plettenberg Bay, South Africa Population 2024. https://worldpopulationreview.com/cities/south-africa/plettenberg-ba



Annex 1: Literature review on the components of the DPSIR framework assessed in Plettenberg Bay.

Frame-work	Findings	References
Drivers	Population growth: Increasing numbers of residents and post-pandemic migration strain resources, infrastructure and natural ecosystems. More people depend on marine resources for food and income.	Kingwell, 2009; Lamberth *et al*., 2010; BirdLife South Africa, 2014
	Economic activities: Tourism, fishing and business ventures shape social and environmental conditions, affecting conservation efforts, resource sustainability and local livelihoods.	Kingwell, 2009; BirdLife South Africa, 2014
	Employment and livelihoods: Dependence on marine industries like fishing and tourism influences economic stability, food security and resource management decisions.	Lamberth & Pitcher, 2001; DEAT, 2003; Kingwell *et al*., 2020
	Boat tourism: Expansion of boating activities contributes to economic growth but also adds pressure on marine resources and infrastructure.	Lamberth & Pitcher, 2001; DEAT, 2003; Kingwell, 2011
	New developments: Coastal expansion, driven by tourism and population growth, alters land use, degrades ecosystems and creates economic disparities.	DEA, 2014; Kearney, 2020; Kingwell, 2009
	Driving force indicators: Metrics that track changes in social and economic activities affecting the Bay (e.g., changes in the number of commercial, recreational and subsistence fishers; population growth; tourism growth; economic activities. Indicators reflect the socio-economic factors driving environmental pressures, helping assess sustainability and community impacts.	DEA, 2014; Cinner *et al.,* 2012; Kingwell, 2009
Pressures	Fishing activities: Overfishing, including commercial, subsistence and recreational fishing, reduces fish stocks, disrupts biodiversity and alters ecological balance.	Pauly & Zeller, 2016; Cinner *et al*., 2012; Kingwell, 2011; Schroeter, 2015; Pauly & Zeller, 2016
	Seal populations: Perceived as adding pressure to fish stocks, influencing human-wildlife conflict.	Kingwell, 2009
	Pollution: Agricultural runoff and boat tourism contribute to declining water quality, affecting marine ecosystems.	Martins *et al*., 2012; McLusky & Elliott, 2004
	Habitat destruction: Development leads to land-use changes, habitat loss and increased runoff, harming marine nurseries and biodiversity.	Martins *et al*., 2012, DEAT, 2003; Holling, 1990

	Social pressures: Economic inequality and resource competition impact local communities dependent on marine and coastal ecosystems.	Kingwell, 2009; Ostrom 1990; Tallis & Lubchenco, 2014
State	Degradation of marine habitats and declining fish stocks.	Pauly & Zeller, 2016; Chadwick *et al.*, 2014;
	Deteriorating water quality and increased strain on local resources.	DEAT, 2003
	Social inequality affecting community resilience.	Berkes, 2009; Ostrom, 1990; Cinner *et al*., 2012; Berkes, 2009; Tallis & Lubchenco, 2014
Impacts	Biodiversity loss and economic impacts on fisheries.	Crutzen, & Stoermer 2000; Martins *et al*. 2012; Steffen *et al*., 2011
	Social conflicts between conservationists, fishers and user groups.	Atkins *et al*. 2011
	Effects on cultural practices and traditional access to marine resources.	Berkes, 2009; Tallis & Lubchenco, 2014
	Reduced tourism income.	Kingwell, 2009
Responses	Implementing fishing quotas, protected areas and conservation initiatives.	DEAT, 2003; McLusky & Elliott, 2004
	Providing through tools for building community capacity at scale, enabling them to manage resources more effectively.	Berkes & Folke, 1998; Tallis & Lubchenco, 2014
	Supporting collaborative research initiatives that leverage community rights and co-management of marine resources.	Berkes, 2009; Ostrom, 1990; Cinner *et al*. 2012; Tallis & Lubchenco, 2014
	Developing best practice knowledge management systems and participatory monitoring and evaluation frameworks for users to track resource usage and measure impacts.	Atkins *et al*., 2011; BirdLife South Africa, 2014; McLusky & Elliott, 2004
	Promoting marine resource self-management by users to enhance sustainability and resilience.	Duarte *et al.,* 2020; Atkins *et al.,* 2011
	Improving governance, research monitoring and stakeholder collaboration.	Ostrom, 1990; Berkes, 2009; Cinner *et al*., 2012
Recovery	Improved marine habitat health and fish stock recovery.	Kingwell, 2009; Chadwick *et al*., 2014; DEAT, 2003; McLusky & Elliott, 2004

Enhanced water quality and improved stakeholder collaboration to manage these systems.	Berkes & Folke, 1998; Tallis & Lubchenco, 2014
Restoration of cultural practices and local access to resources.	Kingwell, 2009; Berkes, 2009; Ostrom, 1990; Cinner *et al*., 2012



A visitor-based tool for assessing the social and physical carrying capacity for sustainable tourism in protected areas

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How to cite this article:

Soares, C., Owusu, A., Neidert, P., Musekwa, K., Wilson, J., Klaver, K., & Molitor, H. 2026. A visitor-based tool for assessing the social and physical carrying capacity for sustainable tourism in protected areas *Nature's Letters* **3**:136-151. URL: https://nvt.org.za/natures-letters/

Abstract

Growing visitor numbers in protected areas challenge both recreational quality and infrastructure sustainability. Addressing these challenges requires integrated tools to assess social and physical carrying capacities while accounting for the relationship between people and protected environments. This study proposes and validates a novel, replicable visitor-based tool, grounded in a transdisciplinary framework, for assessing social and physical carrying capacity in protected areas. The 16-item tool was developed through literature review, expert and stakeholders' input, pre-tests, and piloted at Robberg Nature Reserve (n = 134) with field audits. Internal validation confirmed a five-factor structure – Parking/Restrooms, Crowding, Trail Safety, Width/Maintenance, and Signage/Seating – with reliability (α = 0.73–0.78) and adequacy (Kaiser–Meyer–Olkin (KMO) = 0.57–0.70). External validity analyses provided preliminary, hypothesis-generating evidence that scores vary with visitor origin and visit frequency, aligning with site assessments and carrying capacity studies. Results show the tool offers a concise, evidence-based dashboard for managers to detect social or infrastructure thresholds, enabling timely interventions across protected areas.

Introduction

Protected area management must consider the intersection of social, economic, and ecological priorities, demanding approaches that address the complex interplay between nature and society (West *et al.,* 2006; Oldekop *et al.,* 2016; Ma *et al.,* 2020). These socio-ecological systems often defy clear institutional boundaries, necessitating transdisciplinary research that integrates multiple disciplines and actively engages stakeholders, including managers, communities, policymakers, and visitors (Lang *et al.*, 2012).

Nature-based tourism exemplifies these complexities. While it can generate economic and conservation benefits, it also creates environmental pressures and social tensions (McCool & Mandic, 2018; Gupta *et al.,* 2023). The core challenge is preserving the very landscapes that attract visitors, balancing ecological limits with social and economic value (McCool & Lime, 2001; Xu & Li, 2025).

Such "wicked problems" (Corlett, 2024) demand integrative strategies that manage trade-offs (Frodeman *et al.*, 2017).

In South Africa, these tensions are heightened by growing tourism in an ecologically diverse network of protected areas. While some sites have adopted carrying capacity approaches (iSimangaliso Wetland Park Authority, 2022), many provincial protected areas lack the resources, institutional capacity, and tools to do so systematically (Patel *et al.,* 2023). This limits their ability to prevent environmental degradation and maintain visitor satisfaction (Spenceley *et al.*, 2017). Against this backdrop, practical and efficient tools for localised data collection offer particular value in under-resourced contexts such as South Africa, where scalable and replicable approaches can strengthen management effectiveness.

Carrying capacity refers to thresholds beyond which negative environmental, experiential, or infrastructural impacts occur. Two dimensions are central: physical capacity, the tangible limits of infrastructure and facilities, and social capacity, the point where crowding or overuse diminishes visitor experience (Williams, 2015; Gonson *et al.*, 2018).

Despite its importance, practical, visitor-based instruments for measuring these capacities remain scarce in South Africa (Leung & Marion, 2000; Wilson & Anthony, 2023). We present a validated, replicable tool for assessing social and physical carrying capacities in protected areas using a transdisciplinary framework, enabling evidence-based management that sustains social and physical integrity.

Methodology

Study area

The tool was piloted at Robberg Nature Reserve, a key ecological, archaeological, and tourist site known for its thriving Cape fur seal (*Arctocephalus pusillus pusillus*) colony (Huisamen *et al.,* 2011), situated within the United Nations Educational, Scientific and Cultural Organization (UNESCO)-proclaimed bio-culturally diverse Cape Floral Region Protected Areas World Heritage Site and Garden Route Biosphere Reserve. As a high-traffic destination, ranked fifth among the Western Cape province's attractions in 2024, with over 77,000 visitors per year (Wesgro, 2024), Robberg faces management challenges tied to both physical and social carrying capacity. Its fragile vegetation, rugged terrain, and infrastructure are vulnerable to overuse; while crowding and noise can reduce visitor satisfaction and conservation outcomes. These conditions, along with collaborative input from CapeNature[1] and OnEarth[2], made Robberg an ideal site to test and validate the visitor-based carrying capacity monitoring tool.

Approach

Transdisciplinarity was foundational to this study, with stakeholders from CapeNature and OnEarth actively participating at every stage. This collaborative approach ensured the research was rooted in real-world management needs and perspectives (Lang *et al.*, 2012).

[1] CapeNature is the statutory conservation body and provincial authority responsible for biodiversity conservation across 112 protected areas across the Western Cape province of South Africa (more information available here: https://www.capenature.co.za/)

[2] OnEarth ESG Consultants are sustainability assessment practitioners currently undertaking an inclusive carrying capacity assessment of the Robberg Nature Reserve (more information available here: https://www.onearth-esg.com/).

Participants

The study tested the tool with a sample of 134 visitors, selected using stratified convenience sampling to ensure variation across time and day, plus a pre-test group of experts. Specifically, data collection was purposively scheduled across predefined weekday/weekend and morning/afternoon strata to reduce day-of-week and time-of-day sampling bias, while recruitment within each stratum remained convenience-based. Thus, the approach improves coverage of heterogeneous visitor segments but does not constitute a probability sample. Participants represented a mix of local and international visitors, first-timers and repeat visitors, and diverse demographic backgrounds.

Procedure

Through a phased approach, the study developed and validated a practical tool to measure social and physical carrying capacities in protected areas. In Phase 1, the tool's concepts and items were created through literature review and consultations with stakeholders and experts. In Phase 2, researchers used mixed methods to test and refine the tool, with ongoing partner engagement. Ethical clearance was obtained through Stellenbosch University's Social, Behavioural and Education Research Ethics Committee (Project ID: 32802), and standards were strictly followed, with voluntary participation, informed consent, and confidentiality protections.

Data Collection

In Phase 1, the initial tool was developed from scientific literature, internal reports, and preliminary field observations. A draft in Google Forms was pre-tested with 18 research experts from social and environmental sciences[3], yielding nine responses for preliminary psychometric analysis. A cognitive pre-test with two researchers assessed item clarity using think-aloud protocols (Willis, 2005), identifying ambiguous wording and misalignments. Additional feedback from academic and project partners was gathered via ongoing correspondence and meetings. Based on these inputs, the tool was revised for clarity and consistency and transferred to KoboToolbox[4] for field use.

In Phase 2, two pilot studies used stratified convenience sampling to capture diverse visitor profiles, with strata defined by weekday/weekend and morning/afternoon time slots (Etikan *et al*., 2015). Data were collected during Pilot 1 on Monday, May 26, 2025 (08:00–13:30) and Tuesday, May 27, 2025 (13:00–17:00), and during Pilot 2 on Saturday, May 31, 2025 (13:00–17:00) and Sunday, June 1, 2025 (08:00–12:00). Visitors (≥18 years) were approached at the trailhead near the parking lot as their visit to Robberg was completed. Surveys, administered digitally by the research team, included 5-point Likert items from strongly disagree to strongly agree, with the middle point serving as a neutral option. The survey captured demographics, for example, origin, age, group size/composition, activities, and participant opinions on facilities, capacity, perceived crowding, and space perception.

[3] Experts came from Rhodes University, Stellenbosch University, University of Venda, Eberswalde University, the University of Saskatchewan, and Vancouver Island University.

[4] KoboToolbox is a free, open-source suite of tools for data collection and management, especially popular in humanitarian, development, and research contexts. It allows users to design digital forms, collect data offline or online (via mobile devices), and analyse results: https://www.kobotoolbox.org/

Pilot 1 yielded 63 responses, while Pilot 2 yielded 71.

Post-pilot psychometric analyses assessed item consistency and response patterns. Additional field observations documented infrastructure conditions, such as trail erosion, signage, facility accessibility, litter, and visitor density, through structured notes and photographs, contributing to validation.

Data Analysis

Tool validation addressed both internal and external validity. Internal validity assessed how accurately and consistently the tool measured visitor perceptions of social and physical carrying capacity, while external validity examined its applicability across different protected area contexts.

Internal validity was established through cognitive pre-testing of all items, followed by psychometric analyses: Cronbach's α for internal consistency (Cronbach, 1951), item–total correlations to identify poorly performing items (Streiner, 2003), and exploratory factor analysis to confirm that items loaded onto the expected constructs (Fabrigar *et al.,* 1999). Observational data on physical conditions were paired with visitor survey responses for the same indicators to measure convergent item validity.

External validity was examined via known-groups and criterion-related analyses (Rosenbaum, 1989). Known-groups tests compared scores across theoretically relevant categories, such as local versus international visitors, for facility perceptions, and first-time versus repeat visitors for both facility perceptions and perceived crowding (Budruk *et al.,* 2008; Chi, 2012; Seebunruang *et al.,* 2022;). Trail/space perceptions were compared by hiking status and group size (Priskin, 2003). Criterion-related validity used Spearman correlations to test associations between survey responses and these categories. Only items passing internal validation were included, and pairwise deletion handled missing data. Given the exploratory, off-season sample, we report exact p-values and interpret results with p-values between 0.05 and 0.10 as suggestive and as hypothesis-generating rather than confirmatory.

Final validation used triangulation at two levels: (i) quantitative cross-tabulation of observational data with corresponding survey responses, for example, crowding, infrastructure condition, and (ii) qualitative integration of partner and researcher feedback to interpret divergences. Convergence across psychometric results, observational audits, and stakeholder insights confirmed the tool's reliability, adaptability, and effectiveness for measuring carrying capacities in diverse protected area contexts.

Results and discussion

The main text presents only the items that passed all triangulation levels. For instance, several season-specific items met one of the criteria yet failed the quantitative thresholds and were omitted from the validated instrument. Additional items were trial-run for each construct, but the off-season pilots produced too few responses for those questions, leaving their distributions too flat for reliable psychometric testing.

Internal validation

Table 1 summarises descriptive and item-total statistics for the 16 retained items (n = 71). Descriptive statistics are reported as means (M) and standard deviations (SD). Because all items were scored on a five-point Likert scale,

scores around three should be read as neutral rather than positive or negative. Overall, responses were favourable. For instance, visitors rated the parking lot highly (M = 4.45, SD = 0.65) and reported little difficulty enjoying natural sounds (M = 4.42, SD = 0.57). Conversely, they largely rejected statements about feeling unsafe because of other people (M = 1.42, SD = 0.49) or trail conditions (M = 1.66, SD = 0.75). Mid-scale means appeared for some facility items. For instance, the availability of restrooms (M = 3.38, SD = 0.72) and accessibility (M = 3.49, SD = 0.69) suggest neutrality.

Table 1: A tool that assesses a protected area's social and physical carrying capacity.

The items (n = 16) that made up the tool applied to visitors (Regarding your visit to the nature reserve today...).					
Domain (Construct)	**Items**	**Number of Answers (Pilot 2)**	**M**	**SD**	**Item-total Correlation**
Facilities Perception	I think the parking area was well-maintained and in good condition (e.g., no potholes, visible parking lines)	71	4.45	0.65	0.51
	Enough restroom facilities were available when you needed them	71	3.38	0.72	0.89
	The restroom facilities were easily accessible.	71	3.49	0.69	0.88
	The restroom facilities are in good condition.	71	3.40	0.78	0.82
Capacity & Perceived Crowding	Other visitors affected your experience negatively.	71	2.34	0.70	0.85

	You felt comfortable sharing the space with people doing other types of activities.	71	4.36	0.63	0.85
	You were able to enjoy the natural sounds of the environment without being disturbed by noise from other visitors.	71	4.42	0.57	0.74
Trail/ Space Perception	The trails are well-maintained.	71	4.32	0.67	0.68
	The trail is wide enough for safe passage, especially when encountering others.	71	4.0	0.71	0.58
	The directional signs were confusing or hard to follow (e.g., signs on the trails).	71	2.02	0.89	0.42
	There are enough signs to locate facilities (e.g., restrooms, shops, drinking water facilities).	71	3.94	0.84	0.49
	You had difficulty finding available	71	1.63	0.70	0.62

	seating at the viewing decks.				
	The trails are not designed for everyone (e.g., the elderly, children and people with special needs).	71	3.70	0.93	0.54
	You felt unsafe because of other people.	71	1.42	0.49	0.61
	You felt unsafe because of the trail conditions.	71	1.66	0.75	0.73
	You felt unsafe because of the wildlife.	71	1.43	0.57	0.68

Item-total correlations all exceeded the 0.30 retention criterion (range = 0.42–0.89), confirming that each item contributes meaningfully to its construct. The strongest discrimination occurred for the restroom access items (r = 0.88-0.89), whereas the directional-signage item was the weakest yet acceptable contributor (r = 0.42).

The triangulation validation process showed the tool's clear, iterative strengthening (Table 2). In the pre-test, several items lacked variability, yielding marginal internal consistency for Capacity & Perceived Crowding (α = 0.62) and Trail/ Space Perception (α = 0.66), and no reliable estimate for Facilities Perception. Although these pre-test α values were below the conventional 0.70 benchmark, they usefully guided targeted item refinement. By Pilot 2, internal consistency exceeded acceptable thresholds for all constructs. Partners and researchers review, therefore, focused on clarifying wording, adding accessibility prompts and standardising items. After these revisions, Pilot 1 produced modest gains for Facilities (α = 0.69) but exposed new weaknesses in the Crowding (α = 0.53) and Trail (α = 0.43) scales, prompting a second round of item pruning, scale re-balancing and positive reframing. By Pilot 2, all three constructs surpassed the acceptable 0.70 threshold (α = 0.78, 0.73 and 0.75, respectively), indicating satisfactory internal coherence. Concurrent observation corroborated the survey signals: facilities were generally sound but occasionally inaccessible, crowding was localised to a few viewpoints, and trail conditions were good overall yet marred by isolated hazards. Together, these convergent lines of evidence confirm that the refined instrument now provides a reliable, context-sensitive measure of visitor perceptions while also highlighting residual

seasonal and sample-size limitations to be addressed in future wider-season rollout.



Table 2: Validation process (Triangulation).

Validation process -Triangulation	Pre-test (n = 9)			Pilot 1 (n = 63)			Pilot 2 (n = 71)		
Constructs measured	**Facilities Perception**	**Capacity & Perceived Crowding**	**Trail/ Space Perception**	**Facilities Perception**	**Capacity & Perceived Crowding**	**Trail/ Space Perception**	**Facilities Perception**	**Capacity & Perceived Crowding**	**Trail/ Space Perception**
Partners and researchers/ Pre-test	Wording and response options have been refined to improve clarity, inclusivity, and accessibility, ensuring all facility items capture visitor experience consistently.	Scale broadened and standardised to gauge overall crowding pressure and visitor interaction effects with a unified, first-person framing.	Construct expanded to separate maintenance from design, and to incorporate safety, wayfinding, and heritage considerations for a more holistic path assessment.	Wording clarified further for facility-related items.	Items were revised to better reflect subjective perceptions of space and comfort.	Wording clarified.	Stakeholders and researchers drafted the manuscript detailing the validation process, recognized seasonal bias and low-response limitations for a few items, and highlighted the need for wider-season validation in the next rollout.		
Cronbach's alpha (α)	0.00	0.62	0.66	0.69	0.53	0.43	0.78	0.73	0.75

Local Observation	-	-	-	-	-	-	Parking often near capacity yet functional; surfaces mostly good with minor defects; bins clean, almost no litter; toilets, café, water, braai generally good but occasional cleanliness/ window issues; wheelchair access mixed; Nelson Bay Cave closed during visit.	≈ 30–50 visitors; park felt quiet overall, with brief crowding and chatter at first viewpoint, Witsand trailhead and the point; no sustained signs of overcrowding.	Trails rated good; sporadic broken boards and tripping hazards; width varied 0.62–1.72 m; signage clear though some faded; viewpoints sound, seating available; minor litter on beach, light erosion and informal trail widening noted.

The analysis diagnostics confirmed that every item pool was factorable with Kaiser-Meyer-Olkin (KMO) values ranging from 0.57 to 0.70 and Bartlett's test was significant ($p < .001$). The exploratory factor analysis produced five clear factors in the same sequence used in **Table 1**: Parking/ Restrooms captures satisfaction with core facilities; Crowding reflects the degree to which other visitors and noise disturb the outing; and Trail/Space Perception factors, including Trail Safety, Width/Maintenance, and Signage/Seating which respectively summarise, feelings of personal safety, the physical adequacy and upkeep of the path, and the clarity of way-finding together with the availability of rest points (**Table 3**). Loadings positive values indicate that stronger agreement with an item signals better conditions, so factor scores rise as conditions improve.

The extracted factors align closely with the theoretical constructs introduced earlier: Crowding and Trail Safety reflect social carrying capacity, as they capture visitor-to-visitor interactions and subjective safety in shared space. Conversely, Parking/Restrooms, Signage/Seating, and Width/Maintenance correspond to physical carrying capacity, as they assess infrastructure, facility design, and trail conditions. This alignment reinforces the theoretical coherence of the tool and supports its practical application for assessing both capacity types. Monitoring these five scores offers a concise, visitor-based dashboard for spotting when either capacity threshold at a protected area is being approached or exceeded.

External validation

Establishing external validity is important when a tool is intended for deployment in multiple protected areas. By demonstrating that scores vary in sensible ways across independent constructs such as visitor origin and visit history, factors that differ naturally from site to site, we show that, firstly, the items capture genuine experiential differences rather than site-specific artifacts, and secondly, the tool's interpretability is preserved when transferred to new protected areas. In short, the evidence below evaluates whether the Robberg Nature Reserve visitors and experiences captured by the tool are not the idiosyncrasies of Robberg visitors themselves, thereby providing initial indications of broader applicability, to be confirmed in larger, multi-site samples.

Table 3: Exploratory factor analysis summary.

Domain (Construct)	Factor label	k items (Pilot 2)	Salient loading range*	KMO	Bartlett's test p-value
Facilities Perception	Parking/ Restrooms	4	0.25 – 0.97	0.57	< 0.001
Capacity & Perceived Crowding	Crowding	3	0.60 – 0.90	0.65	< 0.001
Trail/ Space Perception	Trail Safety	4	0.76 – 1.03	0.70	< 0.001
	Width/ Maintenance	2	0.48 – 0.70		
	Signage/ Seating	3	0.34 – 0.50		

*Pattern-matrix loadings shown after the single factor-level sign reflection; KMO and Bartlett statistics apply to the full nine-item Trail/Space Condition pool, covering all three sub-factors.

We conducted known-groups and criterion analyses to test whether the tool behaves in theoretically predictable ways across visitor segments. As summarised (**Table 4**), Facility Perception and Capacity & Perceived Crowding comparisons showed suggestive differences in the expected direction (p = 0.058-0.092).

For instance, local visitors rated parking more favourably than international visitors (Δ = 0.30, p = 0.058), and first-time visitors provided higher scores than repeat visitors on parking (*Δ* = 0.27, p = 0.092). Consistently, first-time visitors perceived crowding to be lower than repeat visitors (*Δ* = 0.23, p = 0.082). These patterns are consistent with familiarity/novelty expectations and provide preliminary, hypothesis-generating evidence warranting further investigation. For Trail/Space Perception, no meaningful differences emerged, most likely because evaluations were already near the top of the response scale, leaving little variance to explain. However, these near-ceiling ratings may reflect genuinely strong trail conditions during the off-season pilot, but they could also arise from positive response tendencies (e.g., social desirability or acquiescence) or from the restricted range of a 5-point Likert scale when conditions are generally good.

Table 4: Known-groups and criterion-related evidence of external validity.

Higher-order construct	External variable (coding – Pilot 2)	Mean gap (Δ)*	p	Interpretation
Facilities Perception	Visitor origin (Local = 38 vs International = 33)	+0.30	0.058	Locals rate parking slightly better, consistent with greater familiarity with access points (Seebunruang, Burns & Arnberger, 2022).
	Visit status (First-time = 22 vs Repeat = 49)	+0.27	0.092	Novelty effect: first-timers perceive parking most positively (Chi, 2012).
Capacity & Perceived Crowding	Visit status (First-time vs Repeat)	+0.23	0.082	New visitors feel the reserve is less crowded than those who have experienced past peak periods (van der Merwe, 2023).
Trail/Space Perception	Hiking vs. non-hiking parties/ Group size	–	› 0.23	No between-group differences or criterion correlations were observed for this construct in this sample

				(all p > 0.23), possibly owing to ceiling effects (overall ≈ 4.2/5), which may reflect very good trail conditions, positive response tendencies, or a restricted response range.

*Raw mean gap on the 1–5 Likert metric; positive values indicate the first-named group scored higher. Exact p-values are reported; values between 0.05 and 0.10 should be interpreted as suggestive and hypothesis-generating rather than confirmatory.

Taken together, these patterns suggest that aside from Trail/Space Perception, the tool captures underlying visitor-experience constructs rather than site-specific or seasonal idiosyncrasies, suggesting it retains interpretability across protected areas with differing infrastructure and visitor profiles. However, given the pilot nature of the data collection, conducted during the off-season and at a single reserve, these results should be interpreted with caution and as preliminary and hypothesis-generating, particularly where p-values fell between 0.05 and 0.10. Further validation across sites and seasons is warranted.

The present study demonstrates that the refined visitor-based monitoring tool reliably captures three core dimensions – Facilities Perception, Capacity & Perceived Crowding, and Trail/Space Perception – when deployed at Robberg Nature Reserve. Internal consistency exceeded acceptable thresholds for all constructs, and exploratory factor analysis revealed clear, theory-aligned subfactors, for example, Parking/Restrooms; Crowding; Trail Safety; Width/Maintenance; Signage/Seating. External validity analyses provided preliminary, hypothesis-generating evidence consistent with theoretical expectations: locals rated parking quality marginally higher than internationals ($\Delta = 0.30$, $p = 0.058$) (Seebunruang, Burns & Arnberger, 2022), and first-time visitors reported lower perceived crowding than repeat visitors ($\Delta = 0.23$, $p = 0.082$) (Budruk *et al.,* 2008), reflecting novelty and familiarity effects (Chi, 2012). Although Trail/Space Perception scores showed ceiling effects, leaving little data variance (Priskin, 2003), this pattern may indicate uniformly strong trail conditions during the pilot, or it may also reflect positive response tendencies or limited scale sensitivity under favourable conditions. While ceiling effects suggest high satisfaction levels, they may reduce the tool's sensitivity to detect marginal differences. Future applications should assess whether additional items or expanded response scales could increase discriminatory power. Notably, all items were anchored to current-day evaluations, yet demonstrated sensitivity to broader visitor characteristics, suggesting the scale taps stable perceptions rather than idiosyncratic, site-specific responses.

Several items – including waste management, drinking-water facilities, and heritage/education – showed limited variability or were inactive in the off-season pilot, constraining our ability to assess them under peak-use pressures. Moreover, data collection was confined to a single reserve outside of high-demand periods, and subgroup sizes were relatively small. Near-ceiling scores on some items further reduced discriminatory power.

Therefore, multi-site and multi-season replication is important to establish the instrument's generalisability across different reserves, seasonal contexts, and visitor profiles.

In addition, several survey items were excluded during the iterative validation process due to insufficient response rate across the development stages, highlighting the challenges of capturing low salient or context-dependent attributes under certain conditions. Nevertheless, the triangulation approach employed here, grounded in transdisciplinary practice, aligns with established validation frameworks (Lang *et al*., 2012; DeVellis, 2016; Boateng *et al.,* 2018) and ensures methodological rigour. The outcomes of the process include strong internal consistency across constructs, successful theoretical alignment, and promising evidence for evaluating social and physical carrying capacity, including sensitivity to differences across groups. Overall, the developed tool is robust and offers a novel, practical resource that can be easily employed in future studies and serves as a prototype for protected areas managers.

Conclusion and recommendations

This pilot provides preliminary evidence that the proposed tool offers a reliable measure of key constructs relevant to both social and physical carrying capacities in protected areas. Its construct-specific, theory-driven approach, combining internal psychometrics with known-groups, criterion analyses and transdisciplinarity, supports its interpretability beyond the Robberg Nature Reserve context.

While the developed visitor-based tool demonstrates strong internal consistency, theoretical alignment, and cross-group sensitivity within the initial study context, several limitations should be acknowledged. The tool's performance has so far only been evaluated at a single site, which may limit the generalisability of findings to other protected areas with different infrastructure and visitor demographics. Furthermore, the initial validation was conducted outside of peak visitation periods, possibly overlooking perceptions and challenges that emerge during times of high visitor density. Some survey items also showed borderline reliability or low variability, indicating the need for further refinement of item wording and response options to enhance sensitivity. Additionally, the factor structure was established through exploratory methods, and confirmatory validation is still required. Some KMO values were at the lower bound of acceptability (e.g., 0.57), suggesting that the factor solution should be interpreted as preliminary until replicated. The sample size is modest relative to the number of factors extracted. Therefore, future confirmatory work with larger samples will be important for establishing the stability of the five-factor structure. Evidence for external validity should be regarded as suggestive and hypothesis-generating, pending replication in larger samples and across seasons. Nevertheless, the tool presents significant possibilities as a concise and practical resource for monitoring social and physical carrying capacity, and it serves as a prototype for application by protected areas managers and researchers in similar settings.

To strengthen the tool's utility, future studies should replicate its administration across multiple reserves with varying visitor profiles to assess measurement invariance and item functioning, redeploy it in both peak and off-peak seasons to capture the full spectrum of visitor experiences, and further refine items that demonstrated limited reliability. Once data from these additional contexts are available, confirmatory factor analysis should be conducted to verify the underlying structure and ensure the tool's robustness across key

visitor groups. By addressing these areas, the tool's validity, reliability, and practical value can be further established, supporting its broader integration into adaptive management and monitoring frameworks in protected areas

Acknowledgements

This research was made possible through the generous support of the TRANSECTS project, funded by the Social Sciences and Humanities Research Council, Canada, and the University of Saskatchewan. We extend our sincere thanks to the experts and colleagues from Rhodes University, Stellenbosch University, University of Venda, Eberswalde University for Sustainable Development, the University of Saskatchewan, and Vancouver Island University, whose insights and expertise greatly enriched this work. Special thanks go to CapeNature and OnEarth ESG for their collaboration and support in the field. We are also deeply appreciative of the invaluable participation and local knowledge shared by TILLies (TRANSECTS researchers) and all other community members and participants/Robberg visitors who engaged with and contributed to the research.

References

Boateng, G. O., Neilands, T. B., Frongillo, E. A., Melgar-Quiñonez, H. R. & Young, S. L. 2018. Best practices for developing and validating scales for health, social, and behavioural research: A primer. Frontiers in Public Health, 6, 149.

Budruk, M., Wilhelm Stanis, S. A., Schneider, I. E. & Heisey, J. J. 2008. Crowding and experience-use history: A study of the moderating effect of place attachment among water-based recreationists. Environmental Management, 41(4), 528–537.

Chi, C. G. Q. 2012. An examination of destination loyalty: Differences between first-time and repeat visitors. Journal of Hospitality & Tourism Research, 36(1), 3–24.

Corlett, J. 2024. University Education in a Time of Perpetually Wicked Problems. Education Journal, 13(2), 46–52:

Cronbach, L. J. 1951. Coefficient alpha and the internal structure of tests. Psychometrika, 16, 297–334.

Etikan, I., Musa, S. A. & Alkassim, R. S. 2015. Comparison of convenience sampling and purposive sampling'. American Journal of Theoretical and Applied Statistics, 5(1), 1–4.

Fabrigar, L. R., Wegener, D. T., MacCallum, R. C. & Strahan, E. J. 1999. Evaluating the use of exploratory factor analysis in psychological research. Psychological Methods, 4(3), 272–299.

Frodeman, R., Klein, J. T. & Pacheco, R. 2017. The Oxford handbook of interdisciplinarity. Oxford: Oxford University Press.

Gonson, C., Pelletier, D. & Alban, F. 2018. Social carrying capacity assessment from questionnaire and counts survey: Insights for recreational settings management in coastal areas', Marine Policy, 98, 146–157.

Gupta, A., et al. 2023. Economic impact of nature-based tourism', PLOS ONE, 18(4), e0282912.

Huisamen, J., et al. 2011. Recolonization of the Robberg Peninsula (Plettenberg Bay, South Africa) by Cape fur seals, African Journal of Marine Science, 33(3).

iSimangaliso Wetland Park Authority 2022. Integrated Management Plan, 2022 – 2031. St Lucia, South Africa. Chapter 5 ("Environmental Management Framework") sets "nodal types and carrying capacities for each block, sub-zone, facility and activity," forming the rules that now regulate visitor numbers in ecologically sensitive zones of the World Heritage Site.

Lang, D.J., Wiek, A., Bergmann, M. et al. 2012. Transdisciplinary research in sustainability science: practice, principles, and challenges. Sustain Sci 7 (1), 25–43. (2012).

Leung, Y.-F. & Marion, J. L. 2000. Recreation impacts and management in wilderness: A state-of-knowledge review', in Cole, D. N., McCool, S. F., Borrie, W. T. and O'Loughlin, J. (eds.) Wilderness science in a time of change conference: Volume 5, Wilderness ecosystems, threats, and management. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station, pp. 23–48.

Ma, B. et al. 2020. Do Protected Areas Matter? A Systematic Review of the Social and Ecological Impacts of the Establishment of Protected Areas. International Journal of Environmental Research and Public Health, 17(19), 7259.

McCool, S. F. & Mandic, A. 2018. A social-ecological systems perspective on working toward resilience in nature-based tourism planning', Tourism Planning and Development, 1–23.

Oldekop, J.A. et al. 2016. A global assessment of the social and conservation outcomes of protected areas. The Journal of the Society for Conservation Biology, 30(1), 133–141.

Patel, T., Cowan, O. S., Little, I. & Blackmore, A. 2023. The state of provincial reserves in South Africa: Challenges and recommendations. Johannesburg: Endangered Wildlife Trust.

Rosenbaum, P. R. 1989. Criterion-related construct validity. Psychometrika, 54(4), 625–633.

Seebunruang, J., Burns, R. C., & Arnberger, A. 2022. Is National Park Affinity Related to Visitors' Satisfaction with Park Service and Recreation Quality? A Case Study from a Thai Forest National Park'. Forests, 13(5), 753.

Streiner, D. L. 2003. Starting at the beginning: An introduction to coefficient alpha and internal consistency'. Journal of Personality Assessment, 80(1), 99–103.

Spenceley, A., Rylance, A. & Laiser, S. L. 2017. Protected area entrance fees in Tanzania: The search for competitiveness and value for money. Koedoe, 59(1), a1442.

Van der Merwe, P. 2023. Tourists' feelings and behaviours in crowded areas of the Kruger National Park's southern section. KOEDOE-African Protected Area Conservation and Science, 65(1), 1762.

WESGRO 2024. Tourism Research Overview: August 2024 – Cape Town & the Western Cape. Cape Town: Wesgro.

West, P., Igoe, J. & Brockington, D. 2006. Parks and peoples: The social impact of protected areas. Annual Review of Anthropology, 35, 251–277.

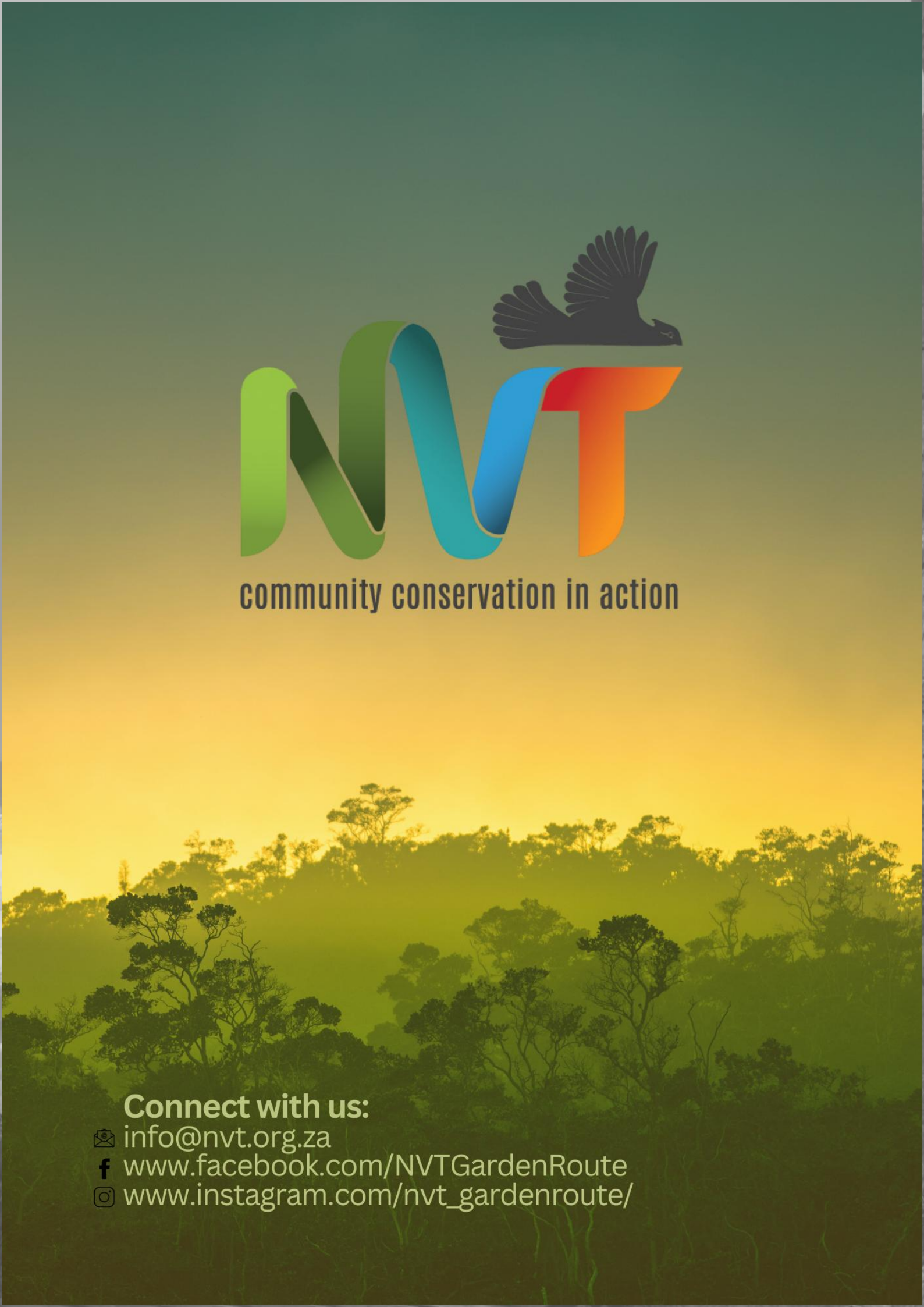
Williams, S. 2015. Tourism geography: Critical understandings of place, space and experience' Routledge.

Willis, G. B. 2005. Cognitive interviewing: A tool for improving questionnaire design. Sage Publications.

Wilson, G. V. E. & Anthony, B. P. 2023. Opportunities and barriers to monitoring and evaluating management effectiveness in protected areas within the Kruger to Canyons Biosphere Region, South Africa. Sustainability, 15(7), 5838.

Xu, N. & Li, H. 2025. Towards management of sustainable tourism development in coastal destinations of the Bohai Rim: insights from a tourism carrying capacity analysis. Discover Sustainability, 6(1), 168.




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