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Quantification of the cost recovery from invasive alien tree clearing for ecological restoration

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Abstract

This study presents a spatially explicit framework that combines remotely sensed, locally validated evapotranspiration and biomass estimates with operational clearing-cost estimation to evaluate the long-term potential for cost recovery of invasive alien tree clearing in South Africa as part of ecological restoration programmes. Using nine case studies and a 50-year discounted cash-flow model, water and biomass benefits were integrated with chemical, transport, personnel and equipment costs to compute site-level net present values (NPVs) and benefit-cost ratios (BCRs). Six of the nine case studies returned positive NPVs of up to R213 million, while the remaining three produced negative NPVs of up to R32 million. BCRs ranged from 0.31 to 4.05 (mean = 2.24), indicating that most sites generate positive returns when the economic value of both water and biomass co-benefits is considered. Cases of positive NPV's are potentially able to self-fund ecological restoration from green biofuel sales and/or water-based financing mechanisms, however, for cases with negative NPVs, we propose that limited public sector funds could be reprioritised to augment budgets. For example, this could free up constrained public funding to prioritise low-density invasions where BCRs are low, but ecological impact is high. Additionally, cost-offset proposals must be managed to avoid perverse incentives for repeated or inefficient clearing. By integrating remote-sensing models into financial analysis at decision-relevant scales, this study maps a realistic, financially feasible path for strategic restoration by landowners, managers and government.

1. Introduction

Invasive alien trees are a growing global environmental problem, degrading ecosystems and causing substantial ecological impacts and multi-billion-dollar economic losses (Brundu et al., 2020; Fernandez et al., 2023; Gioria et al., 2023). Alien trees have been found to consume significant quantities of water in regions already experiencing water scarcity such as the United Arab Emirates, Hawaii, South Africa, and India, exacerbating pressure on freshwater supplies that are under additional stress from anthropogenic climate change (Cavaleri et al., 2014; Howari et al., 2022; Kaushik et al., 2025; Le Maitre et al., 2002, 2015, 2019). Invasive alien trees also severely degrade land productivity worldwide, making agriculture the most impacted sector in terms of recorded economic costs (Fernandez et al., 2023). These financial losses are largely driven by alien tree invasions hindering crop productivity and degrading pastoral rangelands. In addition, they cause significant biodiversity loss in many regions, including the United States, India, Ethiopia, Kenya, South Africa, and Europe (Bawa et al., 2026; Edmund et al., 2019; Latimer et al., 2004; Majumdar et al., 2025; Wohlgemuth et al., 2022). Management of these alien tree invasions is costly, and insufficient funding is a key barrier to scaling invasive alien tree clearing and ecological restoration globally (Ermgassen & Lofqvist, 2024; UNEP, 2026; Witt et al., 2018). Given that mitigating the impacts of invasive species is essential to broader conservation goals, this lack of funding represents a critical bottleneck. With most finance for ecological restoration coming from public budgets (UNEP, 2021), and only one sixth of the funding required for overall global biodiversity conservation provided each year (Beverdam et al., 2025), there is a clear need to mobilize alternative funding streams.

Despite the costs involved in restoration of areas invaded by alien trees, alien clearing interventions present an opportunity to deliver coupled water and biomass co-benefits that could offset management expenses. Given that invasive alien trees generally consume significantly more water than indigenous vegetation overall (Rebelo et al., 2025), their removal as part of an ecological restoration programme can free up water resources (Blignaut et al., 2007; Le Maitre et al., 2015; Moncrieff et al., 2021). The water made available from clearing invasive alien trees represents a potential monetary value depending on the biophysical and socioeconomic context, for

example if upstream of a municipal dam, this could be monetized as increased yield (Marais and Wannenburgh, 2008; Morokong et al., 2016; Nkambule et al., 2017; Preston et al., 2018). In the arid Afar Region of Ethiopia, widespread invasions of *Prosopis juliflora* consume an estimated 3.1 to 3.3 billion m^3/year (Shiferaw et al., 2021). It is estimated that if this lost water were reallocated to regional crop production, it would generate net economic benefits of over $320 million annually. Similarly, invasive trees in Hawaiian forests have been shown to double stand-level water use relative to native vegetation, suggesting that their removal could increase water availability significantly (Cavaleri et al., 2014). Notably, across South Africa and Lesotho the total incremental water use of invasive alien plants has been estimated at ~ 3.3 billion m^3/year across 10 million invaded hectares (Le Maitre et al., 2000). When valued using South Africa's 2025 mean municipal water tariff, this volume translates to an estimated value of over $830 million (Department of Water and Sanitation, 2024).

Felled invasive alien trees also potentially represent a biomass resource that could be leveraged for the generation of various bioproducts (Pirard and Ngwenya, 2025). Studies in Kenya show that using *Prosopis juliflora* biomass in charcoal production significantly increased household incomes for participating communities (Njenga et al., 2023; Tabe-ojong, 2023). Similarly, research from South Africa has highlighted the potential for using genera such as *Pinus, Acacia, Eucalyptus, Neltuma*, and *Populus* for fuelwood, biochar, and bioenergy (Mugido et al., 2014; Mudavanhu et al., 2016; Petersen et al., 2022; Pirard and Ngwenya, 2025; Stafford and Blignaut, 2017; Vera et al., 2022). Beyond direct monetary returns, clearing and restoring invaded landscapes can deliver important non-tangible benefits such as biodiversity recovery, improved ecosystem resilience, as well as beneficial social outcomes, that strengthen the broader case for integrated, multi-stakeholder financing approaches (Boerema et al., 2017; Coldrey et al., 2022; Holden et al., 2022; Pasgaard and Fold, 2024).

Although clearing invasive alien trees can offer potential ecological and monetary benefits, decision-makers and investors face a critical lack of accurate spatial data to determine which types of investment are most appropriate and where they should be applied to ensure successful restoration (Acosta et al., 2018; Poole et al., 2023; Wang et al., 2024; Wilgen and Richardson, 2025). Scalable, real-time approaches are required to translate water and biomass gains as well as restoration costs into cost-benefit analyses (Peacock et al., 2023; Wainaina et al., 2020). In resource-constrained countries like South Africa, addressing this data gap is particularly important, where the equivalent of approximately $19 million in public funds are spent annually on clearing since 2008, yet invasion extent has increased by over 10% by 2023 (van Wilgen et al., 2022; Kotzé et al., 2025). There is a need to better understand how certain practical (e.g. accessibility and remoteness), and ecological (e.g. invasion density and invasion extent) factors can influence the costs of a clearing programme (Dew et al., 2017; Krug et al., 2010; McConnachie et al., 2012). Assessing costs at management-relevant scales (regional to landscape) is, however, inherently complex because invasive alien trees have irregular spatial distributions, and form heterogenous stands (Le Maitre et al., 2002; Richardson et al., 2005). Invasive alien tree stands therefore require different clearing strategies depending on the species, stand characteristics and ecological context (Martens et al., 2021), and have diverse value add bioproduct pathways (Pirard and Ngwenya, 2025).

Recent advances in freely available remotely sensed data and machine learning now make it possible and cost effective to generate spatially explicit, management relevant layers such as invasive alien tree classifications, water use (evapotranspiration), and above-ground biomass across landscapes (Rebelo et al., 2025; Royimani et al., 2019; Zaka and Samat, 2024). When integrated with ancillary layers such as distance to roads, slope, and invasion density, this approach could support spatial prioritisation, more realistic cost and benefit estimation and identification of appropriate financial mechanisms for ecological restoration (Comín et al., 2018; Orsi and

Geneletti, 2010; Prodromou et al., 2025; Wang et al., 2024). This capability is particularly relevant in the growing field of restoration finance, where governments, private investors, and blended finance mechanisms are increasingly exploring nature-based solutions that can sustainably generate both ecological and economic returns (Coldrey et al., 2022; Costadone and Zhang, 2025; Lozano et al., 2025; Strassburg et al., 2020).

Several restoration initiatives have demonstrated promising financial and ecological outcomes for clearing invasive alien trees (Crookes et al., 2020; Morokong et al., 2016; Mudavanhu et al., 2017; Turpie et al., 2008), and ecological restoration in general (Groot et al., 2013; Jahel and Lambin, 2025; Peacock et al., 2023). However, challenges remain regarding financial sustainability, spatial targeting of investments, and governance safeguards to prevent inefficient clearing practices or perverse incentives that could undermine long-term restoration objectives (Cheney et al., 2020; van Wilgen and Wannenburgh, 2016). Improved spatial information is therefore critical for ensuring that restoration investments are directed to locations where they are both ecologically effective and financially viable in resource constrained contexts (Deynze et al., 2022; Río-mena et al., 2023). In this study, we develop a spatially explicit framework to evaluate the potential of cost recovery of invasive alien tree clearing to support ecological restoration for nine case studies across South Africa. Using this framework, we address a key research questions: to what extent can biomass and water co-benefits offset the costs of invasive alien tree clearing within restoration programmes?

2. Methodology

The methodology for this study comprises five parts (Fig. 1). The first part introduces and describes the nine case study sites, outlining their socio-economic and ecological characteristics. The second part provides an applied example of how the spatial layers can be used to inform invasive alien tree management planning and support landowners in decision-making, including cost-benefit considerations. The third part outlines the derivation of water and biomass co-benefit estimations using freely available, satellite-derived models. The fourth part details the estimation of clearing costs for invasive alien trees including follow up maintenance applications as part of ecological restoration programmes. The fifth part presents the cost-benefit analysis over a 50-year horizon.

2.1. Case-study sites

The case study sites are located in invaded regions of catchments in South Africa, namely the Tugela, uMzimvubu, uMngeni, Berg, Luvuvhu, and Olifants-Doring (Fig. 2), all of which include strategic water source areas (Le Maitre et al., 2018). Strategic water source areas are critical high-runoff areas that cover roughly 10% of the country's land area, yet generate approximately 50% of its surface water supply, making them essential for national water security (Le Maitre et al., 2018). Machine learning classifications based on Sentinl-2 multispectral bands and vegetation indices have produced high accuracy maps of invasive alien tree presence in these catchments to the genus level at a 10 m spatial resolution, showing the extent of five dominant taxa: wattle, mesquite, gum, pine, and poplar (Fig. 3 ; Rebelo et al., 2025). To exclude commercial forestry plantations from the analysis, the classification maps were masked using the South African National Land Cover map (DFFE, 2021). These classification maps were used to quantify the composition of invasive alien tree genera as well as the total spatial extent (ha) within each case study region.

The case studies capture contrasting ecological and invasion contexts. These were also selected from areas upstream of reservoirs or dams where potential water savings following clearing could contribute to inflows and

be captured for later use, including in large municipal supply dams and farm dams. As summarised in Table 1, the invaded areas at each site fall into different size classes, ranging from small invasions of approximately 100 ha to larger invasions exceeding 1,000 ha. The biome and bioclimate also differ between sites, representing variation in indigenous vegetation types, and climatic regimes (rainfall, temperature and the seasonality thereof (Köppen and Geiger, 1936)). The case studies additionally represent different beneficiary types: private landowners (i.e. private farmers, private nature/game reserves), communal landholders (i.e. tribal authority regions and communities, etc.), and collectives (i.e. non-governmental organisations, conservancies, etc).

Table 1
Characteristics of the case study sites including details such as the location, invasive alien tree taxa present, invasive alien tree (IAT) cover and the beneficiary class.

Case-study	Catchment	Location	Invasive alien tree taxa dominant	Study area (ha)	IAT cover (ha)	Dominant biome and bioclimate	Beneficiary class
1	Tugela	Winterton: 28° 48' 21.53" S; 29° 29' 08.63" E	Wattle (*Acacia mearnsii, A. dealbata)*, gum (*Eucalyptus* spp.), pine (*Pinus* spp.), poplar (*Populus canescens*)	20667	369	Grassland Humid Subtropical with summer rainfall and warm (warmest month > 22°C)	Private Landowner
2	uMngeni	Karkloof: 29° 19' 37.35" S; 30° 05' 40.91" E	Wattle (*A. mearnsii, A. dealbata*), gum (*Eucalyptus* spp.), pine (*Pinus* spp.)	1889	132	Grassland and Forest Humid Subtropical with summer rainfall and warm (warmest month > 22°C)	
3	Olifants-Doring	Gannabos: 31° 08' 21.43" S; 19° 05' 01.08" E	Neltuma (including *N. juliflora* and *N. glandulosa*), wattle (*A. saligna*), pine (*Pinus* spp.), gum (*Eucalyptus* spp.)	14726	1416	Succulent Karoo and Azonal Semiarid with winter rainfall and warm (average annual temperature > 18°C)	
4	Tugela	Braamhoek: 28° 17' 12.91" S; 29° 34' 27.65" E	Wattle (*A. mearnsii, A. dealbata)*, gum (*Eucalyptus* spp.), pine (*Pinus* spp.), poplar (*Populus canescens)*	54923	387	Grassland Humid Subtropical with summer rainfall and cool (warmest month < 22°C)	Tribal/ Communal
5	uMngeni	Mpophomeni: 29° 32' 48.60" S; 30° 12' 44.41" E	Pine (*Pinus* spp.*)*, wattle (*A. mearnsii, A. dealbata*), gum (*Eucalyptus* spp.)	2107	182	Grassland and Forest Humid Subtropical with annual rainfall and cool (warmest	

Case-study	Catchment	Location	Invasive alien tree taxa dominant	Study area (ha)	IAT cover (ha)	Dominant biome and bioclimate	Beneficiary class
						month < 22°C)	
6	uMzimvubu	Belford: 30° 12' 36.85" S; 28° 45' 55.00" E	Wattle (*A. mearnsii, A. dealbata*), gum (*Eucalyptus* spp.), pine (*Pinus* spp.), poplar (*Populus canescens)*	8313	1079	Grassland Humid Subtropical with summer rainfall and warm (warmest month > 22°C)	
7	Berg/Breede	Wemmershoek: 33° 50' 02.49" S; 19° 08' 08.42" E	Wattle (*A. saligna*), gum (*Eucalyptus* spp.), pine (*Pinus* spp.), poplar (*Populus canescens)*	2738	103	Fynbos Humid Subtropical with winter rainfall and cool (warmest month < 22°C)	Collectives/ Reserves
8	Olifants-Doring	Clanwilliam Dam: 32° 05' 54.46" S; 18° 51' 34.83" E	Wattle (*A. saligna*, gum (*Eucalyptus* spp.), pine (*Pinus s*pp.), *Neltuma* spp. (including *N. juliflora* and *N. glandulosa*)	26922	533	Fynbos and Succulent Karoo Arid with winter rainfall and cool (average annual temperature < 18°C)	
9	Luvuvhu	Soutpansberg: 23° 00' 59.65" S; 29° 41' 50.63" E	Gum (*Eucalyptus* spp.), pine (*Pinus* spp.)	5198	98	Savanna Arid with winter rainfall and warm (average annual temperature > 18°C)	

2.2. Development of a spatial framework application

This study develops a spatial framework for invasive alien tree clearing and management planning by integrating multiple spatial layers within each case study boundary. The framework is applied to selected case studies to demonstrate how it can be used to identify priority areas for intervention based on terrain conditions, invasion characteristics, and potential biomass and/or water-use hotspots. In doing so, the study not only presents a method and its application but also proposes a conceptual approach that could be adapted by other practitioners and researchers working in different contexts, although the underlying workflow would need to be implemented independently. To illustrate its practical use, a representative case study was selected for each beneficiary class:

case studies 3, 4, and 7 represented private landowners, communal/tribal authorities, and collectives, respectively. The resulting spatial outputs included evapotranspiration, biomass, slope, and derived 'sellable units'. These layers were subsequently published within a Google Earth Engine App (**Table S1**). These outputs support more strategic clearing interventions, help prioritise areas where benefits may be greatest, and provide a spatial basis for subsequent cost-benefit analysis.

2.3. Benefit quantification

Two benefits derived from the clearing of invasive alien trees were considered in this study to potentially offset the costs of ecological restoration: water supply and biomass feedstock. Removal of invasive alien trees and their replacement with locally indigenous vegetation through the process of ecological restoration has been shown to increase water supply. In this study, biomass feedstock is considered to be the felled invasive alien trees collected during first clearing only, as part of ecological restoration. The biomass feedstock is defined as the biomass removed from invaded sites, transported off-site, and processed through value-added bioproduct pathways to produce a range of end products, depending on the taxa and the specifications of the wood (Patil et al., 2026; Pirard & Ngwenya, 2025). As part of this framework, follow-up clearing would not represent biomass gains, as follow-up should always be done timeously and therefore would mainly constitute hand-pulling of small seedlings only.

2.3.1. Water supply benefit

The water supply benefit was valued by first applying a satellite-derived evapotranspiration ensemble model developed for southern Africa to the study sites (Cogill et al., 2025a). The ensemble, which is a mean of five high-performing satellite evapotranspiration products, was implemented via the Earth Engine API (Gorelick et al., 2017) and produced monthly evapotranspiration at 100 m spatial resolution (mm $month^{-1}$). For each case study site, we applied the ensemble model and used the invasive alien tree maps (Fig. 3) to estimate evapotranspiration of invasive alien tree classes and adjacent indigenous vegetation classes (such as grasslands, fynbos and indigenous bush types). This approach allows us to quantify differences in water use between invaded and indigenous vegetation states, which can be assumed to approximate the amount of water made available following ecological restoration, given that restoration seeks to remove invasive alien trees and assist natural regeneration of indigenous vegetation. While systematic differences between invaded and indigenous vegetation areas could introduce bias into these estimates, the magnitude of the estimated potential water savings is broadly consistent with previous studies (Le Maitre et al., 2016; Rebelo et al., 2025).

Monthly evapotranspiration totals were summed to produce annual evapotranspiration (mm/a^{-1}) for each class and then aggregated across the study area using zonal statistics from the rasterstats package in Jupyter notebooks (Perry, 2020). This process was repeated over a five-year period from the date of the invasive alien tree classification maps, and mean annual values were calculated per class. Averaging across multiple years was intended to account for interannual variability and to avoid bias associated with relying on a single year that may reflect anomalous conditions, such as a wet or dry year (Bakker et al., 2011; Zhu et al., 2024). In addition, the ensemble modelling approach provides a range of evapotranspiration estimates (minimum, mean, and maximum), capturing model uncertainty, which is presented in **Figure S1**. The difference in evapotranspiration between invasive alien and indigenous classes (mm yr^{-1}) was converted to a volumetric estimate and interpreted as potential surface runoff (m^3 yr^{-1}) for each case study site. This assumes that reductions in evapotranspiration translate directly into increased water availability for downstream systems. However, this represents an

oversimplification, because not all water made available by reduced evapotranspiration will become surface runoff entering dams for example (Sposito, 2017; Demeke, 2025).

In reality, topography, soil and geology will determine partitioning between runoff, soil water, and groundwater, and therefore this approach will be an over-estimation of runoff (Condon and Maxwell, 2019; Hughes et al., 2021; Fang et al., 2022). However, the commonly considered alternative, e.g. hydrological modelling to obtain streamflow (Kusangaya et al., 2017; Mvandaba et al., 2018; Warburton et al., 2010), focusses on streamflow, while groundwater receives less attention (Butterworth et al., 1999). Also, water that percolates into the soil, or groundwater, does not necessarily represent a loss for dams downstream, as it may be that these are fed by groundwater in certain seasons. At this stage, too little is known to draw finite conclusions beyond the fact that these values may be overestimates. However, at least this approach calculates the total impact of alien trees on water resources, not simply on only one part of the water balance. Given that there is insufficient information on runoff/recharge partitioning in most South African catchments, it was not possible to use simple ratios to adjust the results. Should more data become available in future, these calculations can easily be adjusted. Furthermore, to account for sensitivity arising from model uncertainty, the minimum, mean, and maximum potential water benefits for each case study were propagated from the model ensemble and are available in the supplementary material (**Table S2**).

In converting potential surface runoff to a simplified version of reliable reservoir yield, several additional assumptions are made. First, it is assumed that sufficient storage capacity is available within downstream reservoirs to capture the additional inflows. Second, we do not explicitly account for inter and intra-annual variability, including seasonal dynamics or lag effects between dry and rainfall seasons which may influence the timing and reliability of water supply. Third, to convert surface runoff to reliable reservoir yield we applied a conservative retention factor of 0.2 (i.e., 20% of surface runoff is assumed to be stored as reliable yield), to account for losses through evaporation, groundwater recharge and other hydrological processes. This is conservative since other studies report runoff-to-yield factors in the range ~ 40–80%, however the intention is to offset any overestimates that resulted from runoff estimation, intra-annual variability or assuming the availability of storage capacity, therefore arriving at a conservative estimate (Basson and Rooyen, 2001; Stats SA, 2006).

The resulting yield volume ($m^3\ yr^{-1}$) was valued according to the downstream water-use context of each case study. For case studies supplying municipalities as a water-board, we used a bulk water tariff derived from provincial means (Ruiters & Amadi-Echendu, 2023), because this reflects the value of untreated water stored in a bulk reservoir and sold to municipalities or distributors. Municipal retail tariffs are higher and were not used, as they include purification and distribution costs that require additional infrastructure and service delivery beyond the generation of the water resource itself. These costs are not attributable to the water made available through invasive alien tree clearing and would therefore inflate the estimated financial benefit. For private farm dam beneficiaries we used a conservative raw-water tariff (provincial mean) which would be appropriate for direct on-farm abstraction and irrigation (Ruiters & Amadi-Echendu, 2023). Full details of the tariff choice (ZAR m^{-3}) and the valuation applied to each case study are provided in the supplementary material (**Table S3**). An alternative valuation approach, particularly relevant in agricultural settings, would have been to estimate the value of the additional water in terms of the irrigated area it could support for different crop types (D'Odorico et al., 2020). However, given the great diversity in crop types this was not possible in this study but could be a more feasible estimation for private landowners to do in terms of their own costing of the benefits as part of this framework.

2.3.2. Biomass feedstock benefit

The biomass feedstock benefit of felled invasive alien trees was estimated by first quantifying the aboveground biomass within each case study boundary. This was done using a trained random forest model specifically developed for estimating the biomass of invasive alien trees in southern Africa (Cogill et al., 2026). The model, implemented via the Earth Engine API (Gorelick et al., 2017), uses freely available satellite data (Sentinel-1 textural indices, Sentinel-2 multispectral bands, and Shuttle Radar Topography Mission topography) to produce aboveground biomass estimates (t/ha) at a 25 m spatial resolution, and is applied directly to the invaded areas of each case study by using MapWAPS layers (Rebelo et al., 2021; 2024; Cogill et al., 2024a; 2024b; 2025b; Skosana et al., 2024) to mask out all indigenous vegetation. The economic valuation of the estimated biomass depends on the invasive tree genera present in each case study region, as different genera are suited to different bioproducts (Pirard & Ngwenya, 2025). Each bioproduct is associated with a specific market value (ZAR Mg^{-1}), derived from published literature in South Africa. To estimate economic returns, the biomass of each genus is converted into its corresponding bioproduct using established conversion factors. These conversion factors represent the yield of the target product per unit of raw biomass (e.g., 1 tonne of raw biomass produces 0.3 tonnes of briquettes). The resulting quantity of each bioproduct (in tonnes) is then multiplied by its respective market value to derive the total economic value (**Table S4**).

To model the revenue streams from cleared biomass, ‘sellable units’ were systematically identified from the present invasion stands within each case study. The sellable units are high-density, homogenous stands that could specifically be targeted to produce high-value bioproducts, such as wattle briquettes, gum and mesquite charcoal, and pine pulp (Dovie et al., 2012; Vundla et al., 2016, 2017). The remainder of the invasion stands across the study sites, are typically more scattered, occur at lower densities, and consist of mixtures of different invasive alien tree taxa. These stands were ear-marked to be sold as firewood. The comprehensive valuation breakdown for each case study is available in **Table S5**. Furthermore, to account for the volatility of the bioproduct markets, sensitivity reduction factors of -10% and – 30% were applied to the model (Bergman et al., 2022; Gianvenuti and Vyamana, 2018; Ganzi and Pronti, 2026). These adjustments project the realized yields under constrained conditions, such as lowered timber quality, the impact of natural factors (like wind and floods), or unforeseen productivity issues at the processing plant or depot (Gregersen and Contreras, 1979; Toledo-aceves et al., 2022).

2.4. Cost quantification

The cost calculations for invasive alien tree clearing comprised three components: (i) chemical, (ii) transport, and (iii) personnel and equipment. In addition to initial clearing costs, we also accounted for the costs of follow-up maintenance over time, recognising that repeated interventions may be required to manage regrowth and ongoing recruitment of invasive alien trees. Many cost-benefit studies either omit chemical costs (i.e. herbicides and adjuvants) or apply generalized clearing costs, but this is problematic because different genera require different active ingredients and dosage rates (Weidlich et al., 2020); and this can substantially impact treatment cost. Likewise, access and invasion characteristics (such as distance to roads, slope and stand density) can have marked effects on labour and equipment requirements and hence the overall costs of clearing (de Lange et al., 2022; McConnachie et al., 2012). For this reason, accessibility and density were factored into personnel and equipment cost calculations.

2.4.1. Control methods

Different control methods are required for different taxa and different densities (Martens et al., 2021). For each invasive tree genus occurring in the case study regions, we selected the most appropriate control method: for example, young *Pinus* saplings were allocated the hand-pulling method, while mature pines would be felled using a chainsaw with no herbicide application required (de Lange et al., 2022; Environmental Programmes Pesticide Policy, 2021; Martin et al., 2025; Nunez et al., 2017). In contrast, genera such as *Neltuma* require a cut-stump application with systemic herbicides (for example triclopyr – butoxy ethyl ester) capable of translocating to meristematic tissue and killing the rootstock (Bovey et al., 1983). Similarly, *Eucalyptus* spp. has robust regenerative capabilities and requires a cut-stump application immediately following felling (Little and Eccles, 2010; Montes et al., 2022; Roberts et al., 2018). This treatment typically necessitates a highly concentrated dose of herbicide formulated in a carrier solution of diesel or natural oils; which provide extended humectancy and facilitate deeper penetration of the active ingredient into the stump tissues. For the control of *Acacia* spp., successful eradication also relies on a cut stump application with auxin-mimic herbicides, such as triclopyr, aminopyralid, or fluroxypyr, that are highly mobile and can readily translocate through the vascular system to the root network (Grossmann, 2010; Macdonald et al., 2012; Tu et al., 2001). This requirement for extensive herbicide translocation is equally important when managing *Populus* spp. *Populus* species frequently develop complex, interconnected underground shoot systems, so the applied herbicide (in a cut-stump application) must be capable of translocating to all connected clonal members (Lowenstein et al., 2023).

2.4.2. Chemical costs: herbicide and adjuvant

To estimate invasive alien tree clearing costs, invasion density (percentage cover) was first quantified for each taxon, as clearing costs vary with invasion density. A 1 ha fishnet was created across each study region, and zonal statistics were used to calculate the number of invaded 10 m pixels belonging to each taxon within each polygon. Taxon-specific invasion density was expressed as a percentage of each hectare. Where multiple invasive taxa occurred within a polygon, the percentage cover of each taxon was recorded separately. Five density classes were defined (**Table S6**): 1–5% (very scattered), 5–25% (scattered), 25–50% (medium), 50–75% (dense), and 75–100% (very dense). For herbicide cost calculations, all grid cells under the medium, dense, and very dense categories (i.e., > 25% cover) were assumed to require herbicide application via the cut stump method according to national standards (Environmental Programmes Pesticide Policy, 2021; WfW, 2002). In contrast, very scattered and scattered classes (≤ 25% cover) were assumed to be treatable through manual clearing methods (e.g., hand-pulling or felling) without the need for herbicide. These thresholds and management assumptions were informed by literature on invasive alien plant management (de Lange et al., 2022; Environmental Programmes Pesticide Policy, 2021; Fill et al., 2017; Foxcroft, 2008; Le Maitre et al., 2000; Marais and Wannenburgh, 2008; McConnachie et al., 2012).

Polygons classified as medium to very dense were tallied by genus and multiplied by the per-hectare herbicide rate for that genus to estimate total herbicide volume required. Cut stump treatment assumptions used for costing were as follows: an herbicide spray-mixture requirement of 10 L per hectare, with approximately 100 mL of the spray mixture applied per stump. Basal-bark applications were excluded because our scenario simulated active felling, treatment and biomass removal, rather than in situ basal treatments that require waiting periods and different operational procedures. Chemical costs were estimated using South Africa's RT12 national tender schedule (RT12), which provides standard contract prices for herbicide products (National Treasury of South Africa, n.d.). For each genus, the required dosage (L ha^{-1}) was multiplied by the area (ha) of dense invasion to yield the total litres required. This volume was then multiplied by the RT12 unit price (ZAR L^{-1}) to give the

chemical cost per taxon. Aggregating across taxa provided the total chemical cost for each case study region (**Table S7**).

Human-health and environmental implications including any documented health risks and the required personal protective equipment were also considered when choosing the application methods and products. Where more than one herbicide and adjuvant were registered for a genus, selection favoured the most environmentally compatible product with the lowest human-health risk, informed by the Globally Harmonized System (GHS) and product registration status (Hongoeb et al., 2025; Moreira & da Silva, 2024). Detailed product, dosage and safety information are provided in the supplementary material (**Table S8**).

2.4.3. Personnel and equipment costs: terrain and accessibility adjusted

Personnel and equipment costs were estimated by integrating three factors into a composite “difficulty” index representing how costly and operationally challenging clearing would be. The three factors were distance to road, slope, and density. Firstly, invasion density was included using the 1 ha density metrics described in Section 2.4.1, which reflect the physical difficulty of clearing operations, as denser invasions require more effort to reach and treat individual trees effectively, and may have a higher number of individuals/stems, all of which translate to higher costs.

Distance from each invasive alien tree pixel (derived from the alien tree map of Rebelo et al., 2025), to the nearest road were calculated in Jupyter notebooks by computing each pixel’s proximity to South Africa’s national road network (OpenStreetMap Team, 2017) and then aggregating these values to obtain a mean distance for each invasion within each case study region. The roads dataset was filtered to include only primary, secondary, highway, and residential/urban roads. Road classes such as pedestrian routes, proposed roads, links, construction routes, paths, cycleways and steps were excluded, as these would be difficult to access with standard vehicles or trucks required for field crews and biomass transport. Furthermore, it was assumed that the further from the road an invasion is, the greater the logistical burden and non-productive time. Extended transit distances require field crews to hike further while carrying heavy mechanical equipment and herbicide, which reduces the active clearing hours per shift.

Terrain slope was also incorporated as a measure of accessibility and operational difficulty where steep inclines severely restrict workforce mobility and can amplify the occupational health and safety risks associated with felling timber and handling chemicals. Mean slope values for each invasive alien tree pixel were derived from the Shuttle Radar Topography Mission dataset (Farr et al., 2007) at its native 30 m resolution (Gorelick et al., 2017). Slope values were first calculated for each 1 ha polygon and then aggregated to a mean value for each the invasion stands within the respective case studies.

These three factors were combined into a composite ‘difficulty index’ where firstly, the factors were standardized onto a 1 to 5 scoring scale. Slope values were assigned a score based on five intervals: 0.00–5.00° (score of 1), 5.01-10.00° (2), 10.01-15.00° (3), 15.01-20.00° (4), and 20.01-35.00° (5). Distance from the nearest road was similarly categorized and scored: 0–2.00 km (1), 2.01-4.00 km (2), 4.01-6.00 km (3), 6.01-8.00 km (4), and 8.01-10.00 km (5). Invasion densities (ranging from very sparse to very dense) were also assigned corresponding quantitative scores from 1 to 5. The values from these three factor scores (slope, distance, and density) were then averaged to produce a mean composite difficulty index for each invasion class, with final values ranging

from 1 to 5. A personnel and equipment cost (ZAR ha^{-1}) was then assigned to each difficulty class based on values reported in the literature, where clearing costs accounted for practical constraints such as density, terrain and accessibility (**Figure S3**). Average scores between 1.0 and 1.8 were allocated a rate of R1,000/ha; scores > 1.8 to 2.6 a rate of R5,000; >2.6 to 3.4 a rate of R15,000; >3.4 to 4.2 a rate of R30,000; and scores > 4.2 to 5.0 a rate of R50,000. These literature-derived values were cross-checked against informal quotations from five invasive-tree clearing contractors operating across the Western Cape, Northern Cape, Limpopo and Gauteng provinces in South Africa to ensure that the inflation-adjusted 2025 estimates were within realistic industry ranges.

2.4.4. Transport costs

Transport costs represented the once-off expense of getting to the invasion stands and additionally, moving all felled invasive alien tree biomass to the nearest potential depot for processing and sale. For each case study region, we computed the mean road distance (km) from all invasive alien tree pixels within the study boundary to the nearest established town using the rasterstats package (Perry, 2020). Smaller villages and townships were excluded because it was assumed that they are unlikely to host sophisticated depots or processing facilities. The mean distance (km) was multiplied by the total biomass (tonnes) within the region and by a transport rate (ZAR t^{-1}/km^{-1}) to estimate total transport costs (**Table S9**). The transport rate was based on a weighted-average cost derived from a study that modelled transport of different mixtures of *Acacia*, *Eucalyptus* and *Pinus* biomass under the Vehicle Costing Schedule (Mugido et al., 2014). In addition, a daily driver rate and a truck hire rate for a 24-ton truck were included, based on an informal quote for travel from the nearest town to the case study sites (**Table S9**).

2.5. Cost-Benefit-Analysis

Cost-benefit analyses were conducted over a 50-year time horizon using a discount rate of 6%. Discount rates of 4% and 8% were also computed for sensitivity testing. The 50-year period was selected to represent the full management cycle of invasive alien plant control, encompassing initial clearing, repeated follow-up treatments, and eventual ecosystem rehabilitation to a stable, invasion-free state (Rebelo et al., 2025). This endpoint represents a condition in which regrowth and the recruitment of new invasive alien plants are sufficiently controlled to prevent reinvasion, allowing indigenous vegetation to regenerate and the ecosystem to recover (Holmes et al., 2008). Invasive alien tree clearing is not a one-time operation, and requires ongoing maintenance to achieve true ecosystem rehabilitation (Martens et al., 2021; Duarte et al., 2023). In our analysis, gains were represented by the benefits from water supply and bioproducts, while expenditures included transport, chemical, and personnel and equipment costs.

The timing and magnitude of benefit and cost components were structured to reflect realistic ecological and operational dynamics. The analysis caps water yield benefits at 25 years to reflect a conservative estimate of hydrological recovery. This approach accounts for the natural saturation of catchment yield (Le Maitre et al., 2019; Turpie et al., 2008). Biomass benefits were considered a one-time gain occurring in a single year, reflecting successful clearing without significant subsequent regrowth or reharvesting, as part of an ecological restoration programme. Allowing alien trees to grow back to harvest them again would be illegal without a water use licence, as required for a forestry plantation (Dye and Versfeld, 2007; Pirard, 2023).

Chemical, and personnel and equipment costs, were highest in the initial year, declining to 10% of initial costs during years 2–10 to account for minor follow-up treatments (e.g. hand-pulling of seedlings), and further reduced

to 5% during years 11–50 to represent long-term maintenance under sustained invasion control. These reductions represent simplifying assumptions used to model the expected decline in management costs over time, rather than fixed or universally applicable cost trajectories. In practice, maintenance requirements and associated costs are likely to vary among sites and over time, depending on factors such as reinvasion rates, resprouting, seed-bank persistence and the effectiveness of previous control efforts. The need for an extended maintenance period is further supported by evidence that seeds of invasive alien trees can remain dormant and viable in the soil seed bank for many years (Marais and Wannenburgh, 2008; Richardson and Kluge, 2008; Duarte et al., 2023). The transport costs were treated as a once off initial expenditure, as biomass harvesting and transportation to the simulated depots would not take place after the initial successful clearing operation. The economic performance of invasive alien clearing for each case study was evaluated using two standard decision metrics namely Net Present Value and Benefit-Cost Ratio.

To test these metrics against market volatility and hydrological uncertainty, sensitivity analysis was conducted by defining three distinct scenarios to assess the impact on overall profitability. A worst-case scenario was modelled using a 30% reduction in expected biomass value coupled with the minimum potential water supply benefit. A central scenario applied a 10% reduction in biomass value alongside the mean water supply benefit. Finally, a best-case scenario evaluated profitability using the full, unadjusted biomass value (0% reduction) and the maximum potential water supply benefit. We also acknowledge residual uncertainty in the satellite-derived evapotranspiration and above-ground biomass models used to quantify water and biomass benefits (PBias = -15.47% and − 33.92%, respectively). Both models systematically underestimate observed values (Cogill et al., 2025a, 2026). To illustrate the potential effect of this bias on financial outcomes, the water and biomass benefits were scaled by the inverse of the reported PBias and the resulting NPV and BCR were recomputed. The bias corrected results are presented in the supplementary material.

2.5.1. Net Present Value

The Net present value (NPV) represents the discounted net economic benefit of invasive alien plant clearing over the analysis period, calculated as the sum of annual benefits (B_{t1} (water) and B_{t2} (biomass)) minus the costs (C_{t1} (chemical), C_{t2} (personnel and equipment) and C_{t3} (transport)), discounted (s) over time (t) (Schwab and Lusztig, 1969). The NPV was calculated using the NumPy package (Harris et al., 2020) to create arrays representing annual cash flows over a 50-year horizon. The code manually discounts these flows and then aggregates them using the np.sum() function. The final NPV is obtained by subtracting the total Present Value (PV) of costs from the total PV benefits.

2.5.2. Benefit-Cost Ratio

The Benefit-Cost Ratio represents the efficiency of investment by expressing the ratio of total discounted benefits (water and biomass gains) to total discounted costs (chemical, transport and personnel and equipment), with values greater than one indicating economic viability (Schwab and Lusztig, 1969). The BCR was computed using the NumPy array and aggregation logic established for the NPV calculation. It uses the total Present Value of benefits and costs derived from the np.sum() of the discounted flow. The ratio is calculated using standard division (/) to divide the total of Present Value of benefits by the total Present Value of costs.

2.5.3. Evaluation of benefit and cost intensities and their relationship to variables

Invasion density, slope, and distance from roads are direct inputs to the personnel and equipment cost model through the composite difficulty index. Consequently, examining correlations between these variables and the benefit-cost ratio (BCR) would be partly circular, as these variables already contribute to the estimated costs underlying the BCR. Instead, we calculated the present value of biomass benefits, water benefits, personnel and equipment costs, and total costs per invaded hectare for each of the nine case studies. These unit intensities were then examined descriptively in relation to practical site characteristics (invasion density, slope, remoteness, distance from road, and invasion extent), as well as site-level BCR and net present value (NPV). Invasion density was quantified for each site overall using an Ordinal Weighted Density Index, which assigns sequential integer scores to density classes to produce a standardised score out of 5. A post-hoc statistical power analysis based on Fishers z-transformation was conducted to assess the ability to detect correlations given the small sample size (Faul et al., 2009). Using a significance threshold of $p < 0.05$, the statistical power was low across all variables, ranging from 0.05 for remoteness to 0.40 for slope. This indicates that the limited sample size ($n = 9$) substantially restricts the ability to detect associations. Accordingly, the intensity analysis was treated as exploratory and descriptive, and no formal significance tests were reported.

3. Results

3.1. Cost-Benefit-Analyses

Across the nine case studies, the financial outcomes after the 50-year horizon vary substantially depending on the evaluated scenario (**Table S10** – **Table S12**). In all cases, personnel and equipment costs comprised most of the expenditure (more than 90% of the costs on average), while transport and chemical costs contributed a much smaller portion (approximately 10% or less). Regarding the benefits, biomass dominated the total value generated across all case studies. Biomass benefits constituted between 80% and 99% of total benefits, whereas water gains contributed a far smaller share. The case studies with the highest total costs (particularly case studies 3 and 6) also generated the greatest overall benefits across all evaluated scenarios.

3.1.1. Net Present Value (NPV) Outcomes

Under the central scenario, six of the nine case studies returned a positive NPV. These central NPVs ranged considerably across the sites, from a financial loss of approximately R31.5 million for case study 7, to a substantial net benefit of approximately R212.8 million in case study 6. While overall NPV outcomes were robust across scenarios, case study 8, which had marginal NPVs under the central and best-case scenarios, would only partially support restoration under the worst-case scenario (Fig. 4). For the five case studies that managed to maintain a positive NPV under the worst-case scenario, absolute values ranged from a worst-case loss of R35.2 million to a peak gain of R151.1 million. Conversely, under the best-case scenario, the magnitude of the financial returns increased substantially. While the total number of viable cases with positive NPVs remained at six (identical to the central scenario), the highest returns were amplified, with peak NPVs exceeding R255.1 million.

3.1.2. Benefit-Cost Ratio (BCR) Outcomes

Under the central scenario, six of the nine case studies achieved a BCR greater than one (Fig. 5). The central BCRs varied from 0.31 (case study 7) to 4.05 (case study 6), yielding an average ratio of 2.24 across all case studies (Fig. 5). This means that for every Rand invested in clearing operations under baseline assumptions, a

project generates an average return of R2.24 in combined water and biomass value over the 50-year horizon. The sensitivity of the investment efficiency was further demonstrated in the extreme scenarios. Under the worst-case scenario, average financial performance dropped, yielding an average BCR of 1.73 across all sites, with maximum BCRs peaking at 3.16 (and five sites remaining above the break-even BCR > 1 threshold). Under the best-case scenario, the efficiency of the investment was strongly amplified, driving the average BCR up to 2.59 and reaching a maximum BCR of 4.65. Bias-corrected NPV and BCR values that account for the systematic underestimation in the evapotranspiration and above-ground biomass models, show that BCR values increase by 0.10–1.34 and that the three negative sites improved but stayed below the break-even threshold (**Table S13**).

3.2. Benefit and cost intensities and their association with practical variables

Across the nine case studies, present-value biomass benefits ranged from approximately R83 000 to R252 000 per invaded hectare (median of approximately R115 000), while water benefits per hectare were substantially smaller (median of approximately R5 500) (Table 2). Operational costs per hectare showed far greater variation, ranging from R28 000 to R439 000 per hectare (median of approximately R54 000). The total costs per hectare followed a similar pattern. Sites with BCR > 1 (six sites) had a median operational cost of approximately R42 000 per hectare and a median invasion extent of about 450 hectares, whereas the three sites with the BCR < 1 had an operational cost of approximately R341 000 per hectare and much smaller invasion extents with a median of about 100 hectares. Biomass benefit intensity was relatively consistent across high and low BCR sites; the principal difference seemed to be the unit cost rather unit benefit. Across the case studies, operational costs per hectare varied across sites with different terrain and accessibility characteristics, while invasion density showed no clear pattern in relation to benefit or cost intensity. Additionally, per-hectare benefit and cost intensities showed no consistent pattern across sites of different invasion extents. These patterns indicate that cost recovery is most strongly influenced by the combination of moderate unit operational costs and sufficient biomass volume, rather than by any single practical variable acting in isolation.

Table 2
Present-value benefit and cost intensities (ZAR per invaded hectare) and key practical variables for the nine case studies, ranked by benefit-cost-ratio (BCR). Operational cost is the present value of personnel and equipment costs.

Case	BCR	Biomass benefit /ha	Water benefit /ha	Operational cost /ha	Total cost /ha	Slope (°)	Remoteness	Invasion extent (ha)	Density index
6	4.05	251572	10340	54440	64732	12.5	16553	1332	1079
2	3.84	123202	989	28118	32363	8.2	9647	350	132
1	3.36	113902	558	29525	34025	3.5	7849	574	369
3	3.35	209023	559	50226	62495	3.3	25227	8063	1416
5	3.05	109451	5492	33724	37693	10	8475	158	182
8	1.22	83342	20627	83025	85107	6.3	18642	3396	533
4	0.65	90238	7713	142259	150530	16.7	21374	2246	387
9	0.34	115286	2063	340823	348205	18.2	19674	2695	98
7	0.31	123422	13171	438707	443465	20	8732	5993	103

3.3. Applying the spatially explicit framework for decision support

Here we describe the results of three case studies as an example of how this decision support would look when the framework is applied.

Case study 3 illustrates a private farm where *Neltuma* invasions are concentrated within the riparian zone of the Doring River, an important water source supporting irrigated grain production in the Northern and Western Capes (Fig. 6). Within this landscape, three densely invaded riparian patches were identified as the primary sellable units (encircled in black). These compact, high-density stands represent the most suitable targets for higher-value charcoal production, whereas the more dispersed surrounding invasions would be better suited to firewood harvesting. The evapotranspiration map indicates that sellable units 1 and 3 offer the greatest potential water returns per unit area, with values approaching 1400 mm/annum. The above-ground biomass map similarly shows that these sellable units contain the highest biomass potential per pixel, reaching nearly 20 t/pixel, which likely reflects more mature invasion stands. From a logistical perspective, the slope map suggests that the invaded area is largely situated on gentle terrain of approximately 0–10°, which would facilitate easy access for field crews and enable extraction and transport to roadside collection points for delivery to a depot or processing facility.

Case study 4 represents a municipal and communal landscape above Braamhoek Dam, where invasion is dominated by widespread wattle, with isolated gum stands occurring along slopes and drainage lines. Three sellable units were identified as priority targets for large-scale briquette and charcoal production: two dense wattle patches at the base of the slope (sellable units 1 and 2), and one isolated gum stand along the slope (unit 3). Similar to case study 3, the more scattered invasions are more appropriate for firewood harvesting than commercial biomass extraction. The evapotranspiration and above-ground biomass maps show that these units correspond to areas of relatively high water-return and biomass potential compared to the sparser stands.

However, the slope map highlights an important operational constraint. Although the invaded stands themselves are positioned on relatively accessible ground, units 2 and 3 are surrounded by steep and rugged terrain, meaning that contractors would need to traverse more difficult terrain to reach them.

Case study 7 represents a potential collective restoration effort, such as the Boland Groot Winterhoek Strategic Water Source Area Collective, in a landscape invaded by mixed pine and wattle stands along rivers and drainage zones feeding into Wemmershoek Dam. The invasions seem to be low density across the landscape; however, a few denser stands emerge as priority sellable units and are highlighted by black circles. These units present the best opportunities for maximizing both biomass recovery and water-return benefits. Of the three sellable units, unit 1 is the most strategically located, lying closest to the dam and on relatively flat terrain, which makes it the most accessible initial target for field operations. It also shows the highest water-return potential per unit area across the case study, together with strong biomass potential, with many pixels approaching the maximum of 20 tonnes per pixel. In contrast, sellable units 2 and 3 are located deeper within the valley and would require more difficult access for clearing teams.

4. Discussion

4.1. Patterns of cost recovery: a comparative evaluation of remotely sensed and historical assessments

This spatially explicit framework demonstrates how remotely sensed data can be used by decisionmakers in ecological restoration to guide operational management planning and to inform cost-benefit analyses. This framework can support strategic investment decisions, and it can be easily adapted for diverse practical uses. Overall, the cost-benefit analysis indicated that invasive alien tree clearing can completely recover costs in most settings, with six of the nine case studies producing positive net present values (NPVs) and benefit-cost-ratios (BCRs) greater than one. Thus, the results support the thesis that invasive alien tree clearing may produce benefits that can be used to offset the costs of ecological restoration (Blignaut et al., 2013; Crookes and Blignaut, 2019; McConnachie et al., 2012).

The average performance across sites shows a robust BCR of approximately 2.24, although outcomes vary widely among case studies (ranging from 0.31 to 4.05), reflecting strong site-specific controls. These results align with historical assessments like Hosking and Preez, (1999), who reported BCRs for invasive alien tree clearing in the Eastern Cape Province ranging from 0.85 to over 12.00. The wide variance in their results similarly reflected differences in land-use context and the specific discount rates applied. Personnel and equipment expenditure consistently drove costs across all nine sites (accounting for > 90% of expenditure), and the revenue streams similarly displayed a clear pattern, where biomass was shown to be the dominant contributor to net present value. This finding aligns with earlier studies showing that biomass returns can exceed water-related benefits where local bioproduct markets are well developed or where lower-tier, raw water tariffs apply at the source, particularly in regions where water trading opportunities are limited or water markets are poorly developed due to regulatory constraints (Nkambule et al., 2017; Stafford and Blignaut, 2017; Vundla et al., 2016). This, however, differs from findings at the national level, where water benefits were estimated to represent the largest share of the US$6.6 billion ecosystem service value of clearing invasive alien plants (Stafford et al., 2017). This may indicate that our approach underestimated water benefits due to the challenges from converting runoff to yield at the site level.

The costs of restoration at three sites (case studies 4, 7, and 9) outweighed the income potentially generated by biomass and water using our approach which considered private sector funding alone, producing negative NPVs and BCRs well below the break-even threshold. These outcomes are likely influenced by specific, localized combinations of practical operational characteristics that drive up unit costs without yielding sufficient biomass volumes to offset the initial capital outlays. This highlights the importance of considering local conditions when assessing the potential cost recovery of restoration interventions. Extending this spatially explicit approach across a larger number of restoration sites may therefore provide greater power to identify the operational and environmental factors that most strongly influence financial outcomes. Beyond these site-specific findings, our spatially explicit approach supports the findings of historical assessments, serving to contextualize and operationalize their findings through remote sensing approaches. While our results are consistent with earlier broad-scale assessments showing that invasive alien trees consume significant quantities of water (estimated using biomass-water relationships or mean annual runoff (Le Maitre et al., 2000; Le Maitre et al., 2016)), our spatially explicit workflow goes beyond these coarser regional extrapolations to include important local heterogeneity within each case study. We also improve on previous work by replacing coarse density-to-biomass ratios with remote sensing biomass maps (Nkambule et al., 2017; Versfeld et al., 1998; Vundla et al., 2016; Wise et al., 2012).

Furthermore, our spatially explicit workflow provides a foundational data layer that can be integrated into the risk-reward portfolio mapping framework proposed by Crookes et al., (2013). Our framework highlights the site-specific variability in financial feasibility, whilst the methodology developed by Crookes et al., (2013) offers a useful mechanism to classify these projects as 'pearls' (projects with a high likelihood of success), 'oysters' (projects with a high risk and uncertain merits), 'bread and butter' (essential projects that enterprises cannot do without), and ‘white elephants’ (projects which are best avoided). In their study, water-dominated projects were usually ‘pearls’, while water-only or single-service projects were ‘oysters’. Additionally, crop and grazing benefits were mostly ‘bread and butter’; unless bundled with water and biomass. Our results support the same broad split between market-ready sites and sites requiring additional funding. Six of the nine case studies had positive NPVs and BCRs > 1 under the central scenario and five remained viable under the worst-case scenario. Those sites occupy the pearl space. The sites with negative NPVs and BCRs < 1 resemble white elephants. Unlike Crookes et al., (2013), biomass accounted for more than 80% of the discounted benefits in our analysis, rather than water. This difference may reflect the conservative approach we used to estimate water benefits, which applied a runoff-to-yield conversion factor of 0.2.

4.2. Broader implications for restoration finance and policy

Dense and extensive invasions could deliver higher returns on investment, largely because the water and biomass benefits released through clearing outweigh the higher operational and chemical costs required to remove mature stands (Le Maitre et al., 2016; Marais and Wannenburgh, 2008; Mugido et al., 2014). From a finance perspective, this suggests that targeting such invasions can generate meaningful, defensible returns (Blignaut et al., 2007; Stafford, 2019), particularly where water and biomass values can be realised through existing institutional or market pathways. Recent outcomes-based water bonds demonstrate that private capital can be mobilised for catchment restoration, but they also highlight the challenges of treating water benefits as a fully bankable standalone service. Site-level conversion of reduced water use into actual water yield remains uncertain, while the long-term maintenance of cleared areas is essential to sustaining these gains (Brand et al., 2021; Stafford et al., 2019). These findings support the case for viewing invasive alien tree clearing as a financially viable ecological intervention (Crookes et al., 2020). However, it is important to not focus solely on financial gains which could

offset the costs of ecological restoration. Ecological principles emphasise the importance of tackling sparse and younger invasions early, before they spread and become more costly to remove (Mostert et al., 2017; Martens et al., 2021).

Additionally, in this study, we assumed sparse invasions could be managed via hand-pulling without herbicide. While this may vary in reality, opportunistic management, such as clearing regrowth after a fire, remains a cost-effective strategy for lower density invasions (Martens et al., 2021). Planning management around recurrent fire cycles could further reduce costs by allowing agile funding mechanisms to mobilise clearing teams rapidly when fire creates favourable clearing opportunities. Where feasible, integrating biomass harvesting could provide additional financial and ecological benefits by removing above-ground fuel before fires, thereby reducing fire intensity and limit fire intensity and impacts to soil seed banks (Cheney et al., 2020; Holmes et al., 2020; Kraaij et al., 2018; van Wilgen et al., 2001). For cases where invasions are sparse, and water and biomass benefits would be limited, private sector financing may not be able to be leveraged. However, this does not imply that alien tree clearing cannot be fully funded. In such cases, public sector or donor funds could be leveraged, with early intervention and low-density invasions receiving priority.

We therefore propose a blended finance model, whereby public programmes and conservation agencies prioritise early intervention and low density invasions with high preventative value (and low NPV/CBR), while the private sector or blended-finance models focus on denser invasions where returns on investment are higher and more easily captured (Coldrey et al., 2022). Ultimately, this spatial approach enables limited resources to be allocated more efficiently by identifying where investment is likely to deliver the greatest benefits, across both profitable restoration projects and essential ecological maintenance. Overall, the results highlight the potential for strategic collectives, private farmers, and public agencies to invest in invasive alien tree clearing as a nature-based solution that may also have the potential to unlock funds to support ecological restoration.

Critically, the financial returns identified in this study can only be sustained if follow-up maintenance operations are implemented (Duarte et al., 2023; Turpie et al., 2008). Without it, seedling recruitment and regrowth negates the hydrological gains, and the "water benefits" fail to materialize. As such, the long-term success of these interventions depends not only on the initial clearing effort, but on sustained ecological restoration. This creates an important challenge for ecological restoration finance mechanisms. If landowners are compensated through single-service markets, such as green biofuels, there is a risk that incentives become misaligned with broader restoration objectives. In such cases, financial rewards may be linked to biomass outcomes alone, while the maintenance required to sustain water benefits and restore ecosystems is neglected. This risk can, however, be reduced through the bundling of ecosystem services, packaging multiple environmental benefits (such as harvestable biomass, enhanced water yield, and biodiversity recovery), into an integrated financial mechanism (Dreschler, 2021; Kangas and Ollikainen, 2023). By linking compensation to the delivery of these combined outcomes, bundling ensures that landowners are committed to the necessary long-term follow up clearing and holistic landscape maintenance. Therefore, financial viability should not be interpreted as the sole determinant of action but rather as a tool to guide resource allocation alongside sound ecological management practices (Iftekhar et al., 2016). In addition, it would be important for laws and policies governing restoration programmes to explicitly mandate the inclusion of an appropriate long-term maintenance plan, so that ongoing follow-up work is built into the restoration commitment from the outset and factored into budgets (Baker & Eckerberg, 2013; Richardson et al., 2024).

4.3. Prospects for future research

Future research should prioritise strengthening the linkages between invasive alien tree clearing and realised benefits, particularly by improving the quantification of how reductions in evapotranspiration translate into reliable, valuable water yield (Hanna et al., 2018; Jiao et al., 2025; Rebelo et al., 2025). This should include a better understanding of how these hydrological gains are partitioned between runoff and groundwater recharge, and the extent to which each pathway contributes to realised benefits. Future research should also account for the temporal dynamics of hydrological recovery following clearing (Brown et al., 2005; Filosa et al., 2017). Here, clearing is treated as active restoration towards a target indigenous vegetation cover, rather than an unaided natural recovery. As this cover establishes overtime, evapotranspiration may initially decline further before increasing with vegetation development. There may also be lagged recovery of water supply. Future studies should therefore incorporate these lagged, potentially non-linear changes in evapotranspiration to better quantify the magnitude and timing of water benefits (Fill et al., 2018). In parallel, uncertainty propagation should be expanded to explicitly reflect hydrological seasonality, and the diversity and regionality of water tariffs across different user types (e.g. agricultural, domestic, industrial, etc.) (Crookes et al., 2013; Moncrieff et al., 2021). This would enable financial outcomes to be reported not as single point estimates, but as probability distributions that more realistically reflect real-world decision-making conditions. Ultimately, these components could be synthesized into a decision-support tool that allows land managers to estimate the water benefits of alien tree clearing at their own properties and explore how those benefits could be realised under different end uses, such as irrigation of specific crop types or augmentation of farm reservoir storage. Such an approach would provide a more realistic valuation of water by linking the estimated water gains to their actual use and associated economic value.

Regarding biomass, future studies should incorporate more accurate, locally observed valuation pathways that account for the spatial variability of biomass markets (Pirard and Ngwenya, 2025). Case studies of actual green biofuel harvesting and utilisation would also be valuable for validating whether the biomass yields and associated revenues estimated here are broadly consistent with those under real-world operational conditions. Quantifying the specific transaction costs associated with processing, packaging and storing the biomass products for sale, would also help better capture the true net revenue available (Mugido et al., 2014). Equally important is the rigorous refinement of how clearing and logistical costs are measured. Standardized operational and chemical cost estimates should be systematically ground-truthed against actual expenditure data acquired from private contractors and public clearing programmes that have operated across varied environmental and topographical conditions (Iftekhar et al., 2017; McConnachie et al., 2016). Furthermore, transport cost estimates require substantial improvement to accurately reflect real-world logistical barriers. Future models must transition away from flat-rate distance assumptions by integrating spatial routing variables that account for road infrastructure quality (e.g., paved highways versus degraded dirt tracks), terrain gradients that affect fuel consumption and travel time, specific truck payload capacities, and localized fuel price volatility. To close the gap between modelled potential and real-world performance, future research should also monitor actual restoration programmes to measure realised benefits, as restoration outcomes are still scarcely monitored globally (Lindenmayer, 2020; Nilsson et al., 2016). Such validation is key to building confidence among investors and public agencies. Over time, continued calibration and refinement of both water-use and biomass models, supported by improved spatial and temporal resolution of remote sensing data as well as targeted field measurements will reduce uncertainty and improve transferability across landscapes (Rodríguez-Veiga et al., 2019; Wenger and Olden, 2012).

Finally, there is a strong opportunity to move toward operational decision support. Integrating the spatial layers and cost-benefit outputs into a user-friendly decision-support tool or live application, which could enable policymakers, managers, and funders to explore scenarios interactively and to prioritise interventions more transparently (Forsyth et al., 2012). Finally, future research should also extend this approach to a wider range of ecosystem services arising from invasive alien clearing, to support the bundling of ecosystem services. This could include recreation and ecotourism, erosion prevention, flood regulation, employment creation, grazing improvement, and water quality benefits (Peacock et al., 2023; Stafford and Blignaut, 2017). Assigning monetary values to these services would provide a more holistic assessment of the gains from clearing invasive alien trees

5. Conclusion

This study presents a spatially explicit framework that integrates remotely sensed evapotranspiration, and biomass estimates with other landscape attributes to quantify the long-term Benefit-Cost Ratio (BCR) of invasive alien tree clearing as part of ecological restoration programmes at management relevant scales. Given that BCR may be driven by highly localized conditions rather than uniform physical variables, this spatially explicit approach provides a framework that can unlock private sector funds, as well as guide prioritization of limited public sector funds. Furthermore, the models within this framework can be progressively refined and adapted to account for uncertainty as new data become available. Ultimately, this scalable and updatable approach provides a critical tool to guide restoration financing. By spatially identifying the economic potential of invaded landscapes, the framework demonstrates that in most cases, alien tree clearing can be fully funded through private landowners and/or stakeholders. For the minority of cases where value-added returns do not fully offset operational costs, we propose that limited public sector funds should be strategically prioritized, ensuring a financially sustainable and evidence-based approach for national ecological restoration.

Declarations

Competing interests

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

Author contributions

Liam S. Cogill – Conceptualization, Methodology, Code development and running, Analysis, Writing – Original Draft, Investigation, Data Curation.

Karen J. Esler – Conceptualization, Resources, Writing – Review & Editing, Supervision.

James Blignaut – Conceptualization, Validation, Writing – Review & Editing, Supervision.

Alanna J. Rebelo – Conceptualization, Methodology, Code review, Analysis, Writing – Review & Editing, Supervision, Resources, Funding acquisition

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Data availability

The following repository hosts the code for processing the raw spatial layers to derive final benefit and cost values for a net-present value and benefit-cost ratio analysis for invasive alien plant clearing: https://doi.org/10.5281/zenodo.20629002

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Figures

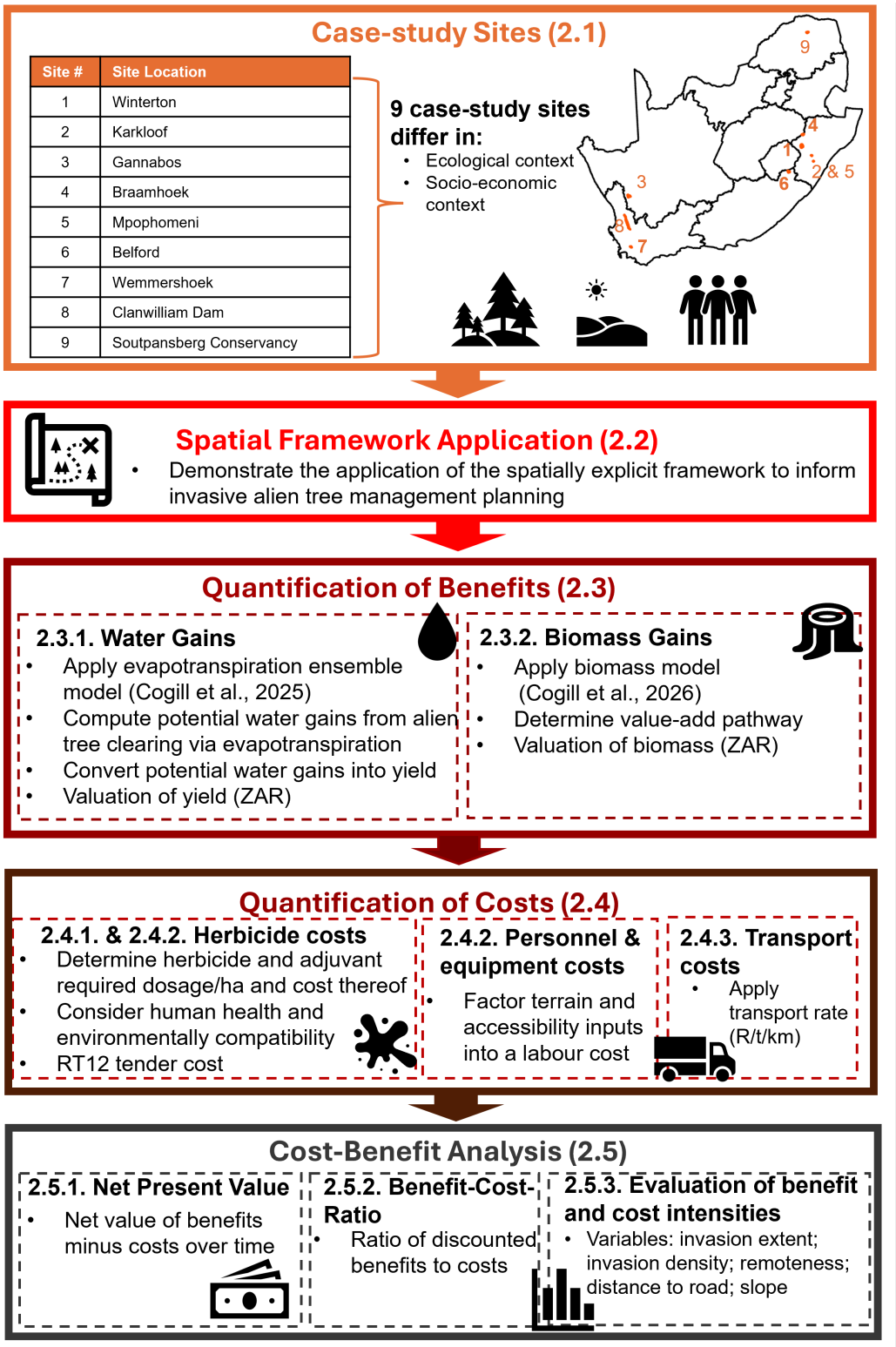


Figure 1

Conceptual framework of the study methodology which includes five parts: (2.1) Outlining the case-study sites; (2.2) Applying the spatial framework; (2.3) Quantifying the benefits of invasive alien tree clearing; (2.4) Quantifying the costs of invasive alien tree clearing; (2.5) Cost-benefit-analysis.

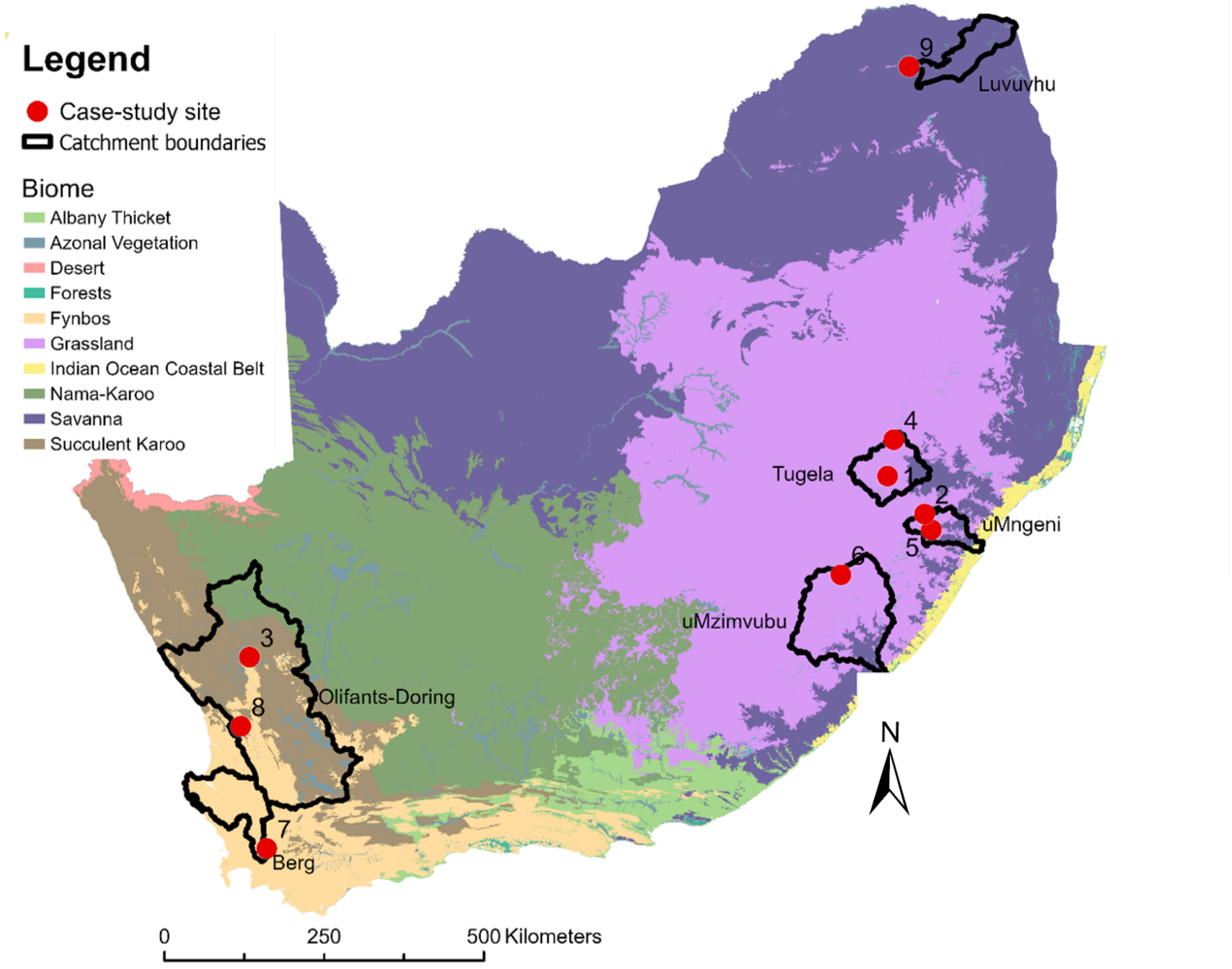


Figure 2

The invasive alien tree case-study sites (1: Winterton; 2: Karkloof; 3: Gannabos; 4: Braamhoek; 5: Mpophomeni; 6: Belford; 7: Wemmershoek; 8: Clanwilliam Dam; 9: Soutpansberg Conservancy) across different catchments and biome types in South Africa (SANBI, 2024). More details of each case-study including the site number, type of invasion, the boundary of each case study, and the beneficiary type are noted in **Table 1.**

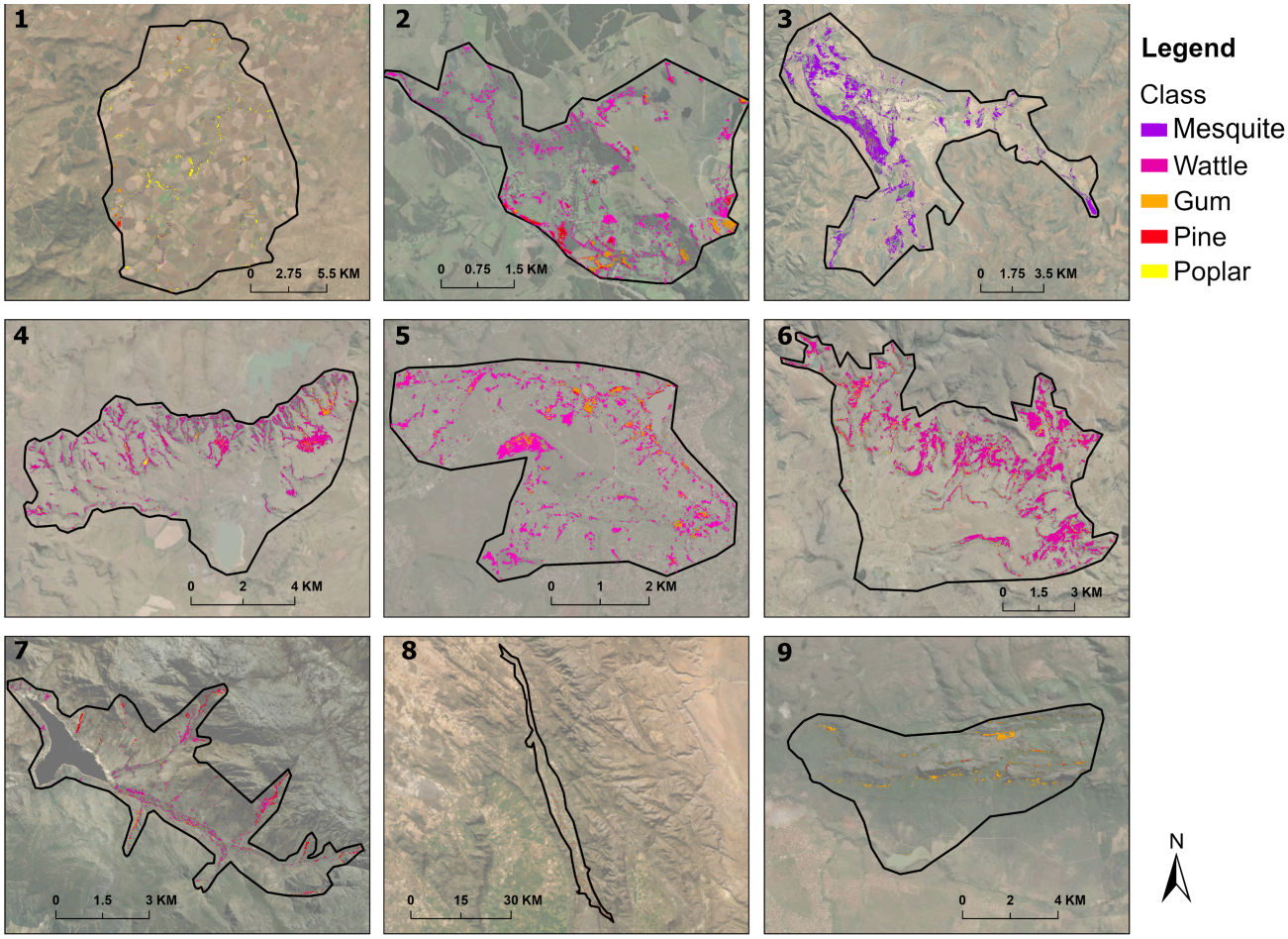


Figure 3

Invasive alien tree distribution across the nine case studies in South Africa (1: Winterton; 2: Karkloof; 3: Gannabos; 4: Braamhoek; 5: Mpophomeni; 6: Belford; 7: Wemmershoek; 8: Clanwilliam Dam; 9: Soutpansberg Conservancy). More details of each case study including the site number, type of invasion, the boundary of each case study, and the beneficiary type are noted in **Table 1**. The classification maps of the invasive alien tree classes and other indigenous vegetation types have been developed and published by the MapWAPS project (Rebelo et al., 2025).

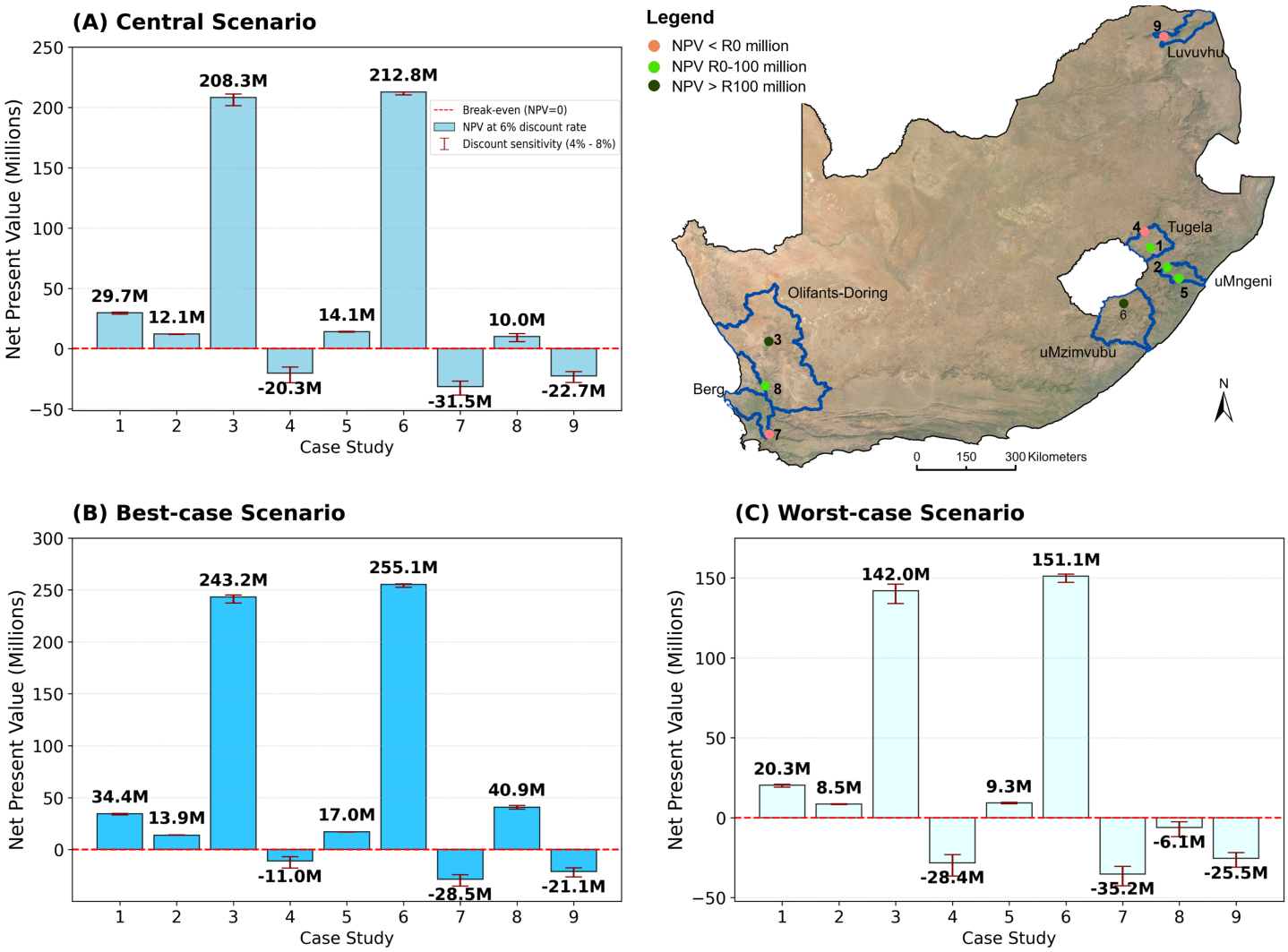


Figure 4

Comparison of the net present values (ZAR) across the nine case studies (1: Winterton; 2: Karkloof; 3: Gannabos; 4: Braamhoek; 5: Mpophomeni; 6: Belford; 7: Wemmershoek; 8: Clanwilliam Dam; 9: Soutpansberg Conservancy) under the (A) central, (B) best-case and (C) worst-case sensitivity scenarios. Whiskers indicate the range of net present values obtained using alternative discount rates of 4% and 8%, illustrating the sensitivity of the results to changes in the discount rate.

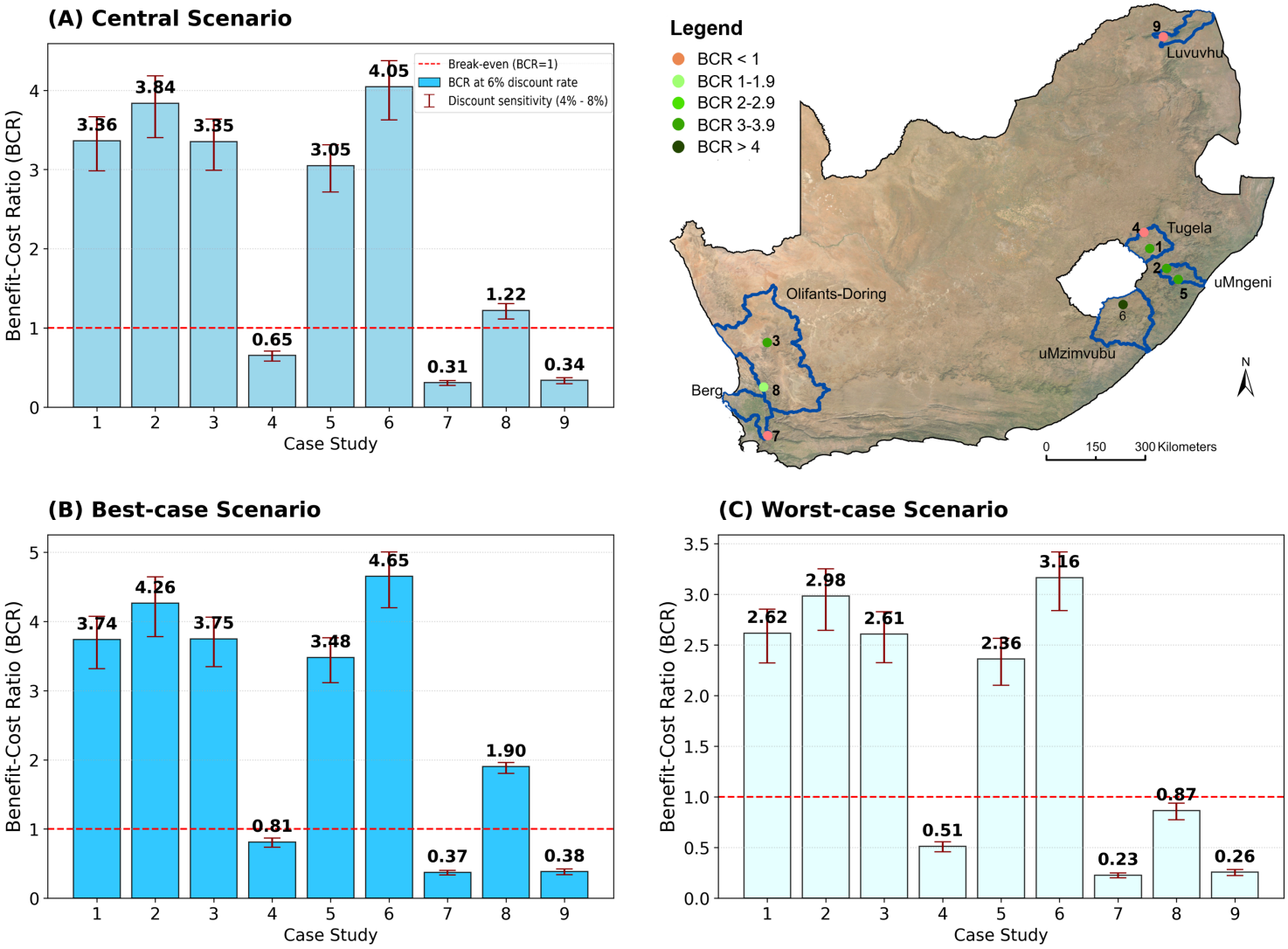


Figure 5

Comparison of the benefit-cost ratios across the nine case studies (1: Winterton; 2: Karkloof; 3: Gannabos; 4: Braamhoek; 5: Mpophomeni; 6: Belford; 7: Wemmershoek; 8: Clanwilliam Dam; 9: Soutpansberg Conservancy) under the (A) central, (B) best-case and (C) worst-case sensitivity scenarios Whiskers indicate the range of benefit-cost ratios (BCRs) obtained using alternative discount rates of 4% and 8%, illustrating the sensitivity of the results to changes in the discount rate. The horizontal dashed red line represents the financial break-even threshold (BCR = 1).

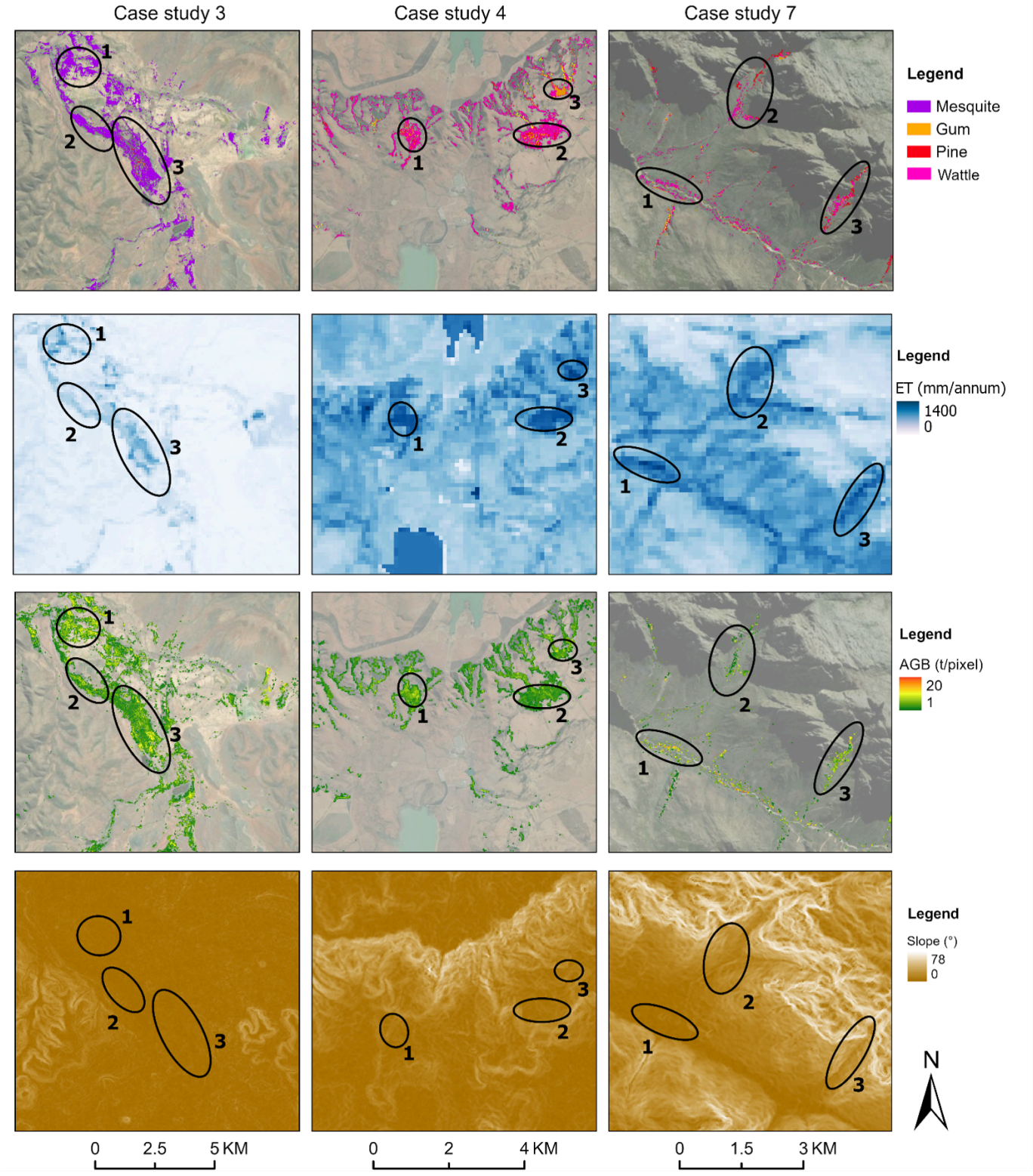


Figure 6

Spatial layers to inform landowner decision-making and cost-benefit analyses for ecological restoration across three distinct operational landscapes (case studies 3, 4, and 7). From top to bottom, the rows display: (1) mapped distribution of dominant invasive alien tree taxa (mesquites, gum, pine, and wattle), with black ellipses highlighting potential "sellable units" targeted for maximum biomass returns; (2) modelled evapotranspiration (ET; mm/annum), (3) Above-Ground Biomass (AGB; t/pixel), and (4) topographical slope (degrees). The

associated Google Earth Engine app links for these case studies can be found in the supplementary material (**Table S1**)

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- SupplementaryMaterial.docx