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CONTRIBUTED PAPER

Comprehensive and representative conservation planning for biodiversity of ice-free Antarctica

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Abstract
Antarctica is designated as a natural reserve devoted to peace and science, with Antarctic Specially Protected Areas (ASPAs) serving as the primary instrument for area-based biodiversity protection. However, the existing ASPA network lacks the comprehensiveness and representativeness required under Annex V, Article 3(2), of the Madrid Protocol. We demonstrated the application of systematic conservation planning to explore the trade-offs underlying protection at a continent-wide scale, focusing on the principles of comprehensiveness and representativeness. These principles are the foundations of designing an effective protected area system that ensures all biodiversity features and their diversity are captured. We investigated how systematic conservation planning applies in the Antarctic and evaluated the impact of alternative species targets and cost measures on performance and area selection. We found that cost assumptions had a greater influence than species target formulation on spatial priorities, and access costs based on proximity to research stations systematically disadvantaged remote species. Furthermore, our results demonstrated that comprehensive and representative protection could be achieved by balancing the placement of protected areas at varying distances from research stations. Finally, stakeholder engagement must occur throughout the process if this approach is to be formally undertaken through the Antarctic Treaty System.

KEYWORDS
comprehensive, continent-wide, protected areas, representative, systematic conservation planning, terrestrial

INTRODUCTION

Protected areas (PAs) play a crucial role in the conservation, maintenance, and rehabilitation of biodiversity and ecosystems and serve as fundamental instruments for conservation (Gaston et al., 2008; Langhammer et al., 2024; Maxwell et al., 2020; Rodrigues et al., 2004). In Antarctica, PAs are a key component of conservation and environmental management activities that protect its unique environments and mitigate human impacts (Shaw et al., 2014). Starting in the late 1960s, PAs were established in terrestrial Antarctica under the Agreed Measures for the Conservation of Antarctic Fauna and Flora (Smith et al., 1994). In 1991, the Protocol on Environmental Protection to the Antarctic Treaty (the Madrid Protocol) designated Antarctica as a "natural reserve, devoted to peace and science" (Antarctic Treaty Secretariat, 1991). The Protocol came into force in 1998 and includes six Annexes, with Annex V implementing Antarctic Specially Protected Areas (ASPAs) for area protection and management. ASPAs are the key instrument for protecting outstanding environmental, scientific, historic, aesthetic, or wilderness values, or any combination thereof, and for supporting ongoing or planned scientific research (Antarctic Treaty Secretariat, 1991).

Antarctica's ice-free areas, which are home to most of its biodiversity, cover approximately 21,745 km^2 (Burton-Johnson et al., 2016), yet ASPAs account for less than 1.5% of this area (Shaw et al., 2014) (Appendix S1). The current distribution of ASPAs is also predominantly concentrated near research stations, with median distances to a research station of less than 37 km (Hughes & Grant, 2017). Furthermore, the rate of ASPA designation slowed in the decade prior to 2017 (Hughes & Grant, 2017) but has since stabilized (Howard-Williams et al., 2021; Secretariat of the Antarctic Treaty, 2023) (Appendix S1). Many terrestrial species (Wauchope et al., 2019), type localities of species (Phillips et al., 2022), and critical biological areas (Howard-Williams et al., 2021) remain underrepresented and insufficiently protected under the current ASPA system (Coetzee et al., 2017; Hughes et al., 2016; Shaw et al., 2014), and Antarctic policy makers have continued to emphasize the need to review and expand PAs in Antarctica (Antarctic Treaty Consultative Meeting, 2019, 2024).

Antarctic ecosystems are under pressure from direct human impacts (Tin et al., 2014), climate change (Lee et al., 2022), pollution (Bargagli, 2008), and invasive species (Chown et al., 2012; Hughes et al., 2025). Given these challenges, the implementation of a systematic environmental and geographic framework for the expansion of the ASPA system, aligned with Article 3(2) of Annex V, is crucial. Such an approach is required to ensure the system's adaptability to environmental changes and cumulative human impacts and promote the design of a future ASPA system that effectively and efficiently conserves biodiversity (Antarctic Treaty Consultative Meeting, 2019; Burrows et al., 2023). To date, this approach has not been implemented for the continent, limiting the evidence base needed to guide future ASPA expansion. Systematic conservation planning (SCP) offers a well-established framework to address this gap.

An SCP framework offers an effective approach for designing and improving PA systems worldwide (Margules & Pressey, 2000; Sarkar et al., 2006) because it provides a transparent, repeatable framework for identifying conservation priorities by considering multiple criteria, including cost-efficient solutions and trade-offs (Margules & Pressey, 2000). The strength of SCP lies in its ability to offer a repeatable, transparent, and equitable process that informs and supports conservation decision-making (Álvarez-Romero et al., 2018; Jung et al., 2025). The SCP framework has been globally adopted by numerous organizations and countries and provides a structured approach to optimizing spatial management for biodiversity conservation in a cost-efficient manner (Pressey & Bottrill, 2009; Schwartz et al., 2018). At its core, SCP follows four key fundamental principles: comprehensiveness, adequacy, representativeness, and efficiency (CARE) (Kukkala & Moilanen, 2013; Margules & Pressey, 2000; Sarkar et al., 2006; Wilson et al., 2009).

In the context of PA systems, the objectives of comprehensiveness and representativeness are to include the full range of biodiversity features (comprehensiveness) and represent the diversity of those features (e.g., phenotypic variation) within PA systems (representativeness) (Kukkala & Moilanen, 2013; Wilson et al., 2009). Adequacy typically concerns setting targets large enough to ensure long-term persistence of features, whereas efficiency relates to achieving those targets with the most economical use of resources (Kukkala & Moilanen, 2013; Wilson et al., 2009). Under Article 2 of the Madrid Protocol on Environmental Protection, the core objective of designing PAs and management plans in Antarctica is the comprehensive protection of the environment and associated ecosystems. However, the CARE principles were not integrated into the identification of existing ASPAs (Coetzee et al., 2017), and a systematic approach incorporating these concepts has not yet been explored at a continental scale in Antarctica (Coetzee et al., 2017; Shaw et al., 2014).

Although there have been calls to adopt a more systematic approach to ASPA expansion, implementing protection can be a slow process due to the unique governance of Antarctica. Under the Antarctic Treaty System, ASPA designation requires consensus among all Treaty Parties (Antarctic Treaty Secretariat, 1991). SCP frameworks have been developed, with key areas identified, to inform marine area protection for the Southern Ocean (Teschke et al., 2021). However, adoption of these Southern Ocean PAs has been slow due to the governance complexities associated with the Convention for Conservation of Antarctic Marine Living Resources (Boothroyd et al., 2023; Brooks et al., 2020). Irrespective of decision-making challenges associated with Antarctic marine and terrestrial PA designations, a necessary precursor to any stakeholder discussion is the development of an evidence base from which to evaluate conservation priorities and trade-offs. Although SCP has recently been applied to Antarctic terrestrial vegetation at the continental scale (SCAR, 2025), no study has evaluated how alternative species targets and cost metrics affect continental spatial priorities across all major taxonomic groups.

We used the most comprehensive biodiversity database currently available (Terauds et al., 2025) to explore whether a more comprehensive and representative ASPA system could be achieved for Antarctica's ice-free areas. Specifically, we assessed the trade-offs in how two planning parameters, species target formulation and distance to research stations (as a cost metric), influence conservation outcomes across the continent. These parameters are central to Antarctic SCP: species range sizes vary widely across taxonomic groups, making target formulation consequential for how biodiversity is represented, whereas access is largely concentrated around research stations, which may introduce spatial bias into access-based costs. We focused on comprehensiveness and representativeness, as they are the most directly aligned with the criteria for area protection under Annex V, Article 3(2), of the Madrid Protocol. Rather than prescribing specific areas for designation, this study offers a transparent, reproducible demonstration of what a comprehensive and representative ASPA system could look like and the different parameters that may influence the outcome.

METHODS

Prioritization problem formulation

We applied the minimum shortfall objective of the reserve selection problem, which aims to identify the set of planning units that minimizes the difference between the targeted level of species protection and the protection amount in the final solution within a specified area-based budget (Arponen et al., 2005; Jung et al., 2021).

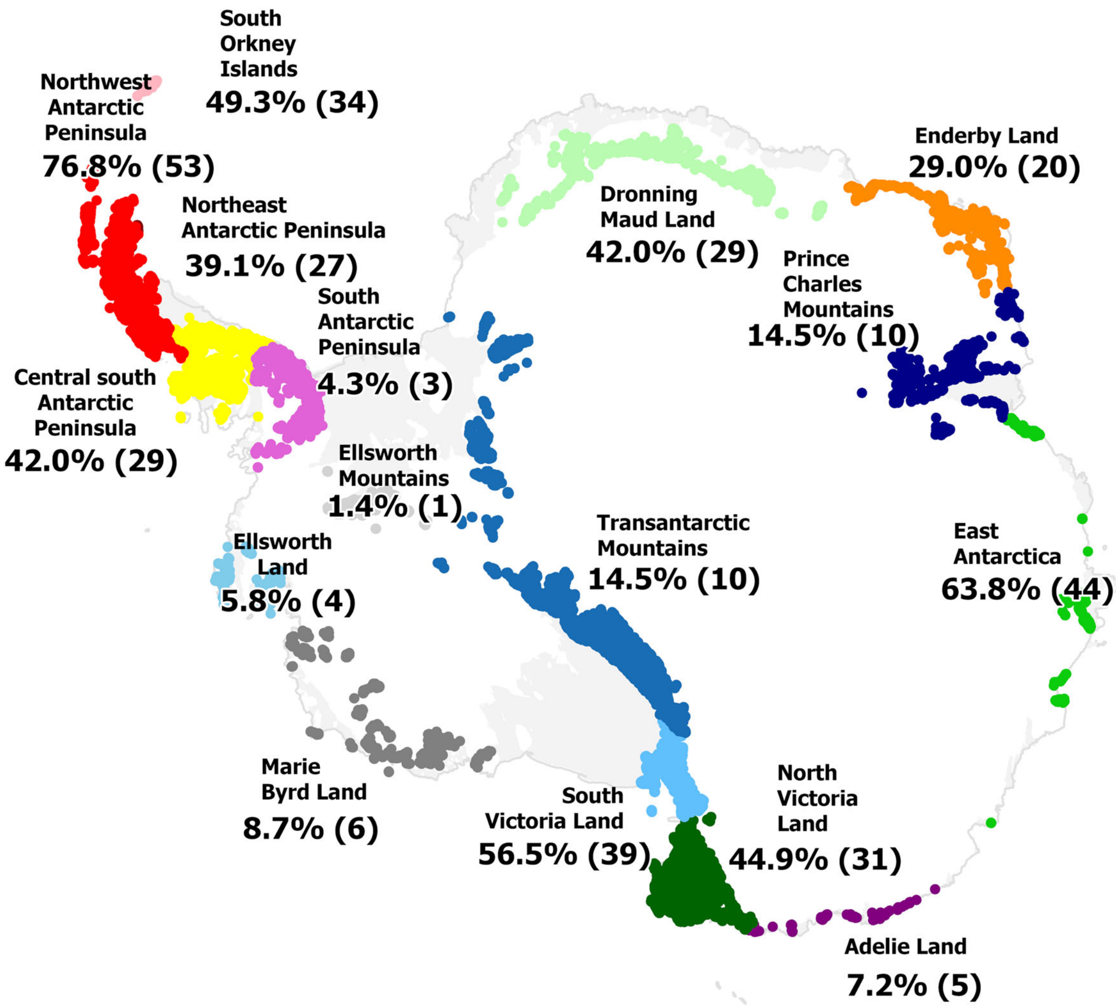

FIGURE 1 Percentage of species groups found in each Antarctic Conservation Biogeographic Region (ACBR) (out of total 69), and the number in brackets indicates the number of species groups recorded within that ACBR (max $n = 69$).

Conservation features and species targets

We used the Biodiversity of Ice-free Antarctica Database (Terauds et al., 2025) to determine species locality points for 69 functional species groups (species groups) of Antarctic terrestrial taxa, as identified previously by experts (Lee et al., 2022). The area of occupancy (area occupied by a species group or its range size) was quantified using an alpha hull using the rangeBuilder R package with an initial alpha value parameter (α) of two (Burgman & Fox, 2003; Chowdhury et al., 2023; Davis Rabosky et al., 2016). Species with fewer than three occurrence records were excluded from the analysis, as the minimum alpha hull requires at least three data points (Burgman & Fox, 2003).

Each species group was subdivided into the 16 Antarctic Conservation Biogeographic Regions (ACBRs), which are distinct biogeographic areas within Antarctica (Terauds & Lee, 2016; Terauds et al., 2012). As not all ACBRs contain each species group (Figure 1), this approach generated a total of 277 subspecies groups (out of maximum 16 ACBRs × 69 species groups). We sought to achieve comprehensiveness (all 69 species groups included) and representativeness (each of the 277 subspecies group captured within at least one ASPA within the ACBR where it occurs).

We assessed how different species targets affected the percentage of targets met using equal costs, and targets ranged from 10% to 100% of each species group, in 10% increments (Appendix S2). We selected 30% as the species target to align with the Kunming–Montreal Global Biodiversity Framework's 30×30 target (Juffe-Bignoli et al., 2016; Watson et al., 2023).

We also assessed the effect of using different species target approaches: 30% for each species group or log-linear targets. A log-linear target form (Jung et al., 2021; Rodrigues et al., 2004) ensured that species groups with smaller range sizes (e.g., 10 km^2) were allocated a target of 100% protection, whereas species groups with more widespread ranges (e.g., 1000 km^2) were allocated a target of 30% of their range size. Targets were scaled using a log-linear function (Rodrigues et al., 2004). Targets of 30% were identified as optimal in the species target comparisons (Appendix S2).

Planning units and existing ASPAs

We divided the study area into 10-km^2 planning units with create grid polygon in QGIS software ($n = 8178$) (QGIS Development Team, 2024). We only included planning units in ice-free areas as these are where most biodiversity and human activities are concentrated (Terauds et al., 2025). We obtained a shapefile of current ASPAs from the Australian Antarctic Data Centre (Terauds et al., 2024). We removed marine areas within the ASPAs because our focus was terrestrial, and we excluded ASPAs designated to protect historic sites and monuments, as these sites were not designated to protect biodiversity. The locations of existing ASPAs were locked in, indicating that they were already considered part of the PA network and thus included in the final solution.

Cost of planning units and research stations

We developed two cost layers for the planning units to test how cost assumptions influenced solutions. First, we generated a cost layer where every planning unit had an equal cost (cost value = 1). This cost metric designated equal costs to both accessible and remote areas without bias toward the location of research stations (Appendix S3). Second, we calculated the access cost of planning units as the distance from each planning unit to the nearest research station. This approach was used as a proxy for the cost of accessing a site (e.g., permit for visitation, management, reporting, and monitoring), where planning units closer to research stations were assigned lower costs than those further away. Access cost metrics are commonly used in SCP to approximate management costs and implementation feasibility, and areas closer to existing infrastructure are typically assigned lower costs (Ban & Klein, 2009; Naidoo et al., 2006). This cost metric was calculated in the R statistical computing environment (R Core Team, 2024) using a linear friction surface based on Euclidean distance across the entire Antarctic continent and then clipped to ice-free areas as in Equation (1):

$$C(x, y) = C_{\min} + (d_e(x, y) \,/\, d_{\max}) \times (C_{\max} - C_{\min}), \quad (1)$$

where C is the friction (cost) value, $C_{\min} = 1$, $C_{\max} = 100$, d_e is the Euclidean distance to nearest research station, and $d_{\max}$ is the maximum distance across the continent. After clipping to ice-free areas, cost values ranged from 1 to 50, where lower values indicate areas closer to research stations and higher values represent more remote locations (Appendix S4). Data on research station locations were obtained from the Council of Managers of National Antarctic Programs (COMNAP) catalog of Antarctic stations (COMNAP, 2017).

Area-based budget threshold

We assessed possible budgets that ranged from 5% to 50% of all planning units, in 5% increments, and compared the cost, percentage of targets met, and total boundary length (Appendix S5). We selected 17% as the area-based budget because our results suggested that 100% of targets were met at 15% with no further benefits by increasing cost (Appendix S5) and because 17% aligns with the Convention on Biological Diversity (CBD) Aichi Target 11.

Spatial prioritization tool

We used the prioritizr R package to identify conservation priority areas (Hanson et al., 2025). This package employs a mixed-integer linear programming (MILP) algorithm to identify optimal solutions (Beyer et al., 2016). We constructed the conservation planning problem in prioritizr using the problem() function. The minimum shortfall objective was implemented via add_min_shortfall_objective() to specify the area-based budget threshold. We used add_relative_targets() or add_loglinear_targets() to set the species group targets. We assigned an equal weight (1) across all species groups and subspecies groups, considering them equally important for biodiversity protection. We selected planning units using add_binary_decisions(), where units were either selected (1) or not selected (0). Optimal solutions were identified using Gurobi solvers (Gurobi Optimization, LLC, 2020) via add_default_solver() with default parameters (gap = 0.1, time limit = none). All analyses were performed in the R statistical computing environment (version 4.5.0) (R Core Team, 2024). The source code is available via a GitHub repository (Appendix S6).

Conservation priority area assessment

We compared the number of planning units selected by each solution, the cost of the solution, the proportion of all species groups that achieved the (30% or log-linear) target, and the total boundary length for evaluating the spatial fragmentation of planning units selected. Zonal statistics were used to conduct this evaluation and implemented in R (R Core Team, 2024), using the terra package (Hijmans, 2024). The proportion of species groups and subspecies groups that met targets was used to assess comprehensiveness and representativeness. A matrix of species group variability in representation across ACBRs was plotted for all solutions to assess the consistency and gaps in representativeness. The results for the selected planning units, which were the most comprehensive (i.e., all 69 species groups must be included within the ASPA system) and representative (i.e., each species group should be captured at least once within an ASPA in each ACBR where it occurs), were plotted as binary decisions for each solution (i.e., targets met or not met). A pairwise correlation test was conducted using the Pearson correlation coefficient (PCC) to assess the consistency of species target achievement between solutions, and a t test was applied to assess the significance of each correlation. Strong correlations indicated that different solutions achieved similar levels of species groups comprehensiveness and representativeness, whereas weaker correlations suggested that differences in target

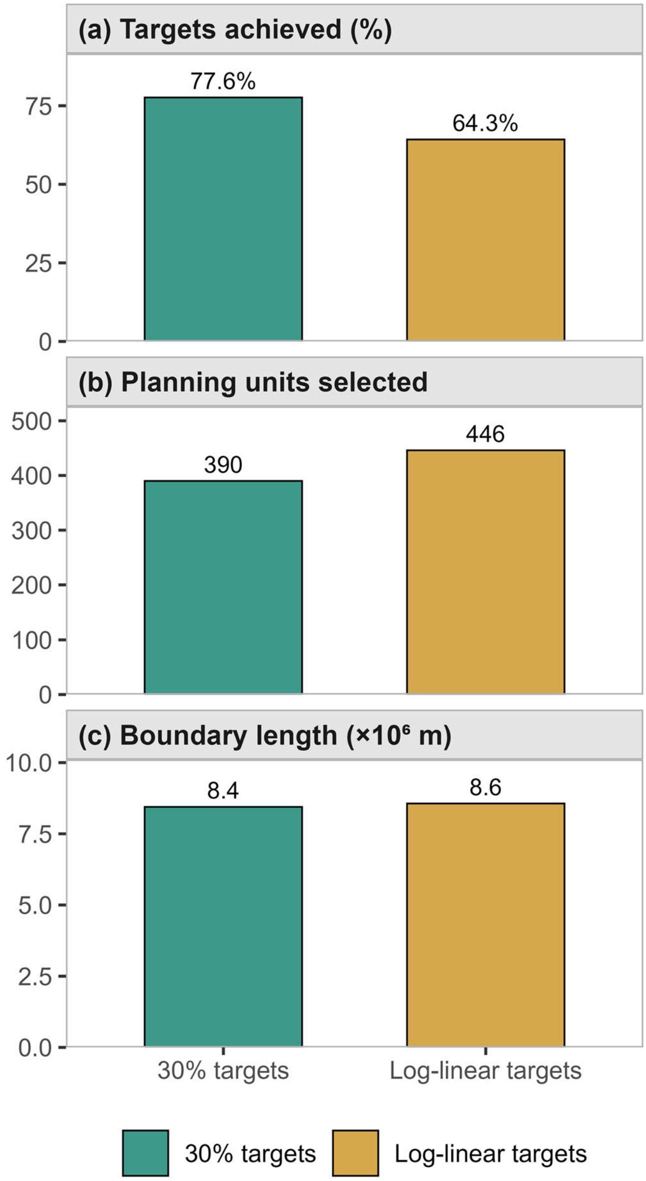

FIGURE 2 Evaluation of performance metrics when two different species targets are applied: (a) percentage of species groups that achieved the target (%), (b) planning units selected, and (c) total boundary length ($\times 10^6$ m).

FIGURE 3 Evaluation of performance metrics when two different cost metrics are applied: (a) species groups that achieved the target (%), (b) planning units selected, and (c) total boundary length ($\times 10^6$ m).

approach or cost metric substantially influenced which species were captured.

RESULTS

Compared with the log-linear targets, the 30% targets for each species group identified planning units that protected more species (Figure 2a) and required fewer planning units (Figure 2b). Regardless of target, species in some remote regions were not captured in the selected planning units when using the access cost metric.

Using the layer of equal costs comprehensively protected all species groups (Figure 3a), including species that were not fully protected under the access cost metric: free-living mites in the Transantarctic Mountains, emperor penguins (*Aptenodytes forsteri*) and xeric moss in Marie Byrd Land, and crustose lichens in Ellsworth Land and the South Antarctic Peninsula (Appendix S7). In contrast, using the access cost metric did not achieve full comprehensiveness or representativeness, failing to meet targets for five species groups and seven subspecies groups. The equal cost metric achieved full representativeness, capturing all 277 subspecies groups. However, compared with the access cost metric, this solution required approximately three times more planning units (Figure 3b) and resulted in greater spatial fragmentation, with the boundary length substantially increased (Figure 3c). Pairwise correlation analysis revealed that using different costs had a greater influence on the spatial configuration of selected planning units than varying species targets given species range sizes (cost metric comparison: $R = 0.74$,

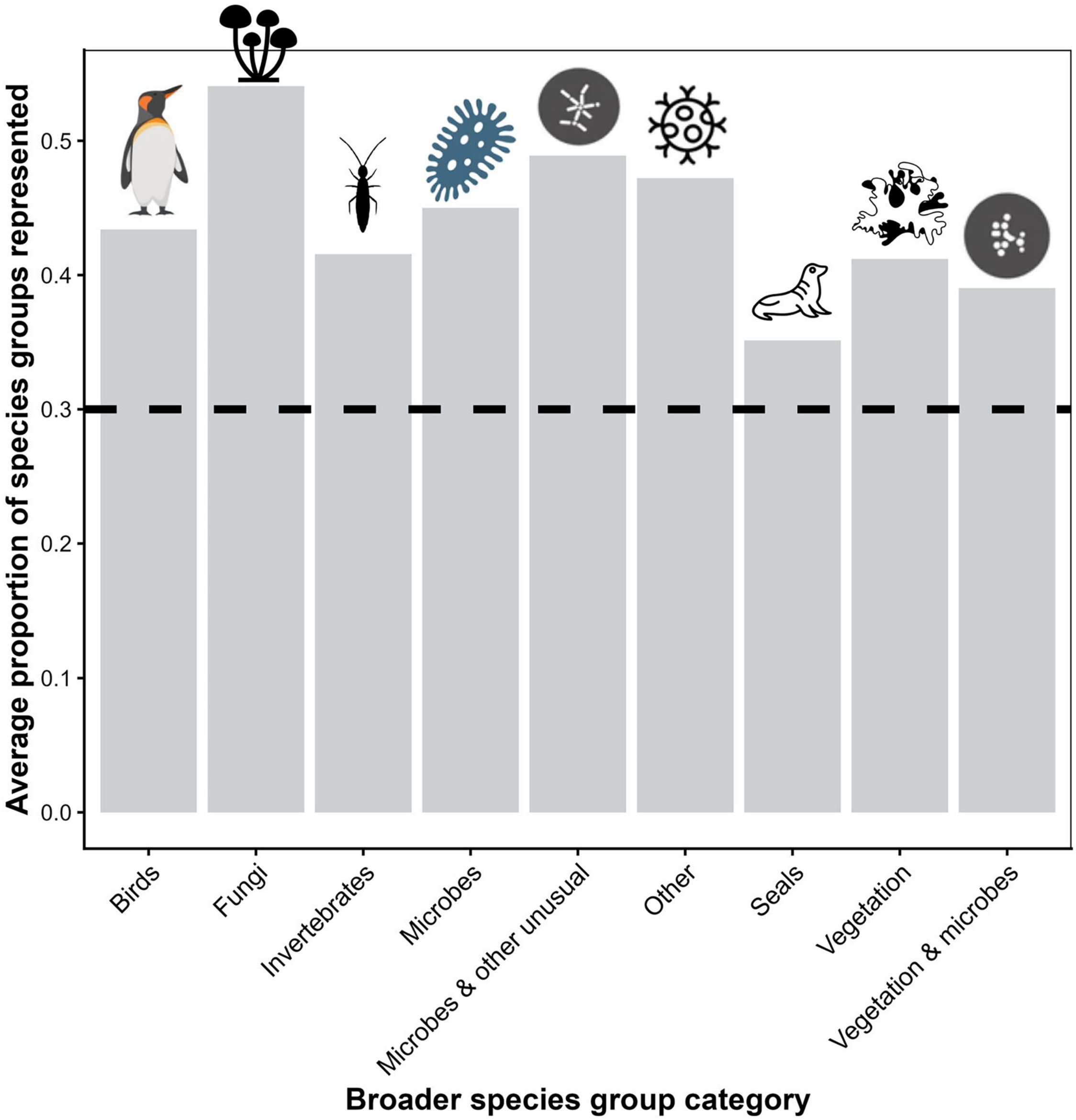

FIGURE 4 Average proportion of broader species group categories represented under the 30% target and equal cost as the most comprehensive and representative solution. The 69 species groups were aggregated into nine broader categories. The dashed line indicates the 30% species target.

$p < 2.2 \times 10^{-16}$; species target comparison: $R = 0.84$, $p < 2.2 \times 10^{-16}$) (Appendix S8).

All broader species group categories met or exceeded the 30% representation target under the equal cost metric, though protection levels varied considerably across groups (Figure 4). Fungi achieved the highest degree of protection, followed by microbes and other unusual categories. Birds, invertebrates, microbes, and vegetation achieved moderate representation, all exceeding the 30% target. In contrast, seals had lower proportions of their range protected, though still meeting the target threshold (30%). These patterns reflected differences in species distribution ranges and sampling intensities across taxonomic groups. A detailed matrix illustrating the comprehensiveness and representativeness of 69 species groups and 277 subspecies groups across 16 ACBRs is provided in Appendix S7.

DISCUSSION

Our study used an SCP framework for biodiversity protection that was both comprehensive and representative at a continent-wide scale in Antarctica. Previous assessments have shown that many species remain underrepresented or inadequately protected in the current ASPA system (e.g., Coetzee et al., 2017; Shaw et al., 2014). Our findings demonstrated that a more comprehensive and representative ASPA network is achievable, with targets met for all 69 species groups and 277 subspecies groups across 16 ACBRs. This finding suggests it is possible to ensure the PA network in ice-free areas of the Antarctic continent reflects the diversity and geographic variation in terrestrial biota. However, achieving full comprehensiveness and representativeness requires careful consideration of trade-offs between species target formulation (Levin et al., 2015) and cost assumptions (Evans et al., 2015).

Compared to the log-linear targets, the 30% targets resulted in more species groups achieving their target in fewer PAs. However, regardless of how the species targets were formulated, using a cost metric that accounted for access (where PAs closer to stations were cheaper) failed to adequately represent priority species groups in poorly represented regions, such as Marie Byrd Land, South Antarctic Peninsula, Ellsworth Land, and parts of the Transantarctic Mountains. These results demonstrated that achieving comprehensiveness and representativeness of coverage across Antarctica's ice-free areas requires expanded protection beyond areas close to research stations, but this approach also would involve trade-offs: more planning units would be needed; spatial fragmentation would increase; and ongoing management of ASPAs in remote locations would be required.

Translating the results of conservation planning analyses into formal protection in Antarctica requires navigating unique governance structures (Hughes et al., 2018; Roura, 2022). Unlike most jurisdictions where SCP is applied, ASPA designation requires proposals be developed by Consultative Parties and adopted by consensus through the Antarctic Treaty Consultative Meeting (ATCM) (Antarctic Treaty Secretariat, 1991). This consensus-based, multi-Party process means that the preanalytical consultation typical in national conservation planning is more complex and protracted. Our analytical approach was designed to support this process by presenting structured analyses that allow decision makers to evaluate some of the trade-offs required when considering comprehensive and representative protection. The sensitivity of conservation priority areas to parameter choices is also informative, revealing which priority areas are consistently selected and which are sensitive to specific target or cost formulations. Several practical obstacles must nonetheless be addressed to achieve comprehensive and representative protection, and these include the logistical demands of developing management plans for remote sites (see Hughes & Grant, 2018), concerns about monitoring capacity within an expanded network, and the tendency for Parties to favor designation near existing research stations, where Parties have operational presence (Hughes & Grant, 2017).

Although achieving comprehensiveness and representativeness is a fundamental goal for systematic biodiversity protection globally, ensuring the adequacy and persistence of all species groups must also be considered, particularly in relation to the minimum viable population sizes required for long-term survival (Kukkala & Moilanen, 2013; Linke et al., 2011; Wilson et al., 2009). Ice-free landscapes that support biodiversity in Antarctica are naturally fragmented and sparse (Lee et al., 2017), which present challenges for designing effective habitat protection. Protecting a single ice-free patch does not necessarily equate to sufficient habitat for certain species groups (Terauds & Lee, 2016), and the size and configuration of PAs can significantly influence species protection. Thus, PAs in Antarctica should not be treated as isolated sites but rather as an interconnected network that incorporates multiple fragmented patches. This network approach can enhance comprehensiveness and representativeness and improve adequacy of the system overall. Although many Antarctic terrestrial invertebrates (e.g., some free-living mites) typically have short-distance movements and are geographically isolated across the continent (Adams et al., 2007; Collins et al., 2019, 2023), certain taxa, such as nematodes and bryophytes, are capable of long-distance wind dispersal (Biersma et al., 2017; Nkem et al., 2006). For species with limited dispersal, a network of PAs is needed to capture isolated populations that a few large PAs might miss. For species capable of long-distance dispersal, a network of PAs can maintain natural connectivity across the landscape (Hawes et al., 2007, 2008; Muñoz et al., 2004). Research on dispersal ecology and species–area relationships of Antarctic terrestrial taxa should inform future applications of SCP for ASPA network design to maximize persistence of biodiversity.

We demonstrated that conservation priority areas should be balanced between those close to research stations and those at remote locations to achieve comprehensive and representative species protection. This approach also supports the need for increased logistical capacity in the future (in terms of budget and technology) to support ASPA designation further from existing research stations (Hughes & Grant, 2018). For example, satellite imagery, drones, and autonomous vehicles could be used for effective monitoring and evaluation in more remote locations (Raniga et al., 2024; Sandino et al., 2023), as these technologies could reduce the need for human visitation and associated disturbance to Antarctic wildlife and terrestrial ecosystems. However, it is important to note that such technologies are not always appropriate for assessing all values designated in the Protocol (i.e., they cannot yet survey specific taxa and their features, such as microfauna abundance).

Currently, there are only two ASPAs that extend 10 km beyond land into the marine environment; there is potential for improved alignment between marine and terrestrial PAs in the future (i.e., an integrated marine–terrestrial PA system). Aligning protection across land and sea would ensure protection of species across their range, such as protection of the nesting and foraging areas of the Adélie penguin (*Pygoscelis adeliae*) where its foraging distance might be up to 100 km from coastlines to open seas (Oosthuizen et al., 2022; Wienecke et al., 2000).

Our analysis was supported by the most comprehensive biodiversity database available to date (Terauds et al., 2025); however, our approach did not account for all values outlined in Annex V of the Environmental Protocol. For example, there are currently no data available on outstanding aesthetic values at a continental scale (Summerson, 2012). Future research could consider incorporating other values to achieve a more inclusive approach to the systematic expansion of PAs, as outlined in the Madrid Protocol (Antarctic Treaty Secretariat, 1991). We also acknowledge that geographic bias in biodiversity data will affect the identification of priority areas for protection. ASPAs are mostly selected near research stations because that is where the biodiversity data are concentrated (Patterson et al., 2025; Terauds et al., 2025), and these data are collected in these areas due to the complexity of conducting scientific research in remote environments of Antarctica, which significantly constrains data collection on terrestrial biodiversity both spatially and temporally (Koerich et al., 2023; Sandino et al., 2023). Nevertheless, ongoing research in Antarctica continues to uncover new terrestrial species (Carapelli et al., 2020), and novel approaches to data uncertainty continue to be developed (Patterson et al., 2025). In the future, we expect these insights will help reduce uncertainty by filling existing knowledge gaps, and in turn, this information can be incorporated into an SCP approach. Our study is one of the first steps toward developing a systematic environmental–geographic framework for the expansion of the ASPA system.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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