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# GREEN INFRASTRUCTURE CONSTRUCTION FOR ADAPTIVE CONSERVATION AND SOCIO-ECOLOGICAL RESILIENCE

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**Abstract**

Protected areas remain crucial for biodiversity protection, but their size and designation status are not sufficient indicators of conservation success. This article examines how spatial prioritization, green infrastructure, ecosystem services, ecological connectivity, uncertainty, and governance can be reconciled in the design of adaptive conservation networks. The review uses a structured critical narrative approach, drawing on the anchor article by Lanzas et al. (2019) and 96 bibliographic records in a single Scopus export. Records were assessed based on thematic relevance and available bibliographic information and abstracts; the procedure is not considered a systematic review or meta-analysis. The synthesis highlights fundamental differences between priority site identification, ecological benefit modeling, policy implementation, and the verification of conservation outcomes. Multiobjective optimization approaches clarify the trade-offs between targets and costs, while modeling connectivity and land-use dynamics exposes the vulnerability of static designs. However, estimates of ecosystem services do not always represent benefits received by society, and increases in spatial indices cannot be treated as evidence of species population change. This article develops an operational framework linking conservation objectives, evidence-based mapping, portfolio selection, implementation agreements, monitoring, and decision revision. This framework represents a conceptual synthesis that still requires empirical evaluation within specific institutional and landscape contexts. Strengthening adaptive conservation, in particular, requires validated ecological indicators, traceable benefit distribution, and iterative learning mechanisms.

***Keywords: Adaptive Conservation; Conservation Network Construction; Green Infrastructure; Ecosystem Services.***

## INTRODUCTION

Protected area planning is often communicated through size and habitat representation. These dimensions are important for illustrating protection commitments, but they do not, by themselves, explain the sustainability of organism flows, the protection of ecological functions, or changes in community benefits. Studies of conservation gaps in upland areas and protected area performance demonstrate why area designation needs to be interpreted in conjunction with the surrounding landscape (Li et al., 2020; Siles et al., 2019).

The anchor article by Lanzas et al. (2019) introduced the issue of designing green infrastructure networks that link protected area conservation with ecosystem functions beyond their administrative boundaries. This idea was then developed through a network approach at the European Union scale, landscape capability mapping, and an inventory of green infrastructure identification methods (Hermoso et al., 2020; Gülçin & Yilmaz, 2020; Honeck et al., 2020). This shift in perspective does not mean that protected areas are becoming irrelevant; rather, they are viewed as nodes in a broader system and are affected by land-use changes.

On the other hand, the term adaptive conservation network risks becoming a normative label if it is not accompanied by applicable decision-making procedures. Priority maps can change depending on target definition, spatial resolution, cost values, or climate scenarios, while their implementation depends on authority, funding, and stakeholder agreement (Tebyanian et al., 2022; Dalla et al., 2026; Losada). This study asks: what do modeling

techniques actually add? At what point should simulation results be interpreted as potentially beneficial? And how are these results translated into verifiable implementation?

The contribution of this paper lies in its synthesis across approaches and the development of decision pathways that differentiate model evidence, intervention implementation, and ecological and social outcomes. The discussion is structured around distinct analytical questions, rather than a repetition of general principles across subsections. The term "construction" is used for the process of designing, agreeing on, implementing, and reviewing conservation networks; it is not intended as a synonym for physical infrastructure development.

## REVIEW METHODS AND LIMITATIONS OF EVIDENCE

### Design, sources, and search boundaries

This study is a structured critical narrative review. The starting materials were a single anchor article, Lanzas et al. (2019), and a set of 96 Scopus export records, dated October 15, 2026, with metadata of title, author, year, DOI, and most abstracts. The TXT and RIS files represent the same export set; they are not counted as separate record groups. The anchor article was examined in manuscript form, while most other publications were read through bibliographic information and abstracts. No comprehensive retrieval of full articles or new, replicable searches of databases were performed during the preparation of this manuscript.

Backward and forward citation searches provide relevant conceptual context, but the available export does not contain the originating markers for each record nor a list of citation relationships sufficient to determine the number of each group. Therefore, the article does not claim that the 96 records represent the results of forward citation searches or the final number of studies in the PRISMA pipeline. Information about the initial Scopus query, the dates of each stage, the total number of records before export, and the reasons for exclusion in the initial stage are not available and cannot be reconstructed speculatively.

### Evidence selection and analysis procedures

Substantive selection focused on studies that explicitly examined at least one of the following elements: protected areas and biodiversity representation; green or blue infrastructure network structures; ecosystem services; connectivity and restoration; systematic conservation planning; land-use change or climate risk; governance, equity, and evaluation. Sources that shared only the term “green” but were not related to landscape conservation were not used as the basis for the argument. A total of 53 articles from exports were selected for citation or directed comparison in this revision; this number represents the number of export sources used in the manuscript, not an estimate of all studies that met the criteria in the Scopus search.

Each source used is classified according to the unit of analysis, method family, type of outcome reported, and inference limitations apparent from the abstract. Findings are then compared thematically across four questions: how conservation targets are set, how spatial relationships and ecosystem services are modeled, how risks and conflicts of interest are considered, and what indicators are needed to assess implementation. Risks of inequity in variable definitions, index weighting, geographic scale, and external validation are treated as part of the synthesis, not eliminated.

### Validity limits and bibliographic audits

A bibliographic check was performed on DOIs, titles, years, and author names as listed in the RIS files. Identical duplications based on DOIs or normalized titles were not observed in the 96 export records. This check could not prove that there was no overlap between the backward and forward citation groups because the identities of the two groups were not yet available. Quantitative claims of the studies were described as results reported in the abstracts, not as effect sizes verified in the full articles. Differences in study designs also precluded a basis for statistically combining the results.

Each record was placed within a single main theme based on its dominant question or contribution, although some studies substantively addressed more than one theme. Themes were structured inductively and directed, referring to conservation planning, green infrastructure networks, ecosystem services, governance, climate adaptation, and methods. Four records were placed in the category outside the core scope during the title and abstract review stage. Therefore, Table 1 below describes the 96 records classified, with 92 potentially substantively relevant; 92 is not a final measure of the verified corpus of citing articles.


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**Table 1.** Grouping of literature records based on seven main themes

| No | Theme groups | Main scope | Number of articles |
|---|---|---|---|
| 1 | Conservation planning and protected areas | Site prioritization; protected area effectiveness; multi-purpose zoning; habitat, biodiversity, and representativeness of protection. | 20 |
| 2 | Green infrastructure network and connectivity | Green network design and optimization; ecological corridors; ecological security patterns; spatial integration of connectivity and ecosystem services. | 25 |
| 3 | Mapping, valuation, and trade-offs of ecosystem services | Mapping the supply and demand of ecosystem services; economic valuation; synergies and trade-offs; land use change. | 22 |
| 4 | Governance, justice and participation | Stakeholder perceptions; participatory design; equitable distribution of benefits; institutional and socio-ecological dimensions. | 8 |
| 5 | Restoration, climate adaptation, and nature-based solutions | Prioritization of restoration; flood and heat reduction; landscape resilience; urban green infrastructure. | 11 |
| 6 | Methods, monitoring and uncertainty | Monitoring and modeling framework; systematic review; uncertainty; spatial methods and indicators. | 6 |
| 7 | Outside the scope of the core theme | Topics include banking, cultural landscape design, software engineering, and water treatment engineering with no clear connection to the review questions. | 4 |
| | TOTAL | All RIS records; each article is counted once. | 96 |

## CONCEPTUAL BASIS: FROM PROTECTION NODES TO FUNCTIONAL NETWORKS

There is a distinction between formal protection, ecological integrity, and network performance. A site can be protected but remain isolated; conversely, a corridor with high connectivity values in a model may not be accessible to target species. Assessments of connectivity's contribution to green infrastructure have shown that the definition of distance, resistance, and landscape elements can alter the interpretation of connectivity (Bolliger & Silbernagel, 2020; Cheng et al., 2020). Therefore, network design needs to express the ecological processes to be maintained, rather than simply displaying connecting lines between nodes.

From an ecosystem services perspective, the ecological capacity of an area does not always equate to the actual benefits received. Mapping ecosystem services in protected areas still faces indicator inconsistencies and validation issues; spatial assessments can be useful for decision-making, but their results must be differentiated from field-based measurements of benefits (Kalinauskas et al., 2024; Xia et al., 2020). This distinction becomes fundamental when conservation priorities compete with land uses that have different beneficiaries and costs.

In this article, social–ecological resilience is not defined as a single aggregate score. Operationally, the term refers to the capacity of an ecological system to maintain function under stress, the social capacity to negotiate and adapt actions, and the relationship between the two over time. Studies of social–ecological–technological system archetypes and green infrastructure planning for climate adaptation provide a way to separate system capacity from the mere presence of physical components (Wicki et al., 2024; Ramyar et al., 2021).

## SPATIAL OPTIMIZATION AND CONSERVATION PRIORITY DETERMINATION

Systematic conservation optimization allows multiple targets to be considered simultaneously under area, budget, or zoning constraints. In protected area management, zone selection can address trade-offs between conservation and human activities; this objective differs from determining a single site with the highest ecological score (Abarca et al., 2022; Yu et al., 2022). Spatial approaches are informative when the objective function, constraints, and cost sources are made public so users can assess why a particular site was chosen.

Hui et al.'s study on the Loess Plateau places ecosystem services within systematic conservation planning, while Ai et al.'s includes carbon vulnerability risks within multi-zone green infrastructure priorities (Hui et al., 2025; Ai et al., 2026). Both speak of goal-based decision-making, but their outcome variables are not identical: the representation


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of ecosystem services, carbon risk management, and land-use suitability cannot be fused into a single indicator of effectiveness. This difference in objectives is useful in demonstrating why a single map cannot address all policy needs.

Cost uncertainty is an issue that is often overlooked in optimization results maps. A review of alternative cost layers confirms that cost definitions—for example, opportunity costs, land acquisition, restoration, or management—will influence the choice of planning unit (Dalla et al., 2026). Therefore, alternative solutions need to be compared across multiple cost assumptions and weights, especially when local fiscal information is incomplete. A review of optimization uncertainty strengthens the rationale for sensitivity analysis but does not prove that any robust solution will be more ecologically successful (Tebyanian et al., 2022).

Beyond single targets, planning can incorporate gaps in biodiversity protection, human services, and distributional equity. Studies on protected area gaps, human-nature alignment, and the integration of conservation and ecosystem service targets demonstrate this spectrum (Xu & Li, 2025; Perschke et al., 2024; Vaz et al., 2021). However, increased target coverage in models should be reported as calculated decision performance, not as direct evidence of population recovery or well-being.

Table 2 compares decision approaches based on the variables that actually need to be examined before the results are transferred to a new area.

**Table 2.** Comparison of optimization approaches and their inference bounds

| Approach | Purpose/unit | Readable output | Source of uncertainty | Ref. |
|---|---|---|---|---|
| Multi-objective zoning | Allocation of protected area zones | Cross-functional target compromise | Goal weight and zoning compliance | (Abarca et al., 2022) |
| Systematic planning | Landscape conservation unit | Priority to meet target representation | Map costs, features, and resolution | (Hui et al., 2025; Yu et al., 2022) |
| Multizone carbon risk | Spatial units of carbon and ecosystem services | Risk sensitive priority map | Carbon models and future risks | (Ai et al., 2026) |
| Ecological resource optimization | Habitat sources at different scales | Source structure and interconnections | Changes in scale and indicators | (Huang et al., 2026) |
| Cost layer selection | Alternative cost per unit proxy | Differences in priorities between scenarios | Adequacy of actual cost data | (Dalla et al., 2026; Tebyanian et al., 2022) |
| Human-nature integration | Biodiversity targets and social benefits | Simulated equilibrium map | Validity of benefit and fairness indicators | (Perschke et al., 2024; Vaz et al., 2021) |

## ECOSYSTEM SERVICES, LANDSCAPE MULTIFUNCTION, AND BENEFICIARIES

Ecosystem service mapping provides a common language for linking ecological functions to public interests. However, a single land cover indicator may represent potential supply rather than benefit flows. Systematic reviews of ecosystem service mapping in protected areas and studies of green infrastructure service provision highlight the diversity of methods and indicators (Kalinauskas et al., 2024; Xia et al., 2020). Therefore, method descriptions should state whether capacity, actual use, beneficiaries, or monetary valuation are being measured.

Specific examples illustrate this limitation. Arnaiz-Schmitz et al. link recreational supply and demand along a rural–urban gradient, while Goyette et al. model wetland protection and restoration by considering the flow of benefits to beneficiaries (Arnaiz-Schmitz et al., 2021; Goyette et al., 2021). A high-value service supply map can shift in priority once beneficiary location and access are included; thus, the "most valuable areas" always depend on the policy question being formulated.

In urban spaces, increasing green cover is not synonymous with equitable distribution of services. Yu et al. incorporate heat mitigation supply–demand matching into the selection of vegetation interventions, and Daniels et al. address pollinator biodiversity in urban green spaces (Yu et al., 2026; Daniels et al., 2020). Both indicators highlight different functions: thermal comfort relates to human exposure, while pollinator diversity relates to biological communities and ecological interactions. Combining them into a single index without explaining the weights would obscure the trade-offs between objectives.


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Payment for ecosystem services schemes and social preferences can introduce an incentive dimension. Assessments of willingness to pay or accept in wetlands and stakeholder perceptions remind us that conservation benefits and costs are not automatically shared (Kim et al., 2023; Losada-Iglesi et al., 2025). It is inappropriate to infer social consensus from the acceptance of spatial models; the representation of affected groups and alternative options need to be assessed separately.

Table 3 differentiates ecosystem service indicators so that study results are not compared as if they were measuring the same output.

**Table 3.** Ecosystem services, indicators, and limits to their use

| Function | Unit/approach | Reported evidence | Interpretation notes | Ref. |
|---|---|---|---|---|
| Mapping in protected areas | Review of ES maps and indicators | Variations in how to map services | Need for model separation and validation | (Kalinauskas et al., 2024) |
| Recreation | Spatial supply–demand | Matching service location and user | Access and preferences are contextual | (Arnaiz-Schmitz et al., 2021) |
| Wetlands | Protection/restoration and service flow | Choice of area for beneficiaries | The benefit stream is model-dependent | (Goyette et al., 2021) |
| Heat mitigation | Supply–demand gap | Needs-oriented vegetation selection | Not a direct measure of health | (Yu et al., 2026) |
| Urban pollinators | Diversity and interaction | Green space habitat function | Cannot replace area index | (Daniels et al., 2020) |
| Economic awards | Payment preferences and schemes | Distribution of value by stakeholder | Economic value is not the same as political feasibility | (Kim et al., 2023) |

## ECOLOGICAL NETWORK CONNECTIVITY AND RESTORATION

Structural connectivity describes the arrangement of landscape elements; functional connectivity relates to the potential movement of organisms or ecological components, influenced by species characteristics and the habitat matrix. This distinction explains why the results of spatial morphology, least-cost path, or circuit analyses should be interpreted as hypotheses about connectivity when species movement data are not available. Studies on connectivity assessment and network optimization in urbanized areas exemplify these approaches, rather than a single standard (Bolliger & Silbernagel, 2020; Sun et al., 2021; Cheng et al., 2020).

Restoration can be targeted at locations that act as barriers to movement or connect fragmented habitats. Ortega et al. identified restoration priorities through landscape connectivity, while Epele et al. prioritized Patagonian wetland areas based on their conservation importance (Ortega et al., 2023; Epele et al., 2021). The objectives of these two approaches differ, so the number of restoration points or the area selected cannot be compared as a measure of method superiority.

Ecological networks can change when urban expansion reduces or relocates corridors. Li et al.'s simulations incorporate conservation priorities into urban growth, while Zhang et al.'s simulations use temporal trade-offs to re-identify green infrastructure networks (Li et al., 2019; Zhang et al., 2025). The conclusion is that single-point analyses risk missing information about changing pressures; model projections cannot predict actual development trajectories without uncertainty.

Connectivity between freshwater habitats has its own logic: hydrological barriers, changes in flow, and water quality require different indicators than terrestrial corridors. Reviews of freshwater systems and studies of the connectivity value of aquatic ecological services serve as a reminder that network procedures should not be mechanically transferred between ecosystems (Erős et al., 2023; Guo et al., 2024).


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## CLIMATE RISK, LAND USE CHANGE, AND DECISIONS ACROSS TIME

Conservation decisions designed with current maps face two sources of shift: external pressures such as urbanization and climate, and changes in internal processes such as forest growth or vegetation regeneration. Analysis of temporal network changes and the implications of land-use change indicates that priorities need to be tested across multiple scenarios (Zhang et al., 2025; Li et al., 2019; Liang et al., 2024).

Carbon assessments demonstrate why time-based indicators matter. Cheng et al. used forest management simulations to compare carbon sequestration capacity across different scenarios; their approach is sensitive to stand age and management choices (Cheng et al., 2024). Ai et al. added carbon vulnerability to spatial prioritization, while Ou et al. placed multi-hazard risks on social- ecological systems (Ai et al ., 2026; Ou et al., 2022). While both address risk, they do not measure the same ultimate impacts; therefore, their results cannot be pooled into a single benefit measure.

In areas with climate-vulnerable species, habitat suitability projections provide clues to locations that may become important in the future, but they depend on climate scenarios and species distribution models. Similarly, urban adaptation strategies cannot be assessed solely on the presence of green infrastructure without considering exposure, maintainability, and land-use change (Ramyar et al., 2021; Xu & Li, 2025).

Resilience analysis needs to distinguish resilience to a single disturbance from the capacity for sustained institutional learning. A spatially robust portfolio may not be feasible if authority boundaries are not aligned with ecological units. These risks should be reported in terms of a range of options, failure conditions, and revision steps, not as promises of optimal solutions for all futures (Tebyanian et al., 2022; Wicki et al., 2024).

## GOVERNANCE, ECOLOGICAL JUSTICE, AND SOCIO-ECOLOGICAL RESILIENCE

Spatial decisions always have distributive consequences: a corridor can increase habitat connectivity while altering land access, livelihoods, or investment priorities. The ecological justice framework for nature-based solutions emphasizes the need to separate benefit distribution, engagement procedures, and recognition of affected groups (Pineda-Pinto et al., 2022). Conservation studies that integrate human and biodiversity benefits also offer a language for formulating indicators that are not solely ecological (Perschke et al., 2024; Martinez-Harms et al., 2021).

Participation needs to occur before alternatives are decided upon, not just during consultations on outcomes. Participatory cause-and-effect diagrams provide a way to uncover stakeholders' assumptions about interventions, while studies of actor perceptions in green infrastructure planning show that policy understandings and priorities can differ across stakeholders (Coletta et al., 2021; Losada-Iglesias et al., 2025). These methods are valuable for increasing the transparency of reasoning, although they cannot replace actual impact evaluation.

Socio-ecological resilience only becomes operational when there are reviewable indicators. At the institutional level, indicators can include budget continuity, conflict resolution procedures, citizen representation, and the implementation of cross-regional agreements; at the ecological level, indicators can include habitat conditions, indicator species responses, and measurable changes in connectivity. The relationship between the two must be tested through an adequate time series, not assumed from a single-period correlation (Wicki et al., 2024; Ou et al., 2022).

Studies of social valuation and incentives show that different beneficiaries can value the same landscape differently (Kim et al., 2023; Losada-Iglesias et al., 2025). Therefore, assessing success requires examining who benefits and who bears the consequences, including after the plan is implemented. Equity is not an additional variable at the end of the modeling process, but rather a condition for the feasibility of the decision.

Table 4 summarizes the forms of evidence regarding implementation, participation, and fairness along with noteworthy limitations on claims.

**Table 4.** Evidence of implementation, governance, and fairness

| Dimensions | Method/object | Contribution to decisions | Limitations | Ref. |
|---|---|---|---|---|
| Involvement | Participatory feedback diagram | Uncovering assumptions and cause-effect relationships | Perceptual models are not evidence of impact | (Coletta et al., 2021) |
| Regional governance | Analysis of GI actors' perceptions | Shows variation in interests | Representation and local context | (Losada-Iglesias et al., 2025) |
| Justice | Nature-based solutions review | Explaining procedures and distribution | Need for measurable distribution indicators | (Pineda-Pinto et al., 2022) |


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| Dimensions | Method/object | Contribution to decisions | Limitations | Ref. |
|---|---|---|---|---|
| Adaptive system | Social–ecological–technological archetypes | Distinguishing between different types of adaptive capacity | Archetypes do not predict specific outcomes. | (Wicki et al., 2024) |
| Community benefits | Human and natural planning | Entering beneficiaries | Assumptions about access and representation | (Perschke et al., 2024; Martinez-Harms et al., 2021) |
| Incentive | PES assessment of wetlands | Describes the variety of economic preferences | WTP/WTA is not proof of compliance | (Kim et al., 2023) |

## OPERATIONAL FRAMEWORK FOR ADAPTIVE CONSERVATION NETWORK CONSTRUCTION

Based on a literature comparison, this article proposes a decision flow framework that integrates elements previously often treated separately. This framework is the author's synthetic contribution, not an empirically tested tool. Its purpose is to help planners link each map and action to a verifiable performance question. The stages are:

a. **Step 1.** Establish the objectives and legitimacy of the prioritized process, assess the services, beneficiaries, jurisdictional boundaries, and equity indicators through a transparent process. Inputs can include conservation targets, area regulations, social data, and available budgets (Abarca et al., 2022; Pineda-Pinto et al., 2022).
b. **Step 2.** Build the evidence base: integrate data on biodiversity, habitat, land use, ecosystem services, societal needs, climate threats, and costs. For each layer, record the date, resolution, assumptions, and quality of the evidence; distinguish direct observations from model predictions (Kalinauskas et al., 2024; Honeck et al., 2020).
c. **Step 3.** Compare portfolio alternatives: run baseline scenarios with no policy changes, biodiversity priorities, ecosystem service orientations, risk priorities, and trade-off options. Test sensitivity to weights, targets, costs, and land use change scenarios; compare solutions not only through aggregate scores but also through the distribution of consequences (Hui et al., 2025; Ai et al., 2026; Tebyanian et al., 2022; Dalla et al., 2026).
d. **Step 4.** Agree on implementation: translate priority units into actions, implementing actors, legal instruments, funding requirements, and grievance mechanisms. Social feasibility and use rights need to be examined before selecting the final restoration corridor or area (Losada-Iglesias et al., 2025; Coletta et al., 2021; Pineda-Pinto et al., 2022).
e. **Step 5.** Measure and review: Record input indicators such as budget and activity locations; outputs such as restoration extent and agreement implementation; intermediate outcomes such as habitat change; and ecological/social impacts such as species responses or equity of access. Use comparison sites or appropriate evaluation designs, if available. If indicators are not on target, update data, renegotiate trade-offs, and revise the portfolio (Kalinauskas et al., 2024; Wicki et al., 2024).




**Figure 1.** Conceptual Framework Flowchart for Adaptive Conservation Network Construction

The framework has three conditions for implementation. First, data sources must be sufficient to distinguish between target characteristics and relevant pressures. Second, those responsible for implementation must have the

mandate and resources to follow through on the options. Third, the program must provide an evaluation period and a mechanism for changing decisions; without the third condition, the term "adaptive" becomes merely a rhetorical description. When any of these conditions are not met, the appropriate output is a policy alternative with limited confidence, not a definitive recommendation.

## EVALUATION OF EFFECTIVENESS AND RESEARCH AGENDA

Effectiveness should be assessed at at least three levels. The first level is design performance, for example, the achievement of target representations or modeled connectivity. The second level is implementation, for example, how many interventions are actually completed or managed as agreed. The third level is impact, for example, longitudinal ecological responses and changes in benefits across community groups. Assessments of protected area potential over time demonstrate the need to differentiate between design quality and ecological integrity (Barbosa et al., 2019).

Future research priorities should not simply expand the composite index, but rather examine the mechanisms at play across the three levels. Validating connectivity with species movements, assessing benefits through measurements at the service recipient level, and testing scenarios against credible benchmarks are areas often underrepresented in this review (Kalinauskas et al., 2024; Bolliger & Silbernagel, 2020; Goyette et al., 2021). Uncertainty reporting should identify the source of error, rather than simply stating that results are “context-dependent.”

Table 5 formulates a set of indicators that can be used as an evaluation protocol, not a list of indicators that are claimed to have been proven superior in all ecosystems.

**Table 5.** Tiered indicators for adaptive conservation evaluation

| Stage | Example indicators | Data source | Evaluation questions | Limits of evidence |
|---|---|---|---|---|
| Planning | Target coverage; selected corridors | GIS, optimization model | Does the design meet the stated objectives? | No evidence of impact on the ground |
| Implementation | Restoration completed; actual costs | Budget audit and location verification | Are the activities in accordance with the decision? | Not yet showing species results |
| Ecological | Habitat conditions; movement; occupancy | Repeated survey/telemetry | Are ecological processes changing? | Need comparison and time period data |
| Social | Access, distribution of benefits, disputes | Survey and complaint records | Benefit to whom, cost to whom? | Risk of reporting and representation bias |
| Adaptation | Revision of decisions that are followed up | Periodic governance documents | Does learning change action? | Revision frequency does not automatically mean quality |

## CONCLUSION

This synthesis demonstrates that a more comprehensive conservation network design cannot be achieved simply by expanding protected areas or incorporating more map layers. Systematic conservation planning is useful for revealing trade-offs between targets and costs; connectivity analysis highlights the relationships between nodes; ecosystem services approaches help clarify beneficiaries; and dynamic scenarios and governance reveal the limits of static designs. This article's contribution is a five-stage decision framework that explicitly separates model performance, intervention implementation, and measurable conservation impacts. This framework needs to be empirically tested and tailored to the ecosystem, institutional capacity, and distribution of rights and benefits. Conservation can be considered adaptive if monitoring evidence is truly used to improve decisions, not simply because plans use dynamic models.

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