Introduction to Net Gain

Katherine Simpson, Emma Cary, Lydia Cole, Roslyn Henry, Xavier Lambin, Philip Stephens, Chris Sutherland, Flurina Wartmann & Jason Matthiopoulos

Drivers of environmental degradation

We live on a human-dominated planet (Vitousek et al. 1997, Foley et al. 2005). The large and growing human population pursues large-scale activities (agriculture, industry, recreation, commercial transport) that result in global alterations to land-use, planetary biogeochemistry and species abundances.

Spatial alteration: Human alteration of terrestrial ecosystems ranges from the most drastic conversions of wilderness into urban infrastructure and development, or intensive crop monocultures and livestock pasture, to less intensive exploitation of space for harvesting (e.g., artisanal hunting, community forestry) or free-range grazing (grassland). In total, about half of the land area on the globe is used for or dominated by human activity (Vitousek et al. 1997). In the marine environment, the relentless trawling of the sea-bed has rendered it a "marine desert", annually covering an area equivalent to perhaps half of the world's continental shelf, or 150 times the land area that is clearcut yearly (Watling and Norse 1998).

Global biogeochemistry changes: Human activity has altered every planetary environmental cycle on which life relies (Steffen et al. 2015, Richardson et al. 2023). On the carbon cycle, emissions of CO_2 through fossil fuel combustion, with the simultaneous degradation of carbon-fixing ecosystems, have accelerated the ambient density of CO_2 from historical levels of 280 ppm, to 362 ppm in the 1990s (Vitousek et al. 1997), all the way to the current values of 420 ppm. The water cycle has also been dramatically altered by human

activity. We use more than 50% of runoff freshwater, mostly for food production, and have altered the landscape around more than 70% of the rivers around the planet (Vitousek et al. 1997). Mineral cycles have been disrupted by mass-fixation of elements such as nitrogen and phosphorus for use in fertilisers (Foley et al. 2005), as well as elements used in high-tech manufacture (e.g., use of rare and toxic metals for batteries or industrial catalysis).

Direct impacts on wildlife abundance: From historical levels of localised hunting and fishing, we have reached global and industrial-scale harvesting of natural populations. Whereas some of the excesses of the past, such as whaling, have been regulated (Gambell 1993), the overfishing of the seas (Sharma et al. 2025), the hunting of terrestrial populations (e.g., bushmeat practices (Ripple et al. 2016)) and the harvesting of pristine forests for timber are ongoing. In addition, many biological invasions can be traced directly to human activity, including: intentional introductions of plants and animals through the horticultural and pet sectors; accidental introductions through the commercial transport routes; and indirect introductions through landscape alterations or changes in climate envelopes. Infectious diseases are a particular type of biological incursion that often has catastrophic impacts on wildlife abundance, usually precipitated directly or indirectly by human activity (Foley et al. 2005). Overall, human and livestock biomass together, are 17 times as large as all the marine and terrestrial biomass of wild mammals (see Fig. 4 in (Greenspoon et al. 2023)) and human biomass movement exceeds the combined biomass movement of all terrestrial wildlife (including invertebrates) by an order of magnitude (Rosenberg et al. 2025).

The environmental crises

These three categories of environmental degradation have led to two global environmental crises:

Climate Change: Planetary temperature has increased rapidly in the last century (Forster et al. 2025). The evidence for anthropogenic climate change (Lynas et al. 2021) and for its grave consequences on life (Parmesan and Yohe 2003) is unambiguous. Multiple, independent lines of evidence (surface temperatures, ocean heat content, cryosphere loss, sea-level rise and changing extremes) show the climate is warming. Physical understanding of greenhouse-gas radiative forcing, plus formal detection and attribution methods using climate models, show with high confidence that the observed warming and related changes are predominantly caused by human emissions of greenhouse gases and other anthropogenic forcings (Intergovernmental Panel On Climate Change (IPCC) 2023) that bring the earth sequentially beyond irreversible tipping points (Lenton et al. 2025).

The key mechanism of global warming is emissions of CO_2 and other greenhouse gases resulting from combustion of fossil fuels (Allen et al. 2009, IPCC 2023). Around 10-20% of anthropogenic emissions of CO_2 can be attributed to land-use change (Vitousek et al. 1997). In addition to large scale climate change, human activity has finer-grained implications for regional and micro-climates, such as those resulting from engineered alterations of water routes (Vitousek et al. 1997, Foley et al. 2005). Direct human harvesting activities—especially deforestation, forest degradation, and large-scale land clearing for agriculture, timber, and fuelwood—are major contributors to climate change. Burning wood not only releases stored CO_2; the associated deforestation reduces the planet's capacity to absorb future emissions (Erb et al. 2018). Loss of canopy also alters surface albedo and evapotranspiration (Betts 2000), which can amplify regional warming and disrupt rainfall patterns, affecting monsoons and downwind agricultural areas. Logging, peatland drainage, and conversion of mangroves or wetlands likewise release methane and nitrous oxide, further enhancing greenhouse forcing. Warming itself can promote positive feedbacks. For example, the release of vast stores of methane that were locked in peatbogs, permafrost and subsea methane hydrates deposits contributes, in turn, to accelerating greenhouse effects (Schuur et al. 2015).

Biodiversity loss: Biodiversity is an umbrella term that captures the intuitive notion of biotic variety. Beyond the term's strict definition and measurement within the ecological literature, the concept has broad implications for a multitude of tangible ecosystem processes. Loss of species is connected to the reduced ability of biological communities to sequester and recycle limiting nutrients, and leads to instability of ecosystem functions and ecosystem productivity (Cardinale et al. 2012). These effects are subject to abrupt tipping points and are

likely to be particularly severe if the species lost come from different trophic levels and represent unique functional traits (Cardinale et al. 2012).

There has been an acceleration in the rate of species loss across the globe (Butchart et al. 2010). Indicators, including population trends, risk of extinction, habitat availability and community composition, show continued overall loss of biodiversity, even though successes are apparent at a range of scales (Sodhi et al. 2011). The causes of these ongoing losses are typically grouped into four main drivers: habitat loss and degradation (including environmental degradation resulting from pollution and climate change); overexploitation; invasive species; and chains of extinction (Diamond et al. 1989).

Changes to landscape (and seascape) use have direct impacts on the viability of species. Land-use change can convert areas suitable for many species into inviable habitat, corresponding to effective habitat loss (Foley et al. 2005). However, even in the absence of net loss, fragmentation of suitable habitats lowers connectivity, dividing formerly viable metapopulations (Hanski and Ovaskainen 2000), or creates extensive edge effects with potentially unfavourable microclimates, and greater exposure to invasion, disease or human disturbance.

The push for greater agricultural productivity, (e.g., through the use of high-nitrogen fertilisers) often results in the reduction of biodiversity. This can happen either by disruption of the competitive and successional relationships between species in plant communities (Tilman 1987), or by triggering biotoxin pollution events through runoff and eutrophication (Jwaideh et al. 2022). Other pollutants are also problematic for biodiversity: mineral (e.g., toxic, inorganic substances) and organic compounds (plastics, pesticides and volatile chlorofluorocarbons) are frequently implicated in the loss of biodiversity across taxonomic kingdoms and biogeographic realms. The effects of pollutants – including greenhouse gases – leading to climate change have already been discussed.

Around a third of the world's harvested fish populations are currently overfished (Sharma et al. 2025) leading to the collapse of many of the world's stocks (Pinsky et al. 2011). Given that fishing has been historically focused on apex species, these declines are causing cascading effects through marine ecosystems (Steneck 2012). Overexploitation is not restricted to the marine realm; despite clear evidence linking unsustainable offtake by humans to extinctions over tens of millennia (Diniz-Filho 2004, Andermann et al. 2020), many terrestrial taxa remain threatened by hunting for food and medicinal use (Ripple et al. 2016).

Invasive non-native species (INNS) have played a significant role in both historical and contemporary extinctions (IPBS 2023). When introduced into ecosystems where they lack natural predators or competitors, INNS rapidly expand, outcompeting native flora and fauna for resources. Invasive predators such as rats, cats, and mongooses have devastated island ecosystems, preying on ground-nesting birds, reptiles, and small mammals (Doherty et al. 2016). Herbivorous INNS can alter vegetation structure, reducing habitat suitability for native species (Asner et al. 2008). In freshwater systems, invasive fish can be hyper-predaceous (Witte et al. 1992) and, along with aquatic plants, can disrupt food webs and oxygen levels, threatening endemic species (Ribaudo et al. 2018). Beyond direct predation and competition, INNS can introduce novel diseases, hybridize with native species, or alter fire regimes and nutrient cycling. The global movement of goods and people has accelerated these introductions (IPBS 2023).

The foregoing drivers of biodiversity loss seldom act in isolation, and abundant evidence links the impacts of two or more types of driver to past and ongoing extinctions (Sage 2020). Moreover, impacts on the abundance of one species, whether or not they lead to its extinction, can destabilise ecological links and alter the viability of habitats for interacting species. Among species with close ecological ties, this can lead to secondary or co-extinctions (the loss of one species in direct response to the loss of another on which it depends)(Colwell et al. 2012). Even in systems with more complex webs of interactions, species loss can lead to cascading impacts on multiple additional elements of the system, with the potential for knock-on chains of extinction (Sanders et al. 2015).

What is Net Gain?

Mitigating climate change and biodiversity loss (Section 2, this chapter) can only happen indirectly, by regulating the activities in section 1 of this chapter. Net Gain, a rapidly evolving concept, has transcended early aspirations, moving towards recognised policy and practical implementation of such interventions. The concept is rooted in the mitigation hierarchy, the principle of No Net Loss (NNL) (Maron et al. 2018), and the practice of biodiversity offsets. These ideas are the natural starting point in exploring what Net Gain means in theory and practice.

The mitigation hierarchy: is a globally recognised framework to help limit the negative impacts of development on biodiversity (Business and Biodiversity Offsets Programme 2012). The hierarchy is implemented through national planning processes and negotiations between government agencies, conservation actors, and developers, with elements of the process often formalised within an Environmental and Impact Assessment (EIA). The hierarchy follows four sequential steps: avoid, minimise, restore, and offset (Figure 1), that together provide a framework for managing biodiversity impacts. The ecological effectiveness of this process depends on early planning, strict sequencing, and transparent monitoring (Arlidge et al. 2018; Needham et al. 2019).

1) **Avoid.** The first and most effective step is to prevent impacts before they occur. This requires biodiversity to be considered from the very start of a project, with measures taken to avoid creating impacts. Examples include choosing not to develop on sensitive areas, changing site layout, or altering project design. Because it stops harm at the source, avoidance is the most certain and reliable form of mitigation (Phalan et al. 2018).
2) **Minimise.** If some impacts cannot be avoided, the next step is to reduce their duration, intensity, or extent. Examples include phased construction activities to avoid the breeding bird season; building wildlife underpasses or overpasses to allow animals to move safely across roads (minimising fragmentation); and implementing water conservation measures in operations to reduce the impact on local water resources.
3) **Restore.** Restoration focusses on the measures taken to repair damage and degradation to a feature within the development site to the state that was present before impacts (Benayas et al. 2009).
4) **Offset.** Offsetting is a last resort, compensating for residual, unavoidable impacts elsewhere. Types include restoration offsets (restoring degraded habitats, but not at the location of the damage) and averted-loss offsets (e.g., avoiding planned deforestation elsewhere – but see (Moilanen and Laitila 2016) for discussion of the risks of damage leakage by relocation of harmful activities).

The term “mitigation” is used in some contexts specifically to refer to avoidance (the reversal of a damaging action), while in others it is used more broadly to mean any action that improves the state of the environment. However, within the framework of the mitigation hierarchy it represents the structured sequence of avoidance, minimisation, restoration, and offsetting. While the mitigation hierarchy provides the foundational logic for Net Gain, the concept has broadened to include voluntary and policy-driven approaches outside formal compliance frameworks. These are explored in later sections.

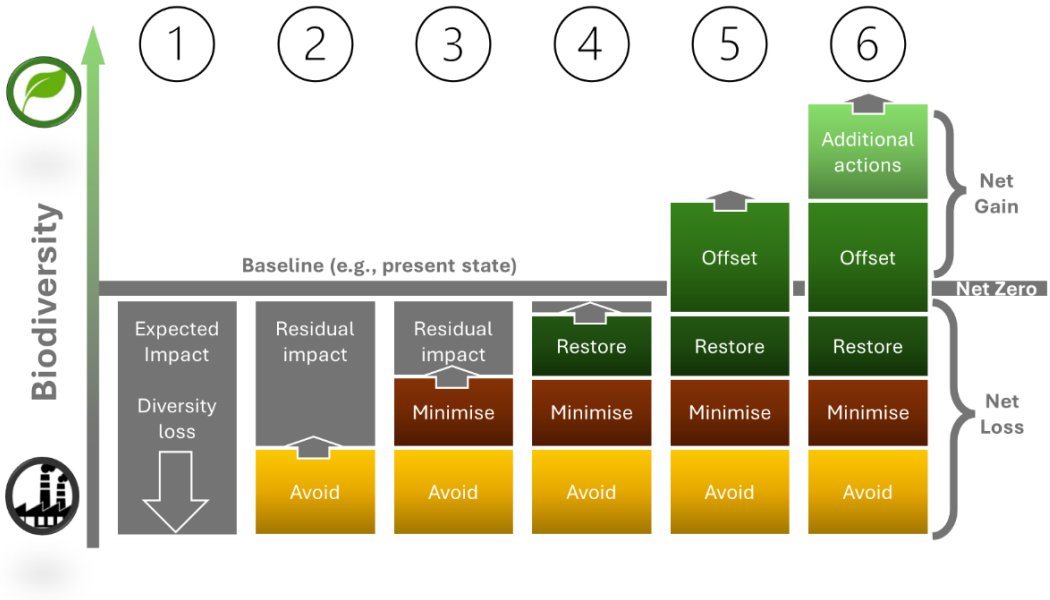


Figure 1: *An example of application of the mitigation hierarchy. The first stage is to determine the negative impact that will be inflicted on an ecosystem by a proposed activity. Avoidance of all or part of that action may reduce the impact (stage 2), and various changes in the implementation of the development may minimise its effect (stage 3). If it is possible, restoration of the habitat can be carried out after the development in-situ (stage 4) but, if not, then offsetting (stage 5) may have a net positive impact overall. In addition to these actions, there may be additional conservation actions that could push the system to higher net gains (stage 6), by improving the biodiversity locally or elsewhere, above its recent historical baseline.*

From no Net Loss to Net Gain: In the context of development projects, the mitigation hierarchy is closely linked to the policy goal of No Net Loss (NNL) which seeks to ensure that biodiversity losses from a project are balanced by equivalent gains elsewhere, relative to a defined baseline (Business and Biodiversity Offsets Programme 2012). Since 2014, there has been a growing shift in policy and practice towards "Net Gain" or "net positive" outcomes (Bull and Brownlie 2017). Under a Net Gain target, restored or recreated habitats must deliver biodiversity benefits that exceed those lost, such that overall gains outweigh losses according to an agreed metric (CIEEM, CIRIA and IEMA 2016, Bull and Brownlie 2017).

To achieve these outcomes, implementation relies on two broad strategies:

- Like-for-like (or 'in kind') compensation, where developers offset losses by overcompensating in the same habitat or biodiversity type.
- Out-of-kind compensation, where additional gains are provided in different biodiversity values (Kiesecker et al. 2010, Bull and Brownlie 2017).

While the overarching goals (NNL or Net Gain) are set by policy or regulatory frameworks, practitioners must implement these goals in specific projects. This requires:

- Establishing metrics and indicators to measure progress (Levrel et al. 2012, Gardner et al. 2013).
- Making explicit assumptions about baselines, counterfactuals (what would have happened without intervention), and timeframes (Maron et al. 2023).

Central to all such efforts is the principle of additionality: only actions that would not otherwise have occurred should count towards Net Gain. The relative merits of like-for-like versus out-of-kind compensation, and how additionality can be demonstrated, are widely debated in conservation science, ecology, and environmental economics. Later chapters will return to this debate in detail, but for now, Needham et al. (2019), Zu Ermgassen et al. (2020), and Maron et al. (2023) provide useful entry points into the literature.

Restoration and Offsetting: Bridging compliance and voluntary action: The transition from NNL to Net Gain is achieved through two closely related mechanisms: restoration and biodiversity offsetting. Both originated within the mitigation hierarchy as means to address the residual impacts of development, yet their application has expanded beyond compliance frameworks into voluntary and other policy-driven initiatives - that is, programmes encouraged or incentivised by government policy but not mandated by law. Restoration generally refers to actions that reinstate or enhance ecosystems affected by human activity, while offsetting involves compensating for residual losses at one site by delivering measurable biodiversity gains elsewhere. Together, they illustrate how practical interventions for Net Gain now span a continuum from regulated project-level requirements to broader, often market- or mission-driven, conservation efforts.

Within the mitigation hierarchy, restoration refers to the reinstatement of biodiversity features directly affected by development, typically on-site and closely tied to compliance obligations. This work usually seeks to return habitats to their pre-impact condition; this might be achieved through soil regrading, hydrological reinstatement, or replanting of native vegetation, for example. Because it occurs at the impact site, restoration is often considered more certain than offsetting, but it is also constrained: ecological recovery beyond the cosmetic can take decades, and restored systems may never fully match their original structure and function. Beyond the mitigation hierarchy, however, restoration is increasingly framed as a proactive conservation and policy goal in its own right. Large-scale initiatives such as the UN Decade on Ecosystem Restoration (2021–2030) seek not only to repair damaged habitats but to create net ecological improvements and social benefits at regional or national scales. This can include habitat creation and enhancement, rewilding, natural approaches to flood risk management and eradication of invasive species from islands (Arlidge et al. 2018, Spatz et al. 2022). In these contexts, restoration is less about compliance with development obligations and more about reversing long-term declines in biodiversity and ecosystem services. It is also important to distinguish between “reinstatement”- the return of a system to a baseline state- and “improvement,” where interventions enhance habitats beyond their prior condition or expand the extent of valuable ecosystems. Restoration therefore operates both as a compliance obligation within development projects and as a voluntary or policy-driven contribution to wider Net Gain goals.

Biodiversity offsetting occupies the final step of the mitigation hierarchy and has been central to the implementation of No Net Loss policies since the 1970s. It seeks to compensate for residual, unavoidable impacts of development by generating measurable biodiversity gains elsewhere, often through ‘habitat banks’ and credit-based systems (Needham et al. 2019). The underlying logic is to align ecological outcomes with economic and regulatory processes: losses at one site are balanced by gains at another, enabling development to proceed while maintaining or enhancing overall biodiversity value.

Historically, biodiversity offsetting can be traced to wetland mitigation banking in the United States under the Clean Water Act (1977), which permitted developers to purchase offset credits from habitat banks to compensate for lost wetlands. Similar approaches were later formalised in Australia under the Environmental Protection and Biodiversity Conservation Act (1999), with state-level schemes such as New South Wales’ BioBanking, and - more recently - its Biodiversity Offsets Scheme. In the United Kingdom, offsetting gained momentum following the Natural Environment White Paper (2011), which initiated government pilots (2012–14) testing habitat-based metrics, spatial multipliers, and habitat banking to support No Net Loss under planning policy. These experiences informed the 25 Year Environment Plan and culminated in the Environment Act (2021) - which, from 2024, made Biodiversity Net Gain (BNG) a mandatory requirement for most new developments in England. Under BNG, developers must deliver at least a 10% Net Gain in biodiversity as calculated by the BNG metric, with flexibility to meet this requirement on-site, through off-site habitat creation or enhancement, or by purchasing offset credits from registered habitat banks managed by landowners.

Collectively, these international trajectories illustrate the formalisation of offsetting within No Net Loss and Net Gain, yet the implementation remains contested. Core debates concern

the ecological equivalence of offset sites compared to those lost, the concept of additionality (ensuring offsets provide benefits that would not have occurred otherwise), and the uncertainty of whether restored or created habitats can replicate the ecological function of natural systems (Bull et al. 2013, Maron et al. 2016, Zu Ermgassen et al. 2021).

Compliance-driven offsets, required as a condition of development consent, must also be distinguished from offset-like voluntary schemes emerging in the corporate and financial sectors. The latter may use similar tools - habitat banking, credits, multipliers - but are often framed as part of ESG (Environmental, Social and Governance) reporting, nature-positive branding, or broader conservation goals. In both forms, offsetting illustrates the tension between aligning development and conservation: proponents highlight the potential for large-scale, coordinated restoration efforts and financial incentives for landowners, while critics caution that offsets risk legitimising biodiversity loss if applied without strict safeguards. These emerging hybrid forms of offsetting blur the boundary between regulatory compliance and voluntary conservation efforts, providing a natural transition to the broader view of Net Gain discussed below.

Multiple Pathways to Net Gain: In its broadest sense, Net Gain means more nature in the future than we have now, implying a reversal of the impacts humanity has already had on ecosystems (Maron et al. 2023). The previous sections explored how restoration and offsetting, initially developed within the mitigation hierarchy, now function across both compliance and voluntary contexts. Here, we take a broader view: while the mitigation hierarchy provides the conceptual and procedural logic underpinning regulatory approaches, the pursuit of Net Gain now extends well beyond these frameworks. Voluntary frameworks (e.g., certification schemes), policy-driven goals (e.g., UN Decade on Ecosystem Restoration (UN 2019)), and market-based initiatives increasingly aim to deliver measurable biodiversity and ecosystem gains independent of formal impact mitigation, motivated by international commitments, corporate responsibility, and emerging nature finance. Here, the terminology shifts: a “restoration action” within a development project is part of a regulated sequence; outside it, restoration may be a conservation goal in its own right. Similarly, “offsetting” in compliance frameworks is distinct from wider habitat recovery programs, though the language and methods can increasingly overlap (Arlidge et al. 2018).

A key difference between voluntary offsets and classical conservation actions lies in their use of quantification and tradeability. Classical conservation, such as protected area management, species recovery programmes, or community-led habitat restoration, has historically sought ecological and social outcomes without translating them into exchangeable units of value. Voluntary offset schemes, by contrast, often seek to apply the same pseudo-quantified metrics and accounting logic as regulatory Net Gain framework with the expectation that biodiversity or carbon gains can be verified and, in some cases, converted into tradeable “nature credits.” This market-oriented framing aims to mobilise private finance for restoration but also inherits many of the same weaknesses and uncertainties, particularly around equivalence, additionality, and long-term outcomes, that affect compliance-based offsetting. Classical conservation, by contrast, has historically focused on ecological and social outcomes without converting success into formal, tradeable units.

The distinction between classical approaches to conservation and the relatively new notion of Net Gain matters because, as Arlidge and colleagues highlight (Arlidge et al. 2018), the mitigation hierarchy has not been widely applied to sectors like agriculture, fisheries, or forestry - where other frameworks dominate. Recognising these sectoral similarities and differences in approach is crucial, since ‘Net Gain’ is not a single framework but a moving boundary concept that encompasses restoration, offsets, and other voluntary conservation actions across fragmented policy arenas ranging from certification, agri-environment schemes and protected areas to TNFD reporting (TNFD 2023), ESG commitments, and philanthropic funding - where terminology and objectives often shift.

These conversations apply equally to biodiversity loss and climate change. Carbon offsets, including voluntary initiatives such as REDD+ (UNFCC 2016), illustrate how ideas of ‘no net loss’ and ‘Net Gain’ are embedded within climate policy. In parallel, ecosystem service

frameworks increasingly emphasise gains in regulating functions such as water quality, soil health, and flood protection, often delivered through agri-environment schemes or large-scale restoration programmes (McVittie and Faccioli 2020). Alongside these, emerging nature markets and corporate reporting frameworks (e.g. (TNFD 2023)) sit next to public investment and philanthropic funding, creating a patchwork of approaches.

Taken together, these developments show that Net Gain is increasingly being applied to the full suite of benefits nature provides, raising questions about how different services are measured, valued, and traded across policy and market contexts.

Key Principles of Net Gain

- "Net" requires accounting for all gains and losses, across spatial and temporal scales.
- "Gain" needs to be defined by weighing short-term improvements against long-term declines, and risks (Damiens et al. 2021)
- Baselines and counterfactuals are crucial: "beyond pre-intervention conditions" only makes sense relative to a clear reference point.

Why now?

Human responses to environmental risks have so far been limited and delayed. Precipitating action in this area is difficult because, historically, we have always prioritised immediate human needs over the capacity of the biosphere to provide goods and services in the long term (Foley et al. 2005) - effectively a form of discounting the future in favour of the present. Beyond such social inertia phenomena, many of the principles laid down for the local and global management of ecosystems rely on the presumption that these systems respond proportionately to incremental human perturbation. However, many earth systems operate non-linearly. For instance, the effects of biodiversity loss may be underestimated when such losses occur locally because these effects grow stronger with time (see extinction debt literature (Figueiredo et al. 2019, Haddou et al. 2022)) and increase disproportionately with spatial scale (Cardinale et al. 2012). Further, many planetary subsystems have tipping points: thresholds that - once exceeded - can pose threats to human viability. Such planetary boundaries are important (Rockström et al. 2009, Steffen et al. 2015) not only because they form the safe operating space for humanity, but also because, once crossed, recovery becomes difficult or impossible. Humanity is currently confronted with serious risks from further inaction.

Inaction on halting and reversing biodiversity loss poses profound societal risks (Pörtner et al. 2023). Biodiversity decline directly threatens food security, water purification, and climate regulation as services provided by the healthy ecosystems on which societies across the globe rely (Hassan et al. 2005, Loreau et al. 2022). When these ecosystem services decline, communities are forced to find costly alternatives that many cannot afford. For instance, where medicinal plants provided free treatment, declining biodiversity may compel individuals to purchase over-the-counter medicines beyond their financial reach.

The defining societal challenge of biodiversity loss is that its impacts fall disproportionately on vulnerable populations. Indigenous communities and the rural poor, who depend most directly on ecosystem services for their livelihoods, nutrition, and cultural practices, face the earliest and harshest consequences. They typically lack the financial resources to substitute natural systems with technological solutions or to relocate when local environments collapse. Marginalised people in urban environments may experience nature loss and a loss of nature-related mental and physical health effects more acutely. Biodiversity loss thus exacerbates existing inequalities, creating an environmental injustice whereby the effects of ecosystem degradation are not equally distributed across society (Armstrong 2024). Without urgent and concerted action, biodiversity loss will deepen social divides and destabilise communities already struggling with poverty and marginalisation.

A combination of scientific insight, policy innovation, and financial pressure has begun to shift the context in which decisions about nature are made. Global ambitions to halt biodiversity loss and take urgent action to combat climate change, reflected in the

Sustainable Development Goals and the UN Decade of Restoration are increasingly shaping national policy agendas such as the UK Environment Act 2021 and UK Net Zero Strategy. International agreements now recognise planetary boundaries as real constraints on economic activity, and businesses are under growing pressure to disclose and reduce their impacts. The adoption of the Kunming–Montreal Global Biodiversity Framework (GBF) in 2022 is considered a landmark international agreement aiming to halt and reverse nature loss. Under Target 15, companies including transnational corporations and financial institutions are required to assess, disclose and reduce biodiversity-related impacts and risks. This is supported by the TNFD, that provides a global framework, developed with input from markets, science, and governments, that enables companies and financial institutions to integrate nature considerations into their decision-making (TNFD 2023). By mid-2025, more than 1,800 organisations had joined the TNFD Forum and of these, 620 institutions, collectively responsible for over £14 trillion in assets under management, had pledged to adopt TNFD-aligned reporting (TNFD 2025). Early adoption has been dominated by financial and corporate actors in the Global North, reflecting both the concentration of reporting capacity and the Western origins of many disclosure frameworks. At the same time, investors and regulators are converging on frameworks that reward nature-positive outcomes. For example, the ENACT (Enhancing Nature-based Solutions for an Accelerated Climate Transformation) initiative, launched at COP27, calls for the mobilisation of private finance to support action on nature and climate goals accompanied by robust environmental and social safeguards (ENACT 2022). Governments are also advancing the development of high integrity frameworks to support the development of markets for biodiversity, carbon and ecosystem services (collectively termed 'nature markets') enabling investment in standardised units of environmental gain. Much of this momentum is channelled into landscape scale conservation and restoration, recognising that collaborative action across large scales can deliver greater ecological connectivity and wider benefits for people and nature (Donaldson et al. 2017). Taken together, these developments suggest that the long-standing inertia in responding to biodiversity loss may finally be giving way to a period of accelerated action, with Net Gain emerging as a practical and supported vehicle for progress.

The need for a holistic approach

Complex challenges such as Net Gain are contingent on the interests of many countries, different human activities and various academic disciplines. They therefore require holism in all three respects.

A multinational approach: Although the twin environmental crises of climate change and biodiversity affect all countries, Multilateral Environmental Agreements (MEAs) are notoriously difficult to achieve and monitor. Goodwill towards international collaboration will always be tempered by inter-nation competition. This is a global-scale version of the tragedy of the commons (Hardin 1968), often evaded by legislative frameworks where multiple actors agree to refrain from maximising pursuit of self-interest. However, at the international stage, different countries or alliances may operate under diverse legal frameworks, making alignment difficult. At the same time as pushing resolution of global problems with high-level summits and agreements, our traditional perceptions about how to manage these crises as a community may need to be challenged. Instead of relying solely on a central authority that enforces an internationally agreed treaty (theory of collective action) we may emphasise more the role of self-organised polycentric systems (Ostrom 2010). Understanding the natural emergence of such systems (e.g., the emergence of groundwater governance (Langridge and Ansell 2024)), may allow us to facilitate their operation with associated benefits of increasing social awareness and stakeholder responsibility.

A multisector approach: Dealing with the twin environmental crises requires us to navigate multiple trade-offs. First, we need to reconcile the priorities of economic growth with sustainability. Second, we need to trade-off the need for sustainability with solutions to many other often interlinked but nevertheless competing crises, such as global health, refugee and population movements, human rights, and living standards of LMICs (Kates and Parris 2003). Finally, environmental priorities are not independent. For instance, some technological solutions for alleviating CO_2 emissions may come at a cost to biodiversity. A particular example of these paradoxical conflicts is the development of marine renewable installations,

which are often blocked by real or suspected costs to wildlife (Virah-Sawmy and Sturmberg 2025).

A multidisciplinary approach: The level of technical depth and expertise required of academics to allow them to operate competitively at the forefront of their field often leads to 'siloing' (the isolation of disciplines from each other), and the alienation of scientific fields from the societal and political processes that they are ultimately intended to serve (Kates et al. 2001). Single-discipline interventions will unavoidably fall into the trap of optimising siloed priorities (e.g., ecological conservation) at the expense of competing concerns (e.g., human livelihoods). This prompts the need for a multidisciplinary approach, a forum within which different specialists can exchange concepts and synchronise their terminology. Going well beyond this are the concepts of interdisciplinarity (i.e., multidisciplinary approaches where methodological convergence happens in-tandem with conceptual exchange) and transdisciplinarity (i.e., interdisciplinary approaches that involve other, non-academic sectors of society, such as government or industry). The need for a transdisciplinary approach has led to the founding of sustainability science, drawing from various strands of literature on collaborative research approaches as well as on practical experience (Spangenberg 2011, Lang et al. 2012). The ingredients of a successful transdisciplinary approach involve gathering a collaborative research team, creating a joint understanding of the problem to be addressed and the concomitant research objectives, and co-designing the methodological frameworks for reaching solutions. When solutions are worked on, or reached, a key challenge is to balance scientific credibility with practical applicability, a task often conducted under strict limits of time and budget. The eBook approaches Net Gain from the perspective of five nominal academic fields: ecology, economics and social sciences, law, and engineering. We introduce the key action levers and challenges within each discipline below but will revisit those in more detail in the following five sections of the book (Part I: Primers).

Ecological and environmental aspects of Net Gain

Ecological and environmental interventions are most proximate to the causes of biodiversity loss and climate change. They are therefore most likely to attract public attention as the "actions on the ground". For an organised approach, we can align them with the three broad causes of the environmental crises discussed above: Spatial alteration, biogeochemical change and direct interventions on biological abundance and biodiversity. There are however additional dimensions to characterising the tools of ecological and environmental intervention. One useful distinction is between intervention and compensation. Interventions may be targeted directly at the causes of the problem and involve either a cessation or reversal of damaging activities. For instance, ecological conservation, in its strictest form, attempts to halt anthropogenic disturbance in pristine ecosystems. Similarly, a strict interpretation of restoration (Ockendon et al. 2018) aims to return ecosystems to some known or hypothetical pristine ancestral baseline. However, we have the option to instigate compensatory actions, that counteract the undesirable effects (of species loss or climate change) by acting at points other than the direct causes of the problems. For instance, CO_2 increases might be compensated by planting forests, instead of cutting emissions. This forms the basis for offsetting (Bull et al. 2013), the idea that a damaging but necessary action can be compensated by an active improvement at some other point of the system. Offsetting prompts questions about the spatio-temporal dimensions of ecological interventions, describing whether compensation happens close to the place and time of the damaging action and, more broadly, the capability of an offset to compensate in a like-for-like manner (Pope et al. 2021).

Ecological interventions via spatial management usually begin by prioritising units of space according to their ecological value (Moilanen et al. 2008), e.g., by some measure of biodiversity, and attempting to reconcile protection of high priority spatial units with the constraints or conflicting priorities of human space use. As spatial prioritisation frameworks have become more ecologically sophisticated, they recognise that spatial units cannot usefully be treated independently, but should rather be considered from the perspective of fragmentation and connectivity (Fahrig 2017) as well as from the perspective of spatial complementarity where the total ecological value of the preserved units of space is considered (Faith and Walker 2002). This question of spatial mosaics prompts the question

of land sharing and sparing (Balmford et al. 2005, Phalan et al. 2011, Fischer et al. 2014): these describe a spectrum of possibility, from biodiversity-friendly, low-intensity farming, where agricultural land also provides good biodiversity habitat (sharing), to the extreme maximisation of yields through intensive farming, enabling land to be set aside purely for biodiversity (sparing).

Ecological interventions via biogeochemical regulation can help ameliorate climate change and prevent biodiversity loss. For the carbon cycle, strategies like wetland restoration, especially rewetting riparian zones, can reduce net CO_2 emissions by ~39% in the short term, increase surface organic carbon and nitrogen stores, and improve soil moisture retention—key for both climate regulation and habitat restoration (Schuster et al. 2025). Interventions in the nitrogen cycle, such as reducing or optimizing fertiliser use, can increase soil bacterial diversity and stabilize microbial networks. Moreover, high nitrogen inputs reduce diversity of nitrogen-fixing plants, shifting plant communities and ecosystem function. In the phosphorus cycle, recycling phosphorus from wastewater, manure or human waste helps close the loop and reduce eutrophication, while interventions like riparian buffer zones, constructed wetlands and sediment treatments mitigate runoff and internal phosphorus loading in aquatic systems.

Ecological interventions via direct manipulations of species abundances include additions such as planting trees for CO_2 sequestration, direct species reintroductions or abundance augmentation and reductions, such as invasive non-native species eradication or control (Pepin et al. 2020). More subtle, indirect interventions on abundance are possible. For instance, we may implement mechanical interventions that directly benefit the demography of threatened species, such as the construction of artificial breeding habitat (Burke et al. 2022) or the avoidance of bycatch and collision risk (Bruinderink and Hazebroek 1996, Lewison et al. 2014, Furness et al. 2018). A more contemporary approach of rewilding (Corlett 2016, Mutillod et al. 2024, Faure et al. 2024) attempts to restore, not the exact ancestral composition of ecological communities, but rather the levels of biodiversity and community function.

There are several challenges to delivering Net Gain from an ecological perspective. The first relates to our scientific ambition and the ease with which we can communicate to non-scientists. In thinking about Net Gain targets, it is important to avoid the narrowest definitions of biodiversity, as a simple one-dimensional measure of environmental health (very much like IQ is not an adequate measure of human intelligence and GDP is not an adequate measure of quality of life). Although the ease of use of such one-dimensional measures makes them tempting to fixate upon, to help us promote the green agenda in public and political fora, the links between biodiversity measures and ecosystem function, stability and health are not easy to establish (Hooper et al. 2005, Tilman et al. 2014, Gonzalez et al. 2020, Wright et al. 2021, Yan et al. 2023).

It is perhaps preferable to use biodiversity as a banner for a broader systems-based approach to ecosystem resilience and recovery. By acknowledging ecological complexity and feedbacks, arising through the dynamical effects of spatial and trophic connectivity (Cromsigt et al. 2018, Perino et al. 2019), we can exploit systems dynamics to amplify the positive effects of modest interventions (a form of ecological “nudging”) potentially even leading to win-win scenarios – where positive effects are achieved at no net monetary cost to human activities (Clough et al. 2011, Jarrett et al. 2025). Failing to adopt such approaches to ecosystem-level management risks turning these potential wins, into losses, for example by allowing short-termist, highly localised, or species-focused interventions to backfire through the indirect effects of spatial or trophic connectivity.

The second challenge is more practical, relating to our ability to monitor negative or positive impacts and quantify success. By definition, monitoring looks at change over time, and therefore requires multiple surveys to be conducted using similar protocols (Moilanen et al. 2024). The gold standard is to have a scheme that includes before-during-after data, ideally with baseline data extending far in the past, follow-up data that extend far enough in the future to capture lagged responses, and control sites to exclude confounding effects. This is known as a BACI design (Before-After, Control-Impact), extended to BACI-R if replication is possible (Moilanen et al. 2024). Under time pressure, and often under pressure from financial

interests, these ideal protocols are difficult to implement, and compliance may become an issue (Bull et al. 2013). In cases where replication and controls are not possible, we are forced to deal with formulating credible counterfactual scenarios – what would have happened if the mitigation action had not been taken (Pope et al. 2021).

Even under ideal monitoring schemes, measuring outcomes and gauging success present separate difficulties (Ruiz-Jaen and Mitchell Aide 2005, Bull et al. 2013). The plethora of biodiversity metrics (Pavoine and Bonsall 2011, McGill et al. 2015) makes it difficult to settle on a single loss-gain currency, prompting the idea of compound measures. It is difficult to define equivalence (Bull et al. 2013) between biodiversity lost and gained (e.g., what is the exchange rate between different endangered species? How many tigers can we lose for every panda gained?). Even if impact and mitigation are currently equivalent at the time of implementation, there is no guarantee that the mitigation effects will last, or that the impacts are not going to worsen due to time lags and extinction debts (Bull et al. 2013, Haddou et al. 2022, Bradfer-Lawrence et al. 2024). Finally, it is quite possible to work towards one Net Gain objective while damaging another. A notable example are reforestation projects, often implemented with fast growing exotic trees that can store carbon but are broadly detrimental to biodiversity (Heilmayr et al. 2020), without always improving carbon sequestration.

Separate mention must be made for problems associated uniquely with the marine environment. Marine ecosystems differ fundamentally from terrestrial ones (Niner et al. 2017). High-resolution data, crucial for assessing baseline conditions and subsequent improvements, are typically less available in marine settings compared to terrestrial ones. The dynamic nature of marine ecosystems, characterised by undefined boundaries and mobile species populations, poses difficulties in establishing permanent or long-term restoration and conservation sites for credit generation. These features complicate the quantification of impacts and mitigation, as well as long-term verification and compliance. The human dimension is equally critical (Voyer et al. 2015). Whilst there is generally broad public support for marine conservation, coastal communities are directly dependent on marine resources for livelihoods and identity, yet do not own or have tenure over the resources. The introduction of financial mechanisms that alter access to or use of those resources must therefore be carefully co-designed to ensure legitimacy and social acceptability. Without this, the risk is that poorly adapted market and legal structures could alienate communities or misallocate resources.

Economic aspects of Net Gain

Economics provides a framework for analysing how human choices over land, labour, and capital affect, and are affected by, the natural environment. Economists study how interactions among firms, consumers, governments, and communities shape the use of scarce resources and determine whether environmental assets are depleted, preserved, or restored. Central to this analysis is the concept of property rights and responsibilities, which define who can use a resource, under what conditions, and with what level of accountability. Well-defined rights create incentives for efficient and sustainable use, but many environmental goods, like the atmosphere or migratory species cannot easily be protected by property rights.

The absence of clear property rights and responsibilities often leads to market failures, where markets alone do not produce socially or ecologically efficient outcomes. These failures take several forms, including externalities, public goods and open access. Negative externalities, such as carbon emissions, impose unpriced costs on others, while positive externalities, like habitat creation, are often undervalued and uncredited. Public goods, such as climate stability or biodiversity, benefit everyone and cannot be restricted to particular users, while open-access resources can be overused, as illustrated by the “Tragedy of the Commons” (Hardin 1968).

A central concern of environmental economics is how to correct these failures. Market-based instruments, such as taxes, tradable permits, and payments for ecosystem services, can align private behaviour with public interests and the objectives of Net Gain. Governments play a crucial enabling role by establishing, monitoring, and enforcing market rules without

directly setting prices (Hanley et al. 2019). We briefly look at some of these regulatory devices in turn.

Pigouvian Taxes: First proposed by economist Arthur Pigou in the 1920s, a Pigouvian tax adds a charge to the market price of a good to reflect the social cost of its negative externalities (Hanley et al. 2019). By increasing the cost of harmful activities, the tax provides a financial incentive for firms and consumers to reduce pollution or switch to less damaging alternatives. A practical example is the UK Climate Change Levy (2001), a tax on commercial electricity and gas use designed to encourage energy efficiency and reduce greenhouse gas emissions across industry, services, and agriculture.

Tradeable Pollution Permits: An alternative to fixed taxation is to set a cap on total emissions or environmental degradation and allow rights to be traded. This system effectively assigns property rights to previously unpriced environmental goods, such as clean air or water (Hanley et al. 2019). Successful tradable pollution permit markets i.e., those that reduce pollution, combine clear caps, rigorous monitoring, and enforcement (Stavins 1998, Schmalensee and Stavins 2013). Under a cap-and-trade system, regulators issue permits up to the environmental limit, creating scarcity and a market price for emissions. Firms that can reduce emissions cheaply sell permits to those facing higher abatement costs, minimising the overall cost of meeting environmental targets. Carbon trading is one of the most mature examples. Mandatory carbon trading systems include the EU Emissions Trading System (Narassimhan et al. 2018)and California's Cap-and-Trade Program (Cushing et al. 2018). These compliance markets operate under regulatory limits on emissions. By contrast, **voluntary carbon markets** allow companies, institutions, and individuals to purchase carbon credits (or permits) outside regulatory obligations to offset or "neutralise" their emissions (Kreibich and Hermwille 2021). Projects, such as reforestation, soil carbon enhancement, or renewable energy, generate credits verified under independent standards. Voluntary carbon markets channel private finance toward mitigation and restoration where compliance regimes are absent, but they face persistent challenges of credibility, additionality, permanence, and equity (Corbera et al. 2009, Miltenberger et al. 2021). Whilst price volatility and inconsistent verification standards can undermine trust, these markets also function as innovation spaces that pioneer methodologies later adopted by mandatory schemes.

From Carbon Markets to Nature Markets: Comparative Insights

In their 2019 paper, Needham et al, argue that much can be learned about designing biodiversity offset markets from the experience of tradable pollution permit and carbon markets**.** Both carbon and biodiversity markets rely on the creation of tradable ecological units, carbon tonnes, habitat hectares, biodiversity credits, underpinned by property or usage rights, measurement standards, and verification systems. As previously outlined, in a pollution permit market, regulators set a fixed cap on emissions and allow firms to trade allowances. The same logic can guide nature markets: the "cap" corresponds to the goal of no net loss or Net Gain in biodiversity. Developers must hold sufficient offset credits to compensate for ecological damage, while landowners create credits by restoring or enhancing habitats. The price of a biodiversity credit should reflect the costs of conservation, and efficiency gains arise from trading across sites with differing conservation costs (Armsworth et al. 2012). However, the analogy between the two markets has limits. Biodiversity is multidimensional and site-specific, making it harder to define standardised units, ensure ecological equivalence, or verify permanence. Carbon is fungible, one tonne emitted is equivalent anywhere, but biodiversity is context-dependent and relational. These complexities open up rich avenues for future research, policy design, and debate, topics that will be explored in later chapters of this book.

Subsidies and Payments for Ecosystem Services: Subsidies, often provided by governments, are financial benefits awarded to individuals, businesses, or institutions. Subsidies may take the form of direct payments or loans, tax exemptions, or market price support, and can be directed toward private actors such as landholders and firms or toward public institutions. In the environment, "nature positive" subsidies aim to correct positive externalities (OECD 2025). When private actions, such as maintaining hedgerows, restoring wetlands, or protecting soil, generate ecological benefits that are not reflected in market

prices, subsidies can help internalise these externalities by rewarding beneficial behaviour. Agri-environment schemes are a long-studied example of these subsidies in relation to agricultural policy and the provision of biodiversity with a focus on how these schemes generate significant environmental improvements for the lowest cost to the public purse. Suggested reading on this topic includes (Nguyen et al. 2022, 2024, Simpson et al. 2023). Payments for Ecosystem Services (PES) schemes extend the logic of government subsidies to more decentralised and voluntary arrangements. PES schemes involve transactions in which ecosystem service users pay providers, typically farmers, forest managers, or landowners, to manage resources in ways that maintain or enhance the delivery of ecosystem services (Wunder 2015). Payments may compensate for restricted use (e.g. reduced logging or pesticide application) or reward new investments (e.g. reforestation, habitat restoration). Operating on a beneficiary-pays principle, PES schemes depend on users' willingness to pay for off-site environmental benefits and providers' willingness to modify practices in return.

Beyond classical market-based instruments, market-enabling mechanisms address the problem of information asymmetry, where the true environmental quality of goods or investments is difficult to observe. For instance, in carbon markets, buyers often lack full information about whether offset projects genuinely deliver long-term carbon sequestration, leading to adverse selection and the proliferation of low-quality credits. To counter this, certification schemes and environmental standards play a crucial role in establishing credible, verifiable information about environmental performance. Likewise, green finance instruments, such as green bonds and sustainability-linked loans, depend on robust standards and disclosure frameworks to guide capital toward genuinely sustainable activities. Together, these mechanisms enhance transparency, strengthen confidence among market participants, and help align financial incentives with authentic environmental outcomes.

Despite their potential, economic instruments for Net Gain face multiple conceptual, practical, and ethical challenges. First, upscaling investment into environmentally beneficial actions carries considerable risks including ecological, political and reputational (Zu Ermgassen et al. 2021). Second, as outlined previously, reliable and widely accepted metrics for biodiversity and the associated baselines are challenging to develop and implement (Needham et al. 2019, Mayfield et al. 2022). Tied to this, is the need and challenge associated with developing effective governance and regulations that supports monitoring and enforcement, as well as ensuring genuine additionality and limiting the impacts of greenwashing. This is perhaps further complicated by the blurring of the lines between voluntary and compliance activities associated with Net Gain and who has responsibility for the development, implementation and enforcement of frameworks. The concept of valuing nature more broadly is challenging: assigning monetary units to complex, interdependent ecosystems is fraught with uncertainty; different valuation methods can produce divergent outcomes, and many ecosystem functions are non-substitutable or remain immeasurable (Paul et al. 2020). Finally, there are intrinsic ecological limits to substitution: while tradable units such as credits can enhance efficiency, not all biodiversity losses are replaceable, and over-reliance on market mechanisms may risk undermining ecological integrity and disproportionately benefit well-resourced actors, highlighting the need to integrate social equity considerations into Net Gain design and implementation. Collectively, these challenges highlight the need to complement economic instruments with regulatory safeguards, robust governance, and ethical reflection to ensure that Net Gain is both ecologically meaningful and socially equitable.

Legal and policy aspects of Net Gain

Where economic routes to Net Gain focus on correcting market failures through prices, incentives, and enabling mechanisms, legal pathways provide the framework that determines who must act, how, and to what standard. Law formalises Net Gain and its associated ambitions by embedding them in statutory duties, planning controls, and environmental permitting systems, ensuring that measurable outcomes towards climate action and biodiversity enhancement are mandated, monitored, and, where necessary, enforced.

Internationally, multilateral environmental agreements establish the cooperative architecture for conservation and sustainable use of environmental resources. Many of the international environmental agreements in action today stem from The United Nations Conference on Environment and Development, also known as the 'Rio Earth Summit' (UN 1992). This aimed to create a new blueprint for international action on sustainable development and developed the foundational treaties that act as a basis for international climate change negotiations, the United Nations Framework Convention on Climate Change (UNFCCC), and the equivalent for biodiversity, the Convention on Biological Diversity (CBD).

On climate change, the two landmark international agreements are the Paris Agreement (UNFCC 2015), and its predecessor the Kyoto Protocol (UNFCC 1997). Under the Paris Agreement (UNFCC 2015) countries agreed to keep a rise in global temperatures to below 2°C from pre-industrial levels, by the end of this century. Additionally, all participating countries are required to report on their emissions and implementation efforts, creating the basis for carbon accounting and compliance. There is also a focus on the mobilization of financial resources to support less developed countries that are most at risk from climate change. Global commitments to halt biodiversity loss and restore ecosystems are exemplified through the UN Sustainable Development Goals (UN 2015), the UN Decade on Ecosystem Restoration (UN 2019), and the Kunming–Montreal Global Biodiversity Framework (GBF 2022). The latter is considered one of the most ambitious environmental agreements, with four goals A) Protect and Restore; B) Prosper with nature; C) Share Benefits Fairly and D) Invest and Collaborate, underpinned by 23 targets that explore the options for conservation, restoration and sustainable use of biodiversity globally (GBF 2022).

Countries participating in the Kunming–Montreal Global Biodiversity Framework and the Paris Agreement commit to setting national targets for biodiversity and climate action. These commitments shape domestic law and policy by embedding conservation, restoration, and climate objectives into environmental regulation, permitting, and planning systems. Beyond formal legal obligations, they also shape market and governance expectations, prompting corporations and financial institutions to align with emerging standards for nature-related risk disclosure (such as those developed through global taskforces on climate and biodiversity) and sustainable resource use. In this way, international environmental law provides both the authority and normative pressure that operationalise Net Gain within national and subnational jurisdictions.

The domestic frameworks countries adopt to achieve their global commitments take varied forms, such as statutory biodiversity duties, licensing and offset regimes, and disclosure obligations, reflecting each nation's legal traditions and governance structures. These instruments vary in degree of enforceability and precision. Statutory duties and permit conditions create binding obligations, generating predictable compliance outcomes and, in some cases, supporting market mechanisms such as tradable biodiversity credits and carbon credits. Planning guidance and discretionary policies, by contrast, rely on negotiation and professional judgment, offering flexibility in how environmental outcomes are achieved. Together, these mechanisms form a continuum, from mandatory legal requirements to policy-led expectations, shaping both the predictability of environmental outcomes and the actors involved in achieving the environmental improvements.

The UK provides a useful illustration of this legal spectrum. The UK policy context for net gain is anchored in the Government's 25 Year Environment Plan (2018), which established natural capital as the foundational framework for environmental decision-making and committed to achieving environmental net gain, ensuring that infrastructure developers leave the environment in a measurably better state compared to the pre-development baseline. This policy vision gained legal force through the Environment Act 2021, which set clear statutory targets for the recovery of the natural world in four priority areas: air quality, biodiversity, water and waste Natural England, whilst enshrining five internationally recognised environmental principles into UK legislation: the integration principle, the precautionary principle, the prevention principle, the rectification at source principle, and the polluter pays principle. The UK Environment Act 2021 applies to England and Wales and established a statutory duty for most developments to deliver a minimum 10% Biodiversity Net Gain (BNG), operationalised through a national biodiversity metric and a statutory register of biodiversity gains (UK Gov

2021). This creates a clear compliance framework and a regulated market for biodiversity credits (Stuart et al. 2024). In contrast, Scotland's National Planning Framework 4 encourages biodiversity enhancement through planning policy rather than statute, embedding the principle of Net Gain and nature positive without a fixed percentage target or mandatory metric: "Development proposals will contribute to the enhancement of biodiversity, including where relevant, restoring degraded habitats and building and strengthening nature networks and the connections between them. Proposals should also integrate nature-based solutions, where possible" (Scottish Government 2023). This approach offers greater local discretion but provides less enforceable demand for measurable uplift.

Comparable statutory mechanisms exist in other countries. The U.S. Clean Water Act (1972, Section 404) established one of the earliest "no net loss" frameworks, requiring developers to offset unavoidable wetland impacts through compensatory mitigation, including wetland mitigation banking (Hough and Robertson 2009, Vaz et al. 2026). Australia's Environment Protection and Biodiversity Conservation Act (1999) similarly underpins biodiversity offset policies, including credit-based schemes that allow landholders to generate and trade biodiversity credits (Maron et al. 2015). These examples illustrate how national legal systems embed measurable ecological improvement into permitting and development processes, even if the mechanisms and metrics vary.

These frameworks are dovetailed with economic instruments. Policy, legal frameworks, and economic instruments are the three pillars which determine how net gain is designed, implemented, and enforced. Policy provides the vision and strategic direction for achieving net gain. It does this by translating broad environmental commitments (for example, the ambition of halting biodiversity decline) into actions which guide public and private sector behaviour. Legal frameworks create the certainty, legitimacy, and accountability that allow economic instruments to function. Markets depend on law to define tradable units, protect and define the scope of property rights, and enforce monitoring and claims. Conversely, economic mechanisms provide regulators with flexible pathways to compliance, lowering costs and encouraging innovation. In this way, legal routes and economic routes are mutually constitutive: law provides the scaffolding, while markets operationalise the outcomes.

Social aspects of Net Gain

A fundamental challenge facing societies today is sustaining and improving the ways nature benefits all people. This goal underpins the work of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), which brings together governments, researchers, and civil society to evaluate and advance understanding of biodiversity, ecosystems, and their societal contributions to support policy development (Díaz et al. 2015). The IPBES conceptual framework introduced the concept of nature's contributions to people (NCP), which expands upon the ecosystem services framework from the Millennium Ecosystem Assessment. The NCP approach differs from previous frameworks in two significant ways: it acknowledges culture as fundamental to all human-nature relationships, and it prioritises indigenous and local knowledge systems in evaluating how nature contributes to human societies (Díaz et al. 2018). Human societies have developed varied ways of understanding and interacting with nature, encapsulating diverse values which extend beyond instrumental values. While natural features can often be quantified by objective measures, social and cultural values are experienced in embodied and relational ways, deeply rooted in specific, unique locations (Stephenson 2010). Despite decades of empirical evidence of how societies and local communities are intricately related to specific places and habitats, net gain approaches and associated policies assume that nature and biodiversity can be financialised and, possibly, traded through market-based mechanisms.

While biodiversity offset instruments aim for no net loss in ecological value, they typically do not consider no net loss in social and cultural values tied to nature in specific places, which often cannot be transferred (Wartmann & Cary 2025, in press). For example, if a reduction of green space in inner-city London is offset with a net gain project in the Lake District, this compensation shift to a different community, possibly with a different socio-economic profile, are risking disproportionate social, recreational, and wellbeing impacts on already marginalised or nature-deprived communities. Where biodiversity offsets are considered at

all, they are often narrowly focused on economic or livelihood concerns, overlooking cultural, spiritual, or recreational values that are tied to nature in specific places. Such place-based social values of nature often reflect both geophysical, ecological and cultural contexts. For example, the Yorkshire Dales encompasses limestone pavements, traditional hay meadows, and drystone walls shaped by centuries of pastoral farming. This landscape is not only biodiverse but embodies cultural heritage through traditional land management practices passed down through generations. If development destroys natural habitats here and biodiversity value is offset elsewhere, the irreplaceable uniqueness and people-place bonds are lost. Thus, understanding people-nature relationships through place reveals a fundamental tension: tradable biodiversity units cannot capture nature's lived, place-based meanings. This tension has already been recognised in much of the critique around the ecosystem service framework (Kirchhoff 2012) but continues to be relevant for biodiversity offsetting and netgain. Research on England's biodiversity offsetting, for example, reveals significant concerns regarding the social and ecological dimensions of compensating biodiversity loss, particularly around distributive justice, equitable access to nature for all people across society, and ownership rights over sites of shared cultural and natural value (Taherzadeh and Howley 2018).

Ethical concerns that have been raised include that biodiversity offsetting can undermine existing positive human-nature relationships and create justice concerns (Karlsson and Edvardsson Björnberg 2021). For example, research from Madagascar shows that gains from development projects fail to offset the costs created by conservation limits, that the people who shoulder these costs are not the ones who receive the benefits, and that the restrictions take effect immediately while the development benefits emerge only after a delay (Bidaud et al. 2017). These problems are significant both for environmental justice and for ensuring the long-term sustainability of the biodiversity outcomes the offset is meant to achieve.

The issue is particularly acute where offsetting is done over great distances and wealth differentials, such as any offsetting projects in the Global South to address biodiversity loss in the Global North. Addressing these ethical dimensions through clearer policy frameworks and enhancing stakeholder engagement is essential to achieve biodiversity net gain that considers human welfare and wellbeing as tied to the biodiversity values that these offsetting frameworks aim to protect.

Jones et al. (2019) propose a framework identifying key challenges and opportunities for achieving improved biodiversity conservation and community wellbeing outcomes. They argue that early coordination of social and ecological expertise alongside meaningful early engagement with local communities can ensure that biodiversity impact mitigation contributes positively to local wellbeing rather than undermining it.

To implement such early engagement, social science methods are essential because they can help us understand how people relate to the nature that will be lost to development, but also how people might relate to the nature that is sought to be improved with a netgain offsetting project, as well as understand people’s attitude to specific aspects of the implementation, ranging from aesthetics of new, ‘messy’ nature (Wartmann and Lorimer 2024) to broader considerations of participation and willingness to offset. For example, a study in East Lothian highlighted that residents mostly agreed about negative effects of housing developments on biodiversity, agriculture, and landscape character, but differed widely in the degree they were willing (and how they would prefer) to offset for these impacts (Scholte et al. 2016).

Despite the importance of understanding social impacts at the site where biodiversity is lost, and where it will be gained, social impact studies are rarely conducted (Tupala et al. 2022). The social sciences, arts, and humanities provide a breadth of different methods and perspectives to analyse, describe and evaluate nature conservation projects and the way they for instance impact people. Such methods are also essential to developing human-centred understanding of conservation policy. At the intersection of conservation biology and social sciences, the subfield of conservation social science emerged, that focuses on understanding the social dimensions of biodiversity conservation and environmental management, describing some of the various methods from social sciences that have applicability for nature conservation (Bennett et al. 2017a, 2017b). While this engagement

with social science from nature conservation scientists was welcome, it was also critiqued as a narrow engagement with certain aspects of mostly quantitative approaches to social sciences that ignored large aspects of qualitative engagements (Moon et al. 2019), and for treating social sciences as one amalgamation, whereas different disciplines can offer different perspectives on nature recovery projects (Cary et al. 2025).

Qualitative social science approaches are of value when exploring a topic while also retaining context. The goal of qualitative methods is to understand the nature of phenomena, as opposed to the magnitude or distribution (Musante and Dewalt 2011). Largely inductive, qualitative approaches often use specific cases representing a non-random sample of a population; the goal being to understand complex social processes, phenomena and attributes from individual perspectives, rather than extrapolating findings to wider populations (Rust et al. 2017). Thus, the emphasis is on depth of understanding rather than breadth.

Qualitative data can be collected through diverse social science methods. These include participant observation to gather participant-derived data. Participant observation is a method whereby the researcher takes part in the daily activities and interactions of the people being studied. Data is collected in naturalistic settings, enabling knowledge to be gained about both the explicit and tacit aspects of participants' lives and practices (Musante and Dewalt 2011). Data is recorded in the form of detailed field notes which included descriptions of conversations and the activities taking place, as well as the researcher's own reflections.

Qualitative data may also be gathered through interviews. Interviews allow the researcher to explore participants' perspectives, map experiences and views and generate reflections and solutions on the phenomenon being studied. Interviews can be structured, semi-structured or unstructured, depending on the level of flexibility the researcher would like to have during the method. Structured interviews may take place in a defined setting and ask the same set of questions to each participant. Less structured interviews may use a topic guide, which is a pre-written interview outline that provides structure alongside the ability to adapt the interview context and content during the participant-interviewer interaction (Knott et al. 2022). Creative and participatory approaches are valuable means to generate qualitative data in conservation. Examples include walking interviews, participatory mapping, future visioning or photovoice (Lembi et al. 2024). Collaboration between social and biodiversity experts in such projects is one way to ensure a wider set of values are included in policy and decision-making.

In conclusion, to address the emergent challenges of biodiversity net gain, ecologists, economists, social scientists and practitioners must collaborate to integrate cutting-edge knowledge from their respective disciplines, ensuring that offsetting mechanisms deliver both robust ecological outcomes and equitable distribution of benefits for affected communities.

Technological aspects of Net Gain

Technologies for avoidance of impacts: A key step in the mitigation hierarchy involves avoidance of damage. Technologies leading to renewable energy production (Panwar et al. 2011, Virah-Sawmy et al. 2014), avoid the use of carbon emissions during operation (although construction stages are still reliant on fossil fuel consumption and production of non-degradable materials). An equally important stage is the minimisation of impacts which is usually achieved from increased efficiency in the use of resources, such as energy or space. For instance, 'Green Revolution' technologies, including high-yielding cultivars, chemical fertilizers and pesticides, and mechanization and irrigation have led to increasing agricultural production per unit land area, per unit fertilizer input, and per unit water consumed (Foley et al. 2005). Biological waste management, smart design, increasing green space in urban areas, thereby reducing runoff and "heat island'' effects, and maintaining soil organic matter in croplands, which is a key to water-holding capacity, nutrient availability, and carbon sequestration are all examples of technological interventions that reconcile development with environmental gain by increasing the efficiency in using energy/space/minerals and minimising the environmental attrition of human developments (Foley et al. 2005).

Technologies for data collection and environmental monitoring: An area where technology plays a unique role towards Net Gain is in monitoring our progress towards Net Gain on the ground. For instance, for the practical implementation of BNG we need to measure biodiversity, an inherently complex task which encompasses not only species richness but also abundance, functional diversity, and genetic variation (Loreau et al. 2022). No single metric can capture this multidimensionality, and trade-offs often emerge when simplifying indicators are used in policy or planning, prompting the need to transcend simplified summaries of biodiversity and obtain species-specific information, as well as information on population trends.

Spatial and temporal mismatches further complicate Net Gain delivery. Habitat losses caused by development occur immediately and often irreversibly, while ecological gains from restoration projects may take decades to materialise (Lira et al. 2019, Haddou et al. 2022). Such lags create a risk that promised improvements will not reflect realities on the ground. This prompts the need to account for transient ecosystem states and to explicitly observe community dynamics, rather than single snapshots of abundance and community composition. Obtaining such detailed time series of biodiversity requires efficient, easily deployable and rigorous monitoring approaches.

Traditional field surveys, conducted by trained ecologists, remain essential for planning, decision-making, and compliance, as they provide detailed, expert insights. However, these "boots on the ground" surveys are resource-intensive and challenging, requiring significant financial and human investments (Henrys & Jarvis, 2019). They can be logistically challenging, hazardous in hostile locations or unavailable during certain parts of the year. More broadly, it is certain that we will always have more species and ecosystems that need monitoring than the resources and scientists we have available to do it (Kindsvater et al. 2018). These challenges highlight the need for complementary technological solutions, be they at the observation stage or at the stage of efficient statistical analysis.

Cutting-edge remote sensing techniques combined with artificial intelligence (AI) are emerging as a potentially useful biodiversity monitoring approach and for BNG site selection (Pettorelli et al. 2025). Remote sensing can overcome challenges of access, health and safety, and seasonal survey limitations, enabling consistent monitoring of dynamic or inaccessible landscapes.

Satellite platforms such as Sentinel and Landsat provide consistent, repeatable coverage of ecosystem structure and function and increasingly on abundance and distribution of biodiversity at regional to global scales (Pettorelli et al. 2016). Advanced sensors, including LiDAR and hyperspectral imaging, enable measurement of vegetation structure, biomass, and even proxies for species composition (Iheaturu et al. 2024). At local scales, drones offer cost-effective tools for high-resolution monitoring of restoration projects, delivering imagery that supports rapid assessment of vegetation recovery and habitat heterogeneity (Robinson et al. 2022).

An altogether different approach to biodiversity monitoring uses environmental DNA (eDNA) sampling from soil, water, or air provides a rapid, non-invasive way to detect species and community composition and could be a promising tool for measuring BNG (Kelly et al. 2014, Sahu et al. 2023). By analysing genetic traces such as skin cells, hair, or pollen, even tiny samples can reveal a broad range of organisms, including elusive or transient species often missed by traditional surveys (Deiner et al. 2017). Genomic tools also extend monitoring to the genetic dimension of biodiversity, which is rarely addressed in BNG assessments. Increasingly, these methods are applied to track species colonisation (Vimercati et al. 2020) and assess the progress of ecological restorations (Yan et al. 2018).

eDNA has limitations suggesting its complementary use alongside conventional ecological methods (Kelly et al. 2014). For instance, it cannot specify exact locations, confirm current presence, or estimate abundance, as DNA may persist in the environment or travel long distances. For example, eDNA from surface waters usually reflects recent site biodiversity, whereas eDNA in surface sediments may represent an unknown length of temporal accumulation (Shaw et al. 2016). Furthermore, while techniques to estimate population size

are emerging, there is still limited evidence that metabarcoding of eDNA can reliably provide abundance information for entire communities (Deiner et al. 2017).

Technological innovation for Net Gain comes with important caveats. The ethics of digital and genomic monitoring tools are particularly critical, especially regarding data ownership and access for Indigenous and local communities whose lands and knowledge may be involved in Net Gain initiatives (Johnson et al. 2021). Ensuring equitable benefit-sharing and safeguarding rights is therefore essential. At the same time, enthusiasm for technological assessments can risk sidelining aspects of biodiversity that cannot be quantified, such as spiritual, cultural, and relational values (Díaz et al. 2018). An exclusive focus on measurable indicators narrows the vision of BNG to what is easily counted, rather than what is ecologically or socially meaningful.

Technologies for processing big data: The growing volume and resolution of remotely sensed imagery are generating vast datasets, making automation algorithms (such as AI) indispensable for managing and interpreting geographical data, classifying habitats or identifying dynamical patterns (Helber et al. 2019, Christin et al. 2019, Lake et al. 2022, Yu et al. 2025, Pettorelli et al. 2025). This reduces dependence on manual interpretation and allows ecologists to focus on modelling and decision-making rather than data pre-processing.

For BNG, the scalability of remote sensing is particularly valuable. For example, satellites combined with deep learning approaches, trained on ground-truthing data in the field (Henrys and Jarvis 2019), could pre-screen sites, produce preliminary habitat maps, or even biodiversity assessments (Rocchini et al. 2018, Helber et al. 2019) at reduced cost, improved efficiency and consistency.

AI also supports biodiversity monitoring beyond remote sensing. In bioacoustics, microphones capture continuous soundscapes, from bird songs to bat calls, producing huge datasets (Mac Aodha et al. 2018, Hill et al. 2018, Lamba et al. 2019). AI-driven automation can analyse data collected by loggers to identify species by their unique acoustic signatures, processing audio data at a scale impossible for human experts (Lamba et al. 2019). Systems such as the BTO Acoustic Pipeline (https://www.bto.org/our-work/science/research-areas/acoustic-monitoring) are already providing automated bird species detection from audio recordings, and are expanding to cover a wide range of taxa. Automated acoustic monitoring often outperforms human observers, particularly for detecting hidden or overlapping calls in dense surroundings (Mac Aodha et al. 2018, Hill et al. 2018).

AI is also transforming camera trap monitoring by rapidly classifying species in images and videos, reducing processing time and minimising human error (Norouzzadeh et al. 2018). At the same time, community-led platforms such as iNaturalist and eBird, are revolutionising biodiversity monitoring. These tools combine millions of citizen science observations with AI-based verification to generate open-access datasets at scale (Kelling et al. 2009). Mobile applications further enhance accessibility by allowing users to collect geo-tagged data while also feeding into AI model training and validation. Such platforms are increasingly useful for democratising data, enabling broader societal participation in biodiversity monitoring, and enriching scientific databases with diverse observations (Sullivan et al. 2018, Lukyanenko et al. 2020, McClure et al. 2020).

However, machine learning is only part of the toolbox available to us for processing and understanding data. The environmental sector is characterised by extreme contrasts in the richness of available information. Although some types of data (e.g., remotely-sensed layers of the planet's surface) are available in almost unwieldy quantities, other data types (e.g., underground or underwater layers) are much harder to observe directly. Furthermore, in the context of conservation, we are dealing with endangered and rare species, so demographic and population data are often lacking. While machine learning is ideally suited to analysing large data sets and describing (spatial, temporal etc.) patterns (Christin et al. 2019), more insightful methods of inference are needed when we are analysing limited data, in search of the underlying ecological mechanisms. Anticipatory management based on understanding mechanisms can be more powerful and targeted than machine learning (Pichler and Hartig 2023). The area of ecological modelling saw a period of rapid development during the 1960s-1990s, influenced by the mechanistic thinking that led to centuries of advancement in

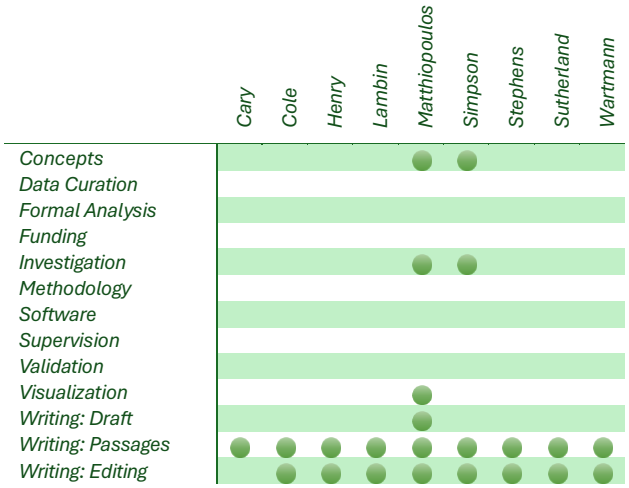
physics. Indeed, much of the language and tools of the discipline were influenced by a reductionist and deterministic search for ecological laws (Berryman and Uni 2003). Methodology remained broadly separated from statistical inference, the field of statistics that brings models together with data. In recent decades, advances in model-fitting methodology, particularly using approximate computation (Beaumont 2010, Bachl et al. 2019, Seaton et al. 2024), Bayesian probability (Clark 2005, Kery and Royle 2020) and computer intensive methods (Newman et al. 2014, Auger-Méthé et al. 2021) have led to a formal melding of theory and data, increasing our ability to bring general principles and underlying mechanisms under empirical scrutiny.

These areas of development benefit the delivery of Net Gain in three ways. First, they maximise the precision and accuracy of estimates generated from sparse data sets (Harwood and Stokes 2003, Kindsvater et al. 2018), allowing us to avoid damaging activities whose effect would otherwise be unknown. Second, they disentangle and quantify the effects of individual drivers of change and can therefore account for the influence of variables, such as climate, that we cannot directly control (Walther et al. 2002) and rank the effectiveness of management levers that we have available (Halpern et al. 2008, Tilman et al. 2017). Third, efficient statistical inference has considerably lower computational costs, both in terms of time and energy. Most obviously, time matters in this instance, because environmental crises are urgent. Perhaps less obviously, energy is becoming a concern since ironically, tools aimed at conservation (such as large deep-learning models) can have substantial environmental costs if not used judiciously (Toderas 2025). Ultimately, inferential techniques from both classical statistics and machine learning are likely to hybridise and converge so that mechanistic insights are gained by high speed computation and predictive power under data-sparse conditions (Linardatos et al. 2020, Han et al. 2023, Wesselkamp et al. 2024).

Take-home messages

- Human dominance transforms space use, biogeochemical cycles, and wildlife abundance globally and irreversibly.
- These changes drive accelerating global warming and biodiversity loss, crossing planetary tipping points.
- Confronting these crises demands urgent, equitable, and coordinated global environmental action. This needs to be multinational, multisector and multidisciplinary.
- Net Gain aims to ensure that human development leaves nature better than before.
- The key conceptual tool in this is the mitigation hierarchy: Sequentially aiming to avoid, minimise, restore, and offset damage, to manage human impacts.
- Thinking has shifted from balancing losses to achieving measurable biodiversity improvements overall by linking regulation and voluntary efforts for ecosystem recovery.
- Ecological interventions include 1) prioritising, connecting, and balancing land use for optimal biodiversity conservation. 2) Restoring carbon, nitrogen, and phosphorus cycles to stabilise ecosystems sustainably and 3) Reintroducing natives, controlling invasives, and rewilding to restore ecological function.
- Economic interventions include taxation of damaging activities, tradeable pollution permits, biodiversity credits, subsidies and payments for ecosystem services, certifications, transparency, and standards to build trust in green finance markets.
- Law defines duties, accountability, and enforceable standards for environmental improvement. Global treaties such as the Paris Agreement and Kunming–Montreal Global Biodiversity Framework coordinate cooperative climate and biodiversity action among nations.
- Biodiversity offsets risk ignoring social, cultural, and place-based values; effective Net Gain requires transparency, community engagement, and recognition of relational connections.
- Technological innovation allows us to extract more efficiencies from our use of resources and to minimise damage. It also allows us synoptic monitoring of global systems and the efficient analysis of available data to maximise precision in ecosystem management.

Author contributions



References

Allen, M. R., D. J. Frame, C. Huntingford, C. D. Jones, J. A. Lowe, M. Meinshausen, and N. Meinshausen. 2009. Warming caused by cumulative carbon emissions towards the trillionth tonne. Nature 458:1163–1166.

Andermann, T., S. Faurby, S. T. Turvey, A. Antonelli, and D. Silvestro. 2020. The past and future human impact on mammalian diversity. Science Advances 6:eabb2313.

Arlidge, W. N. S., J. W. Bull, P. F. E. Addison, M. J. Burgass, D. Gianuca, T. M. Gorham, C. Jacob, N. Shumway, S. P. Sinclair, J. E. M. Watson, C. Wilcox, and E. J. Milner-Gulland. 2018. A Global Mitigation Hierarchy for Nature Conservation. BioScience 68:336–347.

Armstrong, C. 2024. The biodiversity crisis and global justice: a research agenda. Critical Review of International Social and Political Philosophy:1–20.

Armsworth, P. R., S. Acs, M. Dallimer, K. J. Gaston, N. Hanley, and P. Wilson. 2012. The cost of policy simplification in conservation incentive programs. Ecology Letters 15:406–414.

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

Auger-Méthé, M., K. Newman, D. Cole, F. Empacher, R. Gryba, A. A. King, V. Leos-Barajas, J. Mills Flemming, A. Nielsen, G. Petris, and L. Thomas. 2021. A guide to state–space modeling of ecological time series. Ecological Monographs 91:e01470.

Bachl, F. E., F. Lindgren, D. L. Borchers, and J. B. Illian. 2019. inlabru: an R package for Bayesian spatial modelling from ecological survey data. Methods in Ecology and Evolution 10:760–766.

Balmford, A., Rhys. E. Green, and J. P. W. Scharlemann. 2005. Sparing land for nature: exploring the potential impact of changes in agricultural yield on the area needed for crop production. Global Change Biology 11:1594–1605.

Beaumont, M. A. 2010. Approximate Bayesian Computation in Evolution and Ecology. Annual Review of Ecology, Evolution, and Systematics 41:379–406.

Benayas, J. M. R., A. C. Newton, A. Diaz, and J. M. Bullock. 2009. Enhancement of Biodiversity and Ecosystem Services by Ecological Restoration: A Meta-Analysis. Science 325:1121–1124.

Bennett, N. J., R. Roth, S. C. Klain, K. M. A. Chan, D. A. Clark, G. Cullman, G. Epstein, M. P. Nelson, R. Stedman, T. L. Teel, R. E. W. Thomas, C. Wyborn, D. Curran, A. Greenberg, J. Sandlos, and D. Veríssimo. 2017a. Mainstreaming the social sciences in conservation. Conservation Biology 31:56–66.

Bennett, N. J., R. Roth, S. C. Klain, K. M. A. Chan, D. A. Clark, G. Cullman, G. Epstein, M. P. Nelson, R. Stedman, T. L. Teel, R. E. W. Thomas, C. Wyborn, D. Curran, A. Greenberg, J. Sandlos, and D. Veríssimo. 2017b. Mainstreaming the social sciences in conservation. Conservation Biology 31:56–66.

Berryman, A. A., and W. S. Uni. 2003. On principles, laws and theory in population ecology. Oikos.

Betts, R. A. 2000. Offset of the potential carbon sink from boreal forestation by decreases in surface albedo. Nature 408:187–190.

Bidaud, C., K. Schreckenberg, M. Rabeharison, P. Ranjatson, J. Gibbons, and JuliaP. G. Jones. 2017. The Sweet and the Bitter: Intertwined Positive and Negative Social Impacts of a Biodiversity Offset. Conservation and Society 15:1.

Bradfer-Lawrence, T., A. Dobson, T. Finch, E. Fuentes-Montemayor, N. Hanley, J. Matthiopoulos, M. Nthambi, K. Simpson, K. Watts, R. Whytock, and K. Park. 2024, March 18. Spillovers and legacies of land management on temperate woodland biodiversity. In Review.

Bruinderink, G. W. T. A. G., and E. Hazebroek. 1996. Ungulate Traffic Collisions in Europe. Conservation Biology 10:1059–1067.

Bull, J. W., and S. Brownlie. 2017. The transition from No Net Loss to a Net Gain of biodiversity is far from trivial. Oryx 51:53–59.

Bull, J. W., K. B. Suttle, A. Gordon, N. J. Singh, and E. J. Milner-Gulland. 2013. Biodiversity offsets in theory and practice. Oryx 47:369–380.

Burke, B., D. P. O'Connell, D. Kinchin-Smith, S. Sealy, and S. F. Newton. 2022. Nestboxes augment seabird breeding performance in a high-density colony: Insight from 15 years of monitoring data. Ecological Solutions and Evidence 3:e12171.

Business and Biodiversity Offsets Programme. 2012. Standard on biodiversity offsets. Forest Trends, Washington.
Butchart, S. H. M., M. Walpole, B. Collen, A. Van Strien, J. P. W. Scharlemann, R. E. A. Almond, J. E. M. Baillie, B. Bomhard, C. Brown, J. Bruno, K. E. Carpenter, G. M. Carr, J. Chanson, A. M. Chenery, J. Csirke, N. C. Davidson, F. Dentener, M. Foster, A. Galli, J. N. Galloway, P. Genovesi, R. D. Gregory, M. Hockings, V. Kapos, J.-F. Lamarque, F. Leverington, J. Loh, M. A. McGeoch, L. McRae, A. Minasyan, M. H. Morcillo, T. E. E. Oldfield, D. Pauly, S. Quader, C. Revenga, J. R. Sauer, B. Skolnik, D. Spear, D. Stanwell-Smith, S. N. Stuart, A. Symes, M. Tierney, T. D. Tyrrell, J.-C. Vié, and R. Watson. 2010. Global Biodiversity: Indicators of Recent Declines. Science 328:1164–1168.
Cardinale, B. J., J. E. Duffy, A. Gonzalez, D. U. Hooper, C. Perrings, P. Venail, A. Narwani, G. M. Mace, D. Tilman, D. A. Wardle, A. P. Kinzig, G. C. Daily, M. Loreau, J. B. Grace, A. Larigauderie, D. S. Srivastava, and S. Naeem. 2012. Biodiversity loss and its impact on humanity. Nature 486:59–67.
Cary, E., K. Jones, V. Thomas, S. Brieghel, A. Payo Payo, and F. M. Wartmann. 2025. Five critical questions we should ask of rewilding projects—And that social science can help us answer. People and Nature 7:2119–2135.
Christin, S., É. Hervet, and N. Lecomte. 2019. Applications for deep learning in ecology. Methods in Ecology and Evolution 10:1632–1644.
CIEEM, CIRIA and IEMA. 2016. Biodiversity Net Gain: Good practice principles for development.
Clark, J. S. 2005. Why environmental scientists are becoming Bayesians. Ecology:2–14.
Clough, Y., J. Barkmann, J. Juhrbandt, M. Kessler, T. C. Wanger, A. Anshary, D. Buchori, D. Cicuzza, K. Darras, D. D. Putra, S. Erasmi, R. Pitopang, C. Schmidt, C. H. Schulze, D. Seidel, I. Steffan-Dewenter, K. Stenchly, S. Vidal, M. Weist, A. C. Wielgoss, and T. Tscharntke. 2011. Combining high biodiversity with high yields in tropical agroforests. Proceedings of the National Academy of Sciences 108:8311–8316.
Colwell, R. K., R. R. Dunn, and N. C. Harris. 2012. Coextinction and Persistence of Dependent Species in a Changing World. Annual Review of Ecology, Evolution, and Systematics 43:183–203.
Corbera, E., M. Estrada, and K. Brown. 2009. How do regulated and voluntary carbon-offset schemes compare? Journal of Integrative Environmental Sciences 6:25–50.
Corlett, R. T. 2016. Restoration, Reintroduction, and Rewilding in a Changing World. Trends in Ecology & Evolution 31:453–462.
Cromsigt, J. P. G. M., M. Te Beest, G. I. H. Kerley, M. Landman, E. Le Roux, and F. A. Smith. 2018. Trophic rewilding as a climate change mitigation strategy? Philosophical Transactions of the Royal Society B: Biological Sciences 373:20170440.
Cushing, L., D. Blaustein-Rejto, M. Wander, M. Pastor, J. Sadd, A. Zhu, and R. Morello-Frosch. 2018. Carbon trading, co-pollutants, and environmental equity: Evidence from California's cap-and-trade program (2011–2015). PLOS Medicine 15:e1002604.
Damiens, F. L. P., A. Backstrom, and A. Gordon. 2021. Governing for "no net loss" of biodiversity over the long term: challenges and pathways forward. One Earth 4:60–74.
Deiner, K., H. M. Bik, E. Mächler, M. Seymour, A. Lacoursière-Roussel, F. Altermatt, S. Creer, I. Bista, D. M. Lodge, N. De Vere, M. E. Pfrender, and L. Bernatchez. 2017. Environmental DNA metabarcoding: Transforming how we survey animal and plant communities. Molecular Ecology 26:5872–5895.
Diamond, J. M., N. P. Ashmole, and P. E. Purves. 1989. The Present, Past and Future of Human-Caused Extinctions [and Discussion]. Phil. Trans. R. Soc. Lond. B 325:469–477.
Díaz, S., S. Demissew, J. Carabias, C. Joly, M. Lonsdale, N. Ash, A. Larigauderie, J. R. Adhikari, S. Arico, A. Báldi, A. Bartuska, I. A. Baste, A. Bilgin, E. Brondizio, K. M. Chan, V. E. Figueroa, A. Duraiappah, M. Fischer, R. Hill, T. Koetz, P. Leadley, P. Lyver, G. M. Mace, B. Martin-Lopez, M. Okumura, D. Pacheco, U. Pascual, E. S. Pérez, B. Reyers, E. Roth, O. Saito, R. J. Scholes, N. Sharma, H. Tallis, R. Thaman, R. Watson, T. Yahara, Z. A. Hamid, C. Akosim, Y. Al-Hafedh, R. Allahverdiyev, E. Amankwah, S. T. Asah, Z. Asfaw, G. Bartus, L. A. Brooks, J. Caillaux, G. Dalle, D. Darnaedi, A. Driver, G. Erpul, P. Escobar-Eyzaguirre, P. Failler, A. M. M. Fouda, B. Fu, H. Gundimeda, S. Hashimoto, F. Homer, S. Lavorel, G. Lichtenstein, W. A. Mala, W. Mandivenyi, P. Matczak, C. Mbizvo, M. Mehrdadi, J. P. Metzger, J. B. Mikissa, H. Moller, H. A. Mooney, P. Mumby, H. Nagendra, C. Nesshover, A. A. Oteng-Yeboah, G. Pataki, M. Roué, J. Rubis, M. Schultz, P. Smith, R. Sumaila, K. Takeuchi, S. Thomas, M. Verma, Y. Yeo-Chang, and D. Zlatanova. 2015. The IPBES Conceptual Framework — connecting nature and people. Current Opinion in Environmental Sustainability 14:1–16.
Díaz, S., U. Pascual, M. Stenseke, B. Martín-López, R. T. Watson, Z. Molnár, R. Hill, K. M. A. Chan, I. A. Baste, K. A. Brauman, S. Polasky, A. Church, M. Lonsdale, A. Larigauderie, P. W. Leadley, van Oudenhoven, S. Demissew, G. Erpul, P. Failler, C. A. Guerra, C. L. Hewitt, H. Keune, S. Lindley, and Y. Shirayama. 2018. Assessing nature's contributions to people.
Diniz-Filho, J. A. F. 2004. Macroecological analyses support an overkill scenario for late Pleistocene extinctions. Brazilian Journal of Biology 64:407–414.
Doherty, T. S., A. S. Glen, D. G. Nimmo, E. G. Ritchie, and C. R. Dickman. 2016. Invasive predators and global biodiversity loss. Proceedings of the National Academy of Sciences 113:11261–11265.
Donaldson, L., R. J. Wilson, and I. M. D. Maclean. 2017. Old concepts, new challenges: adapting landscape-scale conservation to the twenty-first century. Biodiversity and Conservation 26:527–552.
ENACT. 2022. ENACT - Enhancing Nature-basedSolutions for an AcceleratedClimate Transformation.
Erb, K.-H., T. Kastner, C. Plutzar, A. L. S. Bais, N. Carvalhais, T. Fetzel, S. Gingrich, H. Haberl, C. Lauk, M. Niedertscheider, J. Pongratz, M. Thurner, and S. Luyssaert. 2018. Unexpectedly large impact of forest management and grazing on global vegetation biomass. Nature 553:73–76.
Fahrig, L. 2017. Ecological Responses to Habitat Fragmentation Per Se. Annual Review of Ecology, Evolution, and Systematics 48:1–23.
Faith, D. P., and P. A. Walker. 2002. The role of trade-offs in biodiversity conservation planning: Linking local management, regional planning and global conservation efforts. Journal of Biosciences 27:393–407.
Faure, E., H. Levrel, and F. Quétier. 2024. Economics of rewilding. Ambio 53:1367–1382.
Figueiredo, L., J. Krauss, I. Steffan-Dewenter, and J. Sarmento Cabral. 2019. Understanding extinction debts: spatio–temporal scales, mechanisms and a roadmap for future research. Ecography 42:1973–1990.
Fischer, J., D. J. Abson, V. Butsic, M. J. Chappell, J. Ekroos, J. Hanspach, T. Kuemmerle, H. G. Smith, and H. Von Wehrden. 2014. Land Sparing Versus Land Sharing: Moving Forward. Conservation Letters 7:149–157.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global Consequences of Land Use. Science 309:570–574.

Forster, P. M., C. Smith, T. Walsh, W. F. Lamb, R. Lamboll, C. Cassou, M. Hauser, Z. Hausfather, J.-Y. Lee, M. D. Palmer, K. Von Schuckmann, A. B. A. Slangen, S. Szopa, B. Trewin, J. Yun, N. P. Gillett, S. Jenkins, H. D. Matthews, K. Raghavan, A. Ribes, J. Rogelj, D. Rosen, X. Zhang, M. Allen, L. Aleluia Reis, R. M. Andrew, R. A. Betts, A. Borger, J. A. Broersma, S. N. Burgess, L. Cheng, P. Friedlingstein, C. M. Domingues, M. Gambarini, T. Gasser, J. Gütschow, M. Ishii, C. Kadow, J. Kennedy, R. E. Killick, P. B. Krummel, A. Liné, D. P. Monselesan, C. Morice, J. Mühle, V. Naik, G. P. Peters, A. Pirani, J. Pongratz, J. C. Minx, M. Rigby, R. Rohde, A. Savita, S. I. Seneviratne, P. Thorne, C. Wells, L. M. Western, G. R. Van Der Werf, S. E. Wijffels, V. Masson-Delmotte, and P. Zhai. 2025. Indicators of Global Climate Change 2024: annual update of key indicators of the state of the climate system and human influence. Earth System Science Data 17:2641–2680.

Furness, R. W. R. W., S. Garthe, M. Trinder, J. Matthiopoulos, S. Wanless, and J. Jeglinski. 2018. Nocturnal flight activity of northern gannets Morus bassanus and implications for modelling collision risk at offshore wind farms. Environmental Impact Assessment Review 73:1–6.

Gambell, R. 1993. International Management of Whales and Whaling: An Historical Review of the Regulation of Commercial and Aboriginal Subsistence Whaling. Arctic:97–107.

Gardner, T. A., A. Von Hase, S. Brownlie, J. M. M. Ekstrom, J. D. Pilgrim, C. E. Savy, R. T. T. Stephens, J. Treweek, G. T. Ussher, G. Ward, and K. Ten Kate. 2013. Biodiversity Offsets and the Challenge of Achieving No Net Loss. Conservation Biology 27:1254–1264.

GBF. 2022. Kunming–Montreal Global Biodiversity Framework.

Gonzalez, A., R. M. Germain, D. S. Srivastava, E. Filotas, L. E. Dee, D. Gravel, P. L. Thompson, F. Isbell, S. Wang, S. Kéfi, J. Montoya, Y. R. Zelnik, and M. Loreau. 2020. Scaling-up biodiversity-ecosystem functioning research. Ecology Letters 23:757–776.

Gordon Jones, J. P., J. W. Bull, D. Roe, J. Baker, V. F. Griffiths, M. Starkey, L. J. Sonter, and E. J. Milner-Gulland. 2019. Net Gain: Seeking Better Outcomes for Local People when Mitigating Biodiversity Loss from Development. One Earth 1:195–201.

Greenspoon, L., E. Krieger, R. Sender, Y. Rosenberg, Y. M. Bar-On, U. Moran, T. Antman, S. Meiri, U. Roll, E. Noor, and R. Milo. 2023. The global biomass of wild mammals. Proceedings of the National Academy of Sciences 120:e2204892120.

Haddou, Y., R. Mancy, J. Matthiopoulos, S. Spatharis, and D. M. Dominoni. 2022. Widespread extinction debts and colonization credits in United States breeding bird communities. Nature Ecology & Evolution 2016.

Halpern, B. S., S. Walbridge, K. a Selkoe, C. V. Kappel, F. Micheli, C. D'Agrosa, J. F. Bruno, K. S. Casey, C. Ebert, H. E. Fox, R. Fujita, D. Heinemann, H. S. Lenihan, E. M. P. Madin, M. T. Perry, E. R. Selig, M. Spalding, R. Steneck, and R. Watson. 2008. A global map of human impact on marine ecosystems. Science (New York, N.Y.) 319:948–52.

Han, B. A., K. R. Varshney, S. LaDeau, A. Subramaniam, K. C. Weathers, and J. Zwart. 2023. A synergistic future for AI and ecology. Proceedings of the National Academy of Sciences 120:e2220283120.

Hanley, N., J. Shorgen F., and B. White. 2019. Introduction to environmental economics. Oxford University Press.

Hanski, I., and O. Ovaskainen. 2000. The metapopulation capacity of a fragmented landscape. Nature 404:755–8.

Hardin, G. 1968. The Tragedy of the Commons: The population problem has no technical solution; it requires a fundamental extension in morality. Science 162:1243–1248.

Harwood, J., and K. Stokes. 2003. Coping with uncertainty in ecological advice: Lessons from fisheries. Trends in Ecology and Evolution 18:617–622.

Hassan, R., R. Scholes, and N. Ash. 2005. Ecosystems and Human Well-being: Current State and Trends, Volume 1. Island Press, Washington.

Heilmayr, R., C. Echeverría, and E. F. Lambin. 2020. Impacts of Chilean forest subsidies on forest cover, carbon and biodiversity. Nature Sustainability 3:701–709.

Helber, P., B. Bischke, A. Dengel, and D. Borth. 2019. EuroSAT: A Novel Dataset and Deep Learning Benchmark for Land Use and Land Cover Classification. IEEE Journal of Selected Topics in Applied Earth Observations and Remote Sensing 12:2217–2226.

Henrys, P. A., and S. G. Jarvis. 2019. Integration of ground survey and remote sensing derived data: Producing robust indicators of habitat extent and condition. Ecology and Evolution 9:8104–8112.

Hill, A. P., P. Prince, E. Piña Covarrubias, C. P. Doncaster, J. L. Snaddon, and A. Rogers. 2018. AudioMoth: Evaluation of a smart open acoustic device for monitoring biodiversity and the environment. Methods in Ecology and Evolution 9:1199–1211.

Hooper, D. U., F. S. Chapin, J. J. Ewel, A. Hector, P. Inchausti, S. Lavorel, J. H. Lawton, D. M. Lodge, M. Loreau, S. Naeem, B. Schmid, H. Setälä, A. J. Symstad, J. Vandermeer, and D. A. Wardle. 2005. EFFECTS OF BIODIVERSITY ON ECOSYSTEM FUNCTIONING: A CONSENSUS OF CURRENT KNOWLEDGE. Ecological Monographs 75:3–35.

Hough, P., and M. Robertson. 2009. Mitigation under Section 404 of the Clean Water Act: where it comes from, what it means. Wetlands Ecology and Management 17:15–33.

Iheaturu, C. J., S. Hepner, J. L. Batchelor, G. A. Agonvonon, F. O. Akinyemi, V. R. Wingate, and C. Ifejika Speranza. 2024. Integrating UAV LiDAR and multispectral data to assess forest status and map disturbance severity in a West African forest patch. Ecological Informatics 84:102876.

IPBS. 2023. Thematic Assessment Report on Invasive Alien Species and their Control of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Page 952. IPBES secretariat, Bonn, Germany.

IPCC. 2023. Climate Change 2021 – The Physical Science Basis: Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. First edition. Cambridge University Press.

Jarrett, C., L. L. Powell, T. T. R. Claire, C. Kowo, D. F. Ferreira, A. L. S. Quiñones, A. J. Welch, D. T. Haydon, and J. Matthiopoulos. 2025. Food webs can deliver win–win strategies for tropical agroforestry and biodiversity conservation. ECOLOGICAL APPLICATIONS.

Johnson, N., M. L. Druckenmiller, F. Danielsen, and P. L. Pulsifer. 2021. The Use of Digital Platforms for Community-Based Monitoring. BioScience 71:452–466.

Jwaideh, M. A. A., E. H. Sutanudjaja, and C. Dalin. 2022. Global impacts of nitrogen and phosphorus fertiliser use for major crops on aquatic biodiversity. The International Journal of Life Cycle Assessment 27:1058–1080.

Karlsson, M., and K. Edvardsson Björnberg. 2021. Ethics and biodiversity offsetting. Conservation Biology 35:578–586.
Kates, R. W., W. C. Clark, R. Corell, J. M. Hall, C. C. Jaeger, I. Lowe, J. J. McCarthy, H. J. Schellnhuber, B. Bolin, N. M. Dickson, S. Faucheux, G. C. Gallopin, A. Grübler, B. Huntley, J. Jäger, N. S. Jodha, R. E. Kasperson, A. Mabogunje, P. Matson, H. Mooney, B. Moore, T. O'Riordan, and U. Svedin. 2001. Sustainability Science. Science 292:641–642.
Kates, R. W., and T. M. Parris. 2003. Long-term trends and a sustainability transition. Proceedings of the National Academy of Sciences 100:8062–8067.
Kelling, S., W. M. Hochachka, D. Fink, M. Riedewald, R. Caruana, G. Ballard, and G. Hooker. 2009. Data-intensive Science: A New Paradigm for Biodiversity Studies. BioScience 59:613–620.
Kelly, R. P., J. A. Port, K. M. Yamahara, R. G. Martone, N. Lowell, P. F. Thomsen, M. E. Mach, M. Bennett, E. Prahler, M. R. Caldwell, and L. B. Crowder. 2014. Harnessing DNA to improve environmental management. Science 344:1455–1456.
Kery, M., and J. A. Royle. 2020. Applied Hierarchical Modeling in Ecology: Analysis of Distribution, Abundance and Species Richness in R and BUGS: Volume 2: Dynamic and Advanced Models. Academic Press.
Kiesecker, J. M., H. Copeland, A. Pocewicz, and B. McKenney. 2010. Development by design: blending landscape-level planning with the mitigation hierarchy. Frontiers in Ecology and the Environment 8:261–266.
Kindsvater, H. K. H. K., N. K. N. K. Dulvy, C. Horswill, M. J. M.-J. M. J. M.-J. Juan-Jordá, M. Mangel, and J. Matthiopoulos. 2018. Overcoming the Data Crisis in Biodiversity Conservation. Trends in Ecology and Evolution 33:676–688.
Kirchhoff, T. 2012. Pivotal cultural values of nature cannot be integrated into the ecosystem services framework. Proceedings of the National Academy of Sciences 109.
Knott, E., A. H. Rao, K. Summers, and C. Teeger. 2022. Interviews in the social sciences. Nature Reviews Methods Primers 2:73.
Kreibich, N., and L. Hermwille. 2021. Caught in between: credibility and feasibility of the voluntary carbon market post-2020. Climate Policy 21:939–957.
Lake, T. A., R. D. Briscoe Runquist, and D. A. Moeller. 2022. Deep learning detects invasive plant species across complex landscapes using Worldview-2 and Planetscope satellite imagery. Remote Sensing in Ecology and Conservation 8:875–889.
Lamba, A., P. Cassey, R. R. Segaran, and L. P. Koh. 2019. Deep learning for environmental conservation. Current Biology 29:R977–R982.
Lang, D. J., A. Wiek, M. Bergmann, M. Stauffacher, P. Martens, P. Moll, M. Swilling, and C. J. Thomas. 2012. Transdisciplinary research in sustainability science: practice, principles, and challenges. Sustainability Science 7:25–43.
Langridge, R., and C. Ansell. 2024. How did we get here? The evolution of a polycentric system of groundwater governance. Ecology and Society 29:art34.
Lembi, R., C. Wentworth, and J. Hodbod. 2024. Recipe for a scenario: Moving from vision to actionable pathways towards sustainable futures. Progress in Environmental Geography 3:89–114.
Lenton, T. M., Milkoreit, M., Willcock, S., Abrams, J.F., Armstrong, McKay, D.I., Buxton, J.E., Donges, J.F., Loriani, S., Wunderling, N., Alkemade, F., Barrett, M., Constantino, S., Powell, T., Smith, S.R., Boulton, C. A., Pinho, P., Dijkstra, H., Pearce-Kelly, P., Roman-Cuesta, and R.M., Dennis, D. (eds). 2025. The Global Tipping Points Report 2025. University of Exeter, Exeter, UK.
Levrel, H., S. Pioch, and R. Spieler. 2012. Compensatory mitigation in marine ecosystems: Which indicators for assessing the "no net loss" goal of ecosystem services and ecological functions? Marine Policy 36:1202–1210.
Lewison, R. L., L. B. Crowder, B. P. Wallace, J. E. Moore, T. Cox, R. Zydelis, S. McDonald, A. DiMatteo, D. C. Dunn, C. Y. Kot, R. Bjorkland, S. Kelez, C. Soykan, K. R. Stewart, M. Sims, A. Boustany, A. J. Read, P. Halpin, W. J. Nichols, and C. Safina. 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proceedings of the National Academy of Sciences 111:5271–5276.
Linardatos, P., V. Papastefanopoulos, and S. Kotsiantis. 2020. Explainable AI: A Review of Machine Learning Interpretability Methods. Entropy 23:18.
Lira, P. K., M. de Souza Leite, and J. P. Metzger. 2019. Temporal Lag in Ecological Responses to Landscape Change: Where Are We Now? Current Landscape Ecology Reports 4:70–82.
Loreau, M., y Hector, and F. Isbell. 2022. The Ecological and Societal Consequences of Biodiversity Loss. First edition. Wiley.
Lukyanenko, R., A. Wiggins, and H. K. Rosser. 2020. Citizen Science: An Information Quality Research Frontier. Information Systems Frontiers 22:961–983.
Lynas, M., B. Z. Houlton, and S. Perry. 2021. Greater than 99% consensus on human caused climate change in the peer-reviewed scientific literature. Environmental Research Letters 16:114005.
Mac Aodha, O., R. Gibb, K. E. Barlow, E. Browning, M. Firman, R. Freeman, B. Harder, L. Kinsey, G. R. Mead, S. E. Newson, I. Pandourski, S. Parsons, J. Russ, A. Szodoray-Paradi, F. Szodoray-Paradi, E. Tilova, M. Girolami, G. Brostow, and K. E. Jones. 2018. Bat detective—Deep learning tools for bat acoustic signal detection. PLOS Computational Biology 14:e1005995.
Maron, M., S. Brownlie, J. W. Bull, M. C. Evans, A. Von Hase, F. Quétier, J. E. M. Watson, and A. Gordon. 2018. The many meanings of no net loss in environmental policy. Nature Sustainability 1:19–27.
Maron, M., J. W. Bull, M. C. Evans, and A. Gordon. 2015. Locking in loss: Baselines of decline in Australian biodiversity offset policies. Biological Conservation 192:504–512.
Maron, M., C. D. Ives, H. Kujala, J. W. Bull, F. J. F. Maseyk, S. Bekessy, A. Gordon, J. E. M. Watson, P. E. Lentini, P. Gibbons, H. P. Possingham, R. J. Hobbs, D. A. Keith, B. A. Wintle, and M. C. Evans. 2016. Taming a Wicked Problem: Resolving Controversies in Biodiversity Offsetting. BioScience 66:489–498.
Maron, M., F. Quétier, M. Sarmiento, K. Ten Kate, M. C. Evans, J. W. Bull, J. P. G. Jones, S. O. S. E. Zu Ermgassen, E. J. Milner-Gulland, S. Brownlie, J. Treweek, and A. Von Hase. 2023. 'Nature positive' must incorporate, not undermine, the mitigation hierarchy. Nature Ecology & Evolution 8:14–17.
Mayfield, H. J., J. Bird, M. Cox, G. Dutson, T. Eyre, K. Raiter, J. Ringma, and M. Maron. 2022. Guidelines for selecting an appropriate currency in biodiversity offset transactions. Journal of Environmental Management 322:116060.
McClure, E. C., M. Sievers, C. J. Brown, C. A. Buelow, E. M. Ditria, M. A. Hayes, R. M. Pearson, V. J. D. Tulloch, R. K. F. Unsworth, and R. M. Connolly. 2020. Artificial Intelligence Meets Citizen Science to Supercharge Ecological Monitoring. Patterns 1:100109.

McGill, B. J., M. Dornelas, N. J. Gotelli, and A. E. Magurran. 2015. Fifteen forms of biodiversity trend in the Anthropocene. Trends in Ecology & Evolution 30:104–113.
McVittie, A., and M. Faccioli. 2020. Biodiversity and ecosystem services net gain assessment: A comparison of metrics. Ecosystem Services 44:101145.
Miltenberger, O., C. Jospe, and J. Pittman. 2021. The Good Is Never Perfect: Why the Current Flaws of Voluntary Carbon Markets Are Services, Not Barriers to Successful Climate Change Action. Frontiers in Climate 3:686516.
Moilanen, A., J. Jalkanen, P. Halme, E. Nieminen, J. S. Kotiaho, and H. Kujala. 2024. Monitoring in biodiversity offsetting. Global Ecology and Conservation 54:e03039.
Moilanen, A., and J. Laitila. 2016. FORUM: Indirect leakage leads to a failure of avoided loss biodiversity offsetting. Journal of Applied Ecology 53:106–111.
Moilanen, A., A. Wilson, Kerrie, and H. P. Possingham. 2008. Spatial Conservation Prioritization: Quantitative Methods and Computational Tools. Oxford UP.
Moon, K., D. A. Blackman, V. M. Adams, R. M. Colvin, F. Davila, M. C. Evans, S. R. Januchowski-Hartley, N. J. Bennett, H. Dickinson, C. Sandbrook, K. Sherren, F. A. V. St. John, L. Van Kerkhoff, and C. Wyborn. 2019. Expanding the role of social science in conservation through an engagement with philosophy, methodology, and methods. Methods in Ecology and Evolution 10:294–302.
Musante, K., and B. R. Dewalt. 2011. Participant observation: a guide for fieldworkers. Rowman Littlefield, Md, Lanham, Md.
Mutillod, C., É. Buisson, G. Mahy, R. Jaunatre, J. M. Bullock, L. Tatin, and T. Dutoit. 2024. Ecological restoration and rewilding: two approaches with complementary goals? Biological Reviews 99:820–836.
Narassimhan, E., K. S. Gallagher, S. Koester, and J. R. Alejo. 2018. Carbon pricing in practice: a review of existing emissions trading systems. Climate Policy 18:967–991.
Needham, K., F. P. De Vries, P. R. Armsworth, and N. Hanley. 2019. Designing markets for biodiversity offsets: Lessons from tradable pollution permits. Journal of Applied Ecology 56:1429–1435.
Newman, K. B., S. T. Buckland, B. J. T. Morgan, R. King, D. L. Borchers, D. J. Cole, P. Besbeas, O. Gimenez, and L. Thomas. 2014. Modelling Population Dynamics: Model Formulation, Fitting and Assessment Using State-Space Methods. Springer, New York.
Nguyen, C., U. Latacz-Lohmann, and N. Hanley. 2024. Agri-environmental Schemes Require Improved Design for Better Outcomes. EuroChoices 23:14–21.
Nguyen, C., U. Latacz-Lohmann, N. Hanley, S. Schilizzi, and S. Iftekhar. 2022. Spatial Coordination Incentives for landscape-scale environmental management: A systematic review. Land Use Policy 114:105936.
Niner, H. J., B. Milligan, P. J. S. Jones, and C. A. Styan. 2017. A global snapshot of marine biodiversity offsetting policy. Marine Policy 81:368–374.
Norouzzadeh, M. S., A. Nguyen, M. Kosmala, A. Swanson, M. S. Palmer, C. Packer, and J. Clune. 2018. Automatically identifying, counting, and describing wild animals in camera-trap images with deep learning. Proceedings of the National Academy of Sciences 115.
Ockendon, N., D. H. L. Thomas, J. Cortina, W. M. Adams, T. Aykroyd, B. Barov, L. Boitani, A. Bonn, C. Branquinho, M. Brombacher, C. Burrell, S. Carver, H. Q. P. Crick, B. Duguy, S. Everett, B. Fokkens, R. J. Fuller, D. W. Gibbons, R. Gokhelashvili, C. Griffin, D. J. Halley, P. Hotham, F. M. R. Hughes, A. A. Karamanlidis, C. J. McOwen, L. Miles, R. Mitchell, M. R. W. Rands, J. Roberts, C. J. Sandom, J. W. Spencer, E. Ten Broeke, E. R. Tew, C. D. Thomas, A. Timoshyna, R. K. F. Unsworth, S. Warrington, and W. J. Sutherland. 2018. One hundred priority questions for landscape restoration in Europe. Biological Conservation 221:198–208.
OECD. 2025. Scaling Up Biodiversity-Positive Incentives: Delivering on Target 18 of the Global Biodiversity Framework. OECD Publishing.
Ostrom, E. 2010. Polycentric systems for coping with collective action and global environmental change. Global Environmental Change 20:550–557.
Panwar, N. L., S. C. Kaushik, and S. Kothari. 2011. Role of renewable energy sources in environmental protection: A review. Renewable and Sustainable Energy Reviews 15:1513–1524.
Parmesan, C., and G. Yohe. 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature 421:37–42.
Paul, C., N. Hanley, S. T. Meyer, C. Fürst, W. W. Weisser, and T. Knoke. 2020. On the functional relationship between biodiversity and economic value. Science Advances 6:eaax7712.
Pavoine, S., and M. B. Bonsall. 2011. Measuring biodiversity to explain community assembly: a unified approach. Biological Reviews 86:792–812.
Pepin, K. M., T. J. Smyser, A. J. Davis, R. S. Miller, S. McKee, K. C. VerCauteren, W. Kendall, and C. Slootmaker. 2020. Optimal spatial prioritization of control resources for elimination of invasive species under demographic uncertainty. Ecological Applications 30:e02126.
Perino, A., H. M. Pereira, L. M. Navarro, N. Fernández, J. M. Bullock, S. Ceauşu, A. Cortés-Avizanda, R. Van Klink, T. Kuemmerle, A. Lomba, G. Pe'er, T. Plieninger, J. M. Rey Benayas, C. J. Sandom, J.-C. Svenning, and H. C. Wheeler. 2019. Rewilding complex ecosystems. Science 364:eaav5570.
Pettorelli, N., M. Wegmann, A. Skidmore, S. Mücher, T. P. Dawson, M. Fernandez, R. Lucas, M. E. Schaepman, T. Wang, B. O'Connor, R. H. G. Jongman, P. Kempeneers, R. Sonnenschein, A. K. Leidner, M. Böhm, K. S. He, H. Nagendra, G. Dubois, T. Fatoyinbo, M. C. Hansen, M. Paganini, H. M. De Klerk, G. P. Asner, J. T. Kerr, A. B. Estes, D. S. Schmeller, U. Heiden, D. Rocchini, H. M. Pereira, E. Turak, N. Fernandez, A. Lausch, M. A. Cho, D. Alcaraz-Segura, M. A. McGeoch, W. Turner, A. Mueller, V. St-Louis, J. Penner, P. Vihervaara, A. Belward, B. Reyers, and G. N. Geller. 2016. Framing the concept of satellite remote sensing essential biodiversity variables: challenges and future directions. Remote Sensing in Ecology and Conservation 2:122–131.
Pettorelli, N., J. Williams, H. Schulte To Bühne, and M. Crowson. 2025. Deep learning and satellite remote sensing for biodiversity monitoring and conservation. Remote Sensing in Ecology and Conservation 11:123–132.
Phalan, B., G. Hayes, S. Brooks, D. Marsh, P. Howard, B. Costelloe, B. Vira, A. Kowalska, and S. Whitaker. 2018. Avoiding impacts on biodiversity through strengthening the first stage of the mitigation hierarchy. Oryx 52:316–324.
Phalan, B., M. Onial, A. Balmford, and R. E. Green. 2011. Reconciling Food Production and Biodiversity Conservation: Land Sharing and Land Sparing Compared. Science 333:1289–1291.

Pichler, M., and F. Hartig. 2023. Machine learning and deep learning—A review for ecologists. Methods in Ecology and Evolution 14:994–1016.
Pinsky, M. L., O. P. Jensen, D. Ricard, and S. R. Palumbi. 2011. Unexpected patterns of fisheries collapse in the world's oceans. Proceedings of the National Academy of Sciences 108:8317–8322.
Pope, J., A. Morrison-Saunders, A. Bond, and F. Retief. 2021. When is an Offset Not an Offset? A Framework of Necessary Conditions for Biodiversity Offsets. Environmental Management 67:424–435.
Pörtner, H.-O., R. J. Scholes, A. Arneth, D. K. A. Barnes, M. T. Burrows, S. E. Diamond, C. M. Duarte, W. Kiessling, P. Leadley, S. Managi, P. McElwee, G. Midgley, H. T. Ngo, D. Obura, U. Pascual, M. Sankaran, Y. J. Shin, and A. L. Val. 2023. Overcoming the coupled climate and biodiversity crises and their societal impacts. Science 380:eabl4881.
Ribaudo, C., J. Tison-Rosebery, D. Buquet, G. Jan, A. Jamoneau, G. Abril, P. Anschutz, and V. Bertrin. 2018. Invasive Aquatic Plants as Ecosystem Engineers in an Oligo-Mesotrophic Shallow Lake. Frontiers in Plant Science 9:1781.
Richardson, K., W. Steffen, W. Lucht, J. Bendtsen, S. E. Cornell, J. F. Donges, M. Drüke, I. Fetzer, G. Bala, W. Von Bloh, G. Feulner, S. Fiedler, D. Gerten, T. Gleeson, M. Hofmann, W. Huiskamp, M. Kummu, C. Mohan, D. Nogués-Bravo, S. Petri, M. Porkka, S. Rahmstorf, S. Schaphoff, K. Thonicke, A. Tobian, V. Virkki, L. Wang-Erlandsson, L. Weber, and J. Rockström. 2023. Earth beyond six of nine planetary boundaries. Science Advances 9:eadh2458.
Ripple, W. J., K. Abernethy, M. G. Betts, G. Chapron, R. Dirzo, M. Galetti, T. Levi, P. A. Lindsey, D. W. Macdonald, B. Machovina, T. M. Newsome, C. A. Peres, A. D. Wallach, C. Wolf, and H. Young. 2016. Bushmeat hunting and extinction risk to the world's mammals. Royal Society Open Science 3:160498.
Robinson, J. M., P. A. Harrison, S. Mavoa, and M. F. Breed. 2022. Existing and emerging uses of drones in restoration ecology. Methods in Ecology and Evolution 13:1899–1911.
Rocchini, D., S. Luque, N. Pettorelli, L. Bastin, D. Doktor, N. Faedi, H. Feilhauer, J. Féret, G. M. Foody, Y. Gavish, S. Godinho, W. E. Kunin, A. Lausch, P. J. Leitão, M. Marcantonio, M. Neteler, C. Ricotta, S. Schmidtlein, P. Vihervaara, M. Wegmann, and H. Nagendra. 2018. Measuring β-diversity by remote sensing: A challenge for biodiversity monitoring. Methods in Ecology and Evolution 9:1787–1798.
Rockström, J., W. Steffen, K. Noone, Å. Persson, F. S. Chapin, E. F. Lambin, T. M. Lenton, M. Scheffer, C. Folke, H. J. Schellnhuber, B. Nykvist, C. A. De Wit, T. Hughes, S. Van Der Leeuw, H. Rodhe, S. Sörlin, P. K. Snyder, R. Costanza, U. Svedin, M. Falkenmark, L. Karlberg, R. W. Corell, V. J. Fabry, J. Hansen, B. Walker, D. Liverman, K. Richardson, P. Crutzen, and J. A. Foley. 2009. A safe operating space for humanity. Nature 461:472–475.
Rosenberg, Y., D. Wiedenhofer, D. Virág, G. Bar-Sella, L. Greenspoon, B. Herrnstadt, L. Akenji, R. Phillips, and R. Milo. 2025. Human biomass movement exceeds the biomass movement of all land animals combined. Nature Ecology & Evolution.
Ruiz-Jaen, M. C., and T. Mitchell Aide. 2005. Restoration Success: How Is It Being Measured? Restoration Ecology 13:569–577.
Rust, N. A., A. Abrams, D. W. S. Challender, G. Chapron, A. Ghoddousi, J. A. Glikman, C. H. Gowan, C. Hughes, A. Rastogi, A. Said, A. Sutton, N. Taylor, S. Thomas, H. Unnikrishnan, A. D. Webber, G. Wordingham, and C. M. Hill. 2017. Quantity Does Not Always Mean Quality: The Importance of Qualitative Social Science in Conservation Research. Society & Natural Resources 30:1304–1310.
Sage, R. F. 2020. Global change biology: A primer. Global Change Biology 26:3–30.
Sahu, A., N. Kumar, C. Pal Singh, and M. Singh. 2023. Environmental DNA (eDNA): Powerful technique for biodiversity conservation. Journal for Nature Conservation 71:126325.
Sanders, D., R. Kehoe, and F. J. F. van Veen. 2015. Experimental Evidence for the Population-Dynamic Mechanisms Underlying Extinction Cascades of Carnivores. Current Biology 25:3106–3109.
Schmalensee, R., and R. N. Stavins. 2013. The SO_2 Allowance Trading System: The Ironic History of a Grand Policy Experiment. Journal of Economic Perspectives 27:103–122.
Scholte, S. S. K., B. T. Van Zanten, P. H. Verburg, and A. J. A. Van Teeffelen. 2016. Willingness to offset? Residents' perspectives on compensating impacts from urban development through woodland restoration. Land Use Policy 58:403–414.
Schuster, L., S. Trevathan-Tackett, P. Carnell, K. Morris, B. Mole, and M. E. Malerba. 2025. Restoring riparian wetlands for carbon and nitrogen benefits and other critical ecosystem functions. Journal of Environmental Management 391:126433.
Schuur, E. A. G., A. D. McGuire, C. Schädel, G. Grosse, J. W. Harden, D. J. Hayes, G. Hugelius, C. D. Koven, P. Kuhry, D. M. Lawrence, S. M. Natali, D. Olefeldt, V. E. Romanovsky, K. Schaefer, M. R. Turetsky, C. C. Treat, and J. E. Vonk. 2015. Climate change and the permafrost carbon feedback. Nature 520:171–179.
Scottish Government. 2023. National Planning Framework 4.
Seaton, F. M., S. G. Jarvis, and P. A. Henrys. 2024. Spatio-temporal data integration for species distribution modelling in R-INLA. Methods in Ecology and Evolution:2041–210X.14356.
Sharma, R., M. Barange, V. Agostini, P. Barros, N. L. Gutierrez, D. Fernandez Reguera, M. Vasconcellos, C. Tiffay, and P. Levontin. 2025. Review of the state of world marine fishery resources – 2025. FAO.
Shaw, J. L. A., L. J. Clarke, S. D. Wedderburn, T. C. Barnes, L. S. Weyrich, and A. Cooper. 2016. Comparison of environmental DNA metabarcoding and conventional fish survey methods in a river system. Biological Conservation 197:131–138.
Simpson, K., P. R. Armsworth, M. Dallimer, M. Nthambi, F. P. De Vries, and N. Hanley. 2023. Improving the ecological and economic performance of agri-environment schemes: Payment by modelled results versus payment for actions. Land Use Policy 130:106688.
Sodhi, N. S., R. Butler, W. F. Laurance, and L. Gibson. 2011. Conservation successes at micro-, meso- and macroscales. Trends in Ecology & Evolution 26:585–594.
Spangenberg, J. H. 2011. Sustainability science: a review, an analysis and some empirical lessons. Environmental Conservation 38:275–287.
Spatz, D. R., N. D. Holmes, D. J. Will, S. Hein, Z. T. Carter, R. M. Fewster, B. Keitt, P. Genovesi, A. Samaniego, D. A. Croll, B. R. Tershy, and J. C. Russell. 2022. The global contribution of invasive vertebrate eradication as a key island restoration tool. Scientific Reports 12:13391.
Stavins, R. N. 1998. What Can We Learn from the Grand Policy Experiment? Lessons from SO2 Allowance Trading. Journal of Economic Perspectives 12:69–88.

Steffen, W., K. Richardson, J. Rockström, S. E. Cornell, I. Fetzer, E. M. Bennett, R. Biggs, S. R. Carpenter, W. De Vries, C. A. De Wit, C. Folke, D. Gerten, J. Heinke, G. M. Mace, L. M. Persson, V. Ramanathan, B. Reyers, and S. Sörlin. 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347:1259855.
Steneck, R. S. 2012. Apex predators and trophic cascades in large marine ecosystems: Learning from serendipity. Proceedings of the National Academy of Sciences 109:7953–7954.
Stephenson, J. 2010. The Dimensional Landscape Model: Exploring Differences in Expressing and Locating Landscape Qualities. Landscape Research 35:299–318.
Stuart, A., A. Bond, A. Franco, C. Gerrard, J. Baker, K. Ten Kate, T. Butterworth, J. Bull, and J. Treweek. 2024. How England got to Mandatory Biodiversity Net Gain: A Timeline. SSRN.
Sullivan, D. P., C. F. Winsnes, L. Åkesson, M. Hjelmare, M. Wiking, R. Schutten, L. Campbell, H. Leifsson, S. Rhodes, A. Nordgren, K. Smith, B. Revaz, B. Finnbogason, A. Szantner, and E. Lundberg. 2018. Deep learning is combined with massive-scale citizen science to improve large-scale image classification. Nature Biotechnology 36:820–832.
Taherzadeh, O., and P. Howley. 2018. No net loss of what, for whom?: stakeholder perspectives to Biodiversity Offsetting in England. Environment, Development and Sustainability 20:1807–1830.
Tilman, D. 1987. Secondary Succession and the Pattern of Plant Dominance Along Experimental Nitrogen Gradients. Ecological Monographs 57:189–214.
Tilman, D., M. Clark, D. R. Williams, K. Kimmel, S. Polasky, and C. Packer. 2017. Future threats to biodiversity and pathways to their prevention. Nature 546:73–81.
Tilman, D., F. Isbell, and J. M. Cowles. 2014. Biodiversity and Ecosystem Functioning. Annual Review of Ecology, Evolution, and Systematics 45:471–493.
TNFD. 2023. Recommendations-of-the-Taskforce-on-Nature-related-Financial-Disclosures.
TNFD. 2025. TNFD-Status-Report.
Toderas, M. 2025. Artificial Intelligence for Sustainability: A Systematic Review and Critical Analysis of AI Applications, Challenges, and Future Directions. Sustainability 17:8049.
Tupala, A.-K., S. Huttunen, and P. Halme. 2022. Social impacts of biodiversity offsetting: A review. Biological Conservation 267:109431.
UK Gov. 2021. Environment Act 2021.
UN. 1992. Rio Earth Summit.
UN. 2015. UN Sustainable Development Goals.
UN. 2019. UN Decade on Ecosystem Restoration.
UNFCC. 1997. The Kyoto protocol.
UNFCC. 2015. The Paris Agreement.
UNFCC. 2016. Key decisions relevant for reducing emissions from deforestation and forest degradation in developing countries (REDD+).
Vaz, J., J. Coria, and V. Inkinen. 2026. Environmental offsetting: What drives the choice of offset mechanism in the US Wetland Mitigation Program? Ecological Economics 239:108755.
Vimercati, G., M. Labadesse, T. Dejean, and J. Secondi. 2020. Assessing the effect of landscape features on pond colonisation by an elusive amphibian invader using environmental DNA. Freshwater Biology 65:502–513.
Virah-Sawmy, D., and B. Sturmberg. 2025. Socio-economic and environmental impacts of renewable energy deployments: A review. Renewable and Sustainable Energy Reviews 207:114956.
Virah-Sawmy, M., J. Ebeling, and R. Taplin. 2014. Mining and biodiversity offsets: A transparent and science-based approach to measure “no-net-loss.” Journal of Environmental Management 143:61–70.
Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human Domination of Earth’s Ecosystems. Science:494–499.
Voyer, M., N. Gollan, K. Barclay, and W. Gladstone. 2015. ‘It’s part of me’; understanding the values, images and principles of coastal users and their influence on the social acceptability of MPAs. Marine Policy 52:93–102.
Walther, G.-R., E. Post, P. Convey, A. Menzel, C. Parmesan, T. J. C. Beebee, J.-M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein. 2002. Ecological responses to recent climate change. Nature 416:389–395.
Wartmann, F. M., and J. Lorimer. 2024. Messy natures: The political aesthetics of nature recovery. People and Nature 6:2564–2576.
Watling, L., and E. A. Norse. 1998. Disturbance of the Seabed by Mobile Fishing Gear: A Comparison to Forest Clearcutting. Conservation Biology 12:1180–1197.
Wesselkamp, M., N. Moser, M. Kalweit, J. Boedecker, and C. F. Dormann. 2024. Process-Informed Neural Networks: A Hybrid Modelling Approach to Improve Predictive Performance and Inference of Neural Networks in Ecology and Beyond. Ecology Letters 27:e70012.
Witte, F., T. Goldschmidt, J. Wanink, M. Van Oijen, K. Goudswaard, E. Witte-Maas, and N. Bouton. 1992. The destruction of an endemic species flock: quantitative data on the decline of the haplochromine cichlids of Lake Victoria. Environmental Biology of Fishes 34:1–28.
Wright, A. J., K. E. Barry, C. J. Lortie, and R. M. Callaway. 2021. Biodiversity and ecosystem functioning: Have our experiments and indices been underestimating the role of facilitation? Journal of Ecology 109:1962–1968.
Wunder, S. 2015. Revisiting the concept of payments for environmental services. Ecological Economics 117:234–243.
Yan, D., J. G. Mills, N. J. C. Gellie, A. Bissett, A. J. Lowe, and M. F. Breed. 2018. High-throughput eDNA monitoring of fungi to track functional recovery in ecological restoration. Biological Conservation 217:113–120.
Yan, P., M. Fernández-Martínez, K. Van Meerbeek, G. Yu, M. Migliavacca, and N. He. 2023. The essential role of biodiversity in the key axes of ecosystem function. Global Change Biology 29:4569–4585.
Yu, Z., C. Bu, and Y. Li. 2025. Machine learning for ecological analysis. Chemical Engineering Journal 507:160780.
Zu Ermgassen, S. O. S. E., S. Marsh, K. Ryland, E. Church, R. Marsh, and J. W. Bull. 2021. Exploring the ecological outcomes of mandatory biodiversity net gain using evidence from early-adopter jurisdictions in England. Conservation Letters 14:e12820.