nature sustainability

ARTICLES

https://doi.org/10.1038/s41893-021-00692-9

Check for updates

The economic consequences of conserving or restoring sites for nature

Richard B. Bradbury[1,2] ✉, Stuart H. M. Butchart[2,3], Brendan Fisher[4], Francine M. R. Hughes[5], Lisa Ingwall-King[6], Michael A. MacDonald[7], Jennifer C. Merriman[8], Kelvin S.-H. Peh[2,9], Anne-Sophie Pellier[3], David H. L. Thomas[10], Rosie Trevelyan[11] and Andrew Balmford[2]

Nature provides many benefits for people, yet there are few data on how changes at individual sites impact the net value of ecosystem service provision. A 2002 review found only five analyses comparing the net economic benefits of conserving nature versus pursuing an alternative, more intensive human use. Here we revisit this crucial comparison, synthesizing recent data from 62 sites worldwide. In 24 cases with economic estimates of services, conservation or restoration benefits (for example, greenhouse gas regulation, flood protection) tend to outweigh those private benefits (for example, profits from agriculture or logging) driving change to the alternative state. Net benefits rise rapidly with increasing social cost of carbon. Qualitative data from all 62 sites suggest that monetization of additional services would further increase the difference. Although conservation and restoration did not universally provide greater net value than the alternative state, across a large, geographically and contextually diverse sample, our findings indicate that at current levels of habitat conversion, conserving and restoring sites typically benefits human prosperity.

Recent decades have seen increasing recognition of the economic and human well-being consequences of degradation of nature[1–5]. However, the degradation continues, perhaps in part because inadequate steps are taken to ensure that planning and management decisions are informed by estimates of their net consequences for benefits (ecosystem services) to different stakeholders[6]. Although criticisms of valuation are well rehearsed, from the ethical to the analytical[7,8], cost–benefit and cost-effectiveness analyses are demanded in many regulatory contexts and provide a useful, if partial, lens on the impacts of decisions on human prosperity. An early review[9] found only five site-level studies worldwide comparing the aggregate economic value of flows of ecosystem services delivered by the site when relatively intact with its potential economic value when converted to more human-dominated forms of use. Although tiny, this sample suggested retention of (or sustainably managing) areas of natural habitat typically delivered net economic benefits to people. While striking, this result was almost certainly conservative, given that assessments of service flows at one point in time tend to fail to consider whether those flows can be maintained sustainably into the future[10]. Despite growing understanding of the economic consequences of conserving or restoring nature[11–15] and development of new tools for ecosystem service assessment[16], remarkably few additional studies[3,17,18] have investigated this key question of the net economic value of conserving (or restoring) individual sites.

A new data synthesis on the net benefits of conservation

We addressed this lack of evidence by synthesizing data from a relatively large sample of published and unpublished studies that used the framework of the Toolkit for Ecosystem Service Site-Based Assessment (TESSA; http://tessa.tools)[19] to develop the earlier review[9], evaluating the net consequences of plausible changes in habitat state on the benefits provided by particular sites. TESSA provides relatively simple methods, within a consistent framework, for evaluating the difference in ecosystem service flows, in biophysical and (where possible) economic terms, provided by a site under contrasting states. The resulting analyses do not claim to be full economic valuations but do aspire to cover as many of the main services provided by a site as possible, in either state, and always include the services driving state change. The toolkit emphasizes broad stakeholder participation—including those benefiting most from the change in state—to identify the main ecosystem services and plausible alternative land uses and to facilitate local data collection. Our literature review yielded information on 15 sites (13 in International Scientific Indexing (ISI) journal papers) that met our criteria (Methods) for analysis. Unpublished studies provided information from 47 additional sites (Supplementary Data). The combined set of 62 sites spanned six continents (Supplementary Table 1), contrasting (1) a nature conservation state with a more human-modified state (for example, protected area versus conversion to agriculture; 44 sites) or (2) an ecological restoration state with the pre-restoration (human-modified) state (for example, restoration to intertidal habitat versus coastal area claimed for agriculture; 18 sites). Henceforth, we refer to nature conservation and ecological restoration states as 'nature-focused' and the contrasting states as 'alternative'. These studies provided data on multiple services, including the most important private and toll (club) benefits

[1]RSPB Centre for Conservation Science, The Lodge, Sandy, UK. [2]Conservation Science Group, Department of Zoology, The David Attenborough Building, Cambridge, UK. [3]BirdLife International, The David Attenborough Building, Cambridge, UK. [4]Environmental Program, Gund Institute for Environment—Rubenstein School of Environment and Natural Resources, University of Vermont, Burlington, VT, USA. [5]Global Sustainability Institute, Anglia Ruskin University, Cambridge, UK. [6]United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC), Cambridge, UK. [7]RSPB Centre for Conservation Science, RSPB Cymru, Cardiff, UK. [8]WSP, Cambridge, UK. [9]School of Biological Sciences, University of Southampton, Southampton, UK. [10]The Cambridge Conservation Initiative, The David Attenborough Building, Cambridge, UK. [11]Tropical Biology Association, The David Attenborough Building, Cambridge, UK. ✉e-mail: Richard.bradbury@RSPB.org.uk

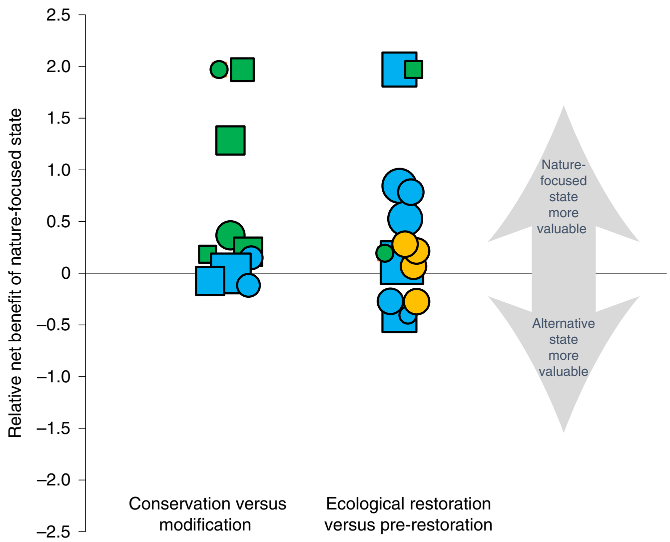

Fig. 1 | Relative net benefits of the nature-focused state at 24 sites. The NPV for the nature-focused state was divided by that for the alternative state and the result logged. Positive values therefore reflect greater net benefit in the nature-focused state while negative values reflect greater net benefit in the alternative state. Cases where the NPV of either state was negative, such that a ratio could not be calculated, were allocated values equivalent to either the largest ratio in the dataset (alternative state NPV negative) or the smallest ratio (nature-focused state NPV negative). For ease of interpretation, data points with very similar response values have been offset along the horizontal axis to minimize overlaps. Green, forest; blue, wetland; gold, other habitats. Squares represent sites with full costs included; circles represent sites with only partial costs included. Symbol size is scaled to the number of services measured. Time horizon 50 years; inter-state change period 20 years; social cost of carbon in year of study; discount rate 2%.

from marketed goods (see the following), as well as one or more public goods or common pool resources delivering unmarketed local or global benefits (Supplementary Data).

We first compared the net economic value of the nature-focused and alternative states at 24 sites that provided economic data (mean = 3.75 services valued per site, range 2 to 7). Service values were adjusted to international dollars for the year of study, one-off values were annualized over 20 years (over which we assumed the change in state would happen) and all service values of a given state were then summed and expressed as net present value (NPV) over a 50-year time horizon. We contrasted NPVs between nature-focused and alternative states, including restoration costs and management costs in both states for the ten sites that provided full cost data. We assumed (1) a social cost of carbon of \$31.21 per tonne CO_2-equivalent (tCO_2e) (ref. [20]), inflation-adjusted to the year of study (plus sensitivity tests of \$5; \$100), (2) a 20-year time horizon for state change (plus 10 years; 30 years), (3) a 50-year time horizon for accrual of costs and benefits (plus 30 years; 100 years) and (4) a discount rate of 2% (plus 1%; 10%).

Net economic benefits of conservation and restoration

Conservation tended to provide greater NPV than the alternative state (Fig. 1: conservation ≥ human-modified state for 8 of 10 sites, including 5 of 6 with full costs), especially for forests. Ecological restoration was also associated with greater NPV than the alternative state (ecological restoration > pre-restoration for 10 of 14 sites, including 3 of 4 with full costs). The state that had the higher NPV was not significantly associated with type of state change, country income class, broad habitat class, number of services quantified per site, whether full costs were included or whether the study was published (Supplementary Table 2).

We found that even with a social cost of carbon of \$0 per tonne, the nature-focused state NPV was greater than the alternative state NPV in 42% of cases (Supplementary Fig. 1), rising to almost 60% of cases when using the low-end social cost of carbon of \$5 per tonne and over 70% with \$31 per tonne. At \$31 per tonne, the nature-focused state NPV was greater than the alternative state NPV at 100% of forest sites. Altering the duration of the state change, the time horizon or the discount rate made negligible differences to our overall findings (Supplementary Fig. 2).

We then looked at how state changes affect benefit distribution by distinguishing whether the benefits of each service could be captured privately or could be enjoyed more widely. Services were divided into non-excludable goods (public goods and common pool resources), where the wider public can typically benefit, and excludable goods (private and toll goods (for example, charges to enter a protected area)), where the benefits are captured by relatively few (Supplementary Data). This is more comprehensive (and representative in our cases) than a simple public–private dichotomy. For both state change contrasts, the NPV of non-excludable goods tended to be greater in the nature-focused state than in the alternative state (Fig. 2: conservation versus degradation, 9 of 10 cases, $\chi^2_1 = 6.400$, $P = 0.022$; ecological restoration versus pre-restoration, 13 of 14 cases, $\chi^2_1 = 10.286$, $P = 0.002$). Contrary to what might be expected, given that change is usually driven by extraction of private goods, the NPV of excludable goods was regularly higher in the nature-focused state than in the alternative state (conservation versus degradation, 5 of 10 cases, $\chi^2_1 = 0.000$, $P = 1.000$; ecological restoration versus pre-restoration, 5 of 14 cases, $\chi^2_1 = 1.143$, $P = 0.424$).

Further support for conservation and restoration

Although these results suggest greater net value of the nature-focused state, our analysis may still be conservative because of lack of monetization of several other services at each site. We investigated whether this might systematically bias our results by using information in the original studies on the direction of change of these services. We did this first for the 24 sites included in the economic analysis, determining whether annual flows of each non-monetized service were qualitatively greater under the nature-focused state, greater under the alternative state or approximately equal (Methods) under the two states. We did the same for the remaining 38 sites, aggregating results across all services to determine whether benefits were (1) apparently greater under the nature-focused state (more services were greater in the nature-focused state than were equal or lower), (2) greater under the alternative state or (3) approximately equal under the two states.

This direction-of-change analysis suggests that the economic analysis may be conservative. First, across the main 24 sites, of 19 cases where a service was assessed for directional change but not monetized (15 of these relating to water quality, water provision or flood risk mitigation), 13 were greater in the nature-focused state, 4 were equal between states, and only 2 were greater in the alternative state. Second, among the 38 remaining sites, overall provision of services in the nature-focused state was greater than or approximately equal to that in the alternative state in all cases, suggesting that the 24 main sites are broadly representative of a wider pool (Fig. 3 and Supplementary Fig. 3). Among the 38 sites, the more services inspected at a site, the higher the probability that more services were greater under the nature-focused state (Supplementary Table 3: $\chi^2_1 = 5.760$, $P = 0.016$). The outcome was not significantly associated with country income class, habitat type, type of state change or whether the study was published (Supplementary Table 3).

For decision makers aiming to maximize profit, contrasting the net benefits of two states is informative. However, if working on a fixed budget with costs as the main consideration, decisions about

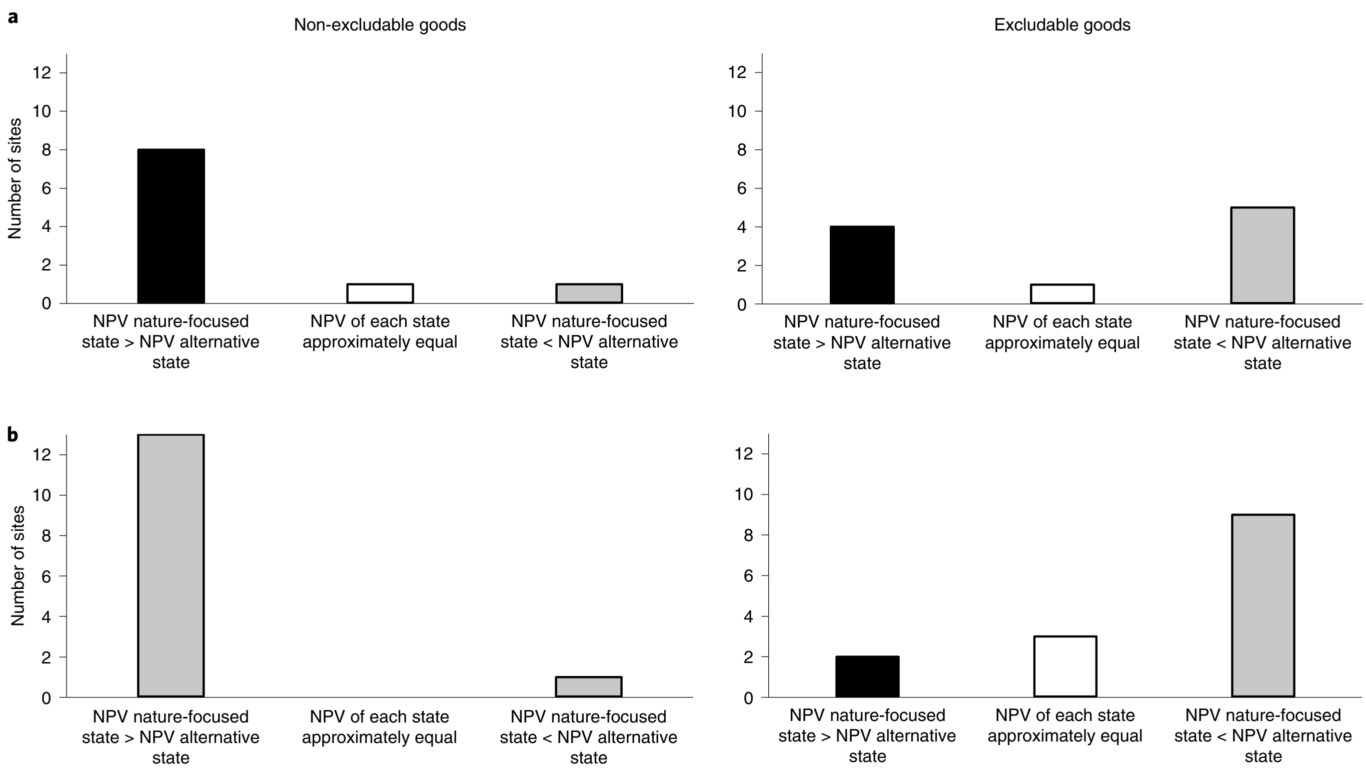

Fig. 2 | Summary of the number of sites at which NPV was greater in the nature-focused state, greater in the alternative state or approximately equal in the two states, for non-excludable and for excludable goods. a, Conservation versus modification ($n = 10$). **b**, Ecological restoration versus pre-restoration ($n = 14$).

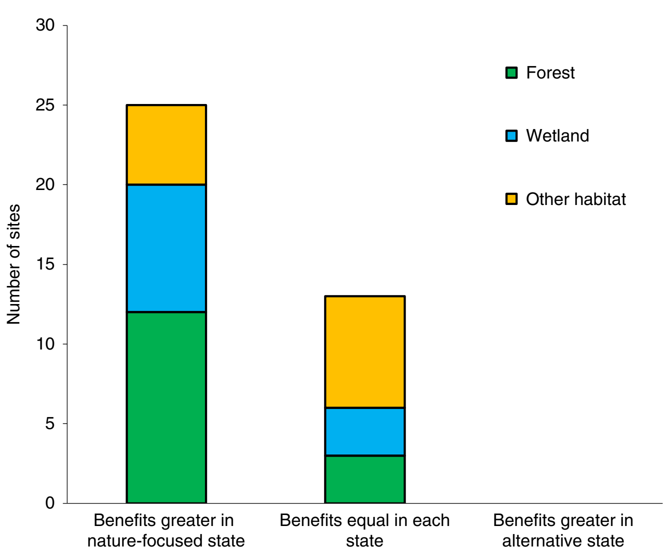

Fig. 3 | Benefits of conservation or ecological restoration at 38 sites without economic data, displayed by habitat. Bars show the number of sites at which overall benefits were greater in the nature-focused state (most services assessed to have qualitatively higher flows in the nature-focused state than in the alternative state), greater in the alternative state or equal in the two states.

site management will also reflect the benefit–cost ratio of alternative options. Of the ten sites that provided full costs, the benefit–cost ratio of the nature-focused state was greater than the alternative at five sites and equal to the alternative at one site (Fig. 4). Of the four sites at which the benefit–cost ratio was greater in the alternative state, two also had greater net benefits in the alternative state.

Conclusions and caveats

Our analysis shows that both conservation and ecological restoration bring considerable net benefits in terms of public goods and common pool resources, regardless of the habitat or type of ecosystem state change being considered. Non-excludable benefits typically outweigh the excludable benefits that normally drive change to an alternative state, such that the total value of the nature-focused state is greater, even when accounting for all costs. For four of the six studies where excludable benefits outweighed non-excludable benefits, the result was driven by commodity crops (cereals, sugar) with a relatively high price. However, commodity crop-driven changes were not always sufficient for excludable benefits to outweigh non-excludable benefits. Commodities (cereals, rubber, tea, cocoa) were also the main driver of change at 10 of the 18 sites where the nature-focused state provided greater overall value. Moreover, our qualitative data suggest these results are conservative: those services whose value could not be monetized were very likely to be greater in the nature-focused state, and the more services considered, the more likely that the nature-focused state delivered greater benefits than the alternative. Benefit–cost ratios also generally favour both conservation and ecological restoration as much as the alternative state. However, while these patterns hold for all goods combined and for non-excludable goods, the alternative state was often more valuable when only excludable goods were considered. Our findings thus provide a strong economic justification for incentives to encourage private landowners towards decisions that favour nature-focused land management to enhance overall social value. This could include negative incentives such as taxes and regulation[21] or positive incentives such as subsidies and payments for

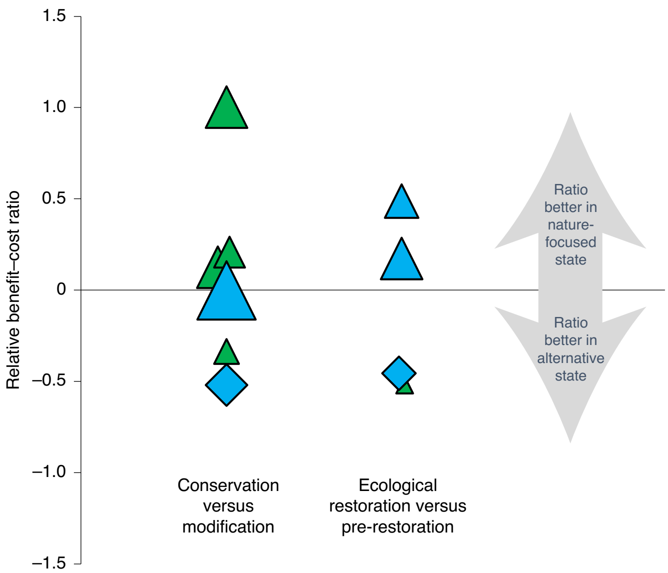

Fig. 4 | Relative benefit–cost ratio of the nature-focused state compared with that of the alternative state at the ten sites providing full cost data. The benefit–cost ratio in the nature-focused state was divided by that in the alternative state and the result logged (*y* axis). Positive values therefore reflect a higher benefit–cost ratio in the nature-focused state while negative values reflect a higher benefit–cost ratio in the alternative state. For ease of interpretation, data points with very similar response values have been offset along the horizontal axis to minimize overlaps. Green, forest; blue, wetland. Symbol size is scaled to the number of services measured. Triangles represent sites where net benefits were greater in the nature-focused state or equal between states; diamonds represent sites where net benefits were greater in the alternative state. Time horizon 50 years; inter-state change period 20 years; social cost of carbon in year of study; discount rate 2%.

ecosystem services, as advocated, for example, in England's new Environmental Land Management scheme[22] and practiced in Costa Rica's payment for forest service schemes[23,24] and the US conservation reserve programme[25].

This analysis updates initial suggestions[9] that conserving remaining natural habitats benefits human well-being as well as biodiversity but is based on a larger, more geographically, ecologically and contextually diverse sample. It also confirms the benefits of complementing conservation of existing habitats with investment in ecological restoration[11]. Importantly, our results characterize the outcomes of marginal changes: the costs and benefits of plausible changes in the state of individual sites[26]. As such, they should not be used to argue for widespread abandonment or restoration of human-dominated land uses. Moreover, we did not find that conservation and restoration have universally greater net value than alternative land uses, so more work is needed to identify the situations when a nature focus does and does not pay. However, our findings do suggest that, within the broad habitat and geographic range present in our data, we have typically passed the point where the benefits of further change from nature towards human-modified uses exceed the costs to society.

There are obvious limitations to this analysis. Benefit values were estimated by quantifying costs (for example, damage cost avoided and travel cost), estimates of producer surplus and, in one case, a replacement cost (Methods and Supplementary Data). As such, our estimates are not strict measures of economic value but rather estimations of ecosystem service values consistently applied to alternative states of landscapes. Case studies did not estimate consumer surpluses, and therefore our benefit values are conservative (underestimates). Uncertainty in some estimates of service values is likely to be high (for example, due to small sample sizes or the use of look-up tables for carbon estimates at some sites), although there is no reason to expect these uncertainties to be systematically biased in favour of the nature-focused state. It is conceivable that unpublished studies might contain biases in which services are selected and valued. However, in our sample, the unpublished studies on average included slightly fewer services than published studies, while including the most common driver of state change (cultivated goods) just as frequently (Supplementary Table 4). It is therefore unlikely that biased service selection in unpublished studies could have favoured the nature-focused state in our results. Moreover, we consistently found no difference in results between published and unpublished studies. One further potential limitation is that we have characterized decision-making as a simple dichotomy, when instead it normally involves choices among a broader and subtler range of options. Nevertheless, the simple contrasts drawn in this study throw light on the net benefits to society that are frequently ignored when land- and water-use decisions are made.

However, there are additional reasons why our results may be conservative. As yet, TESSA does not provide methods for all services. Several omitted services, such as air quality regulation, are also typically provided to a greater extent by natural than human-dominated land covers[27,28]. Also, the data available for economic analysis covered fewer services per site than those used in our simple directional analysis. While we found no evidence that the number of services assessed per site influenced the results of the economic analysis, statistical power was low and, given the results in the directional analysis, omission of some services was possibly important. Services omitted from the economic analysis, especially water services that users found difficult to quantify or monetize, were consistently better provided in the nature-focused state. Further, we used a (year-adjusted) value for the social cost of carbon of \$31.21 (ref. [20]), but others have argued that this should be \$62 (ref. [29]) or even higher[30]. Such values would greatly strengthen the case for nature-focused management. Furthermore, it can be expected that a flows-based assessment such as this will always under-represent the full natural capital value of the ecosystems from which services derive[10]. Our data represent snapshots of two states and assume constant rates of service flow and the same discount rates and time horizons for different services. This accounts for neither different sustainability of flows in the different states[31,32] nor the contention that private goods should be discounted more heavily and over a shorter horizon than public goods[33–36].

The heterogeneity and distribution that we found in the economic merits of nature-focused versus alternative states underscores the importance of using local engagement and local data to assess impending management decisions. We urge researchers and managers undertaking such studies to make their results and underlying data publicly available, to enable a broader understanding of when conservation or restoration is and is not likely to generate net benefit to society. However, at an action level, removing the information deficit will not alone result in better decisions, especially when private landowners bear the costs while benefits are received by various other stakeholders. On occasion, broader non-utilitarian values of landowners (such as wider civic duties and responsibilities or the cultural and social values they attach to land and place) may sway their decision. In other cases, as discussed earlier, financial and other incentives will be necessary to nudge behaviour change. To turn persuasive evidence into effective incentivization requires focused and committed engagement and relationship-building between data gatherers and decision makers[6] to maximize trust and knowledge legitimacy[37]. This includes both individuals and communities most affected by the decision and the policy actors who can provide incentives, influencing downstream operational decisions by altering their decision-making context. Such multi-stakeholder engagement was a vital component of the TESSA assessment process on which the case studies in this analysis are based. Communicating more effectively the full range of benefits and beneficiaries, through such an engagement process, is essential if future land and water management decisions are to stem the ongoing loss of biodiversity while supporting human prosperity.

Methods

TESSA (http://tessa.tools)[19] provides a consistent conceptual framework and relatively simple methods for estimating and valuing ecosystem services, combined with broad stakeholder participation, to contrast explicit alternative states of a site. A site is taken to be a single governance unit of between 100 and 10,000 ha. TESSA currently includes methods for coastal protection, global climate regulation, pollination, harvested wild goods, cultivated goods, water-related services (flood detention, water supply and quality), nature-based recreation and tourism, and cultural services. In addition to costs associated with production of specific services (see the following), users are guided to include both one-off costs of any restoration activity (for example, land purchase, fencing, re-engineering work, infrastructure, casual labour) and annual management costs. Annual management costs include salaries, capital expenditure (equipment, facilities) and operating costs (transport and subsistence, fuel, equipment repair, casual labour, advertising, training, fence maintenance, rental costs and veterinary fees). This enables estimation of the direction of change of key benefit and cost flows under different management or land-use options considered plausible by stakeholders. Recognizing that attributes and perceptions of knowledge and co-production of that knowledge are important determinants of whether it leads to action[37], wide stakeholder engagement in assessments is encouraged from the start. This includes those who benefit most from the change to an alternative state, to prevent systematic bias in selection of services to measure. TESSA helps users to explore the consequences not just of changing natural habitats to human-dominated uses, but of increasing the conservation value of currently converted sites through ecological restoration. Results from using TESSA have been used to help guide site management and land-use planning decisions[38].

Dataset compilation. We first conducted a Google Scholar search for articles since 2010 (when TESSA was launched) containing the terms 'TESSA', 'Peh' and 'ecosystem' anywhere in the article title or text. The search was performed on 30 March 2020 and yielded 243 results. After removing duplicates, we were left with 153 results, which we screened for relevance as case studies. We rejected articles that had not contrasted two alternatives states of a site or had simply cited the TESSA webpage or the original TESSA paper[19]. Following Balmford et al.[9], we included sites only if studies included full stakeholder engagement to identify the range of services present in each state and then gave data on the services providing the most important privately captured benefits (for example, timber, cultivated goods), as well as one or more services delivering local or global public benefits (for example, carbon sequestration, harvested wild goods, flood protection). We included the benefits of housing/infrastructure development in our definition of private goods because they were the driver of change at a notable subset of sites. Although TESSA provides methods for some supporting services, such as pollination, we focused here on final goods and services to avoid double counting service values[39]. This left 14 articles (12 in ISI journals), presenting information on 15 sites (13 in ISI journals), that we deemed suitable for analysis.

In addition to these published studies, we assessed all unpublished studies of which we were aware. After equivalent screening, this left an additional 47 sites (giving 62 sites in total, see Supplementary Data for details). The 62 site studies fell into two categories: a conserved state was contrasted with a modified state at 44 sites (6 in ISI journals); an ecological restoration state was contrasted with the pre-restoration (modified) state at 18 sites (7 in ISI journals). We refer to the conservation or ecological restoration state collectively as the nature-focused state. Sites were also classified by their main habitat (forest, wetland or other) and by World Bank income group classification[40].

Four main valuation approaches were used to estimate service values in the case studies we used (see Supplementary Data for details of the valuation approach for each service/benefit for each case study). Cost-based approaches included 'travel cost' and 'avoided damage cost' methods and were widely used to impute values for recreation and regulating services, respectively. This includes the value of carbon storage at each site using Nordhaus's[20] revised social cost of carbon, via damage cost and abatement cost estimates. For the benefits stemming from harvesting wild goods, livestock rearing, cultivation and timber production, values imputed were estimates of producer surplus (the net benefit a producer receives from the production of a set quantity of output). All case studies collected production costs for this set of goods using a local market price and used these costs to derive the producer surplus. We were not able to impute values for consumer surplus, as such, so for these benefits, the values used in our analyses are clearly underestimates. One study used a replacement cost approach (to estimate the cost of replacement of livestock feed at a site in Nepal). The few values of water provisioning services were based on market prices.

Between-state comparison of NPV of services. Our main analysis focused on an assessment of the net economic consequence of conservation or ecological restoration. A subset of 24 sites had monetized values for the flow in each state of enough services (see the preceding), to allow economic assessment of the consequences of habitat conservation or ecological restoration. The mean number of services monetized per site was 3.75, compared with 4 per site in the original study[9]. Except for carbon and greenhouse gas flux data, data for all services at all sites were primary data collected from the site. For carbon and greenhouse gas fluxes, biophysical data from seven sites were solely from look-up tables, while data for another 13 sites were a mix of look-up table data and locally collected data. Most differences between the alternative states were expressed as annual service flows/costs, but some (carbon and timber) were expressed as a one-off stock change. None of the studies presented data on time horizons for the potential state changes or discounted any of the values presented. Only 10 of the 24 sites provided full annual management costs in each state and, in the case of restoration sites, one-off restoration costs. Multiplier and knock-on economic returns were not considered.

To express monetary values of services in each state at each site in a common currency, we applied the following steps:

1. For services other than carbon (which was always expressed in US dollars (USD)), the monetary value of individual service flows or stock in each state was expressed in local currency units (LCU) at the date of study. If the study had originally presented values in USD, values were converted to LCU using the USD–LCU exchange rate at the date of submission of the study[41]. If only the year of publishing was provided, then the date was set at 1 July, the mid-point of the year.
2. The LCU value was then adjusted to an international dollar value for each state, using year- and country-specific purchasing power parity (PPP) conversion factors. Year-specific PPP conversion factors were extracted from ref. [42], with data for the UK Overseas Territory of Montserrat taken from ref. [43]. Gross domestic product (GDP), in LCU, was extracted from ref. [44]. Then, PPP was calculated as the ratio between GDP (LCU) and GDP–PPP.
3. We did not deflate year-of-study values to a consistent year across all studies because we were ultimately not interested in between-site contrasts, only within-site contrasts of the nature-focused and alternative states (see the following).
4. One-off values (CO_2 stock flux, timber harvest and restoration cost) were annualized into equal amounts over a standard period over which we assumed the state change is completed[45]. Here we used a 20-year time horizon for the state change but tested the sensitivity of our results using 10- and 30-year periods (see the following).
5. For sites providing data on carbon stocks, these quantities were multiplied by 44/12 (to convert atomic mass of carbon to that of carbon dioxide) to convert them to 'CO_2 stock flux' that would occur between the site and the atmosphere if any of that stock was lost.
6. For both one-off carbon stock losses and ongoing flux of CO_2 equivalents, we calculated the year-specific USD value of CO_2e fluxes on the basis of the social cost of carbon value of \$31.21 per tCO_2 (from 2010)[20], adjusted to year-specific values using the US inflation calculator at https://www.usinflationcalculator.com/. As a sensitivity test, we also explored the effects of using CO_2 prices of \$5 or \$100 per tonne, representing the range of values suggested in the original studies (see also ref. [46]).
7. For contrasts of a conservation state with a modified state, annual benefit flows in the conservation state were assumed to be constant over time, while benefit flows in the modified state started in year zero at the conservation state rate before changing steadily (equal annual increments) to the modified state rate by year 20. When contrasting an ecological restoration state with a pre-restored state, benefit flows were assumed to be constant over the whole time horizon in the pre-restored state, while benefit flows in the ecological restoration state started in year zero at the pre-restoration state rate before changing linearly to the ecological restoration state rate by year 20. When contrasting two different restored states, benefit flows were assumed to be zero in year zero (in all cases, these were quarry sites at that point), with benefit flows in each state changing steadily to the rate measured for that state by year 20. In all cases, flows beyond 20 years were assumed to be constant, reflecting the rate observed in whichever state the site was then in.
8. All benefit flows were then converted to values per hectare per year.
9. NPV of each site in each state was determined by setting an appropriate discount rate and time horizon for accrual of the benefits over time. Standard cost–benefit analysis practice is to use a time horizon over which the benefits or costs of a project will occur[47]. This simple rule of thumb is complicated for environmental and sustainability issues where costs or benefits change at varying rates, where private benefits (costs) may have short project horizons and environmental benefits (costs) much longer[48]. In the absence of reported time horizons in the studies we found, we used the central figure (50 years) of those studies presented in the original study[9]. As a sensitivity test, we also used 30 and 100 years: the range of values in the case studies[9]. Discounting all services at the same rate is conservative, not reflecting the contention that public benefits should be discounted at a lower rate than private benefits and

over a much longer time horizon (for example, 100 years (refs. [33–36])). To be conservative, we used a constant rate of 2% (with 1% and 10% as sensitivity tests). These span best-practice rates for environmental projects encompassing the typical consumption rate of interest and returns to private capital[49,50].

10. For each site in each state, we summed the values over the whole time horizon to give the NPV. For the ten sites where this was possible, NPV included full restoration and management costs.
11. We did this first for our chosen values of carbon price, time for state change, time horizon and discount rate. We then ran a series of sensitivity tests manipulating each of these variables in turn. In summary, our analysis (+ sensitivity tests) therefore contrasts NPV between nature-focused and alternative states, assuming (1) a year-specific social cost per tCO_2e ($5, $100), (2) a 20-year time horizon for state change (10 years, 30 years), (3) a 50-year total time horizon for accrual of costs and benefits[51] (30 years, 100 years) and (4) a discount rate of 2% (1%, 10%).
12. To further allow inspection of how changes in the state of a site affect benefit distribution, service flows and costs arising in each state were classified along the dichotomy: excludable or non-excludable goods. Supplementary Table 1 shows which services fell into each category at each site. NPV values for each class were then summed separately for each site in each state to see how the change in state affected the distribution of value between these classes. We chose to use a rival/excludable framework to organize our benefits because we were interested in how the innate characteristics of the benefits affected stakeholders and outcomes.

For presentation purposes, the NPV in the nature-focused state was divided by that in the alternative state and the result logged. Positive values therefore reflect greater net benefit in the nature-focused state while negative values reflect greater net benefit in the alternative state. Note that if the NPV for both cases was negative, then the sign of the resulting ratio was changed to reflect accurately which state provided the greater benefit. Cases where the NPV of either state was negative, such that a ratio could not be calculated, were allocated values equivalent to either the largest ratio in the dataset (alternative state NPV negative; four cases) or smallest ratio (nature-focused state NPV negative: one case). To categorize each site according to which state gave greater overall value, we ascribed values >0.05 as flows greater in the nature-focused state, $-0.05 < \text{value} < 0.05$ as equal between states and scores < -0.05 as flows greater in the alternative state.

To test whether the NPV tended to be greater in the nature-focused than the alternative state, we first specified a binary dependent variable: (1) NPV of nature-focused state ≥ NPV of alternative state versus (2) NPV of alternative state > NPV of nature-focused state. We then used binary logistic regression (two-sided chi-squared tests) to test the odds of the nature-focused state having greater value than the alternative state, on the basis of the values of several candidate independent variables that may affect per hectare values[51]. These include type of stage change (two-level factor), country income class (four-level factor) and broad habitat class (three-level factor). We also tested sensitivity to the number of services inspected per site (ranging from two to seven), whether full costs were included (two-level factor) and where the study was reported (ISI journal or not). For completeness, given the high ratio of parameters to degrees of freedom, we ran tests (1) for univariate models only, (2) in a full model and (3) by stepwise deletion from the full model.

Tests of whether the nature-focused state produced greater value for either excludable or non-excludable goods were made with two-sided binomial tests, with the dependent variable for each site again specified as (1) NPV of nature-focused state ≥ NPV of alternative state versus (2) NPV of alternative state > NPV of nature-focused state. All tests were conducted in SAS Enterprise Guide v7.1[52].

Benefit–cost ratios of nature-focused and alternative states. To determine the benefit–cost ratio of the nature-focused and alternative states, we reran steps 1–10 for each of the ten sites that provided full cost data, separately for benefits and for costs. For each site, we calculated the benefit–cost ratio separately for the nature-focused and alternative states. For presentation purposes, the benefit–cost ratio in the nature-focused state was divided by that in the alternative state and the result logged. Positive values therefore reflect a higher benefit–cost ratio in the nature-focused state while negative values reflect a higher benefit–cost ratio in the alternative state.

Between-state directional change in service provision. For each non-monetized service at each site, we classified the difference in flows between the two contrasting states as (1) flows greater in the nature-focused state, (2) flows equal in the nature-focused and alternative states or (3) flows greater in the alternative state. This directional assessment was therefore undertaken for a mean of 0.8 services per site among the 24 sites included in the monetary analysis. For the remaining 38 sites, such data were available for a mean of 5 services per site, and each of the 38 sites was assigned to one of the following three categories;

1. Benefits apparently greater under the nature-focused state; that is, compared with the alternative state, more services had higher values in the nature-focused state than had equal or lower values
2. Benefits apparently equal under the two states; that is, compared with the alternative state, more services had equal values in the nature-focused state than had higher or lower values and/or the number of services that had lower values in the nature-focused state compared with the alternative state was the same as the number that had higher values
3. Benefits apparently greater under the alternative state; that is, compared with the alternative state, more services had lower values in the nature-focused state than had equal or higher values

Here, the binary dependent variable was specified as (1) benefits apparently greater under the nature-focused state versus (2) benefits approximately equal under the two states. The impact on the dependent variable of type of stage change (two-level factor), country income class (four-level factor), broad habitat class (three-level factor), number of services inspected per site (ranging from two to seven) and where the study was reported (ISI journal or not) was tested in a binary logistic regression. There were no cases where the benefits were greater under the alternative state. Tests were again conducted in SAS Enterprise Guide v7.1[52] and (1) for univariate models only, (2) in a full model and (3) by stepwise deletion from the full model.

Reporting Summary. Further information on research design is available in the Nature Research Reporting Summary linked to this article.

Data availability
The data that support the findings of this study are available in the supplementary information. Source data are provided with this paper.

Code availability
No code was used during the preparation of this paper.

Received: 25 July 2020; Accepted: 11 February 2021;

References
1. Costanza, R. et al. The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260 (1997).
2. Daily, G. C. et al. Ecosystem services in decision making: time to deliver. *Front. Ecol. Environ.* **7**, 21–28 (2009).
3. Bateman, I. J. et al. Bringing ecosystem services into economic decision-making: land use in the United Kingdom. *Science* **341**, 45–50 (2013).
4. Zhang, X., Estoque, R. C., Xie, H., Murayama, Y. & Ranagalage, M. Bibliometric analysis of highly cited articles on ecosystem services. *PLoS ONE* **14**, e0210707 (2019).
5. Díaz, S. et al. (eds) Summary for Policymakers. In *Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (eds Brondizio, E. S. et al.) (IPBES Secretariat, 2020); https://doi.org/10.5281/zenodo.3553579
6. Mandle, L. et al. Increasing decision relevance of ecosystem service science. *Nat. Sustain.* https://doi.org/10.1038/s41893-020-00625-y (2020).
7. Norgaard, R. B. Ecosystem services: from eye-opening metaphor to complexity blinder. *Ecol. Econ.* **69**, 1219–1227 (2010).
8. Blomqvist, L. & Simpson, R. D. in *Effective Conservation Science: Data not Dogma* (eds Kareiva, P. et al.) Ch. 3 (Oxford Univ. Press, 2018).
9. Balmford, A. et al. Economic reasons for conserving wild nature. *Science* **297**, 950–953 (2002).
10. Bateman, I. J. & Mace, G. M. The natural capital framework for sustainably efficient and equitable decision-making. *Nat. Sustain.* https://doi.org/10.1038/s41893-020-0552-3 (2020).
11. Rey Benayas, J. M., Newton, A. C., Diaz, A. & Bullock, J. M. Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science* **325**, 1121–1124 (2009).
12. Bullock, J. M. et al. Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends Ecol. Evol.* **26**, 541–549 (2011).
13. Shimamoto, C. Y., Padial, A. A., da Rosa, C. M. & Marques, M. C. M. Restoration of ecosystem services in tropical forests: a global meta-analysis. *PLoS ONE* **13**, e0208523 (2018).
14. *The Rewards of Investing in Sustainable Land Management* Interim Report (ELD Initiative, 2013); www.eld-initiative.org
15. *The IPBES Assessment Report on Land Degradation and Restoration* (eds Montanarella, L. et al.) (IPBES Secretariat, 2018).
16. Neugarten, R. A. et al. *Tools for Measuring, Modelling and Valuing Ecosystem Services: Guidance for Key Biodiversity Areas, Natural World Heritage Sites, and Protected Areas* (IUCN, 2018).
17. Nelson, E. et al. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Front. Ecol. Environ.* **7**, 4–11 (2009).
18. Ruckleshaus, M. et al. Notes from the field: lessons learned from using ecosystem service approaches to inform real-world decisions. *Ecol. Econ.* **115**, 11–21 (2015).

19. Peh, K. S.-H. et al. TESSA: a toolkit for rapid assessment of ecosystem services at sites of biodiversity conservation importance. *Ecosyst. Serv.* **5**, 51–57 (2013).
20. Nordhaus, W. D. Revisiting the social cost of carbon. *Proc. Natl Acad. Sci. USA* **114**, 1518–1523 (2017).
21. Goldstein, E. S. & Polasky, S. *Economics and the Environment* (Wiley, 2017).
22. *Environmental Land Management* Policy Discussion Document (Department for Environment, Food and Rural Affairs, 2020).
23. Pagiola, S. Payments for environmental services in Costa Rica. *Ecol. Econ.* **65**, 712–724 (2008).
24. Hinojosa, C. *Case Study: Payments for Ecosystem Services in Costa Rica* (DCED, 2017); https://www.enterprise-development.org/wp-content/uploads/DCED-GGWG-Case-study-PES.pdf
25. *Conservation Reserve Program* (USDA Farm Service Agency, 2020); https://www.fsa.usda.gov/programs-and-services/conservation-programs/conservation-reserve-program/index
26. Balmford, A. et al. Bringing ecosystem services into the real world: an operational framework for assessing the economic consequences of losing wild nature. *Environ. Resour. Econ.* **48**, 161–175 (2011).
27. Jones, L. et al. *Developing Estimates for the Valuation of Air Pollution Removal in Ecosystem Accounts* Final Report (Office of National Statistics, 2017).
28. Gopalakrishnan, V., Ziv, G., Hirabayashi, S. & Bakshi, B. R. Nature-based solutions can compete with technology for mitigating air emissions across the United States. *Environ. Sci. Technol.* **53**, 13228–13237 (2019).
29. Tol, R. S. J. Targets for global climate policy: an overview. *J. Econ. Dyn. Control* **37**, 911–928 (2013).
30. van den Bergh, J. C. J. M. & Botzen, W. J. W. Monetary valuation of the social cost of CO_2 emissions: a critical survey. *Ecol. Econ.* **114**, 33–46 (2015).
31. Hughes, F. M. R. et al. The challenges of integrating biodiversity and ecosystem services monitoring and evaluation at a landscape-scale wetland restoration project in the UK. *Ecol. Soc.* **21**, 10 (2016).
32. Mace, G. M. The ecology of natural capital accounting. *Oxf. Rev. Econ. Pol.* **35**, 54–67 (2019).
33. *Green Book* (HM Treasury, 2003).
34. Lebegue, D. *Revision du taux d'actualisation des investissem- net publics* Report (Commisariat Generale de Plan, 2005).
35. Howard, G. Discounting for personal and social payments: patience for others, impatience for ourselves. *J. Environ. Econ. Manag.* **66**, 583–597 (2013).
36. Drupp, M. A., Freeman, M. C., Groom, B. & Nesje, F. Discounting disentangled. *Am. Econ. J. Econ. Policy* **10**, 109–134 (2018).
37. Posner, S. M., McKenzie, E. & Ricketts, T. H. Policy impacts of ecosystem services knowledge. *Proc. Natl Acad. Sci. USA* **113**, 1760–1765 (2016).
38. Field, R. et al. *Ecosystem Service Assessment of the Implementation of a Community Conserved Area in the Lower Tana Delta* (RSPB, 2018).
39. Fisher, B., Turner, R. K. & Morling, P. Defining and classifying ecosystem services for decision making. *Ecol. Econ.* **68**, 643–653 (2009).
40. *World Bank Country and Lending Groups* (World Bank, 2020); https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups
41. *XE Current & Historical Rate Tables* (XE.com Inc., 2020); https://www.xe.com/currencytables/?from=USD&date=2017-12-31
42. *IMF Implied PPP Conversion Rate* (International Monetary Fund, 2020); https://www.imf.org/external/datamapper/PPPEX@WEO/OEMDC/ADVEC/WEOWORLD
43. *Central Intelligence Agency World FactBook* (CIA, 2020); https://www.cia.gov/library/publications/resources/the-world-factbook/docs/rankorderguide.html
44. *UN Data: GDP at Current Prices* (United Nations Statistics Division, 2020); http://data.un.org/Data.aspx?d=SNAAMA&f=grID%3a101%3bcurrID%3aUSD%3bpcFlag%3a1
45. Moreno-Mateos, D. et al. Anthropogenic ecosystem disturbance and the nature recovery debt. *Nat. Commun.* **8**, 14163 (2017).
46. Ferraro, P. J. et al. Estimating the impacts of conservation on ecosystem services and poverty by integrating modeling and evaluation. *Proc. Natl Acad. Sci. USA* **112**, 7420–7425 (2015).
47. Turner, R. K., Pearce, D. & Bateman, I. *Environmental Economics: An Elementary Introduction* (Harvester Wheatsheaf, 1994).
48. Turner, R. K. Limits to CBA in UK and European environmental policy: retrospects and future prospects. *Environ. Resour. Econ.* **37**, 253–269 (2007).
49. *Guidelines for Preparing Economic Analyses* (EPA, 2010).
50. Stern, N. & Stern, N. H. *The Economics of Climate Change: The Stern Review* (Cambridge Univ. Press, 2007).
51. ten Brink P., et al. *Estimating the Overall Economic Value of the Benefits Provided by the Natura 2000 Network* Final Report to the European Commission (IEEP, 2011).
52. SAS Enterprise Guide v7.1 (SAS Institute, 2012).

Acknowledgements

We thank B. Balmford, K. Bolt and J. Vause for advice. A.B. was supported by a Royal Society Wolfson Merit award. K.S.-H.P. was supported by an AXA Research Fund (grant no. RG64520) while at the University of Cambridge.

Author contributions

Conceptualization and methodology were performed by R.B.B., S.H.M.B., B.F., K.S.-H.P. and A.B. Investigation, formal analysis and data curation were by R.B.B. The original draft was written by R.B.B. and A.B. Review and editing were done by R.B.B., S.H.M.B., B.F., F.M.R.H., L.I.-K., M.A.M., J.C.M., K.S.-H.P., A.-S.P., D.H.L.T., R.T. and A.B. Supervision was by S.H.M.B., K.S.-H.P. and A.B.

Competing interests

The authors declare no competing interests.

Additional information

Supplementary information The online version contains supplementary material available at https://doi.org/10.1038/s41893-021-00692-9.

Correspondence and requests for materials should be addressed to R.B.B.

Peer review information *Nature Sustainability* thanks Sarah Klain and Louise Willemen for their contribution to the peer review of this work.

Reprints and permissions information is available at www.nature.com/reprints.

nature research

Corresponding author(s): Richard Bradbury

Last updated by author(s): Aug 11, 2020

Reporting Summary

Nature Research wishes to improve the reproducibility of the work that we publish. This form provides structure for consistency and transparency in reporting. For further information on Nature Research policies, see our Editorial Policies and the Editorial Policy Checklist.

Statistics

For all statistical analyses, confirm that the following items are present in the figure legend, table legend, main text, or Methods section.

n/a	Confirmed	
☐	☒	The exact sample size (n) for each experimental group/condition, given as a discrete number and unit of measurement
☐	☒	A statement on whether measurements were taken from distinct samples or whether the same sample was measured repeatedly
☐	☒	The statistical test(s) used AND whether they are one- or two-sided *Only common tests should be described solely by name; describe more complex techniques in the Methods section.*
☐	☒	A description of all covariates tested
☒	☐	A description of any assumptions or corrections, such as tests of normality and adjustment for multiple comparisons
☐	☒	A full description of the statistical parameters including central tendency (e.g. means) or other basic estimates (e.g. regression coefficient) AND variation (e.g. standard deviation) or associated estimates of uncertainty (e.g. confidence intervals)
☐	☒	For null hypothesis testing, the test statistic (e.g. F, t, r) with confidence intervals, effect sizes, degrees of freedom and P value noted *Give P values as exact values whenever suitable.*
☒	☐	For Bayesian analysis, information on the choice of priors and Markov chain Monte Carlo settings
☒	☐	For hierarchical and complex designs, identification of the appropriate level for tests and full reporting of outcomes
☐	☒	Estimates of effect sizes (e.g. Cohen's d, Pearson's r), indicating how they were calculated

Our web collection on statistics for biologists contains articles on many of the points above.

Software and code

Policy information about availability of computer code

Data collection	Microsoft Excel for Office 365
Data analysis	SAS Enterprise Guide version 7.1

For manuscripts utilizing custom algorithms or software that are central to the research but not yet described in published literature, software must be made available to editors and reviewers. We strongly encourage code deposition in a community repository (e.g. GitHub). See the Nature Research guidelines for submitting code & software for further information.

Data

Policy information about availability of data

All manuscripts must include a data availability statement. This statement should provide the following information, where applicable:

- Accession codes, unique identifiers, or web links for publicly available datasets
- A list of figures that have associated raw data
- A description of any restrictions on data availability

Provide your data availability statement here.

Field-specific reporting

Please select the one below that is the best fit for your research. If you are not sure, read the appropriate sections before making your selection.

☒ Life sciences ☐ Behavioural & social sciences ☐ Ecological, evolutionary & environmental sciences

For a reference copy of the document with all sections, see nature.com/documents/nr-reporting-summary-flat.pdf

Life sciences study design

All studies must disclose on these points even when the disclosure is negative.

Sample size	Sample size (shown in Table S1) was determined by the availability of studies that could be used in the analyses. See Methods for more detail.
Data exclusions	We used all the data that met our criteria (see Methods), thus essentially no data exclusions were involved in this study.
Replication	Not relevant as this study does not involve any experiments.
Randomization	Not relevant as this study does not involve any experiments.
Blinding	Not relevant.

Reporting for specific materials, systems and methods

We require information from authors about some types of materials, experimental systems and methods used in many studies. Here, indicate whether each material, system or method listed is relevant to your study. If you are not sure if a list item applies to your research, read the appropriate section before selecting a response.

Materials & experimental systems

n/a	Involved in the study	
☒	☐	Antibodies
☒	☐	Eukaryotic cell lines
☒	☐	Palaeontology and archaeology
☒	☐	Animals and other organisms
☒	☐	Human research participants
☒	☐	Clinical data
☒	☐	Dual use research of concern

Methods

n/a	Involved in the study	
☒	☐	ChIP-seq
☒	☐	Flow cytometry
☒	☐	MRI-based neuroimaging