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To cite this article: Maximilian Nawrath *et al* 2026 *Environ. Res. Lett.* **21** 173002

ENVIRONMENTAL RESEARCH
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OPEN ACCESS

RECEIVED
18 November 2025

REVISED
6 August 2026

ACCEPTED FOR PUBLICATION
18 August 2026

PUBLISHED
9 September 2026

TOPICAL REVIEW

Quantifying temporal changes in the monetary value of forest cultural ecosystem services globally: a meta-analysis

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Keywords: Benefit transfer, biodiversity, non-market valuation, recreation, revealed preference, stated preference

Supplementary material for this article is available online

Abstract

The world's forests, and the biodiversity they contain, provide multiple cultural ecosystem services that are essential to people's well-being. However, interactions between people and forests evolve alongside broader societal and environmental change, raising questions about whether the values associated with forest cultural ecosystem services have shifted over time. To examine this, we conducted a global meta-analysis of 137 economic valuation studies comprising 705 observations gathered over the four decades between 1980 and 2019. Our results did not provide evidence of statistically significant temporal trends for travel cost, contingent valuation, choice experiment, or hedonic pricing estimates. Likewise, a recreation-only subsample did not provide evidence of a statistically significant temporal trend in monetary values over time. Although a diversity of cultural ecosystem service classes and forest types were examined, the existing evidence base is biased towards high-income countries, and valuation studies remain heavily concentrated on recreational interactions with forests, while spiritual and symbolic interactions are rarely quantified. Such gaps in our knowledge limit our ability to quantify the monetary value of forest cultural ecosystem services globally. Although we did not find evidence of temporal trends in the value people place on forest cultural ecosystem services, the substantial heterogeneity and method-specific patterns observed in our models suggest that temporal stability should not be assumed when undertaking benefit transfers. Given these findings, methodological heterogeneity and contextual differences should be considered when applying benefit transfer methods in environmental decision-making. The scarcity of studies in low- and middle-income countries, combined with a strong focus on recreation, underscores the need for broader and geographically diverse valuation research. Addressing these gaps may support more context-sensitive approaches to forest management and conservation.

1. Introduction

Forests provide irreplaceable ecosystem services, including regulation of climate and carbon storage, but their functioning depends on the maintenance of biodiversity and intact ecological communities (Watson *et al* 2018). Despite the multitude of ecosystem services they provide, forests are confronted with numerous substantial threats on a global scale, such as deforestation (Hansen *et al* 2013), habitat degradation (Banks-leite *et al* 2020) and climate change (Keenan 2015). In particular, forests play a vital role in providing a wide range of cultural ecosystem services that contribute to human well-being (Perera *et al* 2018). These encompass the non-material benefits that people obtain from ecosystems, comprising the cultural, spiritual and aesthetic aspects of human-nature interactions (Haines-young and Potschin

2018). Cultural ecosystem services include, but are not limited to, recreation, nature watching, spiritual and religious experiences, aesthetic appreciation, symbolism and the transmission of cultural heritage and knowledge (Haines-young and Potschin 2018).

Many cultural ecosystem goods and services are quasi-public or public goods. They are partly or fully non-rival, meaning one person's use does not reduce availability for others, and partly or fully non-excludable, meaning it is difficult to stop people from using it (Brander *et al* 2007). In the absence of well-functioning markets, cultural ecosystem services can be systematically undervalued in decision-making processes (HM Treasury 2021). As a result, the value of forests and the full range of biodiversity, goods and services they provide are often not adequately considered, potentially leading to suboptimal resource management, unsustainable practices and a failure to recognise the importance of forests for society (European Investment Bank 2022). To rectify this, environmental economists have sought to quantify the non-material benefits provided by ecosystems for which no market exists. This work has been ongoing for at least four decades from the 1980s onwards (Richardson *et al* 2015a, Haines-young and Potschin 2018), a time that has also been characterised by huge societal, environmental and technological change, such as increases in Global Domestic Product (OECD 2023), average age of the global population (Kluge *et al* 2014), impacts of climate change (IPCC 2018), urbanisation (United Nations 2018), continued declines in species populations and habitat loss (Díaz *et al* 2019), shifts in media and research funding priorities (Veríssimo *et al* 2014), and the pervasiveness of the Internet and mobile phone technologies (Proulx *et al* 2014).

Anthropogenic-driven changes in climate and biodiversity loss are affecting ecosystems globally, with projected reductions in ecosystem service delivery (Díaz *et al* 2019, IPCC 2023). Forests play a central role in climate change mitigation and biodiversity conservation, acting as major carbon sinks and supporting over 80% of terrestrial biodiversity (Harris *et al* 2021; FAO 2020). There is thus a greater public and policy awareness of forests. Concurrently, interactions with forests and associated biodiversity are changing. Engagement with nature has declined in some contexts, particularly among younger generations (Gaston *et al* 2020, Truong and Clayton 2020). Increased reliance on digital technologies for entertainment and information, alongside ongoing urbanisation, may further reduce opportunities for direct interactions with forests (Colléony *et al* 2020, Truong and Clayton 2020, Lin and Ardoin 2023).

The scale and pace of societal, environmental and technological change since the 1980s may thus have altered people's preferences for the cultural ecosystem services of forests (Torralba *et al* 2020). However, there is limited evidence on how preferences have changed and to what degree. A growing body of studies have repeated valuation exercises across different years and documented temporal variation in estimated values (Zandersen *et al* 2007, Fetene *et al* 2014). However, studies are fragmented across ecosystems, valuation methods, and geographical contexts, which limits generalisability. This lack of systematic evidence is problematic given the increasing reliance on benefit transfer (the use of existing data or information, in settings other than the purpose of the original data collection; Rosenberger and Loomis 2003) in non-market valuation, where unaccounted-for temporal change could introduce substantial error (Johnston and Rosenberger 2010, Johnston *et al* 2018). Economic theory also suggests that the relative value of ecosystem services may change over time due to increasing scarcity and limited substitutability with market goods. Empirical evidence indicates that the relative price of ecosystem services is increasing, which is consistent with limited substitutability and slower or no growth of ecosystem services relative to market goods (Heckenhahn and Drupp 2024). Quantifying temporal shifts in stated and revealed preferences for biodiversity and cultural ecosystem services is, therefore, highly original and has profound implications for the application of non-market economic valuation techniques.

Here, we conducted a rapid evidence assessment (REA) and meta-analysis to determine temporal trends in the monetary values associated with forest cultural ecosystem services globally. To avoid confounding effects associated with the COVID-19 pandemic, which substantially altered how often people visited forests and how they experienced it (Wunderlich *et al* 2024), we analysed studies with data collection before 2020. We focused on the most widely applied valuation methods in the cultural ecosystem services literature, such as the stated preference methods of contingent valuation and choice experiments, and the revealed preference methods of travel cost and hedonic pricing (Brander and Koetse 2011). Given the substantial temporal span covered by the literature, we expected that valuation estimates would exhibit systematic changes through time. By examining these shifts, we provide a more complete picture of if and how values for cultural ecosystem services have altered through time.

2. Methods

2.1. Collecting and characterising the evidence

We used a (REA; Civil Service 2010) to systematically search the literature, following the Preferred Reporting Items for Systematic Reviews and Meta-Analyses framework (Page *et al* 2021). This requires the *a priori* development of a protocol (supplementary material 1), which defines the bibliographic databases to be searched, the literature search strategy, eligibility criteria for studies to be included in the REA and the data extraction process.

2.1.1. Search strategy

We used two approaches to develop the search string. First, we examined the search strings of existing evidence synthesis reviews on cultural ecosystem services and terrestrial forests, and extracted relevant search terms (Milcu *et al* 2013, Martin *et al* 2016, Dickinson and Hobbs 2017, Hølleland *et al* 2017, Irvine and Herrett 2018, Cheng *et al* 2019, Kadykalo *et al* 2019). We also used the IUCN's search terms for terrestrial forests (IUCN 2006). To capture ecosystem services, we used terms from the Natural England People and Nature Survey (Natural England 2019) and Common International Classification of Ecosystem Services (Haines-young and Potschin 2018). Second, we used a novel, semi-automated approach based on keyword re-occurrence in the R-package '*litsearchr*' to identify other relevant search terms (Grames *et al* 2019). The search strings for each database were then peer-reviewed by a team of environmental and social scientists (all seven co-authors—*redacted for double blind peer-review*). The finalised search strings (table S1) were then used to comprehensively search the electronic databases Web of Science and Scopus for studies published online with data collection given as prior to 2020.

2.1.2. Eligibility criteria

To be included in the analysis, papers needed to be published in a peer-reviewed journal and written in English. Typically, only peer-reviewed studies are recommended for inclusion in an REA, as grey literature tends to be of lower methodological quality and therefore is more likely to contain biased findings (Egger *et al* 2003). We included studies published between January 1980 and November 2022 and did not use any geographical restrictions. To avoid confounding effects stemming from the COVID-19 pandemic, we excluded studies with data collection from 2020 onwards, as this period was marked by exceptional and likely temporary surges in forest visitation and altered public preferences (Venter *et al* 2020, Wunderlich *et al* 2024). This approach allowed us to provide an assessment of long-term trends free from the influence of this short-term disruption. Types of study included were randomised controlled trial studies, cohort studies, case-control studies, cross-sectional studies, controlled before and after studies, before and after studies, time series and longitudinal studies.

The searches targeted papers that provided data on the non-market economic valuation of terrestrial forest and associated cultural ecosystem services (table S1). We defined terrestrial forest as 'land with tree crown cover (or equivalent stocking level) or more than 10% and area of more than 0.5 hectare' (Dudley and Phillips 2006), relying on authors descriptions within the papers. We also included studies on 'farmed' trees such as forest plantations and orchards.

We defined cultural ecosystem services as 'all the non-material, and normally non-rival and non-consumptive, outputs of ecosystems (biotic and abiotic) that affect the physical and mental states of people' (Haines-young and Potschin 2018). We followed the CICES framework for the classification of cultural services.

We selected studies that used stated and revealed preference methods to assess monetary value. Stated preference methods involve individuals responding to a questionnaire (Hanley and Czajkowski 2019), for example, exploring the willingness of a person to pay for goods/services in a hypothetical scenario (e.g. forest recreation close to home; Ezebilo *et al* 2014). In choice experiments, participant preferences for characteristics (attributes) of non-market goods/services are elicited by having them choose between different scenarios with varying attributes (Bateman *et al* 2002, Hensher *et al* 2005, Holmes *et al* 2017). Revealed preference methods use observations of purchases made by people (Herriges and Kling 2009). In travel cost studies, recreational demand is estimated from variations in visitation rates as travel costs increase with distance, where travel costs typically include both out-of-pocket expenditures and the opportunity cost of travel time (Champ *et al* 2017). Hedonic pricing studies infer values from observed market transactions, such as differences in property prices associated with environmental attributes, including proximity to forests (Train 2002).

2.1.3. Exclusion criteria

Case reports, reviews, opinion pieces, editorials, comments, news, letters and grey literature were excluded from the REA. We also omitted studies focusing on domestic livestock, participation in incentive programmes (e.g. payments for ecosystem services), motivations/payment levels associated with incentive programmes, ecosystem service pollutant removal and the transfer of proxies/data from elsewhere.

2.1.4. Data screening and extraction

All studies arising from the search went through a two-stage screening process. In the first stage, the title and abstract of each paper were examined to select studies meeting the stated eligibility criteria and ineligible studies excluded from the REA. In the second stage, the full text of each remaining paper was examined to determine inclusion/exclusion (table S2). Following title and abstract screening, 759 records were assessed at full text. At this stage, studies were excluded if they did not meet the eligibility criteria, including lack of a monetary valuation of forest cultural ecosystem services, use of ineligible valuation approaches, focus on non-terrestrial forest systems, data collection outside the study period, or insufficient quantitative information (e.g. absence of variance estimates required for meta-analysis). In total, 353 studies met the REA inclusion criteria in terms of scope and study design; however, 214 of these were excluded at full-text screening because they did not report sufficient quantitative detail to allow effect size extraction, while two more were excluded for using a mix of methods that precluded extraction of variation estimates. This resulted in a final sample of 137 studies included in the meta-analysis. During both screening stages, a random 10% subset of studies was examined by a second co-author to ensure the criteria were correctly and consistently applied. We assessed interrater reliability through Cohen's kappa analysis (Belur *et al* 2018). Interrater reliability was rated as 'substantial agreement,' based on a Cohen's K value of 0.77. Using the eligibility criteria, we developed a standardised data extraction process. Data were obtained from the text, tables and figures of each paper. If data were presented graphically, we used WebPlotDigitizer to extract them.

All pertinent metadata related to the effect size (monetary value per unit of good/service, converted to 2019 USD to account for purchasing power parity and inflation), study design (e.g. stated/revealed preference, valuation method), class(es) of cultural ecosystem service and study characteristics were extracted and tabulated (table S2). We used country- and year-specific purchasing power parity conversion factors and consumer price index values from the World Bank (World Bank Group 2024a, 2024b). Some studies met the eligibility criteria but did not report variation estimates for the reported effect sizes, such as standard errors or confidence intervals, which are required for computing the meta-analysis. For these studies, we contacted the authors to request the missing variation data and only included those for which variation data were provided, omitting the remainder from the meta-analysis. The majority of effect size metrics were either willingness to pay (WTP) (the amount of money an individual would be willing to give up for a change in ecosystem service provision; Champ *et al* 2017), consumer surplus (the difference between the price actually paid for an ecosystem service, and the maximum amount that an individual is actually willing to pay for it; Herriges and Kling 2009), or price-based estimates in hedonic pricing studies.

For standardising the effect size to 2019 USD, we followed the approach taken by Brander *et al* (2007). First, to adjust for inflation, we used consumer price index values from the World Bank (World Bank Group 2024a, 2024b). We then used the World Bank's purchasing power parity conversion rates to adjust for differences in purchasing power between countries. When data collection in a study spanned multiple years, we calculated the mean purchasing power parity conversion factor for the period. For the standardisation of variation estimates, we followed the same approach.

Studies reported variation estimates of effect sizes as standard deviation, standard error and confidence intervals. We standardised variation estimates to standard deviations following guidance from the Cochrane Handbook for Meta-Analyses (Cochrane 2020). We used the following formula to transform standard errors into standard deviations:

$$\mathrm{SD} = \mathrm{SE} \times \sqrt{N}.$$

For transforming 95% confidence intervals into standard deviations, we applied:

$$\mathrm{SD} = \sqrt{N} \times (\text{upper limit} - \text{lower limit})/3.92.$$

For 90% confidence intervals, we replaced 3.92 with 3.29, and for 99% confidence intervals we replaced 3.92 with 5.15 (Cochrane 2020). Some studies reported multiple effect sizes for the same measure (e.g. having used probit, logit and non-parametric models to compute the data). To avoid double counting in such cases, we calculated mean effect sizes. If studies reported estimates for different subsamples that could not be distinguished by explanatory variables, we also calculated the mean effect sizes.

Additionally, we recorded information on study design/characteristics including the associated CICES cultural ecosystem service class (Haines-young and Potschin 2018), year data collection was started, country, continent (tables S4 and S5), low/middle- or high-income country in the year data were collected, the payment vehicle (units in which payment would be made; Johnston *et al* 2017) and forest type (tables S2 and S3). We categorised countries following the Development Assistance Committee (DAC) classification, into low- and middle-income countries (LMICs) and high-income countries (HICs) (Development Assistance Committee 2020). The payment vehicles recorded in the studies were 'house' (property purchases), 'one-off' (one-time payments), 'tax' (payments in the form of taxes), 'trip' (recreational trip cost) and 'use fees' (fees for accessing forests). The terrestrial forest types were forest (general), forest plantation, deciduous forest, coniferous forest, mixed forest, boreal forest, temperate forest, tropical forest, urban forest, riparian forest, sacred forest and old growth forest. As urban forests are more likely to be visited as they are closer to most participants (Taye *et al* 2019), we investigated whether effect sizes were larger for urban forests than all other forest types by using a dummy variable (other forest = 0; urban = 1) in the meta-analysis. We excluded the forest type variable from the revealed preference models given that it had too few observations ($n = 4$).

2.2. Meta-analyses

We used meta-analyses (Koricheva *et al* 2013) to synthesise monetary values associated with forest cultural ecosystem services across the world. The analyses were conducted on specialist and high-performance computing systems provided by information services at the University of Kent, using RStudio 4.3.1 with Metafor version 4.2.0 (Viechtbauer 2010, R Core Team 2023).

The effect size for each observation was the raw mean monetary value (WTP, consumer surplus or price) converted to 2019 USD purchasing power parity. This was computed using the *escalc* function in the '*metafor*' R-package (Viechtbauer 2010), which yields the sample mean as the effect size. As primary studies report values across heterogeneous temporal and per-capita bases (e.g. per person per trip, per household per year; table S2), we did not standardise values to a common physical or temporal unit. This was because the original studies often did not report the information needed to convert estimates consistently across units, including payment frequency and duration, the number of trips or beneficiaries, the spatial extent of the valued forest, and the scale of the environmental change being valued. Ex-post standardisation would therefore have required assumptions that were not consistently supported by the primary studies and could have introduced additional measurement error. Instead, we treated them as comparable monetary magnitudes for the purpose of analysing relative variation in reported values across studies. Each model used log-transformed mean effect sizes as the response variable with study design characteristics as possible explanatory variables, consistent with standard practice in environmental valuation meta-analysis (e.g. Brander *et al* 2007, Barrio and Loureiro 2010, Brander and Koetse 2011). The coefficients on the continuous moderators, therefore, represent the approximate percentage changes in monetary values, whereas the coefficients on dummy variables represent approximate percentage differences. To permit the log-transformation of the effect sizes, we used the absolute value of negative estimates prior to transformation, such that the dependent variable captures the absolute magnitude of monetary valuation. Given substantial variation in effect sizes, both within and between valuation methods, separate models were estimated for each non-market economic valuation method to avoid extreme coefficient estimates. This precluded pooling of different effect sizes statistically (Brander and Koetse 2011, Koricheva *et al* 2013). As the value estimates derived from different valuation methods may not be directly comparable, any comparisons drawn between different effect size metrics are, therefore, qualitative. Extracted data and R scripts for calculating and log-transforming the effect sizes are publicly available: [link omitted for double-blind review].

We estimated multivariate mixed-effects meta-regression models using the 'rma.mv' function in the '*metafor*' R-package (Viechtbauer 2010). The within-study sampling variances were treated as known and entered as the sampling variation from the effect sizes. Between-study heterogeneity was modelled through a study-level random intercept, which captured systematic differences between the studies beyond that explained by the moderator variables. Furthermore, we applied cluster-robust standard errors at the study level with small-sample corrections.

The explanatory variables included the year that data collection started. Using the year that data collection started, rather than the year that a study was published, allowed us to evaluate the influence of time on monetary values, as well as establish whether any emerging temporal patterns were modified by the other explanatory variables (table 1). Additionally, we included an interaction term between the year of data collection and an indicator for whether the study was conducted in a HIC or LMIC. This enabled us to test whether temporal trends were specific to HICs or more broadly observed across different economic contexts. In practice, this interaction term was only estimated for the stated preference studies given that all but two revealed preference studies were from HICs.

We controlled for the fact that methodological advances in stated preference valuation methods have been marked by the publication of seminal guidelines for studies by the NOAA blue-ribbon panel (Arrow *et al* 1993) and, more recently, by Johnston *et al* (2017) (table 1). These were incorporated into the model as dummy variables.

We did not differentiate between alternative specifications of the travel cost method, such as the treatment of travel time or wage rate assumptions to approximate the opportunity cost of time. These modelling differences likely contribute to residual heterogeneity across studies (Czajkowski *et al* 2019). The opportunity cost of travel time has been recognised as a central component of recreation demand models since at least the early 1980s (e.g. Smith *et al* 1983), and is typically incorporated in some form in travel cost applications. However, information on how travel time costs are specified was not consistently reported across the reviewed studies, which precluded inclusion as a moderator variable in the meta-analysis.

We tested whether effect sizes were sensitive to the scope of changes proposed in each study (Smith and Osborne 1996). Scope-sensitivity, a test of the theoretical validity of stated preference estimates, examines whether respondents were willing to pay more for a higher quality or quantity of the valued good (Rakotonarivo *et al* 2016). Previous meta-analyses often found that effect sizes were sensitive to the size of changes being valued (e.g. Smith and

Table 1. Description of the variables used in the meta-analyses. Values for continuous variables show the range; dummy variables take values of 0 or 1. Non-urban forest types include agroforest, coniferous forest, deciduous forest, forest plantation, forest (general), mixed forest, primary forest, riparian forest and tropical forests, as defined by study authors. The arrow dummy was excluded from the final model because of limited number of studies being published before 1993.

Variable	Description	Type	Values	Number of observations
Year data collection was started	The year that a study started collecting data. First year used if study over >1 year long.	Continuous	1982–2019	705
Arrow	Dummy for studies where data collection started after 1993, when seminal guidelines for stated preference methods were published by Arrow *et al* (1993)	Dummy	0 (before Arrow): 1 (after Arrow):	110 595
Johnston	Dummy for studies where data collection started after 2017, when seminal guidelines for stated preference studies were published by Johnston *et al* (2017)	Dummy	0 (before Johnston): 1 (after Johnston):	663 42
Log sample size	Number of participants per study	Continuous	4.09–10.03	705
In-situ survey	Dummy for studies which used *in-situ* surveys that were administered within the studied forest areas.	Dummy	0 *ex-situ* 1 *in-situ*	411 294
Valuation method	Stated and revealed preference methods	Categorical	Contingent valuation: Choice experiments: Travel cost: Hedonic pricing:	182 398 100 25
Effect size metric	The effect size metric used to assess the monetary value people place on cultural ecosystem services	Categorical	Consumer surplus: Marginal implicit price: Mean WTP: Median WTP: Price: Marginal rate of substitution:	99 3 568 12 22 1

(Continued.)

Table 1. (Continued.)

Variable	Description	Type	Values	Number of observations
Prevent biodiversity loss	Dummy for if the study valued prevention of biodiversity loss. Used to test scope-sensitivity.	Dummy	0 (did not value the prevention of biodiversity loss) 1 (valued the prevention of biodiversity loss)	634 71
Recover/improve biodiversity	Dummy for if the study valued recovering/improving biodiversity. Used to test scope-sensitivity.	Dummy	0 (did not value the recovery/improvement of biodiversity) 1 (valued the recovery/improvement of biodiversity)	394 311
Payment vehicle	Units in which payment was made	Categorical	House (property purchases): Missing/other: One-off (one-time payments including donations): Use fees (fees for accessing forests): Tax (payments in any form of taxes): Trip (recreational trips):	11 15 72 100 303 204
		Country		
High- income	Dummy if the study collected data in a high- or low/middle-income country	Dummy	0 (not high-income): 1 (high-income):	46 659
		Cultural ecosystem service class		
Aesthetic	Dummy if the study was on aesthetic value	Dummy	0 (not aesthetic): 1 (aesthetic):	655 50
Bequest	Dummy if the study was on bequest values	Dummy	0 (not bequest): 1 (bequest):	674 321
Cultural heritage	Dummy if the study was on cultural heritage value	Dummy	0 (not cultural heritage): 1 (cultural heritage):	690 15

(Continued.)

Table 1. (Continued.)

Education	Dummy if the study was on education value	Dummy	0 (not education): 1 (education):	700 5
Existence	Dummy if the study was on existence value	Dummy	0 (not existence): 1 (existence):	590 115
Sacred/ religious	Dummy if the study was on sacred/religious value	Dummy	0 (not sacred/religious): 1 (sacred/religious):	702 3
Scientific	Dummy if the study was on scientific value	Dummy	0 (not scientific): 1 (scientific):	700 5
Symbolic	Dummy if the study was on symbolic value	Dummy	0 (not symbolic): 1 (symbolic):	705 0
Recreation	Dummy if the study was on recreation value	Dummy	0 (not recreation): 1 (recreation):	165 540
Forest type				
Forest type	Dummy if forest type was non-urban or urban	Dummy	0 (not urban): 1 (urban):	669 36

Osborne 1996, Brouwer et al 1999, Brander et al 2007, Brander and Koetse 2011, Ojea and Loureiro 2011, Hjerpe et al 2015). For instance, Brander and Koetse (2011) reported that across the studies in their meta-regression, respondents were willing to pay more for larger areas of open space. While the scope sensitivity of effect sizes may be tested in meta-analyses by including the size of the valued area as a moderator variable, this approach may be hindered by inconsistent reporting across studies or variation in study designs (Brouwer et al 1999, Ojea and Loureiro 2011). For example, in our data, scope-related metrics varied widely, including the number of trees, percentage changes in hectares, and qualitative labels such as 'medium' or 'high.' To investigate scope sensitivity in this context, we used two proxy variables as moderators of effect sizes: first, a dummy variable for studies broadly addressing the prevention of biodiversity loss and, second, a dummy variable for studies focused on biodiversity recovery or improvement (table 1). Although this approach cannot distinguish between the size of changes, Barrio and Loureiro (2010) used it to find that effect sizes were higher for studies valuing improvements in forest conditions compared to those valuing avoiding damage.

Other moderators comprised the cultural ecosystem service class being valued and the type of terrestrial forest (table 1). We included type of payment vehicle (table 1) to examine how the different units of payment may have influenced effect sizes as this is known to influence WTP (Zhang et al 2022). Payment vehicles were coded as categorical dummy variables distinguishing taxes, one-off payments, and use fees. The 'one-off payment' payment vehicle included one study which used a voluntary donation elicitation format, which may not be fully incentive compatible (Champ and Bishop 2001) and could therefore introduce minor unobserved variation in our results, although this is unlikely to substantially affect the results. Additionally, we incorporated sample size to represent the precision of estimates were recovered within studies (Brander and Koetse 2011) and to account for heterogeneity between studies (Koricheva et al 2013). The sample size was log-transformed to reflect the elasticity of effect sizes in response to changes in sample sizes (Brander and Koetse 2011). To account for potential differences in sampling context, we included a moderator distinguishing *in-situ* from *ex-situ* surveys for choice experiments, contingent valuation studies, and travel cost studies. *In-situ* surveys were conducted within the studied forest areas, whereas *ex-situ* surveys were administered outside the study sites, including online, telephone, or mail-based data collection.

To assess whether temporal trends were influenced by changes in the composition of cultural ecosystem services valued across studies, we conducted an additional analysis restricted to studies valuing recreational services only. This model was estimated using the same specification as the main meta-regression and helps to explore whether observed temporal patterns may have been partly driven by shifts in the mix of ecosystem service classes over time.

To verify the use of these explanatory/moderating variables in the models, we report an omnibus test of them against the null hypothesis that the true value of all coefficients tested is zero (Viechtbauer 2010). This test was rejected at the 5% level for all models, validating the use of the explanatory/moderating variables in the models.

The potential for publication bias was explored via funnel plots (figure S1). We used a multivariate linear mixed-effects regression model, which is applied when effect sizes are assumed to be unbiased and normally distributed estimates of the corresponding true effect.

3. Results

3.1. Collecting and characterising the evidence

The search yielded 3457 hits (figure 1). Removal of duplicates, followed by title and abstract screening, reduced this number to 2416. In total, 353 studies met the inclusion criteria (table S2). Of these, 214 did not report variation estimates and could not be included in the meta-analysis, while two more were excluded for using a mix of methods that precluded extraction of variation estimates. The final set of 137 studies comprised 705 observations that were used in the meta-analysis (table S2). These included 49 choice experiments and 54 contingent valuation stated preference methods. Further, 36 of the studies used revealed preference approaches, including hedonic pricing (eight studies) and travel cost (28 studies). Two studies reported multiple valuation methods. Hence the total number of valuation methods exceeds the number of studies.

The number of studies published on the subject matter increased from the 1980s to 2019 (figure 2). Contingent valuation methods were employed consistently throughout. In contrast, choice experiments were deployed at low frequency in the 1980s and 1990s but gained traction between 2010 and 2019.

Most studies assessed experiential interactions with forests, such as their physical use for recreation ($n = 119$; number of observations = 540). Assessing the value of experiencing forests for their aesthetic ($n = 11$; number of observations = 50), educational ($n = 2$, number of observations = 5) and scientific value ($n = 1$; number of observations = 5) was less common (figure 3). Symbolic and spiritual interactions with forests remain understudied, with only

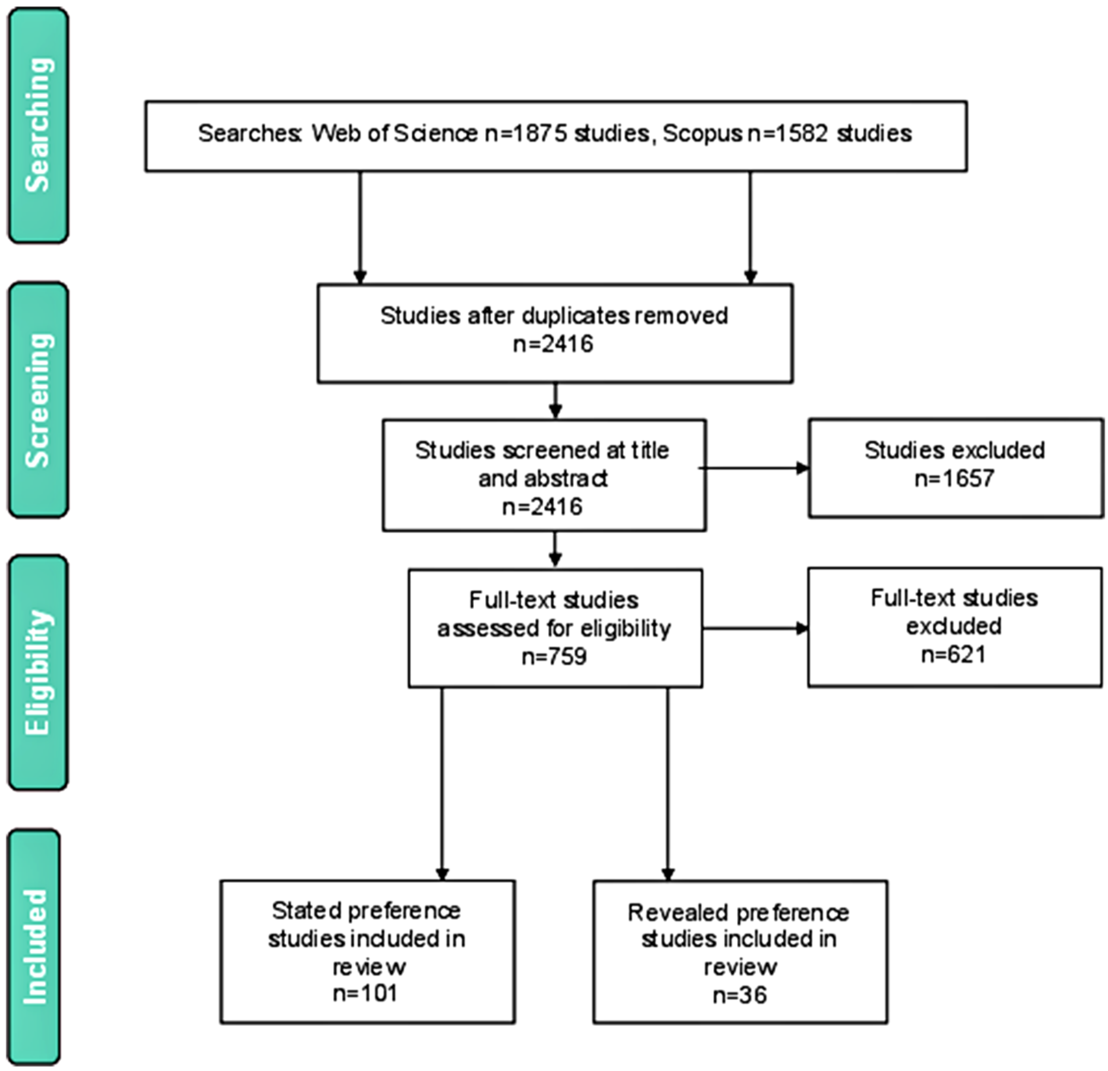

Figure 1. Flowchart showing the systematic collection and screening of studies, following a rapid evidence assessment (REA) approach. Of the 353 studies meeting the REA inclusion criteria at full-text screening, 214 were excluded at full-text screening stage due to missing variation data that were necessary to compute effect sizes, while two additional studies were excluded because mixed methods precluded extraction of variation estimates (see table S2). We included data from 137 studies in the meta-analyses.

a few studies considering cultural heritage ($n = 5$; number of observations = 15) and sacred/religious ($n = 2$; number of observations = 3) monetary values. No study investigated symbolic values. However, a few studies have examined bequest ($n = 7$; number of observations = 31) and existence ($n = 20$; number of observations = 115) values of forests. While recreation has been the most studied class of cultural ecosystem services in the 1980s and 1990s, interest in existence and bequest values has grown in the 2000s and 2010s.

We collated stated and revealed preference studies from 38 countries (figure 4, tables S4 and S5). Our findings demonstrate that the evidence-base is biased towards HICs, with the geographical focus of most work being the USA ($n = 31$), UK ($n = 15$) and Spain ($n = 14$). Africa and South America remain under-represented, with few studies conducted within each continent, whereas more attention has been paid to Asia, covering a range of countries. Only 14% of studies were conducted in LMICs (figure 5).

3.2. Meta-analysis

Although fewer studies used revealed preference methods, average sample sizes were larger than for stated preference non-market economic valuations (table 2). Within stated preferences, total contingent valuation studies were more numerous, although observations in choice experiments were higher due to some individual choice experiments reporting WTP for different attributes.

In the stated preferences studies, for both contingent valuation and choice experiment studies, the year that data collection started was not statistically significantly associated with effect sizes, meaning that temporal trends in associated monetary values were not significant (figure 6; table 3). Log sample size had a significantly positive association with effect sizes

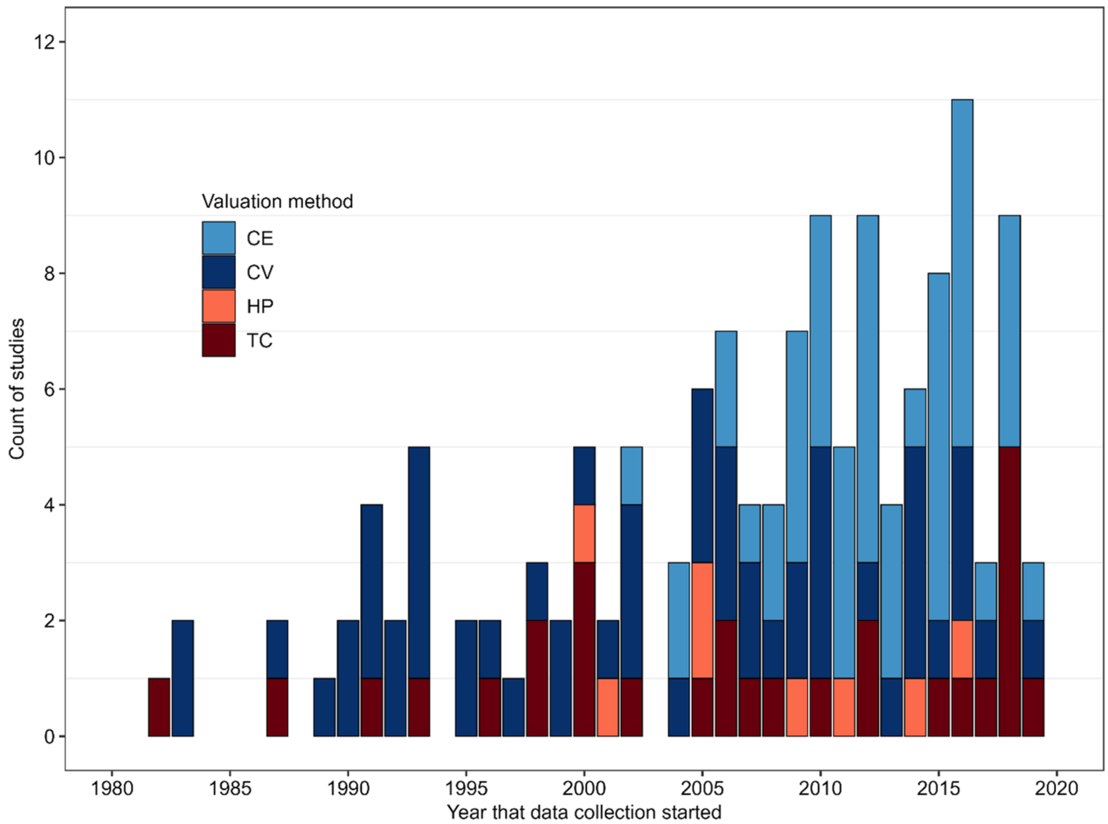

Figure 2. Number of studies ($n = 137$) included in the meta-analysis, grouped by non-market economic valuation method: CE, choice experiments ($n = 49$ stated preference); CV, contingent valuation ($n = 54$; stated preference); HP, hedonic pricing ($n = 8$; revealed preference); TC, travel cost ($n = 28$; revealed preference). Two studies reported multiple valuation methods, so the total number of valuation methods exceeds the number of studies.

for contingent valuation studies. The publication of Johnston *et al* (2017) was positively associated with effect sizes in contingent valuation studies, but not in choice experiments. The interaction between high-income country and year of data collection was not statistically significant for either valuation method, providing no evidence that temporal trends differed systematically between high-income and low- and middle-income country contexts. The *in-situ* survey moderator was not statistically significant for either choice experiments or contingent valuation studies, indicating no evidence of systematic differences in effect sizes between surveys conducted within forest sites and those administered *ex-situ*.

The analysis did not provide evidence of a statistically significant association of the payment vehicle on effect sizes for either choice experiment or contingent valuation studies (table 3). None of the cultural ecosystem services differed significantly in the choice experiment model, whereas in contingent valuation studies, effect sizes were significantly lower when aesthetic and recreation values were considered. Bequest and existence values were not statistically significant in either model. The scope sensitivity variables also showed method-specific patterns. In choice experiments, effect sizes were significantly higher in studies valuing the prevention of biodiversity loss, whereas recovering or improving biodiversity was not statistically significant. In contingent valuation studies, the opposite pattern emerged, with recovering or improving biodiversity being positively associated with effect sizes, while preventing biodiversity loss was not statistically significant.

In the recreation-only models, the year that data collection started was not statistically significant in either the choice experiment or contingent valuation model, indicating no evidence of a general temporal trend in recreational forest values within either stated preference method (table S6). Other moderator effects were method specific. For choice experiments, one-off payment vehicles and prevention of biodiversity loss were positively associated with effect sizes, whereas aesthetic values were negatively associated with effect sizes. For contingent valuation studies, aesthetic values were negatively associated with effect sizes, while prevention of biodiversity loss, recovery or improvement of biodiversity, and urban forest settings were positively associated with effect sizes.

We next present results from separate mixed-effects meta-regression models estimated for the revealed preference studies (table 4). The year that data collection started was not statistically significant

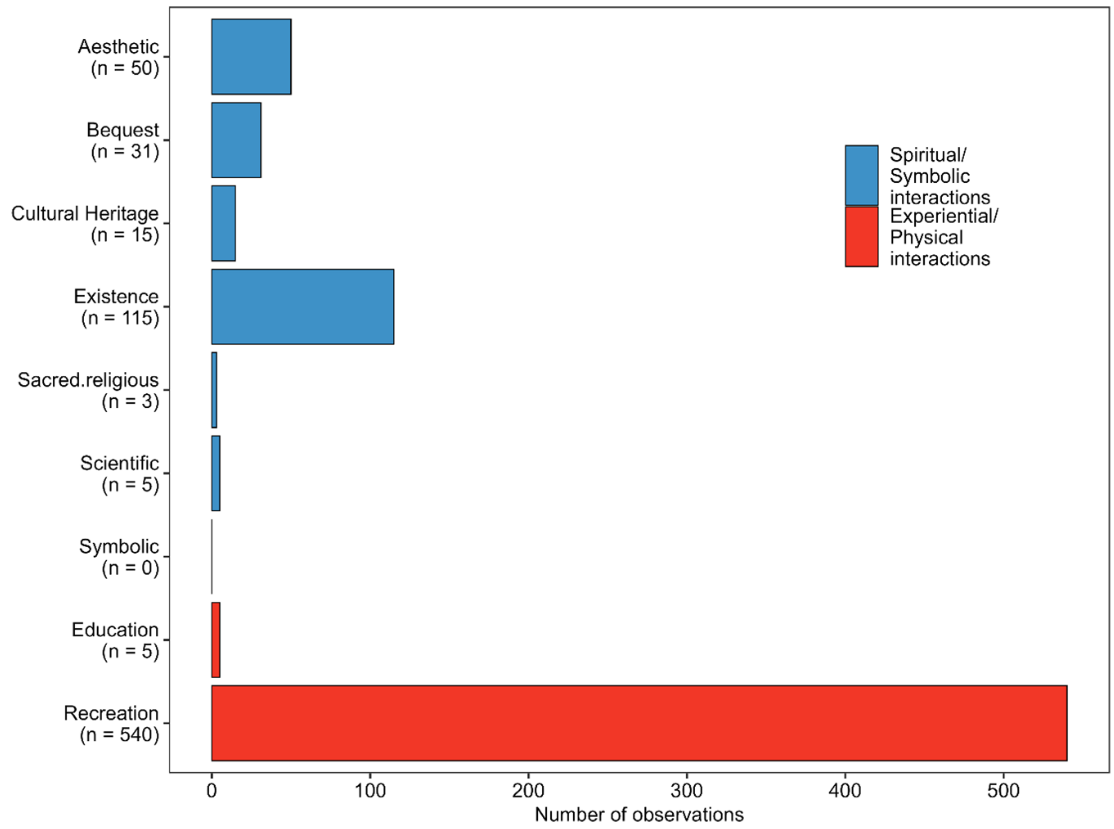

Figure 3. The number of observations for different cultural ecosystem service classes, classified following the common international classification of ecosystem services framework (CICES), assessed in the 137 studies included in the meta-analysis.

for either travel cost or hedonic pricing studies, indicating no evidence of a temporal trend in effect sizes. Log sample size was also not statistically significant in either model. For travel cost studies, the sampling context moderator was not statistically significant, suggesting that whether data were collected *in-situ* or *ex-situ* did not systematically influence estimated values. The cultural ecosystem service class moderators included in the revealed preference models were not statistically significant, with no significant association for existence values in the travel cost model or aesthetic values in the hedonic pricing model. Hedonic pricing observations only came from HICs, and we did not have enough LMIC observations to evaluate whether travel cost effect sizes were sensitive to country income classification. Furthermore, we could not test for the scope sensitivity of effect sizes in the revealed preference studies as they primarily evaluated recreational visits rather than changes to biodiversity. We also excluded terrestrial forest type from the revealed preferences studies due to insufficient observations for model estimation.

4. Discussion

In this meta-analysis, we assessed the monetary values associated with forest cultural ecosystem services and biodiversity, examining temporal changes in the period from the 1980s to 2019. This allowed us to assess whether monetary values of forest cultural ecosystem services have changed systematically over time. Our meta-regression results do not provide evidence of statistically significant temporal trends in monetary values for either stated preference or revealed preference methods. For contingent valuation and choice experiment studies, the year that data collection started was not statistically significant, indicating no evidence that effect sizes changed systematically over time. Country income status and the interaction between high-income country status and year of data collection were also not statistically significant in the full stated preference models, providing no evidence that temporal trends differed systematically between high-income and low- and middle-income country contexts. In the recreation-only models, the year of data collection was likewise not statistically significant for either choice experiments or contingent valuation studies. For revealed preference studies, the travel cost model showed no statistically significant association between year of data collection and effect sizes. Although fewer studies employed hedonic pricing methods, there was no significant temporal trend in estimated values.

These findings do not support our initial expectation that monetary values associated with forest cultural ecosystem services would have experienced

Figure 4. Geographic distribution studies over the last four decades (based on the year data collection was started) included in the meta-analysis ($n = 137$), grouped by non-market economic valuation method: CE, choice experiments ($n = 49$; stated preference); CV, contingent valuation ($n = 54$; stated preference); HP, hedonic pricing ($n = 8$; revealed preference) and TC, travel cost ($n = 28$; revealed preference). Two studies reported multiple valuation methods, so the total number of valuation methods exceeds the number of studies. Tables S4 and S5 report the number of studies by decade, continent, country, and valuation method.

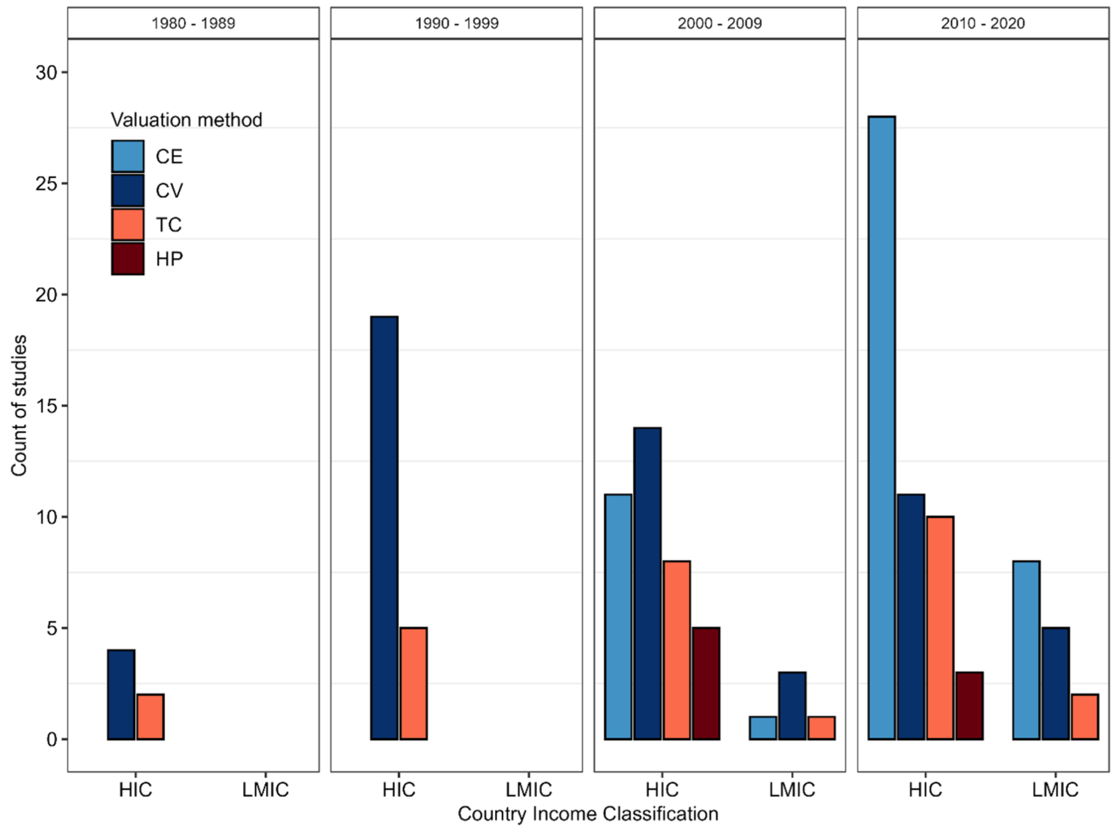

Figure 5. Studies conducted in high- (HICs) and low-/middle-income countries (LMICs) included in the meta-analysis ($n = 137$), grouped by non-market economic valuation method: CE, choice experiments ($n = 49$; stated preference); CV, contingent valuation ($n = 54$; stated preference); HP, hedonic pricing ($n = 8$; revealed preference); TC, travel cost ($n = 28$; revealed preference). Two studies reported multiple valuation methods, so the total number of valuation methods exceeds the number of studies.

consistent substantial alterations through time. Our analysis does not provide evidence for statistically significant temporal trends for either stated or revealed preference methods, nor does it provide evidence for a uniform directional trend across all valuation methods. This should be interpreted as an absence of detectable temporal associations in the available evidence base, rather than as evidence that underlying preferences have remained stable. Given the substantial residual heterogeneity in the models, any temporal signal may have been difficult to identify if it was smaller than, or confounded with, variation arising from study design, valuation context, and the types of cultural ecosystem services valued. The results therefore suggest that methodological and contextual heterogeneity may be more important for explaining variation in monetary values than a consistent temporal trend.

The COVID-19 pandemic was an unprecedented disruption that altered forest visitation patterns and the way people engaged with natural environments. Studies report marked increases in forest use during lockdowns, driven by restrictions on other leisure activities and heightened awareness of well-being benefits (Venter *et al* 2020, Wunderlich *et al* 2024). For example, forest visits rose sharply in Switzerland, and Norway experienced a 291% increase in outdoor recreation. Similar patterns elsewhere suggest temporary increases of the psychological, social, and economic values attributed to forests (Pichlerová *et al* 2023). As these trends have since declined in some regions (Wunderlich *et al* 2024), we restricted our analysis to pre-2020 studies to avoid confounding long-term valuation trends with these short-lived pandemic effects. Future research may assess whether any enduring changes in public preferences have emerged.

The recreation-only models showed the same main temporal pattern as the full stated preference models. Neither specification provided evidence of a statistically significant temporal trend in effect sizes. However, some moderator effects differed between the full and recreation-only models. In the full choice experiment model, none of the cultural ecosystem service classes were statistically significant, whereas in the recreation-only model, studies that additionally valued aesthetic services had significantly lower effect sizes. One-off payment vehicles and prevention of biodiversity loss were also positively associated with effect sizes in the recreation-only choice experiment model, although these effects were not evident in the same way in the full model. For contingent valuation

Table 2. Number of observations and mean log effect sizes by non-market economic valuation method. Two studies reported multiple valuation methods, so the total number of valuation methods exceeds the number of studies. Log-transformed mean effect sizes were used for all studies. In addition, de-logarithmised mean and median effect sizes (with standard deviation) are reported.

	Choice experiments	Contingent valuation	Hedonic pricing	Travel cost
Number of observations	398	182	25	100
Number of studies	49	54	8	28
Mode year that data collection was started (oldest to latest)	2012 (2002–2019)	2014 (1983–2019)	2009 (2000–2016)	1991 (1982–2019)
Log-transformed mean sample size (min to max)	6.39 (4.50–7.60)	6.45 (4.09–9.03)	6.16 (5.42–9.68)	7.36 (4.72–10.03)
Log-transformed mean effect size (min to max)	2.67 (−3.68–5.93)	3.56 (−0.70–11.06)	4.48 (−8.99–10.65)	4.63 (−1.05–8.02)
Mean effect size (min to max)	24.78 (−115.42–377.86)	2739.45 (0.50–63 277.83)	−4636.41 (−42292.68–4703.79)	363.60 (0.35–3034.14)
Median effect size (Std. Dev.)	11.86 (62.27)	24.69 (8704.13)	−0.00 (12544.59)	96.51 (666.60)

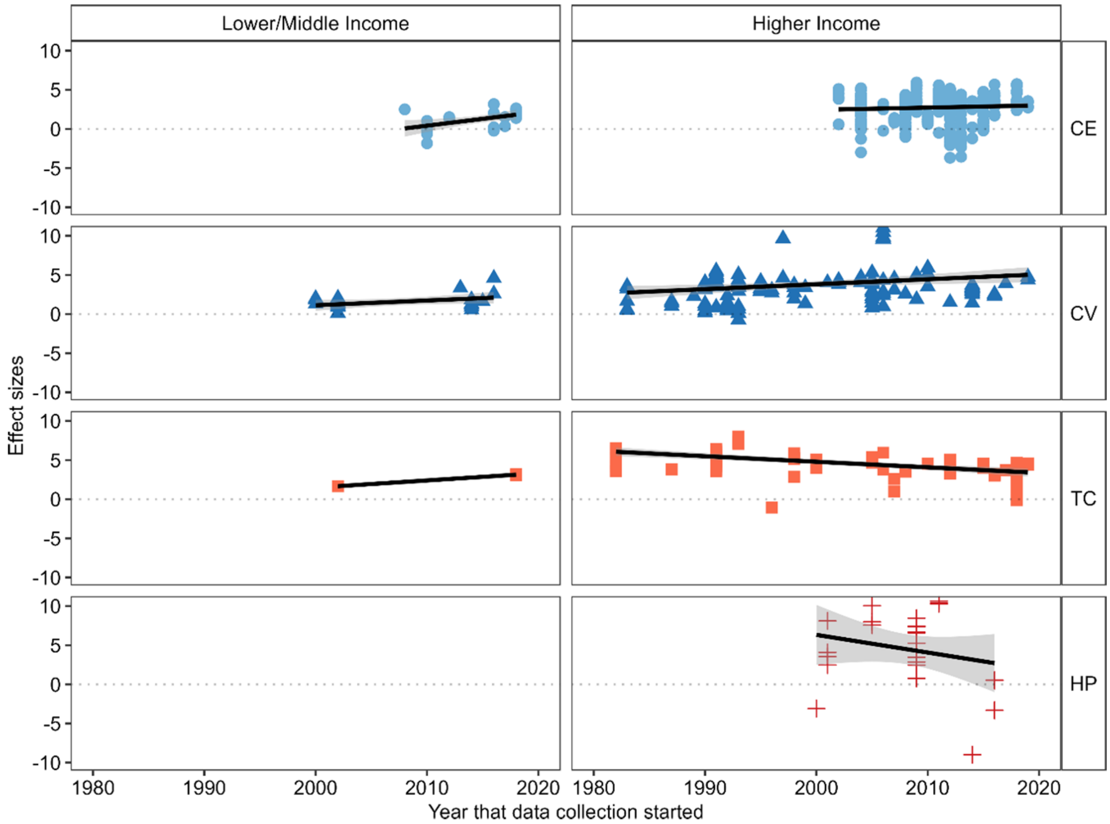

Figure 6. Log-transformed effect sizes against the year that data collection was started for each non-market economic valuation method. Trend line plotted using a generalised linear model of effect size versus year of data collection. Stated preference methods include choice experiments (CE; number of observations = 398) and contingent valuation (CV; number of observations = 182), while revealed preference methods include travel cost (TC; number of observations = 100) and hedonic pricing (HP; number of observations = 25).

studies, the recreation-only model partly reflected patterns observed in the full model but also showed additional moderator effects. Aesthetic values were negatively associated with effect sizes, while recovery or improvement of biodiversity and urban forest settings were positively associated with effect sizes. In addition, prevention of biodiversity loss was positively associated with effect sizes in the recreation-only contingent valuation model, although this effect was not statistically significant in the full contingent valuation model. These differences suggest that restricting the analysis to recreation does not reveal temporal trends but rather indicates differences in study design. This is consistent with the broader valuation literature, which shows that stated preference estimates are sensitive to attribute framing, scenario specification, and the definition of the valued good (Hensher *et al* 2005, Johnston *et al* 2017).

The effect sizes from choice experiment and contingent valuation studies provide indicative evidence consistent with scope sensitivity, as they were significantly associated with the type of biodiversity change being valued. This aligns with previous meta-analyses indicating that effect sizes can be sensitive to the size of changes being valued (Smith and Osborne 1996, Brander *et al* 2007, Brander and Koetse 2011, Ojea and Loureiro 2011, Hjerpe *et al* 2015). However, it is important to note that our variables represented indirect proxies for scope and do not constitute a direct test of scope sensitivity. In choice experiments, effect sizes were significantly higher for studies focusing on preventing biodiversity loss, while in contingent valuation studies, effect sizes were significantly higher for studies focusing on recovery or improvement of biodiversity. This could be due to the inherent differences in methodological approaches between contingent valuation and choice experiments. Choice experiments typically involve trade-offs between multiple attributes, which may lead respondents to consider biodiversity changes more holistically, potentially masking scope effects (Dugstad *et al* 2020). In contrast, contingent valuation often focuses on specific scenarios, allowing for a more direct comparison between prevention and improvement, which may enhance scope sensitivity (Lopes and Kipperberg 2020).

Across the reviewed studies, we did not find evidence that those conducted in HICs reported larger effect sizes compared to those in LMICs, either in the full model or in the recreation-only model. However, the dataset still showed uneven coverage

Table 3. Meta-analysis estimates (standard errors in parentheses) for choice experiment ($n = 49$) and contingent valuation ($n = 54$) studies. Values highlighted in bold indicate statistical significance ($p^* < 0.05$, $p^{**} < 0.01$, $p^{***} < 0.001$). The payment vehicle estimates tax (payments in the form of taxes), one-off (one-time payments), and use fees (fees for accessing forests) are relative to that for trip (recreational trip cost). The high-income country estimate is in comparison to low/middle-income country. We estimate the effect of cultural ecosystem service class relative to studies not including that service (i.e. the estimate for recreation relates to the change in effect sizes from studies that value recreation versus those that do not). We only included cultural ecosystem services that had sufficient observations to allow comparisons to be made. The urban forest estimate was compared to all other forest types (agroforest, coniferous forest, deciduous forest, forest plantation, forest (general), mixed forest, primary forest, riparian forest, tropical forests) combined. The intercept represents the predicted log-transformed effect size for the reference categories when continuous covariates equal zero. Because the model is estimated on a log scale and predictors are transformed, the intercept primarily reflects model parameterisation. 1985–2019 reflects the temporal range of the year in which data collection was started, even though we included studies from 1980 onwards.

Variable	Choice experiments	Contingent valuation
Intercept	−466.31 (380.04)	−38.31 (131.94)
Year data collection was started (1985–2019)	0.23 (0.19)	0.02 (0.07)
Arrow (1 = post-publication)	—	1.20 (0.76)
Johnston (1 = post-publication)	0.97 (0.71)	**3.18*** (1.11)**
Log sample size	0.30 (0.34)	**0.50** (0.20)**
In-situ survey (1 = *in-situ* survey)	0.26 (0.75)	−0.42 (0.61)
Payment vehicle		
Tax (1 = tax as a payment vehicle)	0.83 (0.55)	−0.50 (0.55)
One-off (1 = a one-off payment as the vehicle)	1.01 (0.78)	1.10 (1.88)
Use fees (1 = payment vehicle was a fee to use the good)	−0.44 (0.36)	−0.42 (0.62)
Country		
High-income (0 = country was low/middle-income)	403.21 (387.93)	101.11 (136.65)
Data collection year*High-income interaction	−0.20 (0.19)	−0.05 (0.07)
Cultural ecosystem service class		
Aesthetic (0 = did not value aesthetic services)	−0.70 (0.42)	**−2.01*** (0.49)**
Bequest (0 = did not value bequest services)	0.22 (0.63)	−0.77 (1.72)
Existence (0 = did not value existence services)	0.01 (0.44)	−1.31 (1.60)
Recreation (0 = did not value recreation services)	0.40 (0.44)	**−1.32** (0.59)**
Scope of change		
Prevent biodiversity loss (0 = did not value prevention of loss)	**0.62*** (0.00)**	−0.91 (1.73)
Recover/improve biodiversity (0 = valued other changes to biodiversity)	0.60 (0.56)	**1.21*** (0.36)**
Forest type		
Urban forest (0 = study area was any other greenspace)	−0.55 (0.69)	**1.54* (0.85)**
Diagnostics		
Observations	398	182
Studies	49	54
Residual heterogeneity test	**3267.50*** **	**842.66*** **
Omnibus moderators test (Log test statistic)	**7879.64*** **	**175.51*** **

across country-income contexts and ecosystem service classes. There were no estimates for services other than recreation in LMICs, for instance. This reflects a predominant focus on assessing recreational values in most studies, rather than other cultural ecosystem services such as religious, cultural heritage or symbolic values. In the context of this paper, the absence of a statistically significant country-income effect should not be interpreted as evidence that cultural values and preferences for ecosystem services are comparable across regions and socio-economic contexts. Rather, it highlights the limited and uneven evidence base available for assessing such differences. Factors such as cultural norms and socio-economic conditions can shape the relative importance attributed to different types of cultural ecosystem services (Li 2014, Nawrath *et al* 2021a). For example, preferences for interactions with ecosystems in many Asian countries are shaped by religious beliefs (Nawrath *et al* 2021b). More efforts should be made to explore a broader range of cultural ecosystem services in future studies, beyond just recreation, particularly in LMICs where other types of cultural values may be of greater importance to people. It is important to acknowledge, however, that our study was restricted to studies published

Table 4. Meta-analysis estimates (standard errors in parentheses) for travel cost ($n = 28$) and hedonic pricing ($n = 8$) studies. Values highlighted in bold indicate statistical significance ($p^* < 0.05$, $p^{**} < 0.01$, $p^{***} < 0.001$). Blanks indicate that a variable was automatically dropped during the estimation due to too few observations. We estimate the effect of cultural ecosystem service class relative to studies not including that service (i.e. the estimate for recreation relates to the change in effect sizes from studies that value recreation versus those that do not). We only included cultural ecosystem services that had sufficient observations to allow comparisons to be made. The intercept represents the predicted log-transformed effect size for the reference categories when continuous covariates equal zero. As the model is estimated on a log scale and predictors are transformed, the intercept primarily reflects model parameterisation. 1985–2019 reflects the temporal range of the year in which data collection was started, even though we included studies from 1980 onwards.

Variable	Travel cost	Hedonic pricing
Intercept	125.29 (81.84)	451.30 (592.94)
Year data collection was started (1985–2019)	−0.06 (0.04)	−0.22 (0.30)
Log sample size	−0.00 (0.35)	−2.10 (1.79)
In-situ survey (1 = *in-situ*)	−0.59 (1.36)	—
Cultural ecosystem service class		
Existence	0.55 (1.07)	
Aesthetic		4.02 (5.14)
Diagnostics		
Observations	100	25
Studies	28	8
Residual heterogeneity test	**5985.04*****	NA
Omnibus moderators test (Log test statistic)	**23.57***	0.97

in English, potentially omitting relevant evidence available in other languages (Amano *et al* 2021, 2023, Arenas-castro *et al* 2024).

In terms of cultural ecosystem service classes, we found that a large majority of studies assessed recreation as experiential interactions with forests. This may be because recreation is more straightforward to quantify and price compared to other classes of services (Chan *et al* 2012). It may also reflect to some extent that most non-market economic valuation method studies have been conducted within HICs and recreation is a cultural practice associated with such countries. In contrast, little attention has been paid to symbolic and spiritual interactions with forests in valuation studies. This may be because less-tangible elements are frequently left out of traditional monetary valuation frameworks, which reflects that they are intangible and subjective by definition and, since the demand for such services is not obvious, quantifying their supply is difficult (Small *et al* 2017). The focus of the valuation literature on recreation may lead to an overemphasis on conserving forests with high recreational value, while forests that provide fewer recreational opportunities, but do deliver other important cultural ecosystem services may be neglected by conservation policies (Hermes *et al* 2018). Furthermore, assigning monetary values to symbolic and spiritual interactions is considered problematic for both methodological and ethical reasons by some (Bartkowski 2017). In the literature on non-monetary values, on the other hand, symbolic and spiritual interactions are well represented (e.g. the concept of Nature's Contributions to People, (Díaz *et al* 2018) or relational values, (Chan *et al* 2018)). By integrating plural values into valuation efforts, policies can be more accurately tailored to achieve equitable outcomes for all segments of society, particularly for LMIC populations (Ihemezie *et al* 2023). This approach may help avoid policies that disproportionately benefit high-income groups while neglecting marginalised groups who might value aspects of forests that are more difficult to quantify economically (Ihemezie *et al* 2023).

The mounting interest in existence and bequest values since the 2000s may reflect growing public awareness of the role forests play in climate change mitigation and biodiversity conservation, fuelled by increasing media coverage on environmental challenges (Hase *et al* 2021). Public values for forest biodiversity are heterogeneous and shaped by personal experience (Austen *et al* 2023), which highlights the importance of assessing monetary values associated with symbolic and spiritual interactions with forests. Without such valuation, these benefits risk being overlooked in decision-making processes such as cost-benefit analyses that typically inform environmental policy (Hanley and Barbier 2009, OECD 2018). Consequently, future research could investigate those classes of cultural ecosystem services that have been studied least, including cultural heritage, spiritual and religious interactions, symbolic, and scientific values.

We found that cultural ecosystem service class mattered in contingent valuation studies, but not in choice experiments or revealed preference models. In contingent valuation studies, aesthetic and recreation values were associated with lower effect sizes. This lack of consistent pattern across valuation methods could potentially be due to the small sample size of studies assessing aesthetic values, or because of methodological differences, as only choice experiments can tease apart values for separate attributes (Hensher *et al* 2005). Although existence values were not associated with effect sizes in our models, the small number of studies assessing non-use values limits our ability to draw conclusions about how these values compare with cultural ecosystem service classes representing direct-use value.

The results presented here emerge from a meta-analytical framework that integrated valuation estimates derived from diverse study designs, modelling assumptions, and empirical contexts. Differences in sampling, valuation implementation, and model specification were treated as between-study heterogeneity within the mixed-effects framework rather than as bias. Consequently, the estimated associations represent aggregate tendencies across heterogeneous valuation approaches, not precise estimates tied to any single methodological protocol. Accordingly, because primary studies reported values using heterogeneous units (e.g. per trip, per household per year), the effect sizes should be interpreted as relative monetary magnitudes rather than directly comparable welfare measures.

Methodological practices within valuation approaches have also evolved over time. For example, contingent valuation studies increasingly employ refined elicitation formats and survey protocols designed to reduce bias. Likewise, travel cost studies differ in modelling assumptions, such as the treatment of travel time costs or wage rate parameters, which may influence estimated values. Because such within-method design features were not coded explicitly in the analysis, these differences are absorbed as residual heterogeneity in the meta-analytical models in this paper. Differences in sampling context may also contribute to variation in valuation estimates, as *in-situ* surveys tend to capture active forest users, who typically report higher WTP than broader population samples (Freeman *et al* 2014, Johnston *et al* 2017). However, our results did not indicate that sampling context exerted a systematic influence on effect sizes, suggesting that such methodological differences likely contribute to between-study heterogeneity rather than representing a consistent driver of temporal trends (Bell *et al* 2011). Future meta-analyses that explicitly code methodological refinements could help disentangle changes in valuation practice from shifts in underlying preferences.

Nearly two-thirds of the studies meeting the inclusion criteria could not be included in the meta-analysis because they did not report variance data required for effect size estimation. This meant that these studies could not be incorporated into a variance-based meta-analysis, and statistical inference should be interpreted with caution, as it is based on a reduced sample that may not fully represent the underlying literature. This is a well-documented barrier in meta-analytic practice, and previous reviews have called for stronger reporting standards (Kambach *et al* 2020). We echo these calls and recommend that future studies report variance estimates as standard practice.

Because temporal trends were not statistically significant in either the full stated preference models or the recreation-only models, the absence of detectable temporal patterns is unlikely to be explained solely by variation in the cultural ecosystem service classes included across studies (Brander and Koetse 2011). At the same time, the persistence of method-specific patterns and substantial residual heterogeneity indicates that temporal variation in valuation estimates is likely shaped by changes in research focus, methods, and contextual differences across studies, rather than representing a consistent directional change in underlying preferences (Johnston *et al* 2018).

Our findings have important implications for the practice of benefit transfer, where existing data are used to calculate values in settings other than for what they were originally collected (Rosenberger and Loomis 2003). Although we did not find evidence of temporal trends in the value people place on forest cultural services, the substantial heterogeneity and method-specific patterns observed in our models suggest that temporal stability should not be assumed when undertaking benefit transfers. This is critical to improve the precision of monetary estimates, which appear to be dependent on the non-market economic valuation method used. The potential for over- or under-estimation due to methodological and contextual mismatches, including possible temporal instability, can compromise the validity and reliability of estimates (Rakotonarivo *et al* 2016, Rolfe and Dyack 2019), potentially eroding the credibility of benefit transfers (Bauer and Johnston 2013), which may undermine the integration of ecosystem service values into policymaking (Richardson *et al* 2015a). While consensus exists among non-market valuation and benefit transfer practitioners that the reliability of benefit transfers tends to diminish over longer time periods (Johnston *et al* 2015, Brouwer *et al* 2017), the evidence pertaining to value stability remains mixed (Czajkowski *et al* 2016, Rolfe and Dyack 2019). It is important to note that the inclusion of only a limited number of hedonic pricing studies with substantial variation in effect sizes warrants caution in interpreting our findings. Although prior studies have examined valuation stability over shorter or study-specific time horizons, our analysis addressed multi-decade temporal trends. The literature on stated preference methods generally reports short-term stability (Liebe *et al* 2016, Williams 2022, Wunsch *et al* 2022), while the evidence from revealed preference studies suggests greater variability over longer periods (Brouwer and Bateman 2005, Neher *et al* 2017, Price *et al* 2017, Rolfe and Dyack 2019). These bodies of work are therefore best understood as complementary rather than directly comparable. Our findings reflect that evidence on temporal stability remains limited, and more research is needed to further elucidate the dynamics of value changes through time and their implications for benefit transfer accuracy.

The choice of payment vehicle is an important design consideration in environmental valuation studies, as it can affect the monetary estimates that are derived (Ivehammar 2009, Hassan *et al* 2018). In the full stated preference models, however, we did not find evidence that payment vehicle had a statistically significant association with effect sizes in either contingent valuation or choice experiment studies. In the recreation-only models, one-off payments were positively associated with effect sizes in choice experiments. This suggests that payment vehicle effects were not consistent across model specifications and should therefore be interpreted cautiously. The effect of payment vehicle is likely to be context dependent. In particular, coercive payment formats such as taxes are more likely to satisfy incentive compatibility conditions and reduce strategic behaviour, whereas voluntary contributions may be more susceptible to free-riding and hypothetical bias (Carson and Groves 2007, Vossler and Holladay 2018). Thus, the choice of payment vehicle should be understood as contingent on study design and institutional context (Ivehammar 2009). Furthermore, the choice of payment vehicle in LMICs needs particular attention because many government tools, such as tax systems, are still under development, potentially limiting their applicability (Hassan *et al* 2018). Monetary valuations may also be generally less appropriate for cultural reasons (Ihemezie *et al* 2023), due to subsistence lifestyles, lower incomes (Hagedoorn *et al* 2021) or if trading is commonly done via bartering (Meginnis *et al* 2020).

5. Conclusion

This study synthesised four decades of economic valuation research to assess whether monetary values associated with forest cultural ecosystem services have changed over time. Using a global meta-analysis of 137 studies and 705 valuation estimates, we found no evidence for temporal trends across valuation methods. Contingent valuation, choice experiment, travel cost, and hedonic pricing studies did not provide evidence of statistically significant temporal changes in reported monetary values. Likewise, the recreation-only stated preference models did not provide evidence of a statistically significant temporal trend. We also found no evidence that temporal patterns differed systematically between high-income and low- and middle-income country contexts.

These findings suggest that temporal variation in valuation estimates is not consistent across valuation methods and is likely influenced by the heterogeneous contexts in which studies are conducted. The lack of statistically significant temporal patterns thus does not provide evidence of uniform changes in values associated with forest cultural ecosystem services across valuation approaches or geographic contexts over the four decades from the 1980s to 2019. Instead, the observed patterns likely reflect a combination of evolving research interests, methodological variation, and contextual differences across studies.

Our review also highlights important gaps in the evidence base. The literature remains strongly concentrated on HICs and recreational interactions with forests, while symbolic, spiritual, and cultural heritage values remain rarely quantified. Expanding valuation research across underrepresented cultural ecosystem service classes and geographic regions would improve the empirical basis for understanding how people value forests and the associated biodiversity.

From a policy perspective, our results highlight the importance of considering methodological heterogeneity and contextual differences when applying benefit transfer methods in environmental decision-making. Strengthening the geographic and thematic diversity of valuation studies will therefore be essential for improving the reliability of ecosystem service valuation and ensuring that forest management and conservation policies reflect the diverse ways in which societies value forest ecosystems.

Acknowledgments

This research was funded by the European Research Council (ERC) under the European Union's Horizon 2020 research and innovation programme (Consolidator Grant No. 726104) and by the Norwegian Research Council (Grant Contract No. 342628/L10).

Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: https://doi.org/10.22024/UniKent/01.01.501 (Nawrath *et al* 2026).

Supplementary data 1 available at: https://doi.org/10.1088/1748-9326/ae9b25/data1.

Declaration of interests

The authors declare no competing interests.

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