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Climate change mitigation potential of wetlands and the cost-effectiveness of their restoration

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The cost-effective mitigation of climate change through nature-based carbon dioxide removal strategies has gained substantial policy attention. Inland and coastal wetlands (specifically boreal, temperate and tropical peatlands; tundra; floodplains; freshwater marshes; saltmarshes; and mangroves) are among the most efficient natural long-term carbon sinks. Yet, they also release methane (CH_4) that can offset the carbon they sequester. Here, we conducted a meta-analysis on wetland carbon dynamics to (i) determine their impact on climate using different metrics and time horizons, (ii) investigate the cost-effectiveness of wetland restoration for climate change mitigation, and (iii) discuss their suitability for inclusion in climate policy as negative emission technologies. Depending on metrics, a wetland can simultaneously be a net carbon sink (i.e. boreal and temperate peatlands net ecosystem carbon budget = -28.1 ± 19.13 gC m^{-2} y^{-1}) but have a net warming effect on climate at the 100 years time-scale (i.e. boreal and temperate peatland sustained global warming potential = 298.2 ± 100.6 gCO_2 eq^{-1} m^{-2} y^{-1}). This situation creates ambivalence regarding the effect of wetlands on global temperature. Moreover, our review reveals high heterogeneity among the (limited number of) studies that document wetland carbon budgets. We demonstrate that most coastal and inland wetlands have a net cooling effect as of today. This is explained by the limited CH_4 emissions that undisturbed coastal wetlands produce, and the long-term carbon sequestration performed by older inland wetlands as opposed to the short lifetime of CH_4 in the atmosphere. Analysis of wetland restoration costs relative to the amount of carbon they can sequester revealed that restoration is more cost-effective in coastal wetlands such as mangroves (US\$1800 ton C^{-1}) compared with inland wetlands (US\$4200–49 200 ton C^{-1}). We advise that for inland wetlands, priority should be given to conservation rather than restoration; while for coastal wetlands, both conservation and restoration may be effective techniques for climate change mitigation.

1. Introduction

Climate change mitigation is a pressing international need, with many management actions that can contribute to it. The Intergovernmental Panel on Climate Change does not consider it possible to limit global warming to 2°C without the use of negative emissions technologies [1]. Seven categories of negative emissions technology have been identified: bioenergy with carbon capture and storage, biochar, direct air capture, enhanced weathering, ocean fertilization and natural climate solutions (*sensu* [2]) such as afforestation & reforestation and soil carbon sequestration [3]. Most of these techniques require large financial investment before they can be implemented at the global scale [4]. However, the latter two—'afforestation and reforestation' and 'soil carbon

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sequestration'—are available now and may be among the most cost-effective negative emissions technologies [4]. Ecosystem conservation and ecological restoration can play a crucial role in mitigating climate change both now and in the future and are beginning to receive substantial research and policy interest [5–8].

Inland and coastal wetlands have, per unit area, high soil carbon densities relative to other ecosystems [9]. Key wetland types include peatlands (bogs and fens), mineral wetlands (marshes, tundra), seasonal or permanent floodplains and coastal wetlands (e.g. mangroves, saltmarshes). Unlike other ecosystems, carbon storage in wetlands does not reach saturation, as it accumulates primarily in the soil over century to millennial time scales [10]. This makes wetlands an effective and long-term nature-based approach to mitigating climate change, if the carbon they store is greater than the methane (CH_4) they emit in terms of radiative forcing. The effect of CH_4 emissions on the net radiative forcing of wetlands is however unclear. Long-term monitoring sites are still limited and there is a debate on which metric to use to accurately evaluate the radiative effect of wetlands and their possible inclusion in climate change mitigation schemes [11,12], including for cost-effectiveness of restoration.

Under the United Nations Framework Convention on Climate Change (UNFCCC), greenhouse gases (GHGs) are reported in CO_2-equivalent (CO_2-e) emissions using a global warming potential (GWP) over 100 years. Despite being broadly used, three major critiques regarding GWP can be addressed when assessing the carbon dynamics of ecosystems. First, GWP considers GHG emissions as a single pulse while ecosystem emissions are usually continuous throughout time. Second, the initial GWP metric did not consider, until 2013, the indirect effect and feedbacks of non-CO_2 pathways such as CH_4 oxidation and associated CO_2 production (i.e. climate–carbon feedback). This has led to confusion and a development of different GWP values under the same metric name with inconsistencies between publications when referring to GWP [13]. Third, the 100-year time-scale is purely arbitrary and disconnected from policymaker timespan.

Several approaches have been proposed to circumvent these flaws with their own key attributes and limitations [12,14]. Considering the wide range of situations in which a climate metric is required to assess the effect of a system, no single metric can satisfy the need of all applications [14]. For instance, the climate metric required might differ greatly between a life cycle technology, a national emission estimate, an energy system pathway or an ecosystem exchange assessment [15]. In the specific field of ecosystem biogeochemistry, two approaches have attempted to estimate the radiative effect of natural environments by considering their specificities. First, the sustained GWP (SGWP) which takes into consideration the first two caveats mentioned above (i.e. steady continuous emissions and CH_4 to CO_2 when oxidized). Second, the switchover time which allows determining at what age a wetland has a net radiative cooling effect [11]. Given the increasing number of wetland studies that estimate net ecosystem carbon budget (NECB) [16], and the need to accurately assess the role wetlands play in the modern climate by considering the true effect of other GHGs such as CH_4, a consistent global evaluation of wetland radiative forcing is needed.

If wetlands are to be a viable nature-based approach to climate change mitigation, then the management and restoration of these ecosystems, with a view to increasing their ecosystem service provision, must also be cost-effective for decision-makers to support. The costs of wetland restoration are complex and often not communicated, but preliminary costs have been reported in recent meta-analyses for peatlands [15] and coastal wetlands [17]. This is alongside a burgeoning literature on the progress, challenges and lessons learned in wetland restoration more broadly [15,18,19]. However, links between the costs of wetland restoration relative to their effectiveness at mitigating climate change have seldom been made, and the limited scope of previous meta-analyses omits other important wetlands such as freshwater marshes.

We constrained the net ecosystem carbon budget and radiative forcing of different inland and coastal wetland ecosystems and evaluated their cost-effectiveness of restoration for climate change mitigation. Our objectives were to (i) determine the wetland NECB and compare the different recent metrics to evaluate wetlands' radiative effect and their associated role in the modern carbon budget; (ii) assess and discuss the cost of wetlands conservation and restoration; and (iii) address how carbon storage from wetlands could or should be integrated into climate change mitigation policies.

2. Methods

We performed two systematic reviews, which were analysed through a meta-analysis; the first to determine wetland radiative effect and the second to determine wetland restoration costs. Both systematic reviews exclusively collected data published up until 31 March 2020, using the Scopus database by Elsevier. The detailed method is presented in the electronic supplementary material and a summary is presented below.

2.1. Wetland carbon budget and radiative effects

2.1.1. Literature search

We conducted a systematic review of studies that quantified wetlands net carbon budget, including CH_4 fluxes over an annual time-scale (or growing season for high latitude studies). We first developed and tested a search string based on trial and error using a set of relevant studies from three previously published literature reviews [16,20,21].

2.1.2. Critical appraisal and data extraction

The selected papers went through several rounds of quality control, conducted by two of the authors independently. First, titles (round 1) and abstracts (round 2) were screened. A keyword search approach was used. If none of the terms 'carbon', 'CO_2', 'methane', 'budget' or an ecosystem name appeared, the publication was excluded. Selected papers were then downloaded and critically appraised as described in the electronic supplementary material. Only studies from undisturbed and restored or rewetted peatlands were considered, in line with our stated research objectives.

For inclusion in this review, studies had to provide the wetland type description; geographical coordinates; study length duration; measurement technique and equipment used; time period since restoration or rewetted except for undisturbed sites; net ecosystem exchange (NEE) or net ecosystem productivity (NEP), terrestrial CH_4 exchange, total organic carbon from rain (rain TOC), aquatic carbon export including particulate organic carbon, dissolved organic carbon (DOC), dissolved inorganic carbon, dissolved CO_2 and CH_4, aquatic CO_2 evasion, terrestrial CH_4 exchange and aquatic CH_4 evasion, when available. All these components are required to produce the NECB

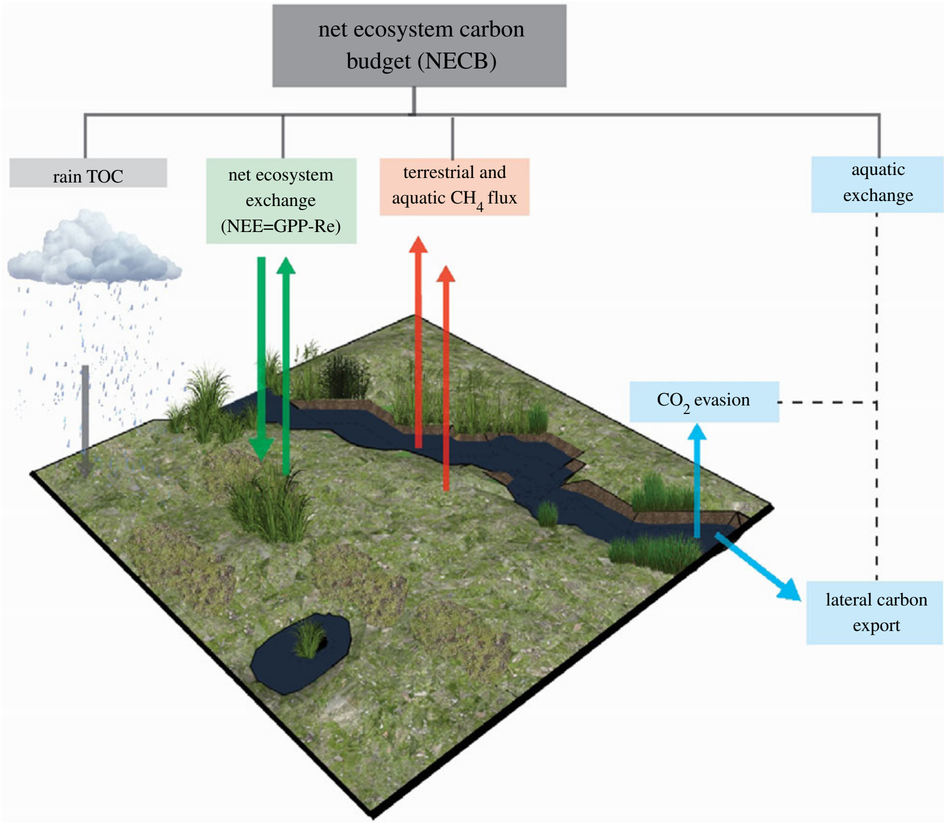

Figure 1. Conceptual model of the NECB that summarizes ecosystem carbon inputs and outputs. Note that if a stream is crossing a wetland (rather than taking its source within it), the import of allochthonous carbon has to be deducted.

as presented in Chaplin *et al.* [22] and summarized in figure 1. Because of the extensive work it represents to measure all the components required in this approach, almost no study has done so. Therefore, we adapted our selection as a compromise as such that NEE (or NEP) and terrestrial CH_4 exchange were the two mandatory variables required to include the study in our meta-analysis. Although not ideal, these two variables were enough to complete our research objective to estimate both NECB and ecosystem net radiative forcing (using SGWP and switchover time).

To clarify common confusion between terms, GPP represents the gross assimilation of CO_2 via photosynthesis and NEE represents the net terrestrial carbon flux (Re-GPP). When negative, NEP would indicate a net uptake or storage of CO_2 from the ecosystem. The only difference between NEE (=Re-GPP) and NEP (=GPP-Re) is the sign [22]. All NEP values were converted to NEE to ensure consistency between studies. Similarly, negative NECB indicates a net carbon sink and positive NECB a net carbon source towards the atmosphere (figure 1).

2.1.3. Data availability

A total of 9390 publications were initially captured from the literature search. This number was reduced to 336 after titles and abstracts were screened. Following a critical appraisal of the full texts, 65 articles remained, together accounting for 64 individual wetland carbon budgets. The second number is lower as some publications presented data from multiple study sites while others published CO_2 and CH_4 budgets independently while conducted at the same site during the same period. The significant reduction in the number of suitable studies pre- and post-screening is similar to other ecosystem carbon studies (e.g. [23]). The meta-analysis reported that 19% of studies integrated aquatic fluxes in their NECB ($n = 12$). Among them, they all quantified lateral export (mainly DOC), and only three studies included CO_2 and CH_4 evasion (see electronic supplementary material for further details). For consistency, we only present results for wetland types that have a restoration cost estimate, except for tundra.

More than half of the selected studies were conducted on boreal and temperate peatlands, either bogs ($n = 15$) or fens ($n = 15$). Freshwater marsh was the second most represented wetland with eight studies, followed by restored peatland ($n = 5$) and rewetted peatland ($n = 5$). Surprisingly, the review did not capture any empirical studies on mangrove wetlands and only two on saltmarshes. While both ecosystems have been the focus of extensive carbon assessments and budget reconstructions over multiple decades (e.g. [24–26]), these studies lack measurements of methane within the budget. This may be due to limited interest to measure these emissions in what are methane-poor environments, relative to terrestrial wetlands. However, since methane emissions from mangroves may partially offset carbon sequestration potential [27,28], we included global estimates compiled by Rosentreter *et al.* [28] for mangroves to compare them with the other terrestrial wetlands presented in this study.

2.2. Wetland restoration costs

2.2.1. Literature search

We conducted a systematic review of studies that reported wetland restoration costs. Similar to the first systematic review presented above (§2.1.1), we used two previously published review to construct our search string [15,17].

2.2.2. Critical appraisal and data extraction

The titles and abstracts of collected articles were screened to remove studies that did not focus on wetland ecosystems (undefined ‘wetlands’ were also excluded). We then searched within individual studies and only included those that reported (i) restoration costs, (ii) the size of the restored area and (iii) the duration of the restoration event,[1] for a specific restoration project or projects. Studies that reported actual costs from completed restoration projects, and those that estimated costs for planned restoration projects,

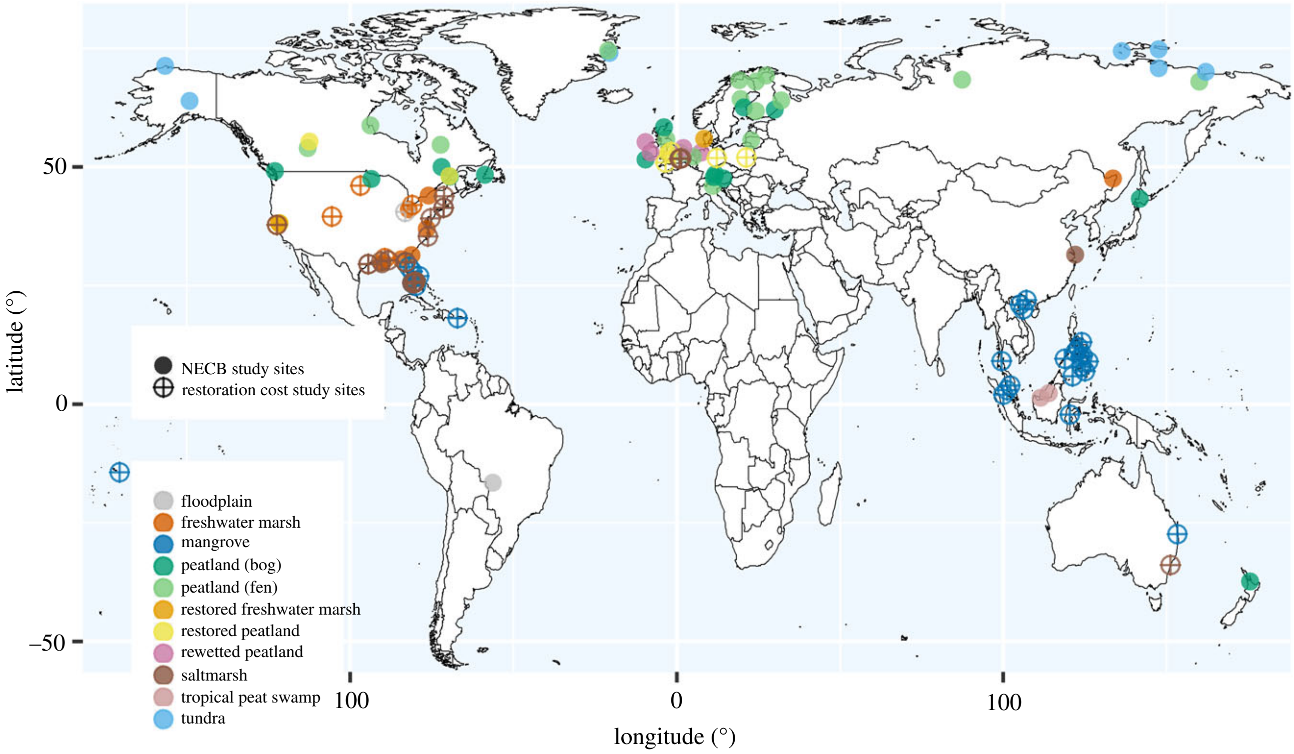

Figure 2. Global distribution of the studies that assessed the net carbon budget by accounting for CO_2, CH_4 (and aquatic lateral export when available) in filled circles and wetland restoration cost in open circles with a cross.

were both included. If a study gave data on multiple restoration projects, data were extracted for each individual project. Care was taken not to double count individual sites that may have featured in multiple different articles. Calculations followed the methodology of Bayraktarov *et al.* [17] and are summarized in the electronic supplementary material. In a final stage, these cost data were combined with the data on wetland carbon budgets and radiative effects, to calculate the cost-effectiveness of wetland restoration for climate change mitigation.

2.2.3. Data availability

A total of 25 658 articles were initially captured from the literature search. These numbers included studies that might have been selected multiple times because of the several search strings used. After all screening and appraisals of the full texts, the final dataset contained restoration costs from 24 articles, covering 63 projects. The breakdown in [articles; projects] format was: mangroves [18; 61], saltmarshes [14; 51], freshwater marsh [4; 15], peatland [4; 7] and floodplain [1; 1]. The majority of datapoints came from tropical coastal ecosystems, many of which had been captured in the dataset compiled by Bayraktarov *et al.* [17]. However, many of the wetland types represented in the carbon budget dataset were also represented in the restoration costs dataset, making comparisons possible. Based on our search, there appears to be no restoration cost data on tundra in the peer-reviewed literature (figure 2).

3. Results and discussion

3.1. The role wetlands play in regulating atmospheric greenhouse gas concentrations and associated radiative effect

3.1.1. Net ecosystem carbon budget

The NECB is a standardized approach to account for carbon gain and loss at the ecosystem scale. Undisturbed inland wetlands included in this study were all small net carbon sinks (figure 3*a*). The median ± standard error NECB for inland wetlands was $-57.2 \pm 27.3\ \mathrm{gC\ m^{-2}\ y^{-1}}$. The lowest reported value of $-978\ \mathrm{gC\ m^{-2}\ y^{-1}}$(sink) was from a freshwater marsh (figure 3*a*). Boreal and temperate peatlands had a median NECB of $-28.1 \pm 19.1\ \mathrm{gC\ m^{-2}\ y^{-1}}$. Values presented here are within the same order of magnitude as previous studies that quantified an average uptake of $-45\ \mathrm{gC\ m^{-2}\ y^{-1}}$ and $-69\ \mathrm{gC\ m^{-2}\ y^{-1}}$ for inland wetlands and peatlands, respectively [29]. Among the 11 wetland types presented in this study, only tropical peat swamps were a net carbon source to the atmosphere (figure 3*a*) but with a standard error seven times greater than its median value ($21.1 \pm 153.0\ \mathrm{gC\ m^{-2}\ y^{-1}}$; figure 3*a*). We also highlight the small number of studies captured for tropical peat swamps ($n = 2$) and for tropical wetlands in general, as opposed to high latitude ecosystems. Mangroves had a NECB of $-235\ \mathrm{gC\ m^{-2}\ y^{-1}}$, which is smaller than the $-1000\ \mathrm{gC\ m^{-2}\ y^{-1}}$ estimated by Webb *et al.* [16]. However, this previous estimate was made using one study site and did not consider CH_4 emissions in their budget, unlike the one presented in this meta-analysis from a global synthesis (i.e. [28]).

Aquatic export was variable between and within wetlands. When reported, it offset 18.8% of the net ecosystem exchange, with values ranging from 0.5% to 158% (electronic supplementary material, figure 1*b*). However, only 12 out of 64 studies included the aquatic component in their budget. The highest aquatic flux over NEE ratio was for studies which reported both DOC export plus CO_2 and CH_4 evasion (instead of just DOC export), regardless of the wetland type. This suggests that NECB studies that did not measure the aquatic component (i.e. lateral export and outgassing) underestimate carbon loss from wetlands [16]. Nonetheless, some studies have questioned the accuracy of considering aquatic export as a net carbon loss as some of this carbon might

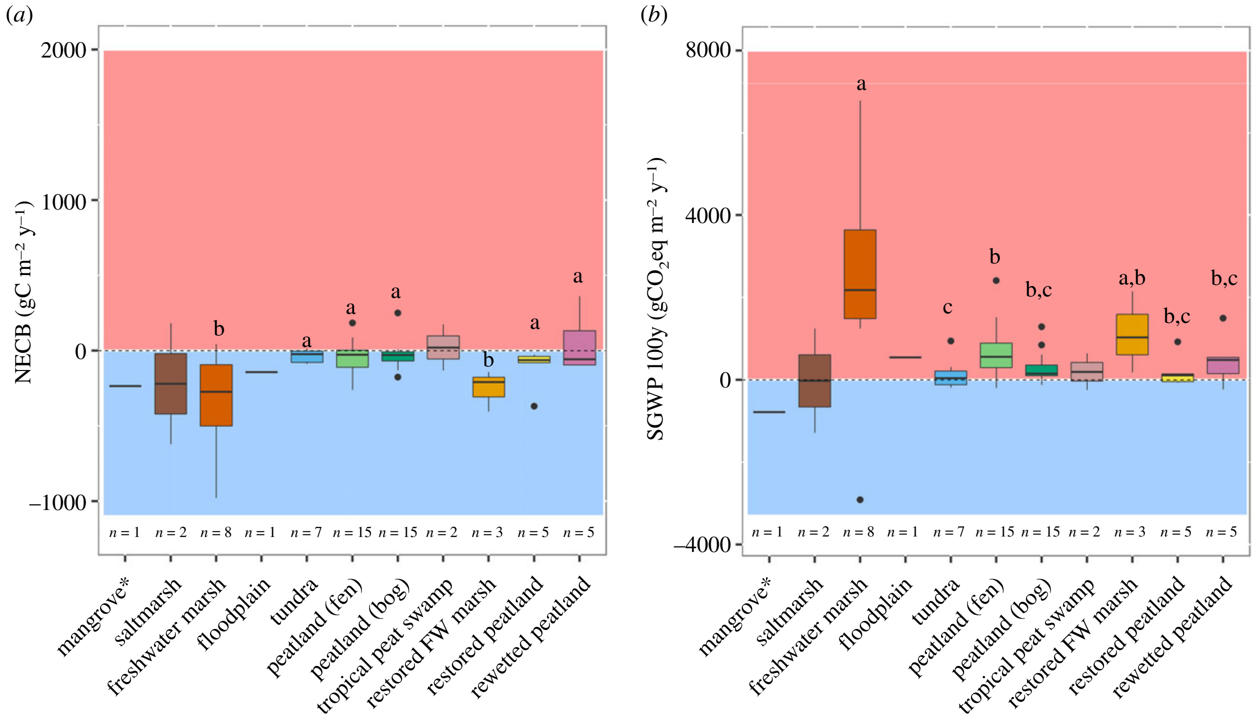

Figure 3. Boxplots for mangrove and inland wetlands presenting (*a*) annual NECB and (*b*) SGWP based on the studies selected for our meta-analysis. Boxes span the interquartile range (25–75% quartiles), whiskers 5–95% of observations, horizontal lines are the medians and circle points represent the outliers. Letters indicate significant differences between wetlands with an $n > 2$ (non-parametric Van der Warden test, $p < 0.05$). *Value for mangrove is not from a particular study site but from a global synthesis from Rosentreter *et al.* [28].

remain trapped in the hydrosphere reservoir rather than be re-emitted to the atmosphere. This is particularly relevant for coastal wetlands, where lateral carbon export could account for a hidden carbon sink. Some studies suggested that this export could represent greater than 50% of ecosystem productivity, and therefore represent the strongest ecosystem carbon output [30,31].

While the NECB metric determines if the ecosystem accumulated or released carbon, it does not directly address the radiative forcing of the studied system and its role in atmospheric GHG mitigation. This is particularly problematic for wetlands as they produce about 30–40% of the global CH_4 emissions (0.2 Gt CH_4 y^{-1}) [32]. Finding a way to determine radiative forcing of CH_4 over CO_2 in wetlands has led to extensive debates in the research community [33–35]. One way to estimate the net radiative effect of an ecosystem is to arbitrarily set a time horizon such as the GWP or SGWP (figure 3*b*).

3.1.2. Sustained global warming potential at the 100 years horizon

In figure 3*b*, we present the SGWP for wetland types using the potential factor of 45 for CH_4 (100-year time horizon) [11]. While most wetland types were net negative using the gC m^{-2} y^{-1} unit (figure 3*a*), all inland wetlands had a net positive sustained radiative forcing using SGWP at 100 years (figure 3*b*). Freshwater marsh was the strongest carbon sink in carbon unit (figure 3*a*) but had a net positive radiative forcing (i.e. warming) using the SGWP metric (figure 3*b*), despite being among the most productive wetland types in terms of NEE (electronic supplementary material, figure 1). Mangroves and saltmarshes were the only two wetland types that were still net negative, because of the limited CH_4 emissions generated in coastal wetlands in comparison with the magnitude of CO_2 uptake.

The SGWP approach showed that inland wetlands have a warming effect rather than a cooling effect on the climate, and therefore coastal wetlands are most appropriate as a climate change mitigation strategy. This is critical because the UNFCCC specifically uses this type of metric. However, we argue that SGWP, or similar ones such as GWP, do not acknowledge the full role that wetlands play in climate regulation, because they do not integrate the age of the ecosystem, which is usually much older than the steady state of CH_4 in the atmosphere (approx. 50 years) [11].

3.1.3. Switchover time of wetlands

Although convenient to align with policy mitigation strategies, using GHG metrics such as SGWP, GWP or GWP* at a fixed time period (i.e. 100 years for UNFCCC) is not fully representative of an ecosystem's warming/cooling effect because it does not consider the period of time since CH_4 emissions and CO_2 sequestration have occurred. This is indeed important as GHGs have different atmospheric lifetimes [11,36]. Assuming that GHG emissions and CO_2 uptake from undisturbed mature wetlands are overall stable over time, methane reaches a steady state after four atmospheric lifetimes (approx. 50 years), meaning that its radiative forcing is stabilized (figure 4*a*). This is because CH_4 emissions are balanced by CH_4 oxidization in the atmosphere [37]. On the other hand, CO_2 never reaches a steady state, because of its infinite time before it equilibrates with external reservoirs including geological scale weathering of continental rocks [13]. Therefore, CH_4 decays much faster than CO_2 equilibrates (figure 4*a*). Consequently, depending on the age since this process has been ongoing, the studied system can have either a warming or cooling effect (figure 4*b*).

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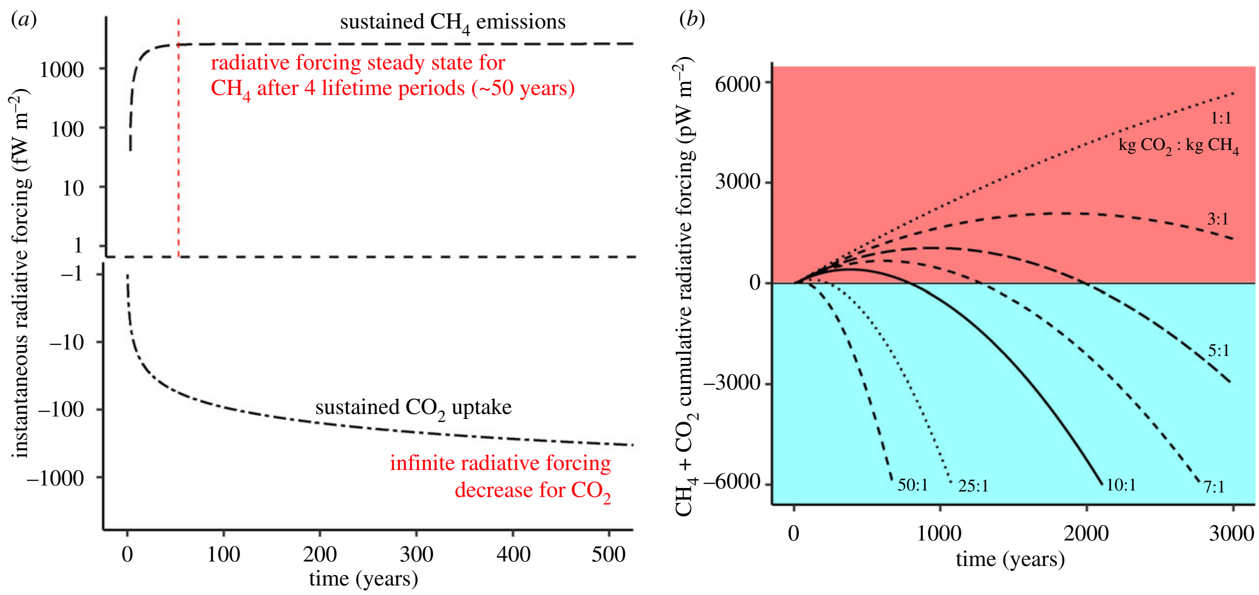

Figure 4. (*a*) Instantaneous radiative effect of CH_4 following a 1 kg CH_4 pulse addition, and CO_2 following 1 kg CO_2 pulse uptake at time 0 and the decay of each gas over a 500-year period. The remainder of the figure shows a radiative effect due to sustained CH_4 emissions or sustained CO_2 uptake. The CH_4 curve includes any radiative effect by CO_2 that was produced from the oxidation of atmospheric CH_4. Note the logarithmic scale on the *y*-axes. fW = 10^{-15}W. Adapted from Neubauer & Megonigal [11]. (*b*) Cumulative radiative perturbation of the integrated lifetime result of warming caused by CH_4 emissions (in kg CH_4) and cooling due to long-term CO_2 sequestration (in kg CO_2). For all the seven scenarios, CH_4 emissions were set at 1 kg CH_4 m^{-2} y^{-1} while CO_2 uptake was modelled from 1 kg CO_2 m^{-2} y^{-1} to 50 kg CO_2 m^{-2} y^{-1}. pW refers to picowatts (10^{-12} W). The red background is where the simulation produces a warming effect, the blue background is where the simulation produces a cooling effect. Adapted from Neubauer & Verhoeven [37].

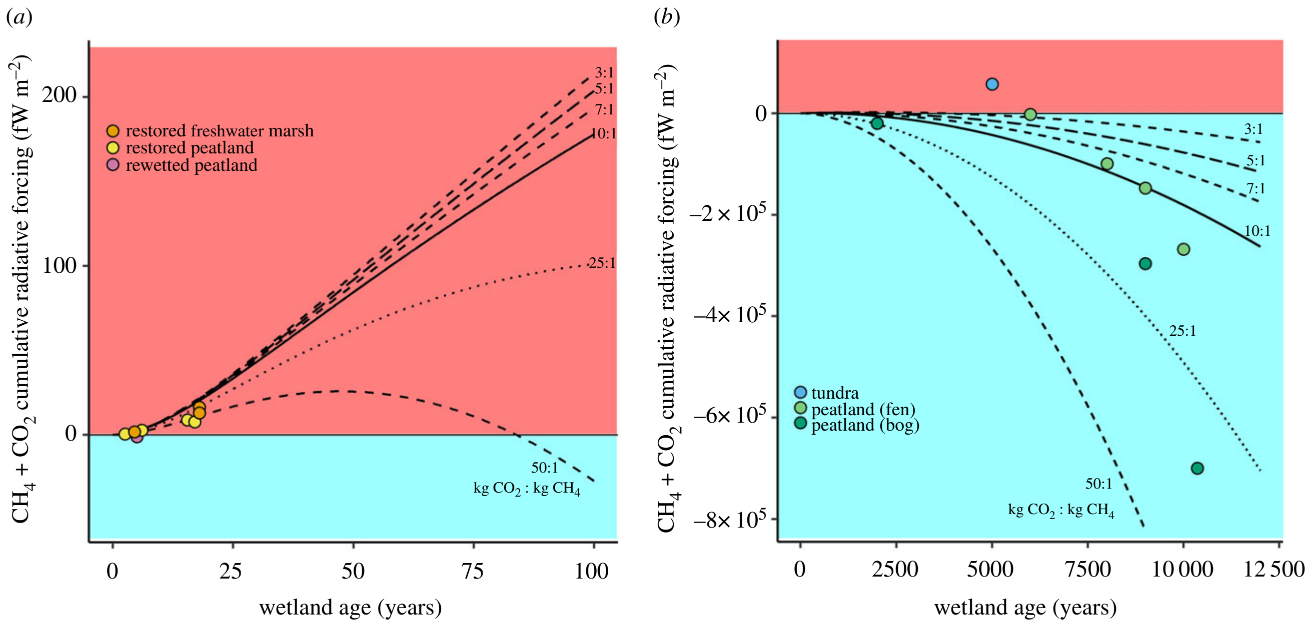

Figure 5. (*a*) Radiative effect of wetlands from our meta-analysis using their radiative balance (kgCO_2:kgCH_4) and age, adapted from figure 4*b*. (*a*) Restored wetlands over a time-scale from 0 to 100 years; and (*b*) the undisturbed wetlands over a time-scale from 0 to 12 500 years.

Using the switchover time approach, we can estimate the contribution of each wetland to the modern climate using (i) the magnitude of CH_4 emissions; (ii) the CO_2 sequestration: CH_4 emission ratio; and (iii) the wetland age. Among the studies that are part of our meta-analysis, only 15 provided the wetland age, along with CO_2 and CH_4 fluxes. All the undisturbed peatlands had a net cooling effect, except for the tundra site (figure 5*b*). Therefore, we conclude that despite steady CH_4 emissions and having a net positive radiative forcing at the 100-year time-scale (figure 3*b*), when integrated with age, peatlands are net carbon sinks and have a net cooling effect on the modern climate (figure 5*a*). Conversely, all restored wetlands had a net warming effect (figure 5*b*), because of their relatively young age and the impossibility for CH_4 emissions to equilibrate in the atmosphere. This suggests that restored wetlands do not directly mitigate radiative warming at the yearly to decadal time-scale, even though they can accumulate carbon in their soil.

Although wetland age measurements were lacking for most of the studies, we were able to calculate switchover times for the other wetland types. This allowed us to evaluate

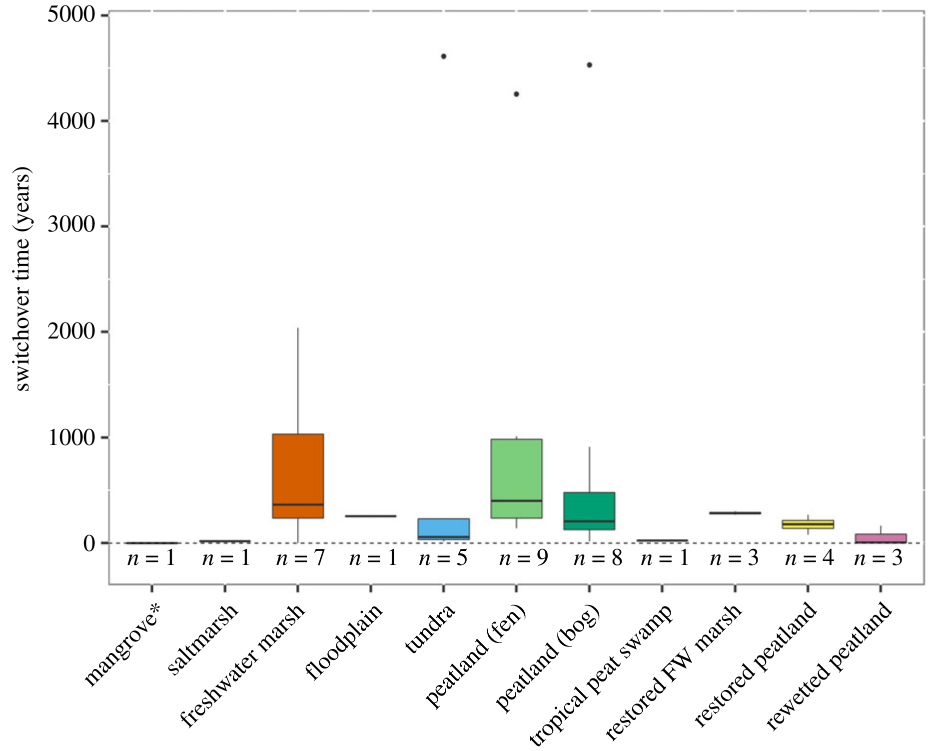

Figure 6. Boxplot of the switchover time when a wetland changes from having a net warming to a net cooling effect. Note that some studies that had a positive NEE (CO_2 emission > CO_2 uptake) were not considered in this model as they cannot have a negative radiative balance. Also note that two outliers from the peatland boxplot are not presented in this figure as their values were 10 608 and 16 354 years.

the range of time necessary for wetlands to have a net negative radiative forcing and also to evaluate how much time is required for a restored or rewetted wetland to help mitigate climate change (figure 6). The shortest time periods calculated were for mangroves (0 year, meaning that mangroves never have a net warming effect) and saltmarshes (17 years). Peatlands (boreal and temperate) and freshwater marshes and had important switchover time variability between study sites with a median value of 298.2 ± 100.6 and 2184 ± 1029 years, respectively (figure 6). Restored freshwater marsh, restored peatlands and rewetted peatlands had an overall short switchover time when compared with natural terrestrial wetlands, ranging from 57 to 299 years (figure 6). While restored and rewetted wetlands induce a re-establishment of CH_4 emissions, it also stops high CO_2 emissions that occurred during the drained and disturbed period [38,39]. Again, the dilemma is to determine if the high radiative effect of CH_4 emissions outcompetes the carbon sequestration potential of restored wetland. The switchover time approach can be seen as the reverse to SGWP and other similar metrics. Instead of knowing what is the radiative effect of a wetland at time *t*, the switchover time tells when exactly the ecosystem will have a net cooling effect. This approach is, therefore, more relevant for determining the role ecosystems play in the modern carbon cycle. However, switchover time estimates must be considered with nuance. The limitation of this approach is that it assumes CO_2 and CH_4 fluxes to be steady over time while they are expected to vary depending on the age of the ecosystem, climatic interannual variability [40] and the effect of climate change [41]. Moreover, land-use change and anthropogenic pressure on the environment might also alter the natural behaviour of wetlands [8,40]. This large variability and potential for wetlands to switch from carbon and GHG sinks to sources mean that close monitoring is needed, with the aquatic component to be integrated into the wetland NECB.

Regarding restored wetlands, fixing the year 0 from when the radiative effect is being assessed is challenging. Two approaches could be used. The first one would consider the wetland age since its natural formation. The second approach would ignore previously undisturbed conditions and consider the restoration starting date as the year 0. Although it risks ignoring a significant contribution of past peatlands to the radiative balance (as demonstrated in figure 5*b*), we recommend using the restoration year as the reference year (i.e. second approach) for two reasons. First, it appears inconsistent to use values from post-restoration state for disturbed or previously undisturbed states. Second, it is the most conservative approach as it does not assume the potential cooling effect a wetland might have had before its disturbance.

3.2. The cost-effectiveness of wetland restoration for climate change mitigation

All total project costs are reported as medians in 2010 US$. Saltmarsh restoration was associated with the highest average total project costs (US$89 660 ha^{-1} y^{-1}; figure 7 and table 1). This was slightly higher than those for freshwater marsh (US$71 221 ha^{-1} y^{-1}) and significantly higher than costs associated with restoring floodplains (US$20 948 ha^{-1} y^{-1}), mangroves (US$4368 ha^{-1} y^{-1}) and peatlands (US$1229 ha^{-1} y^{-1}). Several factors account for the large disparity in restoration costs between ecosystems. Firstly, the restoration of saltmarsh and freshwater marsh may require more hydrological manipulation (e.g. construction of weirs) compared with other ecosystem types, due to the type of historical land-use

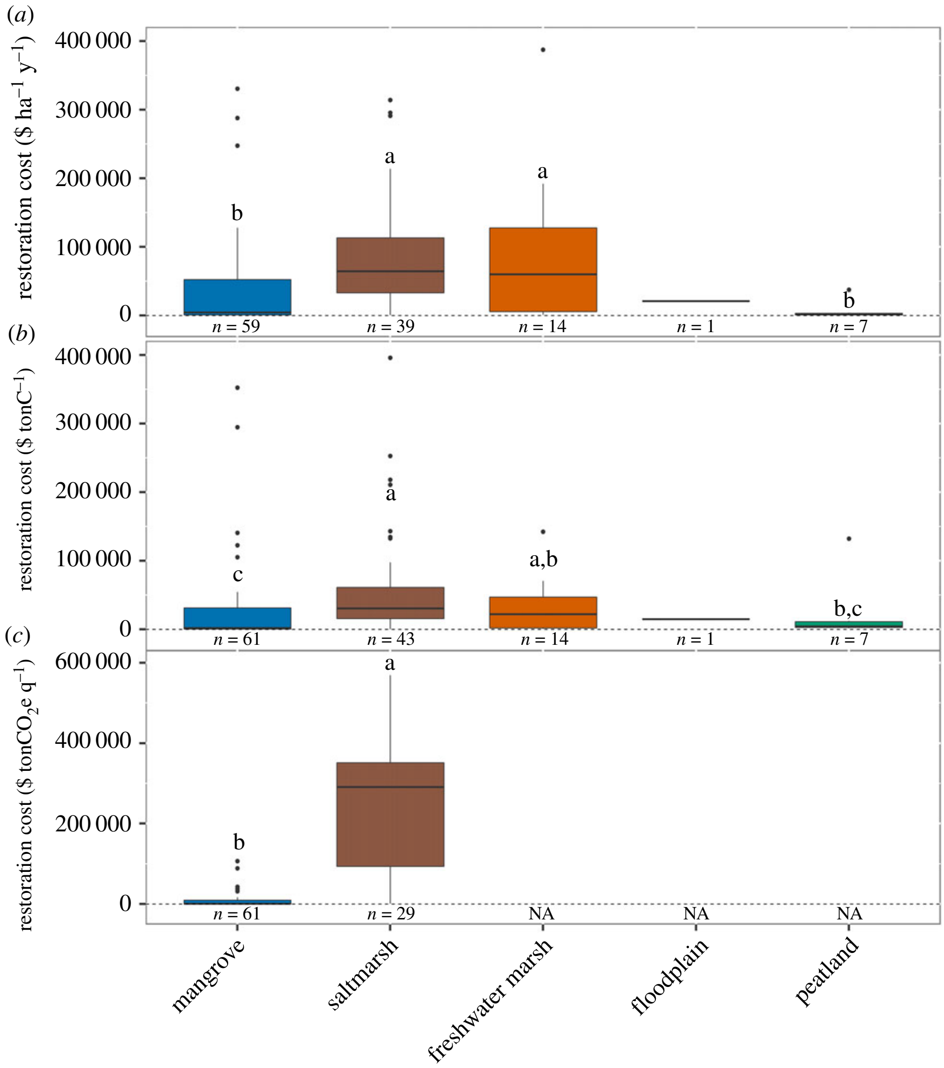

Figure 7. Boxplot of restoration cost per wetland type in (*a*) US\$ ha^{-1} y^{-1}; (*b*) US\$ ton C^{-1}; and (*c*) US\$ ton CO$_2$eq 100 SGWP^{-1}. Note that for (*a*) one outlier from the freshwater marsh (1 733 632 US\$ ha^{-1} y^{-1}) and two from mangrove (828 033 and 692 814 US\$ ha^{-1} y^{-1}) are not presented in this figure. Also note that the lower samples size for (*c*) is because only sites with a negative SGWP-100y could be considered. Letters indicate significant differences between ecosystems (non-parametric Van der Warden test, $p < 0.05$). Values at the bottom of the lower boxplots indicate the sample size for each wetland type.

change affecting these ecosystems (e.g. enclosure and infilling of saltmarsh for during port development, or drainage of freshwater marshes for agriculture), and the specific hydrological requirements of freshwater marsh species [42]. Secondly, in this systematic review, all of the saltmarsh and freshwater marsh restoration projects for which costs could be extracted were based in developed nations, specifically the USA, UK and Germany. Labour, material and engineering costs will be substantially higher in these locations, compared with conducting restoration in developing nations; estimated to be as high as 2010 US\$1.7 M ha^{-1} y^{-1} for one site in Ohio [43]. Conversely, in this review, some mangrove restoration cost data came from projects in the USA, but over half came from those in developing nations (particularly Southeast Asia). Indeed, there is a two order of magnitude difference between the median costs of mangrove restoration in developed (US\$100 861 ha^{-1} y^{-1}) and developing (US\$989 ha^{-1} y^{-1}) countries. This may be explained by the fact that mangrove restoration projects in Southeast Asian countries such as Indonesia often do not involve mechanical earthworks; instead, earthworks are carried out by the local community, using hand tools. This reduces engineering and labour costs, and their community-managed nature may also lower longer-term costs such as maintenance and monitoring [44].

We combined estimates of restoration costs with the potential carbon sequestration and radiative forcing of wetland ecosystems (§3.1) to determine the cost-effectiveness of wetland management as a natural climate solution (table 1). While there are uncertainties in the carbon sequestration values and ecosystem restoration costs reviewed, mangroves appeared to be the most cost-effective wetland type to be restored for climate change mitigation purposes, assessed across all metrics (table 1). Mangroves have an estimated climate change mitigation potential per dollar of restoration of US\$1800 tonC^{-1} y^{-1}. This is due to the relatively low cost of restoration, and the fact that natural mangrove stands have a net cooling effect, as do restored mangroves immediately after a restoration event takes place (figure 6). This suggests that mangrove restoration has high potential as a nature-based solution for climate change mitigation, especially since as much as 810 000 ha of formerly deforested mangrove areas are deemed biophysically suitable across the tropics and sub-tropics [45].

Three inland wetland ecosystems have comparable values of climate change mitigation potential per dollar of

Table 1. Median restoration cost [min; max] presented in different units per wetland types, based on the restoration cost in \$ ha^{-1} y^{-1} multiplied by the NECB, sustained global warming potential at the 100-year time-scale (SGWP-100y), and the switchover time determined by our meta-analysis.

all in 2010 USD	*n*	restoration cost (US\$ ha^{-1} y^{-1})	NECB (gC m^{-2} y^{-1})	restoration cost (US\$ tC^{-1})	SGWP 100y (gCO_2eq^{-1} m^{-2} y^{-1})	restoration cost (US\$ tCO_2eq^{-1})
saltmarsh	43	89 660	−219.5	40 820	−19.1	469 109
		[107; 2 439 305]	[−620; 181]	[17; NA]	[−1284; 1246]	[8; NA]
freshwater marsh	14	71 221	−272.1	26 174	2184.4	NA[a]
		[803; 1 733 632]	[−978; 43]	[82; NA]	[−2912; 6779]	[27; NA]
floodplain	1	20 948	−141.2	14 835	549.7	NA[a]
		[NA; NA]	[NA; NA]	[NA; NA]	[NA; NA]	[NA; NA]
peatland (fen and bog)	7	1229	−28.1	4374	298.2	NA[a]
		[464; 37 173]	[−262; 251]	[177; NA]	[−195; 2413]	[237; NA]
mangrove	61	4368	−235	1782	−776.4	560
		[69; 828 033]	[NA; NA]	[NA; NA]	[NA; NA]	[NA; NA]

[a]No restoration costs are presented for the three inland wetlands based on the SGWP 100y because they all produced a net warming effect using this metric.

restoration between US\$4200 and US\$49 200 $tonC^{-1}$ y^{-1} (table 1). Inland wetland restoration needs to be considered as a long-term investment, as no climate change mitigation benefit can be accrued at the 100 year time-scale, primarily because of the high CH_4 emissions they produced. The issue of time-scale provides a challenge to the restoration of inland wetlands for climate change mitigation, because mechanisms that incentivize restoration and conservation often have much shorter reporting and funding timelines of 10–20 years [46]. Thus, there is a distinct mismatch between the timelines of the carbon budget versus policy and funding. Therefore, the conservation of existing inland wetland ecosystems is more effective than their restoration for less than 100 years climate change mitigation schemes. Yet, both conservation and restoration are effective approaches for the promotion of mangroves (and possibly other coastal wetlands such as temperate and tropical salt marshes) as a nature-based climate solution. Nevertheless, across all wetland types, restoration projects still require subsequent monitoring and operation costs (to conserve the restored site), meaning conservation would likely be more cost-effective overall because it does not require the initial restoration investment. For example, evidence suggests that annual monitoring costs are typically around 20% of the initial restoration investment for floodplain restoration in the USA [47], and mangrove restoration in the Philippines [48] and Vietnam [49].

Mangrove restoration (US\$510 tCO_2^{-1}; table 1) is of the same order of magnitude, yet slightly pricier, than carbon capture cost from other negative emission techniques (US\$15–400 tCO_2^{-1}) presented from Fuss *et al.* [4]. However, the service provided by the other negative emissions technologies is limited to carbon removal and could even create ethical or ecological issues [50,51], whereas mangrove restoration also provides additional ecosystem services that can help meet other United Nations Sustainable Development Goals [52]. Furthermore, although determining wetland conservation or management costs was not possible in this study (because such information is limited in the peer-reviewed literature), it is conceivable that conservation costs could be lower than restoration costs for the same wetland in similar locations [48,49]. Hence, mangrove conservation will likely have a lower carbon capture cost, and, since mangrove deforestation emits at least 26.4–37.3 million tons of CO_2-e per year into the atmosphere globally [53,54], could make its cost-effectiveness even more comparable with other negative emission techniques.

The incurring of costs related to wetland restoration will be context specific. Wetland restoration can be financed through various channels depending on land tenure and actor responsibilities [55]. If actors are deemed responsible for converting or degrading wetlands, they may be legally mandated or voluntarily motivated to finance wetland restoration [18]; for example, major companies are restoring wetlands under the banner of corporate social responsibility [56,57]. If responsibility is difficult to determine, fiscal mechanisms such as tax breaks can be offered to farmers or developers that set-aside wetlands, or else governments may finance the restoration themselves using taxpayer money [55].

Typically, the costs of wetland restoration will be incurred by landowners at local–subnational scales while the benefits of the resulting climate change mitigation accrue globally. As such, incentives are often available to wetland-restoring actors as a reward or compensation for providing these positive environmental externalities. Mechanisms such as Payments for Ecosystem Services offer landowners the financially beneficial option of selling credits through the voluntary carbon market—although such initiatives could lead to a considerable repay period, if the landowner financed the restoration themselves. National governments can also benefit by using restored wetlands in national GHG inventories [58] (or as part of their nationally determined contributions stipulated under the Paris Agreement [5]). Beyond this however, wetland restoration performed with the goal of climate change mitigation will also generate additional ecosystem services to humankind such as flood defence, coastal defence, habitat creation and recreation [59]. As such, wetland-restoring actors may also receive non-financial 'co-benefits' enhanced biodiversity leading to the establishment of ecotourism or non-timber forest product enterprises; as evidenced in the case of mangroves [57] and peatlands [60]. Such co-benefits could be sufficient to motivate landowners to participate in wetland recreation projects even if carbon revenues or contributions to national GHG inventories are low.

3.3. Key messages for successfully integrating wetlands in climate change mitigation strategies

Numerous metrics have been used to determine the role that wetlands may play in climate regulation. As of today, no consensus exists on which metrics to use, while the need for land cover management and GHG emissions mitigation remains urgent. Here, we show that depending on metrics, a wetland can be considered to have a positive or negative effect on the modern climate. However, we also demonstrate that using NECB and SGWP over a fixed period of time is misleading and argue that these metrics are too simplistic to accurately determine the effect of wetlands on climate. Besides, we advocate the use of a switchover time. However, to be able to determine if as of today a wetland has a cooling effect, it is required to know its age using radiometric techniques such as ^{137}Cs, ^{14}C, or ^{210}Pb and compare it with the determined switchover time.

Results from the switchover time approach revealed that any wetland can have a net cooling effect if it is maintained with stable emissions over a period of time. The median switchover time from our meta-analysis was 263 ± 591 years for inland wetlands and 8.5 ± 8.5 years for coastal wetlands (figure 7). Yet, we also highlight that there is an urgent need for more wetland carbon budgets to refine our estimate. Results from our cost-effective analysis revealed that mangrove restoration was the cheapest and most effective per surface of restored area, amount of carbon and CO_2 equivalent stored, when compared with inland wetlands (table 1). Three key messages are coming out of these results that can be addressed to policymakers. First, restoration of disturbed inland wetlands creates a net cooling effect at the decadal to century time-scale only. Thus, inland wetland restoration cannot be included in short-term climate change mitigation strategies but is still essential as restoration of a disturbed site drastically reduces CO_2 emissions and with a long-term net cooling effect [23,39]. Second, the conservation of century- to millenary-old inland wetlands should be of high priority, because they already have a net cooling effect as of today (assuming that their emission and sequestration rates are stable through time) and constitutes irrecoverable carbon stocks at the time-scale set by policymakers [8]. Third, coastal wetland conservation, restoration or creation is likely to be a particularly cost-effective global warming mitigation strategy. Considering the limited extent that these ecosystems can cover, this will always remain a small contribution at the global scale but will likely help some countries meet their climate change mitigation goals at the national scale [5].

Integrating wetlands in climate change mitigation strategy is challenging as the magnitude and direction of their radiative effect is not steady over time and directly reacts to land-use or climate change. Despite this variability, wetland conservation and restoration are effective natural climate solutions since their destruction inevitably leads to GHG emissions. To ensure successful stewardship of wetlands as a negative emissions strategy, close monitoring of wetland NECB should be enforced with the aquatic exchange systematically included. This will help us understand the functioning of wetlands, predict their responses to environmental variability, and better manage and restore those natural carbon sinks.

Data accessibility. Data are available in the electronic supplementary material.

Authors' contributions. P.T. and B.S.T. conceived and designed the study. P.T., B.S.T. and K.T. collected and screened the literature. P.T., B.S.T. and D.A.F. led and structured the writing processes with substantial help from all authors.

Competing interests. We declare we have no competing interests.

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Endnotes

[1]One-off restoration events were recorded as occurring for 1 year, and any unspecified events were clarified with the relevant corresponding authors.

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