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Global patterns in mangrove soil carbon stocks and losses

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Mangrove soils represent a large sink for otherwise rapidly recycled carbon (C). However, widespread deforestation threatens the preservation of this important C stock. It is therefore imperative that global patterns in mangrove soil C stocks and their susceptibility to remineralization are understood. Here, we present patterns in mangrove soil C stocks across hemispheres, latitudes, countries and mangrove community compositions, and estimate potential annual CO_2 emissions for countries where mangroves occur. Global potential CO_2 emissions from soils as a result of mangrove loss were estimated to be ~7.0 Tg CO_2e yr^{-1}. Countries with the highest potential CO_2 emissions from soils are Indonesia (3,410 Gg CO_2e yr^{-1}) and Malaysia (1,288 Gg CO_2e yr^{-1}). The patterns described serve as a baseline by which countries can assess their mangrove soil C stocks and potential emissions from mangrove deforestation.

Mangroves cover just 0.1% of the Earth's continental surface (~81,485 km^2) (ref. 1), but have been identified as some of the most carbon (C)-rich forests on Earth[2–4]. Mangroves differ from terrestrial forests in their ability to store large amounts of C in their soils over millenary timescales. Complex root structures, high sedimentation rates, waterlogged soils free from risk of fires, and anoxic soils in mangroves result in C burial rates that are an order of magnitude greater and soil C turnover rates a thousand times slower than those in terrestrial forests[5,6]. The ability of mangrove ecosystems to store large amounts of soil C (5–10.4 Pg globally)[7,8] for millennia makes these ecosystems important C sinks, and reducing or preventing greenhouse gas (GHG) emissions from the loss of these soil C stocks is a low-cost option for mitigating climate change[9,10]. However, we currently lack robust global estimates for soil C stocks in mangroves, which are required to assess the potential for habitat loss to contribute to annual CO_2 emissions and identify important blue C hotspots requiring conservation.

Mangroves and their associated soil C face a multitude of anthropogenic threats (for example, coastal development, drainage, pollution), leading to large-scale global declines[11,12]. Overall, more than one-third of the world's mangroves have vanished over the past 60 years[1,11]. Despite conservation measures being deployed in many nations (for example, Australia), mangroves continue to be lost at a global rate of about 0.2% per year[1]. Only Bangladesh and Guinea-Bissau, out of the top 15 countries for mangrove area, have experienced no net loss from 2000–2012[1,13]. This global decline in mangroves raises concerns about the fate of the large C deposits stored within their soils. Several studies have suggested that degradation and removal of vegetated coastal habitats have the potential to disturb soil C down to depths of 1 m, leading to its remineralization to CO_2 (refs 14,15). Because mangrove soil C deposits take thousands of years to form, once disrupted they cannot be regained over meaningful human timescales by just restoring the forest. As a result, the remineralization of mangrove soil C may add significantly to the component of anthropogenic GHG emissions designated as 'land-use change' still unaccounted for in global C inventories[16].

Estimates of potential GHG emissions associated with mangrove loss thus far have been derived from global averages in soil C stocks and the global area lost annually[14]. However, these estimates assume losses are randomly distributed relative to soil C stocks, which may not be the case. Moreover, such first-order global estimates provide no guidance as to where mangrove conservation will be most effective in avoiding GHG emissions. Here we provide improved estimates of global soil C stocks underlying mangroves, the susceptibility of these stocks to remineralization, and areas where soil C data are deficient or missing. We resolve the regional

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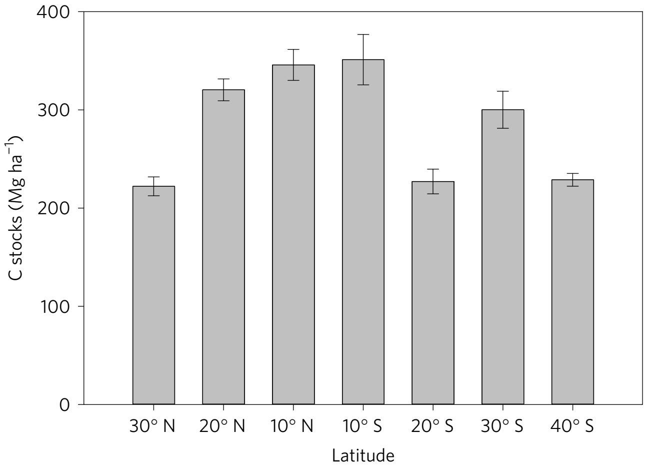

Figure 1 | Latitudinal patterns in mangrove soil carbon. Soil (C) stocks per unit area down to 1 m (mean ± s.e.) across southern and northern latitudes.

variability in mangrove soil C stocks through a combination of estimates from published and unpublished sources as well as regional estimates of annual rates of habitat decline[1]. This country-specific approach allows us to define where mangrove losses may have the highest contribution to annual GHG emissions. Overall our data set includes C stocks for 1,230 distinct sampling locations (Supplementary Data) from 48 countries (out of 105 countries supporting mangrove habitats) encompassing 88% of the global mangrove area (Supplementary Tables 1 and 2)[17], thereby doubling the number of countries in past global estimates[8].

Data quality

Data quality among countries was highly variable (Supplementary Tables 3 and 4). Of the 48 countries with mangrove soil C data, 22 countries scored less than 70% in their total data quality score. Furthermore, 55% relied heavily on the use of a pedotransfer function for estimating bulk density data and 50% of countries were missing estimates of soil C content deeper than 50 cm. In terms of how well the data represented mangrove genera occurring in each country, 15 countries had less than 30% coverage of their mangrove genera. Most of these countries were in areas where mangrove species diversity is high (for example, Asia and Pacific Islands). Coverage of marine eco-regions within countries was overall quite high, and only three countries (Federated States of Micronesia, Papua New Guinea, and Saudi Arabia) had less than 30% of their marine eco-regions with mangroves represented in the data.

Global and national trends in soil C stocks and losses

Mangroves in the Northern and Southern hemispheres were not significantly different in terms of soil C storage per unit area (d.f. 1,227, $t = 1.584$, $P = 0.113$). We did find a significant difference in soil C storage per unit area across latitudinal bands ($F_{6,1222} = 22.5$, $P < 0.001$; Fig. 1), with mangroves between 0° and 10° S having the highest soil C storage per unit area (351 ± 138 Mg C ha^{-1}) and mangroves between 20° and 30° N having the lowest (222 ± 151 Mg C ha^{-1}). However, no pattern between latitude and soil C stocks was detected. The lack of a strong pattern in soil C stocks across latitudes differs from mangrove above-ground biomass, which progressively increases towards the tropics[3,13,18]. This suggests that C hotspots for mangrove soils may not overlap with those for above-ground biomass.

Wetlands provide a wealth of ecosystem services, but often contain low plant diversity with many stands being monotypic[19]. Yet, we found that mixed mangrove stands had 20% higher soil C stocks per unit area than monotypic stands (d.f. 1,122, $t = 5.3149$, $P < 0.001$; Fig. 2a). Soil C stocks differed ~4-fold among genera within monotypic stands ($F_{13,624} = 7.07$, $P < 0.001$; Fig. 2b), with *Laguncularia* and *Rhizophora* forests having the highest stocks (424 ± 262 Mg C ha^{-1}, 388 ± 227, Mg C ha^{-1}, respectively). In mixed mangroves, stands containing 5 genera had 70–90% higher soil C stocks per unit area than all other richness levels ($F_{8,979} = 2.50$, $P = 0.011$; Fig. 2c). This analysis shows only the association between soil C stocks and mangrove genus or genera richness; it does not necessarily imply a causal link between mangrove community composition and enhanced soil C stocks. However, this global trend suggests that research investigating the effects of mangrove community composition and species richness on soil C accumulation and preservation may be warranted, especially when one considers that many mangrove afforestation and restoration programmes plant only one or two species, generally of the genus *Rhizophora* or *Avicennia*[20,21].

We estimate that mangrove soils store ~2.6 Pg C (equivalent to ~9.5 Pg of CO_2) globally. However, several studies have documented that mangrove soils exceed 1 m (refs 2,22); thus, constraining our estimates to a 1 m depth probably underestimates the global soil C stock in mangroves. When combined with above-ground C

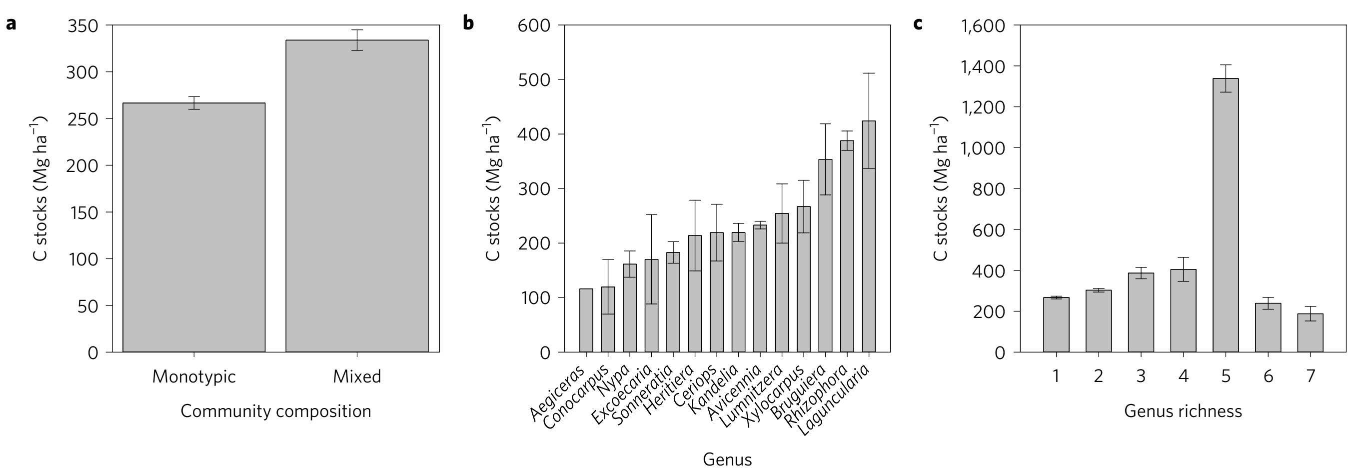

Figure 2 | Mangrove community composition and soil carbon stocks. a, Mean soil C stocks (mean ± s.e.) down to 1 m in global mixed versus monotypic mangrove stands. Monotypic communities are defined as being dominated (>75%) by a single mangrove genus. **b**, Associated mangrove genus and soil C stocks in monotypic stands. **c**, Associated genera richness and mean soil C stocks.

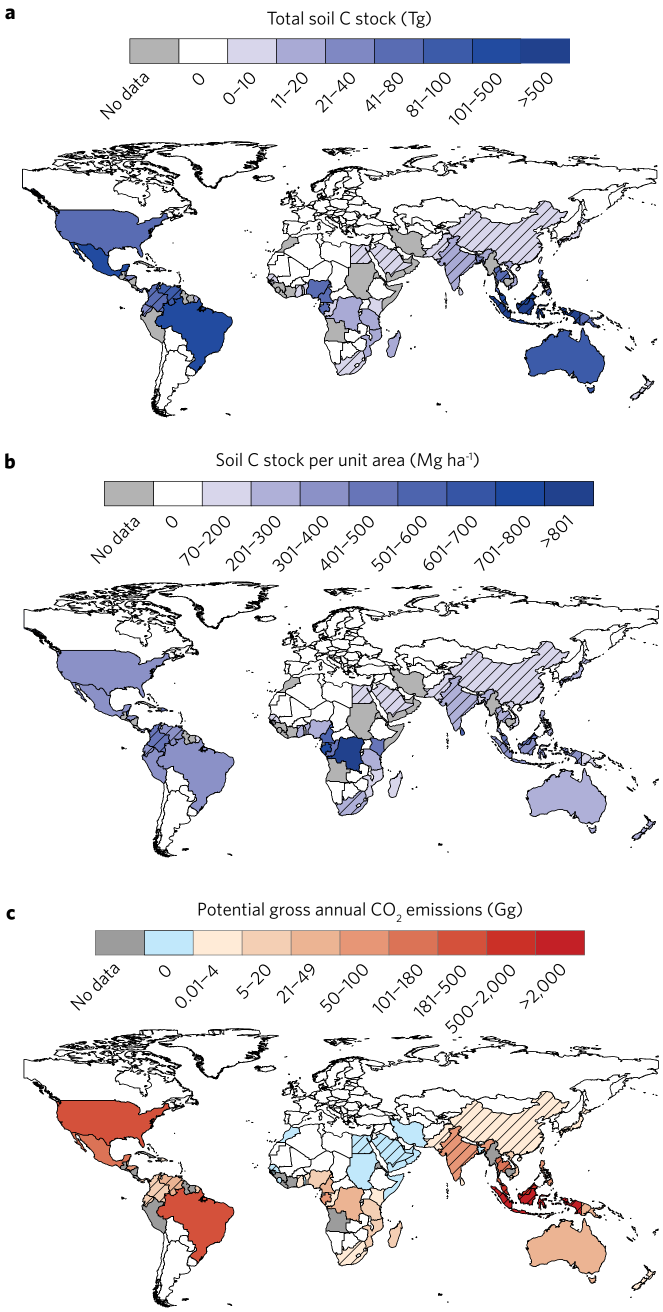

Figure 3 | Soil carbon (C) stocks and potential annual CO_2e emissions from mangrove deforestation. **a**, Total estimated soil C stocks across countries. **b**, Country-specific soil C stocks per unit area down to 1 m. **c**, Potential gross annual CO_2e emissions from soils as a result of annual mangrove habitat declines. Emissions are based on the assumption that 43% of C stocks down to 1 m in the soil are remineralized after deforestation. Grey areas represent countries where mangroves are known to occur, but soil C data were not available. White represents countries where no mangroves occur. Countries with a striped pattern represent those with relatively poor data quality (<70% data quality score).

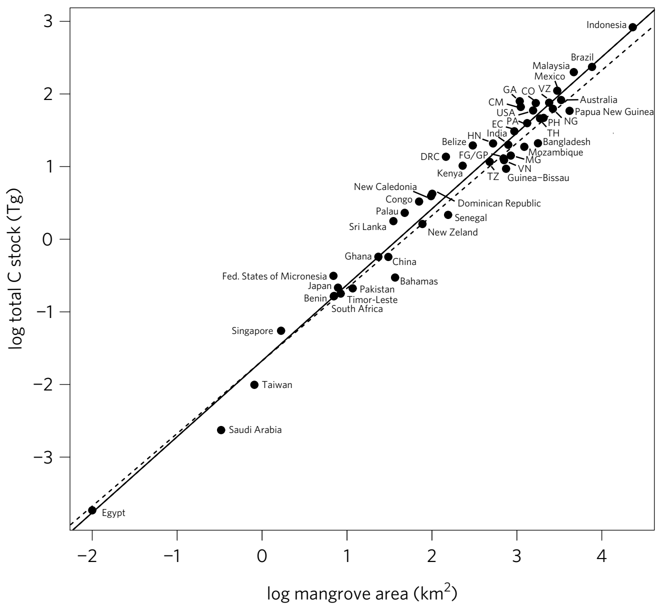

Figure 4 | Relationship between country-specific total C stocks and country-specific mangrove area. The solid line represents the model ($P < 0.001$, $r^2_{adj} = 0.97$) and the dashed line represents the 1:1 slope. The model slope was not significantly different from 1:1 ($f_{1,46} = 4.026$, $P = 0.102$). Only countries with soil C data were included in the model. CO, Colombia; CM, Cameroon; DRC, Democratic Republic of Congo; EC, Ecuador; FG/GP, French Guiana/Guadeloupe; GA, Gabon; HN, Honduras; MG, Madagascar; NG, Nigeria, NZ, New Zealand; PA, Panama; PH, Philippines; TH, Thailand; TZ, Tanzania; USA, United States of America; VE, Venezuela; VN, Vietnam.

biomass[18], mangroves store ~4.4 Pg C. Our global soil C estimate is ~54–78% lower than those previously reported[7,8], despite our average soil C stock per unit area of 283 ± 193 Mg C ha^{-1} (±s.d.) being similar to past estimates[7]. Our lower estimate was largely due to our use of a more recent and conservative estimate of global mangrove area by Hamilton and Casey[1], which was ~39% lower than that reported by Giri and colleagues[13]. The large discrepancies in estimates of mangrove habitat area and its effect on mangrove soil C stocks underscore the need to develop more robust and standardized methods for measuring mangrove habitat area[23].

We found substantial variation across countries in total soil C stocks (Supplementary Table 1 and Fig. 3a), ranging from 831 Tg C in Indonesia to 0.0001 Tg C in Egypt. Nearly all (97%) of the variation in total soil C stocks across countries could be explained by differences in mangrove habitat area ($f_{1,48} = 1{,}755$, $P < 0.001$, $r^2_{adj} = 0.972$; Fig. 4). In fact, none of the top five countries for total soil C stocks was in the top five for C densities; further highlighting the need for improved standardized methods for estimating mangrove area. Nevertheless, we also found large variation in soil C stocks per unit area across countries, which ranged over an order of magnitude from 936 Mg C ha^{-1} in the Democratic Republic of Congo to 72 Mg C ha^{-1} in Saudi Arabia (Fig. 3b). Although the scope of data collection in this study precludes further investigations into the source of this variation, several individual studies suggest that site-specific physicochemical properties (for example, karstic soils, distance from seaward edge, salinity, nitrogen and phosphorus content of the soil) play a major role in soil C storage in mangroves and may be leading to the observed patterns in this study[2,24].

Our study also highlights major gaps and data deficiencies in mangrove soil C data (Fig. 3). We were able to obtain data for 48 countries, encompassing 88% of the global mangrove extent. However, we still lack soil C data for 44 countries where mangroves occur. Although many of these are countries with limited mangrove cover, we currently have no estimates of soil C stocks for mangroves in Myanmar or Cuba, which rank 8th and 14th in mangrove cover and 1st and 18th, respectively for annual declines in mangrove habitat[1]. Additionally, African countries were under-represented, with no data for 17 countries and only sparse data for many others. Furthermore, 22 countries in our study scored less than 70% in their data quality scores, making their contributions to global C

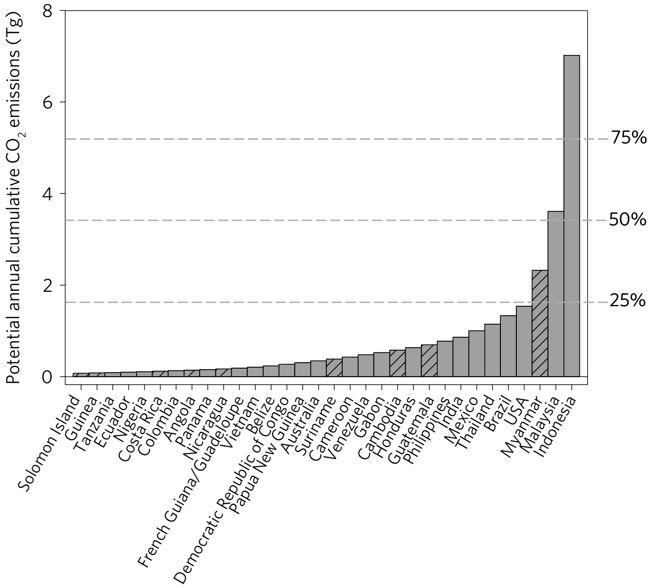

Figure 5 | Cumulative potential annual CO_2e emissions from soils as a result of mangrove deforestation. Percentages represent 25%, 50% and 75% of the total potential emissions of 7.0 Tg CO_2e yr^{-1} resulting from mangrove deforestation. Emissions are based on the assumption that 43% of C stocks down to 1 m in the soil are remineralized. Countries represented in the graph contribute ≥0.1% to total CO_2e emissions. Countries not included in the graph, which contribute <0.01% to total CO_2e emissions include: Mozambique, Trinidad and Tobago, Equatorial Guinea, Madagascar, Cuba, Fiji, Brunei, El Salvador, Congo, New Caledonia, Ghana, Sri Lanka, New Zealand, Dominican Republic, Jamaica, Ivory Coast, Guyana, China, Kenya, Liberia, Palau, Puerto Rico, Haiti, The Bahamas, Timor-Leste, Japan, South Africa, Federated States of Micronesia, Peru, Grenada, Pakistan, Taiwan, Eritrea and Benin. Hashed bars represent countries for which no soil C data existed, as a result potential emissions were calculated using an average global C stock per unit area of 283 Mg C ha^{-1}.

soils stocks and CO_2e (carbon dioxide equivalent) emissions more uncertain. Among these countries were Indonesia, Malaysia and Thailand, which rank in the top five for potential CO_2e emissions from mangrove soils as a result of deforestation. This assessment, therefore, should lead to efforts to address these gaps.

Mangroves support a wealth of ecosystem services, and their decline therefore results in lost socio-economic benefits. Globally, ~131–639 km^2 (Mangrove Forests of the World and Terrestrial Ecosystems of the World-Mangrove Biome estimates, respectively) of mangroves are destroyed annually[1]. When combined with per area soil C stocks, this equates to a potential loss of 2.0–75 Tg C yr^{-1} from soils as a result of mangrove deforestation, which corresponds to ~7.3–275 Tg of CO_2e emissions. The potential gross annual CO_2e emissions from the remineralization of soil C in mangroves, thus, is equivalent to 0.2–6% of those from terrestrial deforestation globally[25]. We estimated the potential gross annual emissions with the assumption that 43% of the soil C stocks down to 1 m are eventually remineralized following mangrove loss (Supplementary Table 5)[14,15,26].

The paucity of research on the fate of soil C after a mangrove ecosystem is disturbed presents some uncertainties in CO_2 emission estimates. Although our approach follows Intergovernmental Panel on Climate Change (IPCC) protocols and common practice in the literature, which standardizes loss to 1 m in the soil, disturbances can influence C stocks at greater depths[27,28]. Since many mangrove ecosystems around the world have soil depths that extend well beyond 1 m (refs 22,29), standardizing losses to this depth may underestimate annual CO_2e emissions. However, it should be noted that a 43% loss of soil C stocks, as was used in this study, would require an average CO_2 emission of ~1.05 Mg CO_2e ha^{-1} d^{-1}. This amount is ~4–15 times higher than reported estimates from direct measures of CO_2 emissions from disturbed mangrove soils[26,28,30]. There are two potential explanations for this discrepancy. First, a large proportion of the soil C could be being transported elsewhere as a result of increased resuspension or erosion. This would explain why soil C stocks are reduced in disturbed areas without a subsequent increase in CO_2 emissions. This process would make C accounting difficult because the fate of the transported C is unknown. Second, many field studies measure CO_2 emissions from disturbed soils years after the event takes place, potentially missing the key period when most of the emissions resulting from a disturbance occur. For example, one study found that CO_2 emissions rapidly increased by ~3 times directly following the disruption of mangrove peat, but then returned to baseline levels after just two days[28]. Nevertheless, refining potential soil CO_2 emission estimates will require further studies on the fate (for example, remineralized, transported, and so on) of soil C stocks after mangrove ecosystems are disturbed or destroyed, a topic urgently in need of research.

The top four countries with the highest potential gross annual CO_2e emissions due to mangrove deforestation are Indonesia (3,511 Gg CO_2e yr^{-1}), Malaysia (1,288 Gg CO_2e yr^{-1}), United States (206 Gg CO_2e yr^{-1}) and Brazil (186 Gg CO_2e yr^{-1}) (Fig. 3c). Together these countries account for ~86% of the total potential emissions due to mangrove deforestation, with Indonesia alone accounting for ~50% (Fig. 5). Brazil and Indonesia have been at the centre of many debates about deforestation, and combined they account for 55% of total emissions from tropical terrestrial deforestation[31]. Soil C stocks per unit area explain only ~9% of the variability in potential gross annual CO_2e emissions ($f_{1,40} = 7.122$, $P = 0.029$, $r^2_{adj} = 0.09$; Fig. 6). These results suggest that relatively low levels of deforestation in countries with high soil C stocks per unit area are currently constraining potential CO_2e emissions. Among these important countries, The Democratic Republic of Congo, Gabon, Cameroon, Belize and Colombia have the highest C stocks per unit area and currently have relatively low or moderate levels of deforestation. However, all five countries have declining ocean health indices[32], and stable (Belize and The Democratic Republic of Congo) or increasing levels of mangrove decline (Colombia, Cameroon and Gabon)[1]. Our results suggest that mangrove conservation efforts should be most effective if targeting the highest potential emitters, along with mangroves with C-rich soils (>500 Mg C ha^{-1}). Furthermore, nations with no available data on soil C stocks may rank among such hotspots representing priority areas for mangrove conservation, providing an additional motivation to address this gap. For example, Myanmar, which has the highest annual rates (~1%) of mangrove deforestation in the world[1], may have the third highest potential annual CO_2 emissions (784 Gg CO_2e yr^{-1}) and would contribute to 18% of the total potential CO_2 emission due to mangrove deforestation.

In general, aquaculture is the major cause of mangrove removal in Asia and South America[21,33,34], and globally it is responsible for 52% of mangrove declines[35]. This is concerning for two primary reasons. First, conversion of mangroves to aquaculture ponds significantly increases CO_2 emissions from the soil[26,28,29], because the excavation and oxidation of soils during pond construction can exceed several metres deep. Furthermore, the construction of aquaculture ponds can lead to further mangrove losses in surrounding areas as the ponds leach high levels of nutrients and alter tidal flow. Second, the demand for seafood will double to 14.8 million tons by 2030, with at least 50% supplied by aquaculture[36]. In response to these projections many countries in Asia and South America are ramping up their aquaculture production, and the Indonesian Ministry of Marine Affairs and Fisheries has set a target to increase aquaculture production over the next 10 years by 61% (refs 33,37). In addition

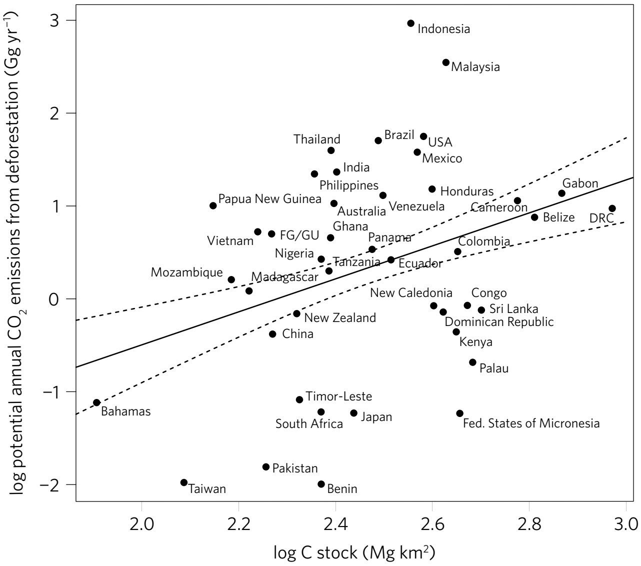

Figure 6 | Relationship between country-specific C stocks per unit area and country-specific potential annual CO_2e emissions. Emissions are based on the assumption that 43% of C stocks down to 1 m in the soil are remineralized. The solid line represents the model and the dashed lines represent the 95% confidence intervals (CIs; $P = 0.029$, $r^2_{adj} = 0.09$). The model slope was not significantly different from 1:1 ($f_{1,40} = 0.971$, $P = 0.334$). Countries above the upper 95% CI represent countries with intense deforestation that results in annual potential CO_2e emissions that exceed what would be predicted from their C stocks per unit area alone. Countries below the lower 95% CI represent countries with relatively more constrained deforestation, resulting in lower annual potential CO_2e emissions than what would be predicted from their C stocks per unit area alone. DRC, Democratic Republic of Congo; USA, United States of America.

to aquaculture, the conversion of mangroves to rice agriculture and oil palm plantations is an emerging problem in many Southeast Asian countries (for example, Myanmar, Malaysia and Indonesia)[38].

Mangroves store an exceptional amount of C in their soils. However, a global estimate alone does not help to target conservation efforts, because soil C stocks are highly variable across hemispheres, latitudes, countries and plant community compositions. Here, we provide estimates for how C stocks and potential CO_2e emissions vary across these variables. In general, our analyses show that protecting large areas of mangrove forests may be more beneficial for preventing GHG emissions from mangrove soils as a result of forest loss than selecting for small areas with high soil C content. Furthermore, global conservation efforts aimed at protecting blue carbon stocks should focus on two strategies. First, reduce intense deforestation in countries with large mangrove area and high soil C stock per unit area. In fact, eliminating mangrove deforestation entirely in just Malaysia and Indonesia would reduce global soil CO_2e emissions from mangrove deforestation by as much as ~70%. Second, maintain low deforestation rates in countries with relatively large areas of mangroves and high soil C stocks per unit area, with a particular focus on The Democratic Republic of Congo, Gabon and Cameroon. As global demand for food and biofuels intensifies, so will social, ecological and economic debates about the future of mangroves. As a result, large-scale estimates on the value of mangroves as C sinks, as well as other ecosystem services, are essential for managers and policymakers to accurately evaluate economic and ecological trade-offs for the management of mangrove forests. The patterns presented in this study provide a baseline assessment of mangrove soil C stocks and potential emissions from mangrove deforestation and degradation, while identifying hotspots for priority conservation and gaps that need to be urgently addressed.

Methods

Methods, including statements of data availability and any associated accession codes and references, are available in the online version of this paper.

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References

1. Hamilton, S. & Casey, D. Creation of high spatiotemporal resolution global database of continuous mangrove forest cover for the 21st century: a big-data fusion approach. *Glob. Ecol. Biogeogr.* **25,** 729–738 (2016).
2. Donato, D. C. *et al*. Mangroves among the most carbon-rich forests in the tropics. *Nat. Geosci.* **4,** 293–297 (2011).
3. Twilley, R. R., Chen, R. & Hargis, T. Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. *Wat. Air Soil Pollut.* **64,** 265–288 (1992).
4. Bouillon, S. *et al*. Mangrove production and carbon sinks: a revision of global budget estimates. *Glob. Biogeochem. Cycles* **22,** GB2013 (2008).
5. Mcleod, E. *et al*. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Front. Ecol. Environ.* **9,** 552–560 (2011).
6. Alongi, D. M. Carbon sequestration in mangrove forests. *Carbon Manage.* **3,** 313–322 (2012).
7. Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I. & Marbà, N. The role of coastal plant communities for climate change mitigation and adaptation. *Nat. Clim. Change* **3,** 961–968 (2013).
8. Jardine, S. L. & Siikamäki, J. V. A global predictive model of carbon in mangrove soils. *Environ. Res. Lett.* **9,** 104013 (2014).
9. Siikamäki, J., Sanchirico, J. N. & Jardine, S. L. Global economic potential for reducing carbon dioxide emissions from mangrove loss. *Proc. Natl Acad. Sci. USA* **109,** 14369–14374 (2012).
10. Murdiyarso, D. *et al*. The potential of Indonesian mangrove forests for global climate change mitigation. *Nat. Clim. Change* **5,** 8–11 (2015).
11. Alongi, D. M. Present state and future of the world's mangrove forests. *Environ. Conserv.* **29,** 331–349 (2002).
12. Lovelock, C. E. *et al*. The vulnerability of Indo-Pacific mangrove forests to sea-level rise. *Nature* **526,** 559–563 (2015).
13. Giri, C. *et al*. Status and distribution of mangrove forests of the world using earth observation satellite data. *Glob. Ecol. Biogeogr.* **20,** 154–159 (2011).
14. Pendleton, L. *et al*. Estimating global 'blue carbon' emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS ONE* **7,** e43542 (2012).
15. Fourqurean, J. W. *et al*. Seagrass ecosystems as a globally significant carbon stock. *Nat. Geosci.* **5,** 505–509 (2012).
16. IPCC *Climate Change 2007: The Physical Sciences Basis* (eds Solomon, S. *et al*.) (Cambridge Univ. Press, 2007).
17. Atwood, T. B. *et al*. *Country-Level Mangrove Soil Carbon Stocks and Losses Dataset*. http://dx.doi.org/10.1594/PANGAEA.874382 (Pangaea, 2017).
18. Hutchison, J., Manica, A., Swetnam, R., Balmford, A. & Spalding, M. Predicting global patterns in mangrove forest biomass. *Conserv. Lett.* **7,** 233–240 (2014).
19. Field, C. B. *et al*. Mangrove biodiversity and ecosystem function. *Glob. Ecol. Biogeogr. Lett.* **7,** 3–14 (1998).
20. Datta, D., Chattopadhyay, R. N. & Guha, P. Community based mangrove management: a review on status and sustainability. *J. Environ. Manage.* **107,** 84–95 (2012).
21. Ferreira, A. C. & Lacerda, L. D. Degradation and conservation of Brazilian mangroves, status and perspectives. *Ocean Coast. Manage.* **125,** 38–46 (2016).
22. Sanders, C. J. *et al*. Are global mangrove carbon stocks driven by rainfall? *J. Geophys. Res. Biogeosci.* **10,** 2600–2609 (2016).
23. Friess, D. A. & Webb, E. L. Variability in mangrove change estimates and implications for the assessment of ecosystem service provision. *Glob. Ecol. Biogeogr.* **23,** 715–725 (2014).
24. Adame, M. F. *et al*. Carbon stocks of tropical coastal wetlands within the karstic landscape of the Mexican Caribbean. *PLoS ONE* **8,** e56569 (2013).
25. van der Werf, G. R. *et al*. CO_2 emissions from forest loss. *Nat. Geosci.* **2,** 737–738 (2009).
26. Lovelock, C. E., Ruess, R. W. & Feller, I. C. CO_2 efflux from cleared mangrove peat. *PLoS ONE* **6,** e21279 (2011).
27. Kauffman, J. B. *et al*. Carbon stocks of mangroves and losses arising from their conversion to cattle pastures in the Pantanos de Centla, Mexico. *Wetl. Ecol. Manage.* **24,** 203–216 (2016).
28. Kauffman, J. B., Heider, C., Norfolk, J. & Payton, F. Carbon stocks of intact mangroves and carbon emissions arising from their conversion in the Dominican Republic. *Ecol. Appl.* **24,** 518–527 (2014).

29. Donato, D. C. *et al*. Mangroves among the most carbon-rich forests in the tropics. *Nat. Geosci.* **4,** 293–297 (2011).
30. Sidik, F. & Lovelock, C. E. CO_2 efflux from shrimp ponds in Indonesia. *PLoS ONE* **8,** 6–9 (2013).
31. Harris, N. L. *et al*. Baseline map of carbon emissions from deforestation in tropical regions. *Science* **336,** 1573–1576 (2012).
32. Halpern, B. S. *et al*. Patterns and emerging trends in global ocean health. *PLoS ONE* **10,** e0117863 (2015).
33. Ilman, M., Dargusch, P., Dart, P. & Onrizal. A historical analysis of the drivers of loss and degradation of Indonesia's mangroves. *Land Use Policy* **54,** 448–459 (2016).
34. Ahmed, N. & Glaser, M. Coastal aquaculture, mangrove deforestation and blue carbon emissions: is REDD+ a solution? *Mar. Policy* **66,** 58–66 (2016).
35. Valiela, I., Bowen, J. L. & York, J. K. Mangrove forests: one of the world's threatened major tropical environments. *Bioscience* **51,** 807–815 (2001).
36. *Fish to 2030: Prospects for Fisheries and Aquaculture*. Discussion paper no. 3 (World Bank, 2013); http://documents.worldbank.org/curated/en/458631468152376668/Fish-to-2030-prospects-for-fisheries-and-aquaculture
37. Armitage, D. Socio-institutional dynamics and the political ecology of mangrove forest conservation in Central Sulawesi, Indonesia. *Glob. Environ. Change* **12,** 203–217 (2002).
38. Richards, D. R. & Friess, D. A. Rates and drivers of mangrove deforestation in Southeast Asia, 2000–2012. *Proc. Natl Acad. Sci. USA* **113,** 344–349 (2016).

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Author contributions
T.B.A., R.M.C., and C.E.L. designed the study. T.B.A., C.E.L., H.A., P.E.C., C.M.D., C.J.E.L., X.I., J.J.K., P.S.L., P.I.M., O.S., C.J.S., I.S. and A.D.L.S. contributed data. T.B.A. analysed the data and drafted the first version of the manuscript. All authors contributed to the writing and editing of the manuscript.

Additional information
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Competing financial interests
The authors declare no competing financial interests.

Methods

We used Google Scholar, Web of Science, personal data sets and published reports to generate the most robust data set to date on mangrove soil C stocks[17]. Studies included in the data set contained both undisturbed and degraded mangrove ecosystems; however, planted mangroves were excluded. At a minimum, studies needed to include latitude and longitude of the sample location, percentage of organic carbon (hereon referred to as C) content of the soil or loss on ignition (LOI), core depth and section thickness. In some cases, the original data were presented as an average across multiple sites; in these cases, GPS (Global Positioning System) coordinates of the coastal middle point were used. Studies reporting LOI were converted to percentage of C by dividing by 2.07, an estimate of the ratio of organic matter to C concentration[39]. Studies that included measures of organic matter concentrations in the soil were converted by multiplying the organic matter content by 0.58. The inclusion of studies reporting LOI or percentage of organic matter allowed us to include data from less developed nations. We standardized soil C stocks down to 1 m in the soil. Studies containing soil depth profiles <1 m were extrapolated by taking the average percentage of C content and dry bulk density from known depths and multiplying it to 1 m. We used the pedotransfer function in equation (1) to estimate dry bulk density in studies that did not report one.

$$\text{dry bulk density} = 1.25 * \%\text{C}^{-0.5163} \tag{1}$$

We quantified data quality for each country by scoring characteristics relating to the quality of individual data points and the overall country-wide data set (Supplementary Table 1). A total of seven categories were included and quality scores were ranked on a scale of 0–3, with a score of 21 being the highest possible. Individual data point categories were related to the quality of the data needed to estimate C stocks down to 1 m in the soil, as well as the publication quality. Individual data point categories include: quality of C stock data, quality of down-core stock data to 1 m, quality of percentage of organic carbon data, quality of bulk density data, and quality of the publication. Individual data point scores for each country were averaged within a category. Overall country-wide data set characteristics were related to the extent of data coverage for that country and included: extent of mangrove genera covered and extent of marine eco-regions covered (MEOW). We used MEOW to qualify data extent because eco-regions represent areas of relatively homogeneous species compositions that have distinct communities and biogeographical forcing agents (for example, nutrient inputs, freshwater influx, temperature, sediments, currents and coastal complexity) compared with adjacent systems. To determine the extent of MEOW regions covered we overlaid MEOW, site locations and mangrove habitat extent GIS (geographic information system) layers. The scoring matrix is presented in Supplementary Table 1.

We investigated how soil C stocks per unit area differed across hemispheres and latitudes. In some cases, studies did not report latitude coordinates, but rather supplied a map of sample locations. In these instances, we used Google Earth to manually obtain a more precise location. To examine the difference between Northern and Southern hemispheres in C stocks per unit area we used a t-test. To examine differences among latitudes in soil C stocks per unit area we used a one-way ANOVA with latitude grouped into 10° latitudinal belts (0–10, 10–20, 20–30, 30–40) either side of the Equator. All analyses were conducted using the statistical programming package R (R Development Core Team 2015).

We also assessed how different aspects of the mangrove community composition affected C stocks per unit area. Studies were included in this analysis if they contained information about the genus or species present in the study location. Stands were considered monotypic if they were dominated ($>75\%$) by a single genus of mangrove. The classification for dominance was mainly based on number of individuals; however, biomass was also commonly used. In cases where quantitative measurements of genera composition were not reported we accepted qualitative statements of dominance by the author of the study. We assessed the effects of mixed versus monotypic stands on soil C storage per unit area using a t-test. We looked at the effects of mangrove genera richness and mangrove genus in monotypic stands on soil C stocks per unit area using linear mixed-effect models with the statistical programming package R. We included study ID as a random factor in models looking at genera richness and genus to account for the fact that in some cases data came predominantly from only one or two studies.

To scale up to country, soil C stocks per unit area were averaged across all sites within a country and then multiplied by the Mangrove Forests of the Word (MFW) country-specific mangrove habitat area for 2014[1]. For the 57 countries that lacked soil C data we used the average global C stocks per unit area of 283 ± 194 Mg C ha^{-1}. We used MFW as it is more conservative than the Terrestrial Ecosystems of the World-Mangrove Biome (TEOW)[1]. The global mangrove soil C stock was estimated by summing all country-specific soil C stocks.

To estimate potential CO_2e emissions as a result of mangrove decline, we compiled published data on the effects of different types of disturbance on soil C stocks or C content. In total we found 19 studies reporting change in mangrove soil C after disturbance (Supplementary Table 5). In all cases, studies compared impacted sites with non-impacted sites, as opposed to pre- and post-disturbance. Disturbances ranged from those that directly affected the soils (aquaculture, agriculture and urban development) to those that directly affected above-ground biomass with potential impacts to the soils (timber harvest, grazing, pollution and water diversions), as well as sites that had a combination of both types of disturbance. In addition, ~70% of these studies quantified impacts down to 80 cm to >100 cm in the soil, while the remaining 30% quantified affect to 20 or 50 cm depth in the soil. Effects on soil C ranged from 10% to 85% (average 43% ± 5%, s.e.). However, we found that there was no statistically significant difference of the affect of disturbance type (directly affected soils, indirectly affected soils, both) on soil C (ANOVA: $F_{3,16} = P = 0.159$). As a result, we used the average loss of 43% for C stocks down to 1 m in the soil for all countries. Propagating losses down to 1 m in the soil is the protocol suggested by the IPCC for estimating CO_2e emissions from mangrove ecosystems[40], despite several studies having shown that disturbances can influence soil organic carbon to depths >1 m (refs 27,28,41), thus underestimating CO_2e emissions. Country-level potential annual CO_2e emissions from mangrove losses were estimated by first reducing the country's C storage per unit area by 43%, then we multiplied a country's reduced C storage per unit area by 3.67 (the molecular weight ratio of CO_2e to C) and then multiplied by its annual mangrove habitat loss[1]. CO_2e emissions are reported as CO_2e (or carbon dioxide equivalents) because CO_2e is the most common and conservative C-based greenhouse gas[39]. Potential global annual CO_2e emissions from mangrove soils were estimated by summing all country-specific potential annual CO_2e emissions. Although in some cases it may be unlikely for the entire 43% of the C stock to be lost in just one year, thus overestimating annual CO_2e emissions, we followed IPCC protocols that state that it should be estimated that all C in the pool is emitted as CO_2e during the year of the land-use conversion[40].

To help determine which countries should be priorities for conservation we also investigated the relationship between total soil C stocks by country and country-specific mangrove area, and country-specific soil C stocks per unit area and potential annual CO_2e emissions using linear regressions. For these regressions we included only countries for which soil C data were available, which excluded the 57 countries where we used the average global soil C stocks per unit to estimate total soil C stocks. For the regression investigating the relationship between country-specific soil C stocks per unit area and potential annual CO_2e emissions we did not include countries with no annual mangrove loss. We tested for statistical differences between the slope of the regressions and a 1:1 relationship using a Wald test in the R package 'car'. Data were log transformed prior to the analyses to achieve normality of residuals and to improve homoscedasticity of variances.

Data availability. Soil C data and mangrove community composition data that support the findings of this study have been deposited in Pangaea with the identifier doi: http://dx.doi.org/10.1594/PANGAEA.874382.

References

39. Kauffman, J. B. & Donato, D. C. Protocols for the measurement, monitoring, and reporting of structure, biomass and carbon stocks in mangrove forests. CIFOR Work. Pap. 86. Cent. Int. For. Res. Bogor, Indones.
40. Kennedy, H., Alongi, D. M. & Karim, A. CHAPTER 4: Coastal Wetlands. 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands (2013); http://www.ipcc-nggip.iges.or.jp/public/wetlands
41. Bhomia, R. K., Kauffman, J. B. & McFadden, T. N. Ecosystem carbon stocks of mangrove forests along the Pacific and Caribbean coasts of Honduras. *Wetl. Ecol. Manage.* **24,** 187–201 (2016).