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Effects of shrimp aquaculture on mangrove soil carbon stocks and sustained-flux global warming potentials

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Mangroves provide a multitude of ecosystem services, including storing up to *ca.* 20 Pg of carbon. Aquaculture driven mangrove degradation results in significant carbon losses associated with forest clearance, drainage, and the removal of surface soils. However, uncertainties exist regarding the magnitude of loss due to environmental factors, land use, study type and aquacultural practices. Sustained-flux Global Warming Potentials (SGWP) during construction, operation and abandonment phases is understudied. Here we used a microcosm simulation in order to better constrain carbon loss pathways and SGWP, inform management practices and highlight knowledge gaps. Biogeochemical emissions during culture and abandonment added *ca.* 25%, suggesting the impact of aquaculture is underestimated. However, soil removal during construction dominated emissions (*ca*. 75%). Thus, long operation duration and re-use of ponds would reduce emission factors, and the fate of removed soils is suggested as a priority for research. Extrapolation suggests emissions could be important for national and regional carbon accounting, but also in relation to global climate mitigation, given the potential for significant underestimates of the impact of aquaculture on mangrove ecosystems.

KEYWORDS
aquaculture, blue carbon and ecosystem services, climate mitigation, greenhouse gases (GHG), mangrove

1 Introduction

Mangrove forests deliver valuable ecosystem services (Zhang et al., 2025), including providing nursery habitat for many species of fish, protection from coastal erosion and supplying fuel and food to sustain local livelihoods (Lee et al., 2014). Mangroves are especially efficient at fixing atmospheric carbon dioxide (CO_2) and storing it as coastal 'blue' carbon, making them among the most carbon abundant ecosystems in the world (Sanderman et al., 2018; Sasmito et al., 2020). Indeed, mangroves have been described as the most important carbon sinks in the tropics, their production exceeding that of terrestrial forests (Alongi, 2014; Dung et al., 2016; Murdiyarso et al., 2015). Estimates of mangrove carbon storage range from 3.7 Pg–6.2 Pg (Ouyang and Lee, 2020) to 4–20 Pg (Donato et al., 2011) with Alongi (2020a) estimating 6.17 Pg and Kauffman et al. (2017) estimating 10.8 Pg C. This equates to ≥ twice the annual quantity of CO_2 emissions produced anthropogenically (IPCC, 2018; Watson et al., 2020). The carbon bank mangroves provide is largely due to deep and organic-rich soils, storing carbon for centuries if left undisturbed

(Saderne et al., 2019) with soil carbon dominating stocks globally (Elwin et al., 2019) and accounting for up to 90% of total ecosystem carbon stock (Hong Tinh et al., 2020; Royna et al., 2024; Sanderman et al., 2018). However, this can differ markedly depending on hydrogeomorphic category (Balke and Friess, 2016). Anoxic conditions are created within inundated soils, limiting decomposition rates and therefore preventing carbon loss to the atmosphere (Cheng et al., 2015; Saraswati et al., 2016).

Mangroves are being lost at an estimated 1%–2% per year (Carugati et al., 2018). Recent estimates (Ju et al., 2025) suggest that global mangrove area decreased from 17.35 million-hectares in 1985 (carbon-storage of 6.84 Pg) to 13.61 million hectares in 2020 (carbon-storage of 5.72 Pg). Major threats include eustatic sea-level rise, pollution and coastal development (Pendleton et al., 2012). The largest driver of mangrove deforestation is agricultural coastal development, with the aquaculture industry alone accounting for 38% of loss (Thomas et al., 2017). In 2018 the market value of the aquaculture industry was estimated to be 250.1 billion USD and demand is increasing by around 10% annually (FAO, 2020). Shimp farming constitutes 18% of the value for all seafood products traded globally. Extensive demand for farmed shrimp has fuelled the destruction of mangroves, which are cleared to facilitate the establishment of aquaculture ponds (Bosma et al., 2012). Converting mangroves to shrimp ponds involves cutting down mangrove trees within the construction area (Rahman et al., 2013). Subsequently, the pond borders are outlined and excavated, often using heavy machinery, to create ditches along the inner margins. Excavated soil is then utilised to construct dykes around the edges of the shrimp pond, in order to exclude tidal water. Additionally, any remaining vegetation is often burnt (Bosma et al., 2012).

There is little doubt there is the potential for large carbon losses to the atmosphere as a result of forest clearance, and removal of surface soils, and many studies emphasise the importance of integrating the reduction of mangrove deforestation into climate change mitigation strategies (e.g., Arifanti et al., 2019; Järviö et al., 2018; Kauffman et al., 2014). However, this mitigation strategy is disputed (Alongi, 2020a) and considerable uncertainties remain particularly in coastal settings (Adame et al., 2024) because: 1. There is a large variation in study types (Sasmito et al., 2019), location (Reithmaier et al., 2020), season (Cabezas et al., 2018), salinity, mangrove species and hydrogeomophology (e.g., Balke and Friess, 2016; Marchio et al., 2016; Sasmito et al., 2020), tidal amplitudes (Rovai et al., 2018), and aquaculture practices (Kosten et al., 2020; Tan et al., 2023; Yuan et al., 2019; Zhang et al., 2022), specificity of land use, etc. Reithmaier et al. (2020) suggest that whether or not mangroves act as greenhouse gas (GHG) sources or sinks varies strongly between sites and Sasmito et al. (2019) recommend paired studies in order to better assess net carbon balances and emissions factors. 2. Many studies have used the Intergovernmental Panel on Climate Change (IPCC) Landuse Carbon Stock (LUCS) approach to estimate carbon losses and potential emissions as a result (Supplementary Text S1) potentially omitting 'biogeochemical' pathways of soil carbon loss and GHG contributions. Similarly, long-abandoned systems are often included (Ju et al., 2025; Sasmito et al., 2019) which may contain regenerated carbon stocks (Elwin et al., 2019), thus, underestimating the impact. 3. Conversely, whilst the operational emissions of shrimp ponds and aquaculture (Kosten et al., 2020) have been studied, Arifanti et al. (2019) recommend that soil loss due to land use conversion should be included in product Life Cycle Assessments. 4. There is a paucity of research on newly abandoned systems despite these representing large numbers (Elwin et al., 2019). The total Sustained-flux Global Warming Potential (SGWP) as a result of construction, operation and abandonment is understudied, despite the potential for significant underestimates of the value of avoiding these emissions in climate mitigation strategies and potential for management opportunities to reduce emissions. Thus, here we performed a microcosm experiment in order to test whether 'biogeochemical' carbon losses as a result of conversion to aquaculture are significant in drainage, operational or abandoned phases compared with 'physical' losses through soil removal and to calculate the SGWP across all phases in order to better inform management practices, research priorities and climate mitigation potential strategies. It was hypothesized that 1. physical losses would dominate emissions through carbon losses, but that 2. overall SGWP would be significantly underestimated as a result of biogeochemical emissions during drainage, operation and abandonment. A strictly paired systematic review and meta-analysis (SR) was also undertaken in order to provide large-scale context with restricted confounding factors.

2 Materials and methods

2.1 Site description and microcosm construction

The site used to collect material for the microcosm study sits within the Estero Bay Aquatic Preserve (see Abeels et al., 2012) near Bonita Springs, Florida, USA. It is a semi-diurnal, micro-tidal system including the estuarine zone of the Imperial River which flows into the Gulf of Mexico. The subtropical climate of Southwest Florida (mean annual temperature 23.6 °C) is characterized by a dry season (October/November to May) and a wet season June to September (Abeels et al., 2012; Collins et al., 2017) when 60%–65% of the 1346 mm year^{-1} precipitation occurs (Marchio et al., 2016). Soil collection took place near the Vester Field Station (N 26° 19.895, W 81° 50.282) where the fluvial mangrove system is dominated by the comsmopolitan species *Rhizophora mangle*.

Microcosms were constructed from polyvinylchloride pipes sitting within an outer reservoir of site water (Supplementary Figure S1; Supplementary Table S1; Fenner et al., 2021) using the tidal simulation method of MacTavish and Cohen (2014) and maintained at field temperature with lighting simulating diurnal conditions (in climate-controlled facilities at Bangor University, UK; Supplementary Text S2) for 2.5 years (including 0.5 years duration for the drainage phase and 1 year each for the operational and abandoned phases. Measurements were taken 4 times per year at midday using 6 microcoms per treatment and averaged over distinct time periods (e.g., operational or abandoned), whilst retaining variability across replicates (Fenner et al., 2021).

Waterlogged soil from 0 (i.e., soil surface) to 0.5 m depth was collected as intact cores to construct 6 reference 'mangrove' microcosms. Twelve more samples were collected in order to construct 6 'drained' and 6 'aquaculture' microcosms. For the latter, the surface 1.0 m was removed in a layer from drained soil simulating typical shrimp pond construction (Kauffman et al., 2014). Peat from 1.0 to 1.5 m depth was used for the aquaculture microcosms. Collection was done with minimal distruption to the surrounding area and avoiding trees to preserve the site. The layer of soil removed was replaced except for a small, exposed, disturbed area which was later re-sampled. Water from White Shrimp (*Litopenaeus vannamei*) monoculture, characterized by high levels of nutrients from commercial aquatic feed pellets (Supplementary Table S2), was added to simulate the operational phase. White shrimp (accounted for 53% of total global crustacean production in 2016, and is one of the most important commercial species in the coastal aquaculture industry (FAO, 2020; Tan et al., 2023). There was no water exchange during operational and abandoned aquaculture phases, simulating a typical post-shrimp harvest stage (Tan et al., 2023) and post-abandonment conditions where sluices and levees continue to block freshwater and tidal inputs (Elwin et al., 2019; Kauffman et al., 2014).

2.2 Biogeochemical measurements

Site hydrochemical measurements were taken before microcosm collection with a YSI ProDSS (Yellow Springs Instruments Digital Sampling System) Multiparameter Meter across a 6 m transect (mean salinity 23.62 ± 0.06 ppt, temperature 23.49 °C ± 0.02 °C, pH 8.15 ± 0.01). Microcosm hydrochemical analysis included the above determinands, inorganic ions (Metrohm 850 IC, Herisau, Switzerland), Total Nitrogen (TN) and Dissolved Organic Carbon (DOC; difference between Total Carbon (TC) and Inorganic Carbon (IC) concentrations in 0.45 µm filtered water) using a Total Organic Carbon/TN analyser (Thermalox, Analytical Sciences, Cambridge, UK).

Soil TOC and IC content was determined according to Marchio et al. (2016) and the difference between TC and IC concentrations. Soil carbon was the focus given its dominance in total ecosystem carbon stock (Hong Tinh et al., 2020; Royna et al., 2024; Sanderman et al., 2018). Bulk density (ρb) was calculated using the method of Marchio et al. (2016), which involved drying to constant weight (60 °C), removal of visible roots and calcareous shell material and homogenization. After drying, 2 cm soil samples were weighed to determine ρb in g cm^{-3}.

GHG flux measurements followed the method of Fenner and Freeman (2013) and Sugiana et al. (2023) using a closed chamber technique to measure whole microcosom fluxes accounting for photosynthesis. Based on the mean gas fluxes of the microcosms, the CH_4 and N_2O fluxes were converted to CO_2-equivalent (CO_2-e) fluxes to indicate SGWP (Neubauer and Megonigal, 2015). The impact of abandoned systems was estimated using SGWP half-life ($t_{1/2}$; Supplementary Text S2).

2.3 Systematic review and meta-analysis

A strictly paired Sytematic Review (SR; Supplementary Text S3; Supplementary Figure S2) was used to collect soil carbon loss and ρb data from the literature minimizing the potential for confounding factors (e.g., differences in reference mangrove system, salinity, etc.) and using only internationally standardized methodology.

2.4 Statistical analysis and extrapolation

The Shapiro-Wilk test was used to test for nomality (Ghasemi and Zahediasl, 2012; Supplementary Text S4). If data were normally distributed, then paired t-tests were conducted between mangrove and aquaculture sites, with regression analysis for the identification of relationships between variables and repeated measures ANOVA for the comparison of phases. If data were skewed, a Wilcoxon signed-rank and Spearman's rank test was undertaken with a median presented. A p-value of $p \leq 0.05$ was used to determine statistical significance. The Global Seafood Alliance estimate for the pond area of the mainland United States and the global total was used for extrapolation (520 and 2,135,110 ha, respectively; Supplementary Text S4).

3 Results

3.1 Carbon losses and bulk density

During the aquaculture construction phase 1 m soil was removed containing approximately 422.1 ± 14.6 Mg C ha^{-1} (1549.11 Mg or tonnes ha^{-1} CO_2e) and resampling suggested a loss of ~242.9 Mg C ha^{-1} or 57.78% (Table 1). The mangrove microcosms contained an estimated 242.1 Mg ha^{-1} in the top 0.5 m. Comparing the mangrove to the operational aquaculture microcosms showed a 30.1% ± 0.8% significant loss of soil carbon (74.67 Mg C ha^{-1} or 274.04 Mg ha^{-1} CO_2e; t-test $p < 0.001$ Figure 1A), with a reduction in soil carbon observed at every soil depth analysed (Figure 1B). Bulk density also decreased (54.59%, Figure 1C; t-test, $p < 0.001$) and a strong correlation with carbon stock was found (Pearson r = –0.885, $p < 0.001$; Figure 1D).

3.2 Potential CO_2e emissions

Under the drainage regime mean SGWP increased by 34,075.38 mg m^{-2} d^{-1} (172.39%) with all GHG species contributing modestly (Figures 2A,B; Supplementary Table S3) compared with the reference mangrove giving a difference of 21,565.39 mg m^{-2} d^{-1} or 0.79 Mg CO_2e ha^{-1} yr^{-1} Mg ha^{-1} (Figure 1E; Table 1). The largest difference in SGWP as a result of GHG emissions was found during the operational phase of aquaculture; 65,343.26 mg m^{-2} d^{-1} (522%) equivalent to 2.39 Mg ha^{-1} yr^{-1} CO_2e or 0.65 Mg ha^{-1} C. Both CH_4 and N_2O effluxes increased significantly, whilst CO_2 uptake was higher than in the reference mangrove systems (Figures 2C,D; Supplementary Table S3). The abandoned pond simulation showed a 393.59% increase in comparison to the reference mangrove (a difference of 49,238.68 mg m^{-2} d^{-1}) after 1 year, i.e., 1.80 Mg ha^{-1} yr^{-1} CO_2e or 0.49 Mg ha^{-1} yr^{-1} C with both CH_4 and N_2O effluxes contributing significantly, whilst CO_2 emissions were again lower than the reference mangroves (Figures 2E,F; Supplementary Table S3).

3.3 Total lifetime emissions and extraploation

Taking 5 years of impact post abandonment a further 5.19 Mg ha^{-1} yr^{-1} CO_2e (2.04 Mg C ha^{-1}) is added from GHG emissions. Assuming 3–9 years of operation (Kauffman et al., 2018) and 5 years of impact post abandonment the total SGWP was 12.35–26.66 Mg ha^{-1} CO_2e or 13.13–27.44 Mg ha^{-1} CO_2e with a

TABLE 1 Carbon losses and potential CO_2e emissions due to simulated shrimp aquaculture compared with mangrove microcosms. Total carbon in removed soil is shown but how much of this is lost is not well constrained - limited measurement of actual loss in italics. Lifetime emissions assuming typical 3–9 year operational duration with 5 years of abandoned emissions, with and without a year's drainage. SGWP denotes Sustained-flux Global Warming Potential. Six microcosms were used per treatment.

			LIFETIME EMISSIONS (Mg ha^{-1} CO_2e)			
POND PHASE	Mg C ha^{-1}	Mg ha^{-1} CO_2e	3+5 years	+ Drainage	9+5 years	+ Drainage
Construction						
Soil Removal	422.10	1549.11	1549.11	1549.11	1549.11	1549.11
Soil removal C loss	*242.09*	*888.45*	*888.45*	*888.45*	*888.45*	*888.45*
Drainage	0.22	0.79				
Operational						
Soil carbon loss	74.67	274.04	274.04	274.04	274.04	274.04
GHG	0.65	2.39	12.35	13.13	26.66	27.44
Abandoned						
GHG	0.49	1.80				
Total						
SGWP	498.13	1167.47	1174.84	1175.62	1189.15	1189.93

year's drainage. Adding biogeochemical soil carbon losses gave 286.39–300.70 Mg ha^{-1} CO_2e or 287.17–301.48 Mg ha^{-1} CO_2e with drainage. Adding the construction soil loss gives 1174.84–1189.15 Mg ha^{-1} CO_2e or 1175.62–1189.93 Mg ha^{-1} CO_2e with drainage (Table 1). The percentage of emissions via the biogeochemical pathways compared with the total was similar without (24.33%–25.24%) and with drainage (24.38%–25.29%).

3.4 Systematic review and meta-analysis

The effect size of aquaculture on soil carbon stocks for each of the 6 paired studies were all similar and could be represented within the overall effect size, except one (Supplementary Figure S3a; Supplementary Table S4; Supplementary Text S5); thus, Bhomia et al. (2016) was excluded from the subsequent meta-analysis. No statistically significant heterogeneity between effect sizes for the 5 remaining 'core' studies was found (overall effect size −1.63, $p <$ 0.001) and this represents a large across-study effect size of aquaculture on mangrove stocks (Sullivan and Feinn, 2012).

The overall difference in soil carbon stocks between mangroves and aquaculture ponds was 46.82% ± 8.70%, with mangroves containing significantly larger carbon stocks (t-test $p < 0.001$; Supplementary Table S5; Supplementary Figure S3b). Mangroves contained on average 819.38 Mg C ha^{-1}, compared to 435.77 Mg C ha^{-1} for aquaculture ponds (t-test, p < 0.001). Aquaculture pond soil carbon stocks were significantly reduced at every soil depth analysed (Supplementary Table S6; Supplementary Figure S3c) when compared to mangrove soils (P < 0.05 for 0–15, 15–30 and 30–50; P < 0.001 for 50–100 and 100+; t-test or Wilcoxon signed-rank test). Differences between 0 and 30 cm belowground were similar, with mangroves containing on average 42.57% ± 11.09% more carbon. Below 30 cm, however, the differences between mangroves and aquaculture ponds were much larger. At 100+cm belowground mangroves contained on average 62.55% ± 12.16% more soil carbon than aquaculture ponds. For both mangroves and aquaculture ponds, the majority of their soil carbon stock was found below 100 cm depth. In mangroves, soil carbon stock 100+cm belowground was found to be 457.52 ± 48.40 Mg C ha^{-1}, compared to a combined average of 337.81 ± 6.32 Mg C ha^{-1} for the first 100 cm. Aquaculture ponds showed a similar pattern, with on average 105.88 ± 4.94 Mg C ha^{-1} in the first 100 cm, and 171.33 ± 43.37 Mg C ha^{-1} below 100 cm. Differences in carbon stock between 0 and 15 cm and 100+cm were significant in both mangroves and aquaculture ponds p < 0.001; Supplementary Figure S3d).

Notable differences in ρb were observed, with mangrove soils having significantly lower ρb values than aquaculture ponds (Wilcoxon signed-rank Z = −4.21, $p < 0.001$, Supplementary Figure S4a; Supplementary Table S7). A Spearman's rank correlation coefficient revealed a negative correlation between soil carbon stocks and ρb ($r_s = -0.490$, $p < 0.001$; Supplementary Figure S4b) whilst the relationship with salinity was also negative but more variable (Supplementary Figure S4c).

CO_2e emissions were estimated in 4 core papers (Supplementary Table S5) based on the LUCS approach (IPCC, 2003; Kauffman et al., 2018). This ranged from 1390 Mg CO_2e ha^{-1} (Kauffman et al., 2018) to between 2244 and 3799 Mg CO_2e ha^{-1} (Kauffman et al., 2014). The mean soil carbon loss (383.61 Mg C ha^{-1}) was equivalent to 1407.85 Mg C ha^{-1} CO_2e. Only one study included N_2O production during aquaculture operation and no studies measured all three major GHG or calculated SGWP.

4 Discussion

4.1 Effects of aquaculture on carbon loss pathways

During the simulated aquaculture operational phase 74.67 Mg C ha^{-1} of soil carbon stock (274.04 Mg CO_2e ha^{-1} y^{-1})

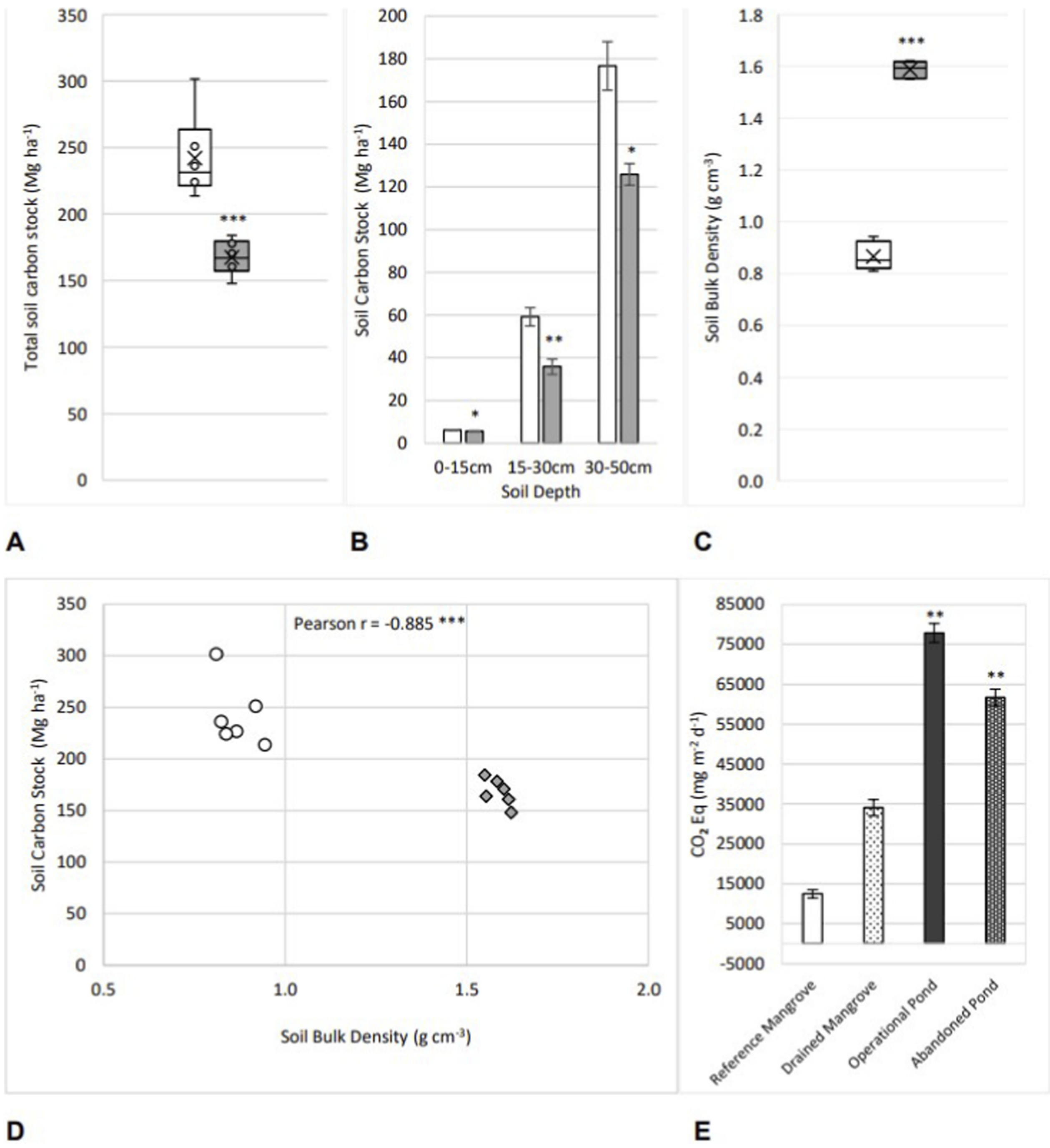

FIGURE 1
(A) Total carbon stock in mangrove (no fill) and aquaculture (fill) microcosms. Lines and cross within the box denote the median and mean value, bars represent the interquartile range, "whiskers" represent range (excluding outliers) and points denote outliers (beyond either the upper quartile + 1.5 x the interquartile range or lower quartile + 1.5 x the interquartile range). **(B)** Carbon stock depth profiles for mangrove (no fill) and aquaculture (fill) microcosms. Error bars denote standard error of the mean. **(C)** Bulk density in mangrove (no fill) and aquaculture (fill) microcosms. Lines and cross within the box denote the median and mean value, bars represent the interquartile range, "whiskers" represent range (excluding outliers) and points denote outliers (beyond either the upper quartile + 1.5 x the interquartile range or lower quartile + 1.5 x the interquartile range). **(D)** Carbon stock as a function of bulk density (aquaculture and mangrove microcosms denoted by filled diamonds and open circles respectively). **(E)** Sustained-flux Global Warming Potentials (SGWP) for each phase of simulated shrimp aquaculture (fill) compared with mangrove (no fill) microcosms. Six microcosms were used per treatment. Significance at $p < 0.05$, $p < 0.01$ and $p < 0.001$ denoted by *, ** and *** respectively.

was lost compared to the reference mangrove systems (Figure 1A), representing 'biogeochemical' pathways such as fine root decomposition (e.g., Balke and Friess, 2016) and stimulated anaerobic biodegradation (e.g., Hu et al., 2012) through nutrient and labile carbon inputs (Supplementary Tables S2, S8). The 1 m soil removed during construction contained *ca.* 422.1 Mg C ha^{-1}

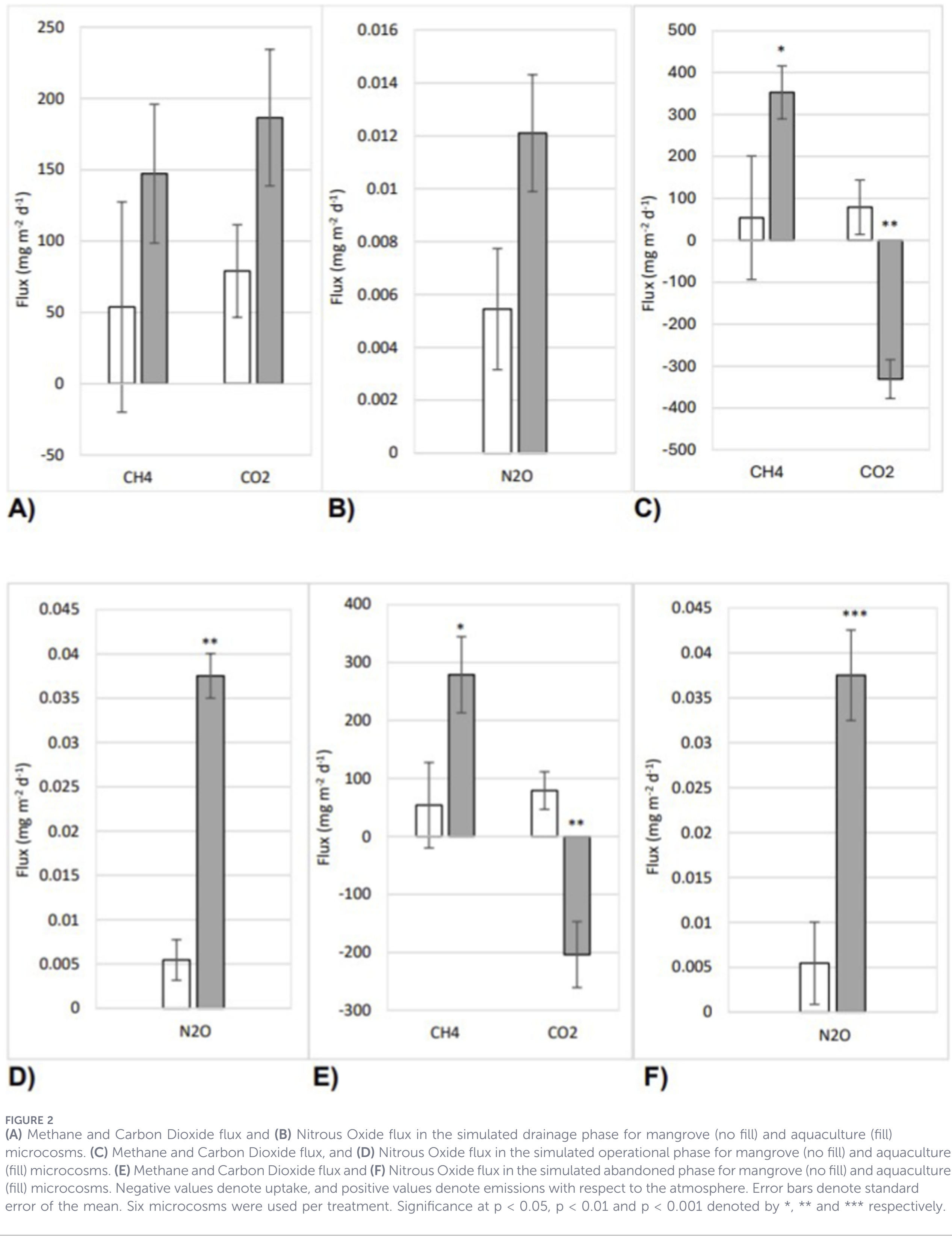

FIGURE 2
(A) Methane and Carbon Dioxide flux and **(B)** Nitrous Oxide flux in the simulated drainage phase for mangrove (no fill) and aquaculture (fill) microcosms. **(C)** Methane and Carbon Dioxide flux, and **(D)** Nitrous Oxide flux in the simulated operational phase for mangrove (no fill) and aquaculture (fill) microcosms. **(E)** Methane and Carbon Dioxide flux and **(F)** Nitrous Oxide flux in the simulated abandoned phase for mangrove (no fill) and aquaculture (fill) microcosms. Negative values denote uptake, and positive values denote emissions with respect to the atmosphere. Error bars denote standard error of the mean. Six microcosms were used per treatment. Significance at $p < 0.05$, $p < 0.01$ and $p < 0.001$ denoted by *, ** and *** respectively.

(Table 1), but how much of this is lost to the atmosphere will likely depend on factors such as how the soil is used (e.g., construction of dykes or dams), sediment type, inundation level and nutrient inputs. Limited measurements here suggest a loss of ~57.35% (242.09 Mg C ha^{-1}) representing 888.45 Mg ha^{-1} CO_2e. The processes responsible require further research but could include oxidation, leaching of DOC (Supplementary Table S8) and POC, and decomposition of fine roots (Balke and Friess, 2016). The total from soil carbon losses was therefore 316.77 Mg ha^{-1} or 1162.48 Mg ha^{-1} CO_2e with 23.57% contributed from biogeochemical pathways.

The effect size of −1.63 (Supplementary Figure S3a; Supplementary Table S4) in the SR is large (Sullivan and Feinn, 2012) and homogenous, indicating the conversion of mangroves to aquaculture ponds causes a significant loss of soil carbon irrespective of differences in salinity and other potentially confounding factors. This pattern may be due to the relatively consistent nature of physical drivers of carbon loss (soil removal, depth, compaction, etc.) in pond construction processes. The physical removal of the top ~1–1.5 m of soil is one of the main factors relating to soil depth reduction in aquaculture ponds (Elwin et al., 2019; Kauffman et al., 2014). The lower quartile of soil carbon values for aquaculture ponds at 1 m + depth is equal to the minimum value of 0 (Supplementary Figure S3d) because of the effect of multiple aquaculture sample sites comprising soil depths of ≤1 m. Elwin et al. (2019) suggest that standardisation to 1 m depth may more accurately compare carbon stocks between sites, as it eliminates soil depth effects on variability. The SR results suggest that the mangrove sites, when standardized to 1m, contained 338 Mg C ha^{-1} (Supplementary Table S9), similar to 329 Mg C ha^{-1} found by Elwin et al. (2019) and in line with that reported for the Americas and globally (Adame et al., 2024). Conversion to aquaculture ponds resulted in a loss of 43% (decreasing the stock to 194 t C ha^{-1}), rising to 45% in the 0–15 cm layer, in line with findings from abandoned ponds (Elwin et al., 2019). Soil carbon stocks decreased by 46.82% (i.e., 383.61 Mg C ha^{-1} or 1407.85 Mg CO_2e ha^{-1}) as a result of convertion from mangroves to aquaculture ponds and the mean mangrove soil carbon stock estimate (819.38 Mg C ha^{-1}) supports previous estimates (Adame et al., 2015; Cameron et al., 2019; Jones et al., 2016; Murdiyarso et al., 2015; Nam et al., 2016; Phang et al., 2015) despite the restricted sample. However, the lack of paired studies from South East Asia should be noted as an important contributor to global aquaculture (Boyd, 2018).

Significantly higher ρb values were observed in aquaculture microcosms (Figure 1C) indicative of degraded soils (Liu et al., 2019) and a significant inverse relationship with soil carbon stocks was observed (Figure 1D). Higher ρb values found in the field (Supplementary Figures S4a,b; Stringer et al., 2016; Xu et al., 2016) have been attributed to the use of heavy machinery during pond excavation, causing compaction and reduced porosity of soil (Arifanti et al., 2019; Castillo et al., 2017). Bulk density is an important factor determining soil quality and mangrove health, as increased ρb can restrict root growth and reduce available water capacity (Ola et al., 2019). How this impacts mangrove forest regeneration therefore is a research priority. However, many additional factors contribute to the large soil carbon losses as a result of conversion to aquaculture. Tidal amplitude is an important regulator of SOC content, with regular flooding of mangrove soils promoting preservation of carbon and extensive below-ground carbon stocks (Lovelock et al., 2015; Rovai et al., 2018; Woodroffe et al., 2016). Allochthonous carbon inputs through flooding are key (Saderne et al., 2019; Sasmito et al., 2020) particularly in maintaining minerogenic mangroves (Balke and Friess, 2016). The building of dykes prevents both freshwater and saline flooding (Elwin et al., 2019; Kauffman et al., 2014), likely explaning the lower carbon stocks of ponds at deeper depths (Supplementary Figure S3c). Since carbon burial occurs slowly, a continual decline after immediate losses may occur for a considerable period (months to years; Arifanti et al., 2019; Breithaupt et al., 2012; Richards et al., 2020), which may suggest carbon stock losses are an underestimate, but this is not well understood (Elwin et al., 2019). Similarly, aquaculture effluents could impact intact mangroves systems downstream through exports of nutrients and organic carbon (e.g., Hargan et al., 2020).

4.2 Effects of aquaculture on potential CO_2e emissions and SGWP

During the construction phase an extra 0.79 Mg CO_2e ha^{-1} annually (172.39% higher than the reference mangrove) was added as a result of drainage (Figure 1E; Table 1), in line with the 0.16–0.44 Mg of CO_2 effluxes found by Sidik and Lovelock (2013) from the floor and walls of Indonesian shrimp ponds, and the 0.29–1.06 range found by Lovelock et al. (2011) in deforested systems. A review by Alongi (2020b) found significantly higher rates of O_2 consumption in exposed compared to inundated mangrove soils in line with the concept of the ‘enzymic latch’ (Freeman et al., 2001) whereby increased enzymic decomposition has been found in temperate peatlands, freshwaters (Fenner and Freeman, 2013) and mangroves (Saraswati et al., 2016) with drainage. From the microcosm results, CH_4 and N_2O effluxes also increased (Figures 2A,B; Supplementary Table S3) suggesting peat degradation (Liu et al., 2019). Whether the exposure of deeper soils with higher carbon concentrations as a result of excavation intensifies the potential for carbon to be lost via biogeochemical oxidation pathways, or is well represented by the LUCS method of estimating potential emissions, is not well understood. Similarly, the temporal extent of drainage effects in these systems, remains poorly characterized, necessitating further investigation.

During the operational phase SGWP increased by 2.39 Mg CO_2e ha^{-1} $year^{-1}$ (Figure 1E; Table 1). Such increases support the concept that aquaculture ponds act as GHG emission hotspots driven by high levels of labile organic matter due to the regular addition of feed or fertiliser (Supplementary Table S8) to stimulate primary production (e.g., Kosten et al., 2020; Yuan et al., 2019). High methanogensis rates are characteristic of the anaerobic conditions that prevail in sediments and high ammonium and nitrate concentrations fuel N_2O production through nitrification and denitrification (Figures 2C,D; Supplementary Table S8; Hu et al., 2012). On the other hand, CO_2 emissions, largely return recently fixed CO_2 (in organic fish feed) back to the atmosphere with some ponds acting as CO_2 sinks (Figure 2C; Supplementary Table S3; Flickinger et al., 2020; Zhang et al., 2022) when algal primary production exceeds CO_2 release from sediments and the water column. However, aeration regimes and drainage for harvesting purposes are important in dictating emissions, particularly due to effects on N_2O (Kosten et al., 2020).

In the abandoned phase an extra 1.80 Mg CO_2e ha^{-1} $year^{-1}$ was produced with a similar pattern of GHG fluxes to the operational phase (Figures 2E,F; Supplementary Table S3), suggesting a legacy of biogeochemical impacts. While it is not known how long abandoned ponds would continue to produce significant emissions, 5 years would give a further 5.19 Mg CO_2e ha^{-1} $year^{-1}$. The length of time systems have been adandoned for before assessment will affect carbon budgets and GHG inventories, along with hydrology, but is not always well documented. Elwin et al. (2019) suggest that comparing carbon stocks in the top 0–15 cm of soil in their abandoned ponds with those at 1.5 m depth in reference mangroves is likely to be more useful because ~1.5 m of soil was removed from the ponds during construction. Soil carbon stocks in the abandoned ponds were ~70% lower than the mangrove sites on this basis and here we found this could be closer to 90% when newly abandoned systems are included (Supplementary Table S9) consistent with hypothesis 2. Elwin et al. (2019) found that carbon stored in the surface soils of aquaponds is comparable to natural mangroves 22 years after abandondment, where tidal flushing occurs. However, continued hydrological isolation is common (Elwin et al., 2019; Kauffman et al., 2014) suggesting constraining legacy effects from degradation of mangrove soils is a research priority.

Assuming 3–9 years of operation and 5 years of impact post abandonment, the SGWP was 12.35–26.66 Mg CO_2e ha^{-1} (or 13.13–27.44 with drainage). Adding operational (i.e., biogeochemical) soil carbon losses gave between 286.39 and 300.70 Mg CO_2e ha^{-1} or 287.17 to 301.48 with a year's drainage. Adding the construction soil loss gives 1174.83–1189.14 Mg CO_2e or 1175.62–1189.93 with drainage. Biogeochemical pathways contributed similarly without (24.33%–25.24%) and with drainage (24.38%–25.29%), again suggesting underestimates of the impact of aquaculture (hypothesis 2). However, soil carbon losses dominate, consistent with hypothesis 1, and should be included in life cycle assessments of aquaculture products (Arifanti et al., 2019). The reuse of abandoned ponds could, therefore, significantly mitigate SGWP if disease and other issues could be minimized, delayed or prevented. Annual emissions are highest for the shortest operational periods, since the greatest loss comes from construction and initial biogeochemical soil carbon loss, thus, longer operational durations would lower emission factors.

4.3 Comparison of the microcosm model experiments and systematic review

Whilst microcosms are a simplification of natural ecosystems they appear to be an appropriate model for investigating the effects of aquaculture on mangrove systems, on the basis of total carbon stock loss, major pathways of loss, depth profile patterns and relationships between OC loss and ρb, at least for systems represented by the core SR studies. Table 2 directly compares soil losses from the microcosms and the SR, where depth could be assigned. The former represents a 'top down' approach where 1 m of soil is removed and the loss from that stock quantified, along with the carbon lost in the remaining depth profile compared to a reference system. The SR represents a 'bottom up' approach which measures soil stock remaining *in situ* after soil removal for construction. The two approaches show very similar losses down to 100 cm depth. However, they diverge when the losses down to 150 cm depth are compared. In the microcoms, the loss is at this depth is calculated by adding biogeochemical losses to those from the removal of 1 m soil layer, hence this is a precisely constrained depth. However, this is much less certain for the SR which includes all depths up to 150 cm and deeper, likely explaining the substantial increase in comparison to the microcosms.

Similarities between microcosm studies and field studies may be due to the hydrologically closed nature of the ponds they simulate (Elwin et al., 2019; Kauffman et al., 2014). However, this needs further research, since the fate of the sediment in the construction phase is likely to impact carbon budgets along with the type of sediment and hydrological regime. Biogeochemical pathways of carbon loss are not measured directly in the SR and may be underestimated in the microcosm study due to the shallow depth of soil analysed, since impacts of aquaculture increase with depth. The effects of aquaculture simulation appears to replicate the effects of aquaculture on bulk density *in situ*, despite the lack of heavy machinery used to construct the microcoms, and this might relate to soil compaction via trampling, excavation, transport and changes in hydrology/saturation. However, this also may suggest degraded soils (Liu et al., 2019) through nutrient pollution (Supplementary Table S8) are a more important driver than physical factors.

The study by Bhomia et al. (2016) showed a considerable difference in soil carbon stock, the effect size being >3x larger than the overall effect size for core papers (Supplementary Figure S3a; Supplementary Table S4) and while this might be a result of low sample size, it may be related to low carbon content due to limited burial of autochthonous carbon in soils transported by rivers relatively recently (Bhomia et al., 2016). Thus, more research is needed to determine if certain sediment/geomorphic types are particularly vulnerable to carbon loss and therefore are priorities for conservation, particularly given that mangroves are presumed to contain high OC levels despite the predominance of minerogenic systems (Balke and Friess, 2016).

While we simulated high nutrient input as feed (much of which remains uneaten; Kosten et al., 2020), live shrimp pathways are not represented, which may affect potential carbon loss or emissions pathways. N_2O production inside the gut of cultured shrimp, for example, may be an important contribution to pond emissions (Kosten et al., 2020). Furthermore, shrimp production increases methanogenesis, but the effect of salinity on CH_4 production and consumption is less well known (Tan et al., 2023). Similarly, potentially important emission pathways, such as ebullition are commonly neglected (Kosten et al., 2020).

Due to the dominance of the physical soil loss pathway and the fact that some biogeochemical losses (e.g., soil) may also be captured, the LUCS approach may be robust and convenient method suitable for large-scale extrapolations, with the advantage of being internationally recognized (Kauffman and Donato, 2012) and allowing biomass carbon stocks to be accounted for. This approach is likely to underestimate the SGWP through short to medium-term pathways being poorly represented but potentially over estimate soil loses from soil removal. Thus, a combined microcosm and SR approach is recommended to ensure a better understanding of the value of mangrove carbon stocks.

4.4 Exploration of the importance of mangrove aquaculture in climate mitigation

Calculations of GHG emissions based on land use are usually reported at country to global scales using units of Pg or gigagrams (IPCC, 2006). The simulated potential emissions from soil loss alone extrapolated to the USA area gave 604,495.8 Mg ha^{-1} CO_2e or around 0.001 Pg (i.e., 0.01% of anthropogenic emissions estimated at 7.34 Pg CO_2e or approx. 2 PgC yr^{-1}; Watson et al., 2020). Potential CO_2e emissions in the SR (Supplementary Table S5)

TABLE 2 Comparison of soil carbon lost (Mg C ha-1) and total Sustained-flux Global Warming Potential (SGWP) from microcosms (top down approach measuring soil removed) and paired Systematic Review (SR; bottom up approach measuring soil carbon stock remaining *in situ*) using the mean for the latter SGWP. Six microcosms were used per treatment.

Effect of aquaculture	Microcosms	Systematic review
Soil C lost 0–50 (Mg C ha^{-1})	74.67	69.44
Soil C lost 0–100 (Mg C ha^{-1})	149.34	144.30
Soil C lost 0–150 (Mg C ha^{-1})	316.76	383.61
Soil carbon emissions (Mg ha^{-1} CO_2e)	1162.49	1407.85
Total SGWP (Mg ha^{-1} CO_2e)	1167.47	

ranged from 1390 Mg CO_2e ha^{-1} (Kauffman et al., 2018) to 3799 Mg CO_2e ha^{-1} (Kauffman et al., 2014). Extrapolating this to the mainland USA gives 0.001–0.002 Pg CO_2e, or around 0.01%–0.03% of anthropogenic CO_2e emissions. The mean soil carbon loss found was equivalent to 1407.85 Mg ha^{-1} CO_2e and extrapolating this gives 0.001 Pg CO_2e, or around 0.01% of anthropogenic CO_2e emissions. Whilst total SGWP at the USA scale gave an extra *ca.* 10,000–20000 Mg CO_2e when accounting for GHG contributions over the lifetime ranges of the ponds (Table 1), on an annual basis, this is similar (0.008%) due to the predominance of soil carbon losses and suggests a modest contribution to climate mitigation potential but significant national and regional contribution (especially in areas with high rates of deforestation/destruction), in line with Alongi (2020a).

Exploring extrapolation at the global shrimp pond area scale suggests that soil loss could be responsible for around 2.48 Pg CO_2e (or 33.81% anthropogenic emissions) and total SGWP gives 2.51–2.54 Pg CO_2e irrespective of drainage, which on an annaual basis would account for around 33.98%–34.59%). Moreover, the SR range was 2.97–8.11 Pg and mean soil carbon loss values gave 3.01 Pg CO_2e (40.95% of anthropogenic emissions), suggesting that integrating the reduction of mangrove deforestation into climate change mitigation strategies is important (Arifanti et al., 2019; Järviö et al., 2018; Kauffman et al., 2014) and in line with studies finding conversion of mangrove forests, especially to aquaculture ponds, cattle pastures and infrastructure upon deforestation, could account for >50% (0.09–0.45 Pg CO_2 $year^{-1}$) of the carbon lost from coastal systems to the atmosphere (Pendleton et al., 2012).

In addition to the considerations of model suitability, caution is required with extrapolation across complex and dynamic systems and the net effect of aquaculture would depend on sediment type, salinity, etc., due to effects on both the source/sink function of the reference mangrove (Cabezas et al., 2018; Reithmaier et al., 2020) and the impact of aquaculture on that system, which requires further research in terms of uncertainty ranges, vulnerability to emissions and conservation priorities (*c.f.* Adame et al., 2024). Without this, extrapolation to national/global scales may exaggerate emissions and sub-optimal or even deleterious management practices may be initiated. However, the importance of including mangroves in mitigation strategies could be underestimated because:

1. Only SOC is included here along with associated GHG, but biomass and other carbon stocks will also be lost.
2. The total SGWP from soil removal is not known since only soil carbon loss emissions were accounted for.
3. Operational phase soil carbon loss may be underestimated since the impact of aquaculture on carbon stocks increases with depth and time.
4. Drainage during aquaculture has been found to increase N_2O emissions by 6–100 fold (Kosten et al., 2020) and little is known about the post-abandonment phase.
5. Only pond areas were used for extrapolation and there are also 980,310 ha of extensive shrimp farming globally (Boyd, 2018).
6. Literature values may be biased towards regenerated carbon stocks in abandoned aquapond systems.
7. High DOC concentrations in aquaponds (Tan et al., 2023) and POM exports (Hargan et al., 2020) in effluents will potentially fuel further GHG emissions downstream.
8. Reference mangrove systems may be impacted by higher riverine N inputs fuelling higher N_2O emissions compared with pristine mangroves (Maher et al., 2018), leading to systematic underestimates of mitigation potential, given the dominance of fluvial systems.

5 Conclusion

Microcosm simulations of shrimp aquaculture suggest biogeochemical pathways could add *ca.* 25% to overall SGWP compared with the LUCS approach across construction, operational and abandoned phases. However, soil carbon loss during construction and operation was found to dominate and therefore 1. reuse of ponds and long operational periods would improve sustainability, and 2. the fate of removed soils is suggested as a priority for future research, along with constraining drainage effects and managing abandoned systems (e.g., through tidal reconnection). The microcosm model was found to be consistent with the results of the SR showing consistent loss of soil carbon across a range of sytems types (geographical region, ρb, salinity), however, the effect of sediment type on impact and regeneration is a research gap. Given that underestimates of SGWP are likely and global extrapolation suggests potential emissions of ~3 Pg CO_2e (2.97–8.11 Pg) representing *ca.* 40% of anthropogenic CO_2 contributions and 60% of the minimum CO_2 removal at 2050 proposed to achieve Paris Agreement targets (5–10 Pg CO_2 $year^{-1}$; Smith et al., 2023), the importance of understanding mangrove carbon sink functions in the face of growing demand for aquaculture products is highlighted; the value of avoiding emissions and integrated aquaculture approaches may not yet be appreciated for mangroves.

Data availability statement

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found in the article/Supplementary Material.

Author contributions

NF: Conceptualization, Data curation, Formal Analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Visualization, Writing – original draft, Writing – review and editing. EH: Conceptualization, Data curation, Formal Analysis, Investigation, Methodology, Validation, Visualization, Writing – original draft, Writing – review and editing. BB: Resources, Validation, Writing – review and editing. SC: Resources, Validation, Writing – review and editing. CD: Resources, Validation, Writing – review and editing. CF: Resources, Validation, Writing – review and editing. NM: Resources, Validation, Writing – review and editing.

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Conflict of interest

The author(s) declared that this work was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

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