RESEARCH ARTICLE

A nature-based solutions approach to managing shrimp aquaculture effluent

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Abstract

While coastal habitat conversion was a primary environmental concern in Asia for the mostly extensive shrimp aquaculture sector in previous decades, the transition towards intensive production is creating new environmental risks, primarily water quality impacts from nutrient-rich effluent. There is a need to compare the performance of conventional and Nature-based Solution (NbS) effluent treatment options given the increasing nutrient loads from more intensive aquaculture and historic loss of ecosystem services from mangrove deforestation. This study evaluates the potential for common and emerging effluent treatment systems to address total nitrogen and total phosphorus effluent from shrimp farms across a spectrum of production intensities. Nutrient waste loading for four stocking density scenarios (7PLm^{-2}, 20PLm^{-2}, 75PLm^{-2}, and 120PLm^{-2}) are estimated to compare the treatment efficiency, economic feasibility, spatial requirements, and ecosystem service provision of conventional and NbS effluent treatment systems. We use secondary data to assess effluent treatment systems applicable for shrimp aquaculture in Asia. Findings provide the conceptual framework for comparing the characteristics and tradeoffs of aquaculture effluent treatment systems. Constructed mangrove wetlands are an NbS approach that can meet the intensification needs of aquaculture producers and reduce negative impacts from aquaculture effluent at competitive costs, while also providing ecosystem service co-benefits.

Author summary

Shrimp aquaculture expansion has been a primary driver of mangrove deforestation and degradation globally. Although deforestation rates have slowed considerably in the last two decades, the loss of these critical ecosystems leaves coastal communities vulnerable to climate change risks, like sea level rise and increasing storm surge intensity. Concurrently, the shrimp aquaculture sector has been trending towards more intensive production, resulting in higher production volumes and the concentrated release of nutrient-rich effluent into aquatic ecosystems. These imminent environmental threats from shrimp aquaculture motivate our study focused on common and emerging effluent management approaches. We model four shrimp production intensity scenarios to estimate nutrient waste loads and use those waste loads as inputs to compare seven conventional and

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Nature-based Solution (NbS) effluent treatment systems. Effluent treatment systems are evaluated based on removal rate efficiencies, equivalent annual costs per kg harvested, spatial footprint requirements, and ecosystem service provision. Our results suggest that constructed mangrove treatment wetlands, a type of NbS approach, are not only an economically viable option for effluent management compared to more conventional approaches, such as settling ponds, but can also provide additional ecosystem services. Our study demonstrates how an NbS approach can be applied to shrimp aquaculture while accommodating commercial production needs and trends towards intensification.

Introduction

The substantial growth of global shrimp aquaculture has often come at the expense of coastal ecosystems, such as mangrove forests, which have been cleared to create new shrimp ponds. Global shrimp aquaculture production has increased by 10,000% in 40 years, from an estimated 74,000 metric tons (t) in 1980 to 7.43 million t in 2020 [1]. The majority of which, 83% (6.13 million t), originated from Asia, with China alone accounting for 2.57 million t and other top producing Asian countries (e.g., Viet Nam, India, Indonesia, Thailand, Bangladesh, Myanmar, Philippines, and Malaysia) contributing a combined 3.46 million t over the same time period [1].

Addressing emerging challenges that come with such growth, such as pollution and degraded water quality linked with intensification, while also rectifying the damage and ecosystem services lost from mangrove deforestation, will require new strategies to align environmental and economic incentives. Approaches that utilize Nature-based Solutions (NbS), in which the designed function of the system (e.g. nutrient effluent mitigation) can also provide ecosystem benefits, show promise in that they employ, *"actions to protect, sustainably manage and restore natural or modified ecosystems that [also] address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits"* [2]. The United Nation's Environmental Programme asserts that NbS are a restorative and cost-effective climate tool [3], but their viability and potential applications in aquaculture systems are nascent and far less understood [4,5].

Conversely, conventional treatment systems in this study are those that are manufactured or built, are static, and cannot adapt as environmental conditions change, nor are they able to recover or grow back after environmental or structural disturbances. These types of systems can be categorized as "gray infrastructure", which have a specific design life, during which they require appropriate maintenance and operation and after which they can no longer be expected to provide the designed service [6].

NbS are gaining support for more conventional applications, but a lack of bankable projects and general unfamiliarity for these types of solutions have limited their use [7]. Conventional built structures, like manufactured commercial filters housed in buildings, have a finite lifespan and limited resilience to a rapidly changing coastal environment. Structures that can incorporate climate-resilience strategies are likely to reduce damages from environmental disturbances and increase longevity [8]. Hybrid approaches, such as constructed treatment wetlands, that can integrate conventional engineering techniques with NbS, may be a strategy that minimizes the limitations of using either approach individually [6]. This type of integration of NbS may benefit coastal shrimp aquaculture and should be explored further.

Shrimp are commonly farmed in ponds and tanks under a range of production intensities, broadly categorized as extensive, semi-intensive, intensive, and super-intensive. These intensities are typically defined by multiple, linked metrics, including stocking density (individual post-larvae (PL) shrimp per m^2 per cycle) and harvested yield (t per hectare (ha) per cycle or per year) [9]. The gradations between these categories of production intensity are not universally defined and vary by farming region and across time [10–14]. Pacific whiteleg shrimp (*Litopenaeus vannamei*) are typically preferred for more intensive production systems [15] but are grown across a spectrum of intensities characterized by different levels of management, stocking densities, and yields (Table 1). Extensive production utilizes few to no inputs (i.e., relying mostly on natural productivity to provide seed, feed, and water treatment) while intensive and super-intensive production are entirely dependent on external inputs (i.e., farmers must provide increasing amounts of seed, feed, and water treatment) [16]. Increasing production intensity requires financial investments and additional technical capacity, both of which are often barriers for farmers to intensify from low (extensive) stocking density upwards to medium (semi-intensive), high (intensive), and very high (super-intensive) stocking densities [15].

While habitat conversion was the primary environmental concern for the mostly extensive shrimp aquaculture sector in previous decades [18–21], the shift to more intensive production creates new environmental risks, such as water quality impacts from nutrient-rich effluent (Fig 1). As production intensifies, supplemental feed is required to meet the metabolic requirements of the shrimp. Excess nutrients that are not utilized to create shrimp biomass, such as

Table 1. Whiteleg shrimp (*Litopenaeus vannamei*) production intensities and characteristics.

	Extensive	**Semi-Intensive**	**Intensive**	**Super-Intensive**
Water Quality Management	Limited to no water quality management interventions, passive tidal flushing.	Use of probiotics, fertilizers, and other inputs. Increased daily water exchange with pumping.	High use of inputs, some have pretreatment ponds.	Precision technology and high use of inputs.
Feed	Manufactured pellet feed rarely used, dependent on natural productivity and/or fertilizers to stimulate natural productivity.	Mostly manufactured pelleted feed or farm-made feeds.	Dependent on manufactured pellets.	Dependent on high-quality manufactured pellets.
Aerators	None	Few	Aerators essential	Many aerators are essential
Stocking Density (*PL* m^{-2})				
Indonesia [10]	1 to 10	–	50 to 150	> 200
Indonesia [13]	4 to 10	10 to 30	60 to 300	300 to 750
Viet Nam [15]	< 10	10 to 29	> 30	–
Viet Nam [11]	–	26 to 31	66	73
Thailand [11]	–	62.5[1]	82	99
Production Yield ($t\ ha^{-1}\ yr^{-1}$)				
Viet Nam [15]	< 0.70	< 3.5	> 3.5	–
Viet Nam [11]	–	0.27 to 3.5	7	11.7
Thailand [11]	–	1.3 to 3.5	7	13.6
Indonesia [10]	< 5	–	5 to 30	30 to 80
Not country-specific [17]	0.25 to 0.5	1 to 3	4 to 20	>20
Not country-specific [12]	< 1	2 to 20	20 to 200	> 200

[1]The stocking density for medium-intensity production from this study is higher compared to other sources but meets the other criteria for semi-intensive production. This can be a result of feeding rate, harvest weight of the shrimp, survivability or other factors.

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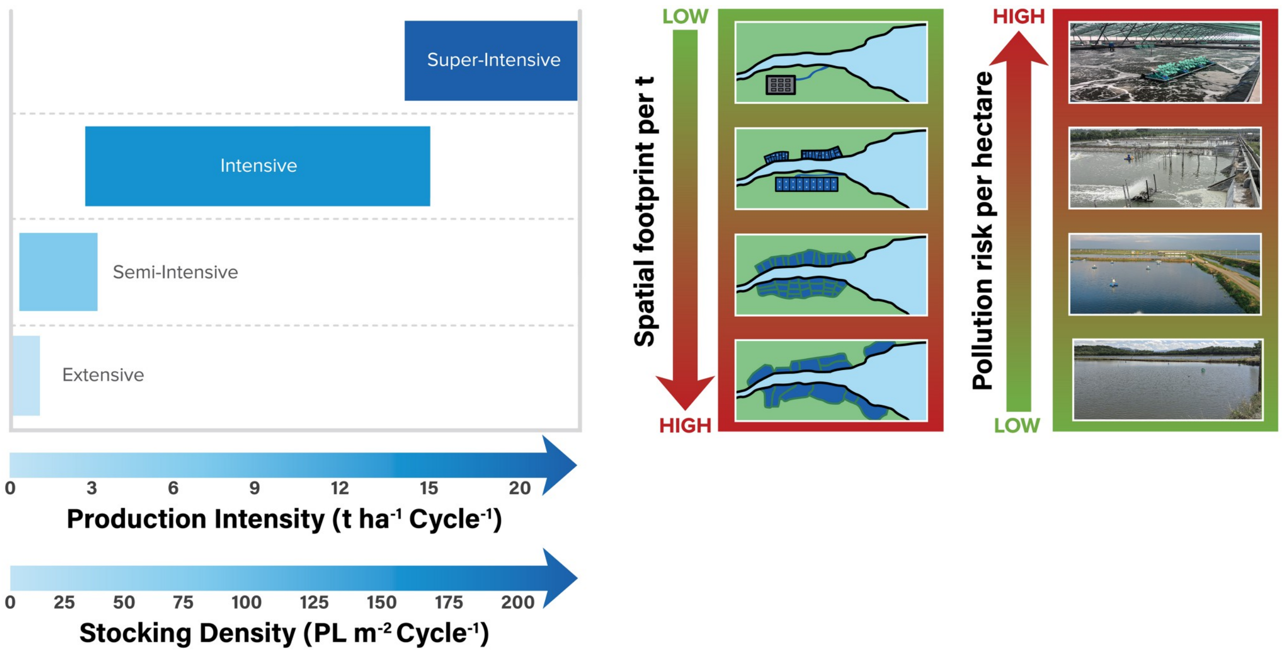

Fig 1. The spectrum of production intensities, expressed as yield in metric tons (t) per ha per cycle and stocking density post-larvae (PL) per m^2 per cycle, associated with four categories of shrimp aquaculture production. Habitat conversion risk per t is relative to the spatial footprint required by each production category. Pollution risk per hectare is relative to the amounts of effluent discharged from production systems. Photo credits from top to bottom: a) Super-intensive indoor shrimp farm in Nha Mat, Bac Lieu, Vietnam. Courtesy of Viet-Uc Seafood Corporation. b) Intensive cement-lined shrimp farm in East Java, Indonesia. c) Semi-intensive earthen shrimp ponds in Guayas, Ecuador. d) Extensive shrimp pond in Guanacaste, Costa Rica.

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uneaten feed or those excreted by the shrimp, can pose an environmental risk when loads are discharged into adjacent coastal waterways without adequate treatment [22]. Feed can contribute 60% to 96% of total nitrogen (TN) and 12% to 95% of total phosphorus (TP) in aquaculture effluents [23–29]. Dissolved nutrients, like an excess of reactive nitrogen, impact the health of coastal ecosystems by disrupting nutrient dynamics, especially in areas with low mixing rates or densely concentrated production regions [30–35]. Settleable solids (SS), a measure of materials that settle from solution in an hour, and total suspended solids (TSS), the weight of filterable solid material in the water column, are together the amount of recoverable particulate matter from effluent through settling or filtration [36], which can impact benthic environments if discharged. Dissolved and particulate effluents from aquaculture have negatively impacted seagrass beds [37], contributed to harmful algal blooms [38], and increased turbidity and sedimentation [39], which have smothered coral reefs [40]. The increased loading of nutrient effluents per unit area of production associated with intensification amplifies the potential for untreated waste to degrade surrounding water bodies and ecosystems [22], such as through acidification, eutrophication, and ecotoxicological impacts [16,41].

The landscape of shrimp aquaculture production is changing to keep up with growing demand for seafood, characterized by increasing production intensification trends [21,42,43] (see [43] for a history of shrimp aquaculture). For example, average production intensity across all of Indonesia, one of the largest shrimp producing countries in Asia [44], increased from 0.17 t ha^{-1} to 1.49 t ha^{-1} between 1986 and 2019 (calculated as production in t divided by shrimp pond area [45,46]). Meanwhile, the real price of farmed shrimp has decreased on average for the last several decades [47], meaning farmers must reduce production costs to remain profitable [43]. Intensification can achieve such cost reductions on a per unit basis [48], largely through improved resource efficiency at the farm level [17]. Increasing production intensity per unit area creates an opportunity to increase overall production yield without expanding direct land requirements [49]. There is an estimated 2.1 million ha of shrimp ponds in production, however the use of high intensity production could meet current demand with just under 43,000 ha, or 2% of today's spatial footprint [50].

Shrimp aquaculture production has seen significant growth in recent decades, but innovative strategies to address nutrient waste from increasingly intensive systems are lagging [51]. Previous studies have evaluated aquaculture effluents systems individually [25,52–60] or compared the general characteristics of systems [51,61], but there is a need to understand system tradeoffs and attributes using uniform inputs. Often, nutrient removal is the primary, or only, characteristic evaluated, but economic and spatial constraints also play a role in farm decision making. Given rapidly changing coastal climate and marine conditions, the shrimp aquaculture sector should consider systems that provide ecosystem services, especially those that can improve coastal adaptation and resilience to natural disturbances [4]. As global demand for shrimp drives more intensive production, and climate change impacts increase in their intensity, there is an opportunity to address both nutrient effluent wastes and climate resilience through the introduction of NbS into shrimp aquaculture effluent treatments.

This study evaluates the potential of conventional and NbS effluent treatment systems for addressing TN and TP from shrimp farm effluent across a spectrum of production intensities. We use secondary data collected from literature review to model the effluent treatment requirements of systems that are prevalent in Asia for shrimp aquaculture. Nutrient waste loading for a range of stocking densities is estimated to compare the treatment efficiency and economic feasibility of conventional and NbS effluent treatment systems for shrimp aquaculture. Effluent treatment systems that convey benefits to farmers and the environment are then highlighted. Results from this study provide a comparison of pollutant removal efficiencies, economic considerations, spatial requirements, ecosystem service provision, and suggested areas of research. These findings provide the conceptual framework for comparing the characteristics and tradeoffs of shrimp aquaculture effluent treatments.

Materials and methods

Research approach

Due to a paucity of publicly available farm-level primary data, this study models shrimp aquaculture effluent waste for four stocking densities based on average *L. vannamei* production parameters [17] and nutrient loading [34] with a focus on production system trends in Asia. The four scenarios include stocking densities of 7 PLm^{-2}, 20 PLm^{-2}, 75 PLm^{-2}, and 120 PLm^{-2} to capture a range of production intensities. Super-intensive ponds systems (e.g., stocking densities of 125 to 252PLm^{-2}) have been reported in Indonesia [62] but are less common and, thus, such a scenario was not included. The modelled TN and TP loads from the four stocking density scenarios are used as inputs for each of the evaluated effluent treatment systems to estimate their pollutant removal effectiveness, farm-level economics, spatial requirements, and ecosystem service provision (Fig 2). Data were drawn from a review of academic, industry, and popular press literature.

Evaluated treatment systems

Effluent treatment systems are categorized as either conventional or NbS if the system is able to "*protect, sustainably manage, or restore natural or modified ecosystems*" [2] (Table 2). Common and emerging effluent treatment systems within both categories were chosen to show a range of potential approaches that have proven some commercial viability. Settling ponds or

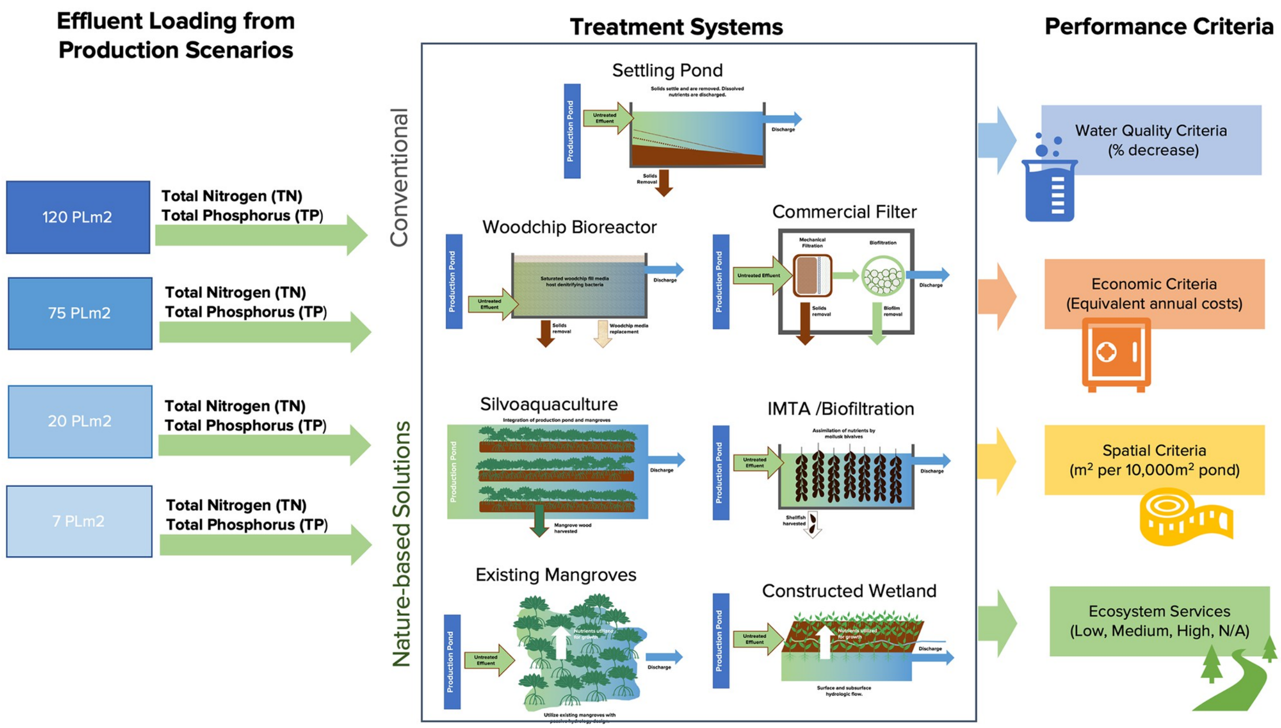

Fig 2. Conceptual analysis diagram for the study. Stocking densities of 7 PLm^{-2}, 20 PLm^{-2}, 75 PLm^{-2}, and 120 PLm^{-2} represent a range of common production intensities in Asia. Effluent, modelled as TN and TP, from each of the stocking densities are used as inputs for conventional and NbS treatment systems. Performance criteria for each treatment system and stocking density combination was evaluated based on water quality, economic, and spatial requirement criterium and ecosystem service provision.

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basins are the most widely used treatment system, and are a requirement for international certifications [63,64]. Commercial filters are typically only used for high- and super-intensive shrimp production [65], while woodchip bioreactors are emerging systems in aquaculture but have proven application in other sectors [66]. Silvoaquaculture, also referred to as mixed mangrove-shrimp farming, is commonly practiced in Vietnam and parts of Indonesia [59]. Biofiltration, or integrated multi-tropic aquaculture (IMTA), as an effluent mitigation approach has been researched [60,67,68], but is limited in its commercial application. The use of existing mangrove forests to treat aquaculture effluent is often a *post facto* decision, where the location and use of the forest is opportunistic, rather than designed [56,69], which poses significant environmental risks [70]. Constructed treatment wetlands can provide many of the same environmental and effluent treatment benefits of existing mangroves, but in a deliberate and designed way for appropriate nutrient loads [71]. Apart from silvoaquaculture (which can include the integration of mangroves within the production pond) the selected treatment systems focus on post-production effluent management, noting that there are a wide range of farm-level practices (e.g., feeding regimes, biofloc, etc.) that can influence water quality but are not within the scope of this study. Table 2 provides a brief description of each system, its mechanism to treat effluent, and prevalence of use in shrimp aquaculture. Additional information and conceptual diagrams provided in Table A in S1 Text.

Data collection and analysis

Effluent composition. Four production intensities, based on stocking densities of 7 PLm^{-2}, 20 PLm^{-2}, 75 PLm^{-2}, and 120 PLm^{-2}, were chosen to represent a spectrum of common

Table 2. Treatment systems categorized by solution type (i.e., conventional or NbS) and their characteristics.

Treatment System	Solution Type	Description	Mechanism	Application	References
Commercial Filters	Conventional	Commercial filtration systems use mechanical filters to remove solid wastes through the physical separation of particle sizes and biofilters to convert toxic nitrogenous waste into non-toxic nitrate via microbial activity.	Microbial biofiltration converts dissolved nitrogen wastes through the breakdown of unionized ammonia to nitrite and then the mineralization of nitrite into nitrate by autotrophic bacteria that colonize a bio-media substrate in a filter, such as a fluidized-sand biofilter [72] and moving bed biofilm reactors.	Widely used in land-based Recirculating Aquaculture Systems (RAS) but are generally cost-prohibitive for pond-based systems.	[54,72–74]
Woodchip Bioreactors	Conventional	Woodchip bioreactors direct effluent through designated carbon-filled trenches that host denitrifying bacteria.	Wood media, often wood chips, enhance the passive treatment of nitrate-nitrogen by hosting microbial biomass [66].	Have been applied in other industries (e.g., to treat agricultural runoff) but are rare in the aquaculture sector.	[55,75–78]
Settling Ponds	Conventional	Settling ponds, or sedimentation basins, are designated areas of a farm that accept effluent discharge from production ponds throughout the production cycle and during harvest.	The physical process of sedimentation is defined by Stoke's Law [79], which calculates the settling velocities of particles suspended in the water, primarily as a function of particle size, density, water temperature, and flow rate [80].	Typical post-production treatment systems for pond aquaculture due to their simple design and ease of use.	[25,52,80,81]
Biofiltration (Integrated Multi-Trophic Aquaculture, IMTA)	Nature-Based Solution	Biofiltration in this context involves nitrogen removal by shellfish when incorporated into shellfish biomass (e.g tissue and shells) and removed during harvest [82].	IMTA has a broad range of applications but can be defined as growing species from two or more trophic levels in the same production area where the wastes from one species provides the nutrient inputs for another species [83, 84].	Gaining popularity in general, but its application for biofiltration to treat effluent is still experimental.	[82–85]
Silvoaquaculture	Nature-Based Solution	Silvoaquaculture integrates mangrove forestry and the cultivation of aquatic species [59]. These systems are primarily extensive but can be semi-intensive when supplemented with hatchery grown seed.	Low stocking densities in these systems usually require few additional inputs, meaning differences in water quality of production ponds and surrounding areas are not significantly different. Water quality changes with the addition of feed and can be adversely affected by decomposing leaf litter [59].	Widely used in Vietnam with limited use in other production geographies.	[59,86,87]
Constructed Treatment Wetlands	Nature-Based Solution	Constructed wetlands can use either, or a combination of, surface and subsurface hydrologic flow to mitigate nutrients and accumulate solids through the use of geomorphic design and vegetation [71].	Constructed wetlands treat wastewater from multiple sources [88] where nutrient removal occurs as a result of biotic and abiotic processes through substrate media, vegetation (macrophytes), diverse microbial communities, and other chemical processes.	There are a wide range of wastewater applications but their use in the tropics, and with coastal species such as mangroves, have largely been focused on field experiments and trials.	[71,89–91]
Existing Mangrove Forests	Nature-Based Solution	Mangrove forests for wastewater treatment utilize existing, matured vegetation and a passive hydrology design.	The nutrient remediation processes in mangrove wetlands include sedimentation, decomposition of organic matter, assimilation of nutrients by plants, bacteria nitrification and dentification, and ion absorption by soil compounds [92].	Studies have evaluated the impacts of aquaculture effluent on mangrove forests, but few have evaluated them as an effluent management tool.	[56,69,93]

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production scenarios to model TN and TP effluent waste loads. Generalized production parameters for *L. vannamei* were used to estimate yield, Y (t ha^{-1} yr^{-1}) (Eq 1) and feed requirements (Eq 1) [17].

$$Y = ([(A \times D \times S \times W) \times 1000^{-1} \times 1000^{-1}] \times C \quad (1)$$

Where A = pond area (m^2), D = stocking density (PL m^{-2}), S = survival (%), W = shrimp weight at harvest (g), and C = harvest cycles (# yr^{-1}). Total feed requirements, F (t yr^{-1}), can be estimated by multiplying yield by an average feed conversion ratio (FCR) as described in Eq 2.

$$F = Y \times FCR \quad (2)$$

Effluent loads for TN (Eq 3) and TP (Eq 4) were estimated assuming an FCR of 1.5 and the difference of nitrogen and phosphorus in feed compared to the weight of harvested biomass [34].

$$N_w = \left[(FCR)\left(N_f\right) - N_c\right] \times 1000 \quad (3)$$

$$P_w = \left[(FCR)\left(P_f\right) - P_c\right] \times 1000 \quad (4)$$

Where N_w and P_w are nitrogen and phosphorus waste loads (kg t^{-1} of cultured species), N_f and P_f are decimal fractions of nitrogen and phosphorus in the feed, and N_c and P_c are the decimal fractions of nitrogen and phosphorus in the live weight of harvested shrimp biomass. Production parameters and assumptions are provided in Table 3. Calculated values for the effluent waste load of TN and TP were used as inputs for each treatment system. Assumptions and detailed calculations provided in S1 Text.

Table 3. Production parameters and assumptions for modelled scenarios [17,34].

Parameter	Unit	Modelled Scenarios		75PLm^{-2}	120PLm^{-2}
		7PLm^{-2}	20PLm^{-2}		
Production Pond Area (this study)	*ha*	1	1	1	1
Production Pond Depth (this study)	*m*	1	1	1	1
Stocking Density (this study)	*PLm^{-2}*	7	20	75	120
Survival	*%*	60	60	70	70
Shrimp Weight at Harvest	*g*	18	18	16	16
Crop Duration	*days/crop*	110	110	80	80
Harvest Cycles	*#yr^{-1}*	2.2	2.2	2.5	2.5
FCR	*kg kg^{-1}*	1.5	1.5	1.5	1.5
Average percentage of nitrogen in whiteleg shrimp	*%*	2.86	2.86	2.86	2.86
Average percentage of phosphorus in whiteleg shrimp	*%*	0.32	0.32	0.32	0.32
Air-dry concentration of phosphorus in grower feeds	*%*	5.33	5.33	5.33	5.33
Air-dry concentration of nitrogen in grower feeds	*%*	1.12	1.12	1.12	1.12
Nitrogen waste load	*kg t^{-1}*	37	37	37	37
Phosphorus waste load	*kg t^{-1}*	11.7	11.7	11.7	11.7
Production Yield	*t ha^{-1} yr^{-1}*	1.66	4.75	21	33.6
Feed Requirement	*t yr^{-1}*	2.5	7.13	31.5	50.4
Total Nitrogen, TN (this study)	*kgN ha^{-1} yr^{-1}*	62	176	777	1,243
Total Phosphorus, TP (this study)	*kgP ha^{-1} yr^{-1}*	19	56	246	393

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Water quality criteria. Nutrient and suspended solids removal rates, expressed as a percent decrease from influent concentrations, are reported as minimum and maximum values based on a literature review. Removal rates can vary across and within treatment types due to biotic and abiotic factors and as a function of hydraulic residence time, or how long the effluent is undergoing the decomposition processes. Commercial filters target the removal of total ammonia nitrogen through an effective nitrification process [54], which resulted in reporting of total ammonia nitrogen removal rates being more common than TN in the literature. TSS was also included as a key indicator of water quality improvement, although reporting on TSS in the literature was less common than for TN and TP. This study assigns pollution reduction thresholds of at least 40% for TN and TP, and at least 70% reduction for TSS based on Low Impact Development and Best Management Practice approaches [94], but actual biophysical thresholds will depend on the conditions and assimilative capacity of the receiving water body.

Aquaculture certification standards, such as those developed by the Aquaculture Stewardship Council (ASC) and Global Seafood Alliance's Best Aquaculture Practices (BAP), set effluent pollution limits and require farm-level management practices to decrease the risk of eutrophication in receiving water bodies [63, 64]. These certification bodies set effluent limits with allowances of nutrients above that of receiving bodies, recognizing the difficulty for farmers to return discharged wastewater to ambient conditions. The impact of such an effluent allowance will differ depending on the conditions of the receiving water body, and thus, certification standards require compliance with a comprehensive list of precautionary water quality parameters and management practices to limit cumulative impacts.

The ASC shrimp standard limits annual effluent to 25.2kgTN and 3.9kgTP for *L. vannamei* on a per t of harvested shrimp basis (Criterion 7.5.1 and 7.5.2) [63]. Conversely, the BAP standard sets effluent concentration limits of less than 5 mg/L total ammonia nitrogen and less than 0.5 mg/L soluble phosphorus (BAP, Appendix B) or compliance with other effluent criteria that can demonstrate limited negative impacts to receiving water bodies (Pillar 3.C. 3.3.2–6) [64]. Mass loading for nitrogen and phosphorus are required to be recorded by auditors (Pillar 3.B. 3.1), but only provisional targets have been set at 15kg total ammonia nitrogen t^{-1} and 1kg soluble phosphorus t^{-1} (BAP, Appendix A). In addition to nitrogen and phosphorus criteria, both ASC and BAP include guidance for other water quality and monitoring criteria to reduce negative impacts on receiving water bodies, such as suspended solids thresholds, limiting changes to dissolved oxygen, and biochemical oxygen demand. Effluent concentrations, nutrient loads, and total discharge volume–during the production cycle and at pulse events, such as harvest–are all critical aspects for a comprehensive understanding of effluent management.

Economic criteria. Capital costs and annual operating costs were collected from literature and expert inputs with costs adjusted for inflation to USD2021. Capital costs include a treatment-specific lifetime depreciation on items such as hardware and infrastructure, which are assumed to be 10 years for earthen settling ponds, commercial filters, woodchip bioreactors [55], and IMTA [95,96]. HDPE-lined settling ponds can have shorter lifespans and increased operating costs, compared to earthen ponds, if liners get punctured, torn, or degraded but could be considerably longer with proper maintenance. The effective lifespans for NbS systems are substantially longer. For example, constructed wetlands were conservatively estimated to last 20 years [97] but commonly last up to 30 years [71]. Lifespans for silvoaquaculture and existing mangroves are estimated to be 25 and 50 years, respectively [98,99].

An equivalent annual cost (EAC) for each production scenario within each treatment system was used to compare the cost-effectiveness across treatment systems with unequal lifespans, which is described by the ratio of its net present value to an annuity factor [100]. The

EAC is described in (Eq 5) as:

$$EAC = NPV\left(A_{N,\,k}\right)^{-1} \tag{5}$$

Where k = the discount rate, here assumed to be 11.73% for crustacean aquaculture in developing countries [101], N = the treatment-specific economic lifespan of the asset in years, and $A_{N,\,k}$ = the present annuity factor. The annuity factor can be calculated as (Eq 6):

$$A_{N,\mathrm{k}} = \left(1 - \left(1*(1+k)^{-N}\right)\right)k^{-1} \tag{6}$$

Simply, the initial capital expenses for the asset are divided by the annuity factor and then added to the expected annual operating costs. Calculated EAC values were then divided by the estimated shrimp yield in kg to standardize outputs for each production scenario (See Table B in S1 Text for complete outputs). This model assumes a break-even earnings before interest and taxes (EBIT) margin of 16% [13]. Using a market price of USD 4.47 per kg [102], a value of USD 0.18 per kg (4% of EBIT) was selected as the threshold EAC per kg of shrimp, allowing for an EBIT margin of 12%.

Spatial criteria. Nutrient remediation capacities, typically estimated in terms of mass per unit area per unit time, were collected through literature review for each of the treatment systems. Nutrient loading rates in Table 3 were used as inputs to determine the total area required for each of the modelled scenarios. Spatial requirements for constructed treatment wetlands used a first-order plug flow kinetic model [58], which required an estimation of influent nutrient concentrations (i.e., into the treatment system) based on total nutrient load, daily water exchange, and pond volumes (detailed calculations in S1 Text—Supplementary Methods). The spatial footprint requirements ranged widely across treatment systems and have been reported as the area, in square meters, proportional to 1 ha of shrimp production pond (i.e., 1,000m^2 = 0.1ha:1ha and 25,000m^2 = 2.5ha:1ha treatment area to pond ratio). Commercial filters were only evaluated at the 120 PLm^{-2} scenario due to the technical and economic requirements of those systems. Silvoaquaculture is defined by extensive culture and was only evaluated for the 7 PLm^{-2} scenario. Treatment systems were categorized as above or below a threshold of 0.6ha based on the area required for a generic sedimentation basin with a hydraulic retention time of 6 hours [52,103]. The hydraulic retention time allows course and medium solids to settle and is calculated as basin volume divided by the incoming flow rate but does not factor in nutrient loads or concentrations. Assuming a basin depth of 1m, and that maximum effluent inflow occurs during harvest over a 10-hour period at a rate of 1,000 m^3 hr^{-1}, the calculated basin area would require 0.6ha.

Ecosystem services. Environmental and social benefits delivered by treatment systems beyond their intended water quality improvements are captured as ecosystem services. Ratings for provisioning services, regulating services, cultural services, and supporting services under different wetland ecosystem types were collected from Ramsar's Global Wetland Outlook 2018, Table 2.7 [104,105]. Ratings for several of the evaluated wetland ecosystems were directly transferable to the treatment systems in the study (i.e., settling ponds = "waste ponds", existing mangrove forest = "Mangrove", biofiltration = "Shellfish Reef"), while assigning hybrid ratings for treatment systems not directly transferable (i.e., silvoaquaculture = "Mangroves" and "Aqua Ponds" and constructed treatment wetlands = "Mangrove" and "Salt Marsh"). Commercial filters and woodchip bioreactors are treatment systems that do not provide ecosystem services. 'High', 'Medium', and 'Low' ratings given in the Global Wetland Outlook table were assigned scores of 3, 2, and 1, based on each systems' ability to deliver on specified ecosystem services, respectively (ratings of 'Not known' and 'Not applicable' given scores of zero). Provisioning services and regulating services had five sub-categories with a possible score of 15 each, while cultural services and supporting services each had four sub-categories with a potential score of 12 each. Scores for each treatment system and across service types were summed and their cumulative scores assigned to one of three equal bins ('High' = 37 to 54, 'Medium' = 19 to 36, 'Low' = 0 to 18) or 'NA' categories. Treatment systems within the 'Low' or 'NA' bins were categorized as below the threshold to deliver ecosystem services. Tabulated scores for all ecosystem services in Table C in S1 Text.

Normalization and aggregation of performance indicators. Overall treatment system performance for each production intensity scenario was compared by normalizing values within specific indicators (e.g., TN, TP, TSS, and Ecosystem Services) where higher values indicated preferred directionality (Eq 7).

$$Z_i = X_i X_{max}^{-1} \tag{7}$$

Where Z_i = the normalized indicator value, X_i = the actual indicator value expressed numerically (e.g., 46% = 0.46), and X_{max} = the maximum value expressed numerically of all values within the indicator category. For the economic criteria and spatial requirement criteria, where the preferred directionality would be to have lower values, normalizing values included taking the absolute value of one minus $X_i X_{max}^{-1}$ (Eq 8).

$$Z_i = |1 - (X_i X_{max}^{-1})| \tag{8}$$

These normalized values for each treatment system's performance criteria indicated that lower values have lower performance, and that higher values achieve higher performance. With normalized scores being a numerical vector between 0 to 1.00, a color gradient was applied to emphasize relative scoring for the evaluated indicator under each scenario. Each performance indicator (e.g., TN, TP, TSS, EAC, m^2, and Ecosystem Services) was weighted equally and multiplied by its normalized score to determine an aggregate indicator for the treatment system-production scenario combinations. However, in practice, the weights for specific criteria would be different depending on the location, farmer, production system, or other factors. This wide diversity of potential preferences suggests that no single combination of weighting performance indicators would be able to adequately characterize shrimp aquaculture in Asia. Thus, assigning equal weights to all the performance criteria may not accurately reflect the relative importance of each indicator as this would be context-dependent and vary by stakeholder preferences. Treatment systems were then ranked by their additive aggregation indicators e.g., the sum of their normalized scores with equal weights. Threshold values for each criterion were also normalized and ranked by their aggregation indicators.

Study limitations. A lack of available farm-level effluent data from commercial operations, especially from more intensive production systems, was a major barrier for this study. Reports of effluent water quality data from commercial shrimp farms were more prevalent in older studies and at lower production intensities. Due to the wide variety of potential nutrient inputs and pond-level management practices across geographies, this study utilized hypothetical scenarios under uniform conditions to evaluate treatment systems. Audit reports on water quality are available for ASC certified farms globally, requiring reports on kgN, kgP, and settleable solids. While informative, these reported effluent loads are those that are being discharged into surrounding waterways and have already undergone some level of effluent treatment and water quality management to achieve ASC's effluent standards. Additionally, predicting TSS was sparse and only found in one incidence, where a "rule of thumb" suggested that suspended solids could be estimated using 25% of the fed quantity of feed [106]. Overall, reporting on TSS was less common than for TN and TP in literature. Improved monitoring, data collection, and transparency of current farm-level effluent loading would help to inform management practices that benefit the environmental sustainability of the sector.

Results

Estimated nutrient loading

Modelled nutrient waste loading scenarios for 7 PLm^{-2}, 20 PLm^{-2}, and 75 PLm^{-2} fell within ranges observed in literature, although reported values of TN and TP effluent per unit area (kg ha^{-1} yr^{-1}) and per t of shrimp varied considerably across studies (Table 4). Higher stocking densities of 125 PLm^{-2} and 252 PLm^{-2} [62], and up to 750, 1000, and 1200 PLm^{-2} [107] were documented in Indonesia, but these studies did not record nutrient effluent loads. The results from Table 4 highlight the wide range of effluent loads collected from different production systems and under varying management regimes. As production intensifies, total nutrient discharge (e.g., kgN ha^{-1} yr^{-1}) would be expected to increase, while nutrient discharge per unit weight of shrimp (e.g., TN kg^{-1}) would generally decrease. While these trends can loosely be observed, there are many cases reported in the literature where values are far from the anticipated range. These findings further emphasize the complexity of aquaculture effluents under real-world conditions.

Treatment system performance

This analysis provides a preliminary comparison of shrimp aquaculture effluent treatment systems across a spectrum of production intensities (Table 5). Ranges of pollution removal collected from literature, expressed as a decrease from initial condition, varied widely within each treatment system and across system types. Constructed mangrove wetlands and existing mangrove forests, types of NbS systems, showed cost-competitiveness with settling ponds based on EAC kg^{-1} values. While NbS systems require much larger spatial footprints, especially at higher production intensities, these areas also provide additional ecosystem services beyond their intended water quality improvements. Target thresholds for each criterion were identified to help define the relative performance under the various scenarios. Although actual pollutant removal would be highly dependent on farm-level design and operation of effluent treatment systems, the economic and environmental results across the production intensity scenarios suggest that NbS can be a viable aquaculture effluent solution.

Water quality criteria

Nutrient and sediment removal rates varied widely, and no system was found to consistently remove 100% of TN, TP, or TSS, suggesting that treatment system design and proper use may be more important factors than system type (e.g., conventional or NbS). Constructed treatment wetlands achieved removal rates above the minimum threshold for all three pollutants, while existing mangrove forests and commercial filters had lower bounds above the threshold for TN (>40%) and TSS (>70%). The lower bound removal rates of the remaining treatment systems were below the minimum threshold for two or more pollutants.

Comparing modelled effluent nutrient loads in this study to ASC certified farms [111] illuminates the potential remediation requirements necessary to achieve certification (Table 6). A modelled nutrient waste load for *L. vannamei*, based on the difference between nutrients provided as feed and incorporated into shrimp biomass, found that average loading was 37.0kgN per t and 11.7kgP per t [34]. This output suggests that average farmers would need to reduce their nitrogen and phosphorus loads by 32% and 67%, respectively, to meet the ASC certification standards of 25.2kgN per t and 3.9kgP per t.

Table 4. Comparisons of effluent waste loading of TN and TP across a range of stocking densities and production yields. Waste load kg shrimp^{-1} calculated by dividing total kg ha^{-1} yr^{-1} by yield (t).

Stocking Density	Yield	Effluent Waste Loading		TP		Reference
		TN				
PL m^{-2}	*t ha^{-1} yr^{-1}*	*kgN ha^{-1} yr^{-1}*	*TN kg shirmp^{-1}*	*kgP ha^{-1} yr^{-1}*	*TP kg shirmp^{-1}*	
4	1.0	128.0	128.0	40.0	40.0	[56]
4	1.0	128.0	128.0	-	-	[108]
6	0.2	220.7	1036.0	28.4	133.2	[109]
6	0.3	250.1	729.3	31.5	91.7	[109]
6	0.4	203.9	504.8	29.7	73.5	[109]
6	0.9	152.5	169.4	25.3	28.1	[109]
7	0.3	78.3	260.8	27.3	90.8	[26]
7	0.5	216.3	468.1	44.4	96.2	[109]
7	1.0	141.2	141.1	14.3	14.3	[109]
7	1.7	62.0	37.0*	19.0	11.7*	This study
8	0.2	215.8	985.5	50.2	229.0	[109]
8	0.6	214.3	338.5	28.8	45.5	[109]
8	0.6	265.3	457.4	43.8	75.5	[109]
9	0.8	298.1	386.6	22.2	28.8	[109]
10	0.1	189.0	1592.1	27.7	233.5	[109]
10	0.5	176.2	381.4	15.6	33.8	[109]
10	0.5	193.2	411.9	14.7	31.4	[109]
10	0.6	304.0	492.8	30.8	49.9	[109]
11	0.1	187.7	1524.5	26.4	214.5	[109]
13	0.9	186.5	207.2	49.8	55.3	[26]
13	3.0	34.1	11.3	-	-	[108]
17	5.8	59.1	10.3	-	-	[108]
19	7.8	90.5	11.5	-	-	[108]
20	-	190.8	-	8.5	-	[110]
20	-	214.3	-	7.8	-	[110]
20	4.8	176.0	37.0*	56.0	11.7*	This study
21	-	177.0	-	7.8	-	[110]
24	6.4	73.8	11.6	-	-	[108]
52	13.8	199.0	14.4	39.0	2.8	[56]
52	13.8	199.0	14.4	-	-	[108]
58	2.0	238.5	119.3	82.5	41.3	[26]
68	21.3	934.5	44.0	302.5	14.2	[27]
71	12.1	934.5	77.4	302.5	25.1	[27]
71	17.8	934.5	52.6	302.5	17.0	[27]
71	19.4	934.5	48.3	302.5	15.6	[27]
75	21.0	777.0	37.0*	246.0	11.7*	This study
120	33.6	1243.0	37.0*	393.0	11.7*	This study

Notes: [108] reports dissolved inorganic nitrogen (DIN) = ([NO2 + NO3] + [NH4]), [109] includes nutrient load from water exchange and pond drainage and calculated assuming 2.2 cycles per year, [110] calculated assuming 2.5 cycles per year, [27] includes nutrient load as total load minus shrimp harvest and calculated assuming 2.5 cycles per year.
*TN and TP load t-shrimp^{-1} for this study taken from [34].

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Table 5. Water quality improvements, equivalent annual costs per kg of harvested shrimp, spatial requirements, and ecosystem services associated with every treatment option across each production scenario. Water quality criteria are expressed as a percentage decrease from initial condition. Equivalent annual costs assess the discounted NPV over the lifespan of each treatment per kg of harvested biomass. Spatial criteria are expressed as the area in m^2 necessary to treat 1 ha of production pond, where 1 ha = 10,000m^2. Select ecosystem services are presented with service value (full assessment in Table C in S1 Text). Threshold target values, shaded green, are removal rates of at least 40%TN, 40%TP, and 70%TSS, EAC less than \$0.18 kg^{-1}, spatial requirement less than 10,000m^2 (1ha), and at least 'Medium' ecosystem service provision. Values below targets are orange and cross-hatched cells indicate 'not applicable'.

Treatment System Type	Treatment System (Stocking Density)	Water Quality Criteria			Economic Criteria	Spatial Criteria	Ecosystem Services				
		Percent decrease from initial condition			Equivalent Annual Cost USD kg^{-1}	Area Required m^2:1 ha of production (1ha = 10,000m^2)	Categorized as providing High, Medium, or Low service provision				
		TN	*TP*	*TSS*	*USD kg^{-1}*	*m^2*	*Cultural Services*	*Regulating Services*	*Provisioning Services*	*Supporting Services*	*Cumulative Score*
Conventional	Commercial Filters (7PL m^{-2} = per m^2)	NA			NA	NA	NA	NA	NA	NA	NA
	Commercial Filters (20PL m^{-2} = per m^2)	NA			NA	NA					
	Commercial Filters (75PL m^{-2} = per m^2)	NA			NA	NA					
	Commercial Filters (120 PL m^{-2} = per m^2)	43–91%	15–84%	92%	0.99	30					
	Woodchip Bioreactors (7PL m^{-2} = per m^2)	49–71%	15–55%	64–93%	0.32	10	NA	NA	NA	NA	NA
	Woodchip Bioreactors (20PL m^{-2} = per m^2)				0.28	20					
	Woodchip Bioreactors (75PL m^{-2} = per m^2)				0.24	100					
	Woodchip Bioreactors (120 PL m^{-2} = per m^2)				0.22	140					
	Settling Ponds (7PL m^{-2} = per m^2)	20–31%	22–55%	60–88%	0.26	70	L	L	L	M	L
	Settling Ponds (20PL m^{-2} = per m^2)				0.11	190					
	Settling Ponds (75PL m^{-2} = per m^2)				0.04	820					
	Settling Ponds (120 PL m^{-2} = per m^2)				0.04	1,320					

(*Continued*)

Table 5. (Continued)

Treatment System Type	Treatment System (Stocking Density)	Water Quality Criteria			Economic Criteria	Spatial Criteria	Ecosystem Services				
		Percent decrease from initial condition			Equivalent Annual Cost USD kg^{-1}	Area Required m^2:1 ha of production (1ha = 10,000m^2)	Categorized as providing High, Medium, or Low service provision				
		TN	*TP*	*TSS*	*USD kg^{-1}*	*m^2*	*Cultural Services*	*Regulating Services*	*Provisioning Services*	*Supporting Services*	*Cumulative Score*
Nature-based Solutions	Biofiltration (IMTA*) (7PL m^{-2} = per m^2)	10–34%	10–44%	10–71%	2.12	1,300	L	L	L	L	L
	Biofiltration (IMTA*) (20PL m^{-2} = per m^2)				2.02	3,540					
	Biofiltration (IMTA*) (75PL m^{-2} = per m^2)				2.19	14,920					
	Biofiltration (IMTA*) (120 PL m^{-2} = per m^2)				1.85	22,830					
	Silvoaquaculture (7PL m^{-2} = per m^2)	2–53%	(-6)-46%	NA	0.39	6,670	L	H	M	M	M
	Silvoaquaculture (20PL m^{-2} = per m^2)	NA			NA	NA	NA				
	Silvoaquaculture (75PL m^{-2} = per m^2)	NA			NA	NA	NA				
	Silvoaquaculture (120 PL m^{-2} = per m^2)	NA			NA	NA	NA				
	Constructed (Mangrove) Wetlands (7PL m^{-2} = per m^2)	46–92%	60–100%	70–95%	0.25	2,710	L	H	M	M	M
	Constructed (Mangrove) Wetlands (20PL m^{-2} = per m^2)				0.17	5,370					
	Constructed (Mangrove) Wetlands (7PL m^{-2} = per m^2)				0.09	12,290					
	Constructed (Mangrove) Wetlands (120 PL m^{-2} = per m^2)				0.06	13,430					
	Existing Mangrove Forests (7PL m^{-2} = per m^2)	43–50%	28–48%	95%	0.00	630	L	H	M	H	M
	Existing Mangrove Forests (20PL m^{-2} = per m^2)				0.00	1,800					
	Existing Mangrove Forests (75PL m^{-2} = per m^2)				0.00	7,960					
	Existing Mangrove Forests (120 PL m^{-2} = per m^2)				0.00	12,740					
	* IMTA generates additional revenue through secondary, or even tertiary, products which were not factored into the calculation. However, these additional products may also incur additional capital and operating costs.										
			TN and TP >40%, TSS >70%		EAC kg^{-1} < \$0.18	Area < 0.6ha		Medium or High Ecosystem Service Score			
			TN and TP <40%, TSS <70%		EAC kg^{-1} > \$0.18	Area > 0.6ha		Low Ecosystem Service Score			
		NA	Not applicable		NA Not applicable	NA Not applicable	NA	Not Applicable			

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Table 6. Comparisons of effluent waste loading of TN and TP across a range of production yields from ASC certified farms. Annual waste load (kg ha^{-1} yr^{-1}) calculated by multiplying yield (t) and nutrient load (kg t-shrimp^{-1}). ASC certification standards require less than 25.2kgTN t^{-1} and less than 3.9kgP t^{-1}. Table data from [111] and rearranged in order of production yield.

Stocking Density	Yield	Effluent Waste Loading from ASC Certified Farms			
		TN		TP	
PL m^{-2}	*t ha^{-1} yr^{-1}*	*kgN ha^{-1} yr^{-1}*	*kgN t-shirmp^{-1}*	*kgP ha^{-1} yr^{-1}*	*kgP t-shrimp^{-1}*
-	1.6	0.0	0.0	0.0	0.0
7*	1.7	62.0	37.0	19.0	11.7
-	3.1	56.2	18.2	6.5	2.1
-	3.7	32.1	8.7	6.6	1.8
-	3.8	45.3	11.9	4.5	1.2
-	4.7	47.4	10.1	1.9	0.4
20*	4.8	176.0	37.0	56.0	11.7
-	5.2	0.5	0.1	0.3	0.1
-	16.6	306.2	18.5	42.0	2.5
-	19.0	17.1	0.9	1.3	0.1
75*	21.0	777.0	37.0	246.0	11.7
-	26.5	392.8	14.8	56.5	2.1
-	29.2	447.1	15.3	47.6	1.6
120*	33.6	1243.0	37.0	393.0	11.7
-	43.6	558.5	12.8	83.8	1.9

* Indicates modelled scenarios and outputs from this study.

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Equivalent annual costs per kg

NbS treatment systems showed cost-competitiveness against conventional systems based on EAC kg^{-1} values. Settling ponds, constructed wetlands, and existing mangrove forests generally had EAC values below the USD 0.18 kg^{-1} threshold, while the low output of the 7 PL m^{-2} scenario resulted in EAC values exceeding the threshold. As production yields and revenues grow with intensification, the capital and operating costs for treatment systems decrease on a per kg basis, indicated by falling EAC kg^{-1} values as stocking density increases. However, the high capital and operating costs of the commercial filters in the 120 PL m^{-2} scenario exceeded the threshold of USD 0.18 kg^{-1}. The use of existing mangrove forests requires little capital costs compared to other treatment systems as the mechanism for nutrient retention would already be in place. Paired with low annual costs to monitor and maintain the forest, overall EAC values for existing mangrove forests are minimal.

Area requirements

Conventional effluent treatment systems demand a smaller spatial footprint compared to NbS systems, which require large areas for complex biotic and abiotic processes necessary for nutrient remediation. Conventional treatment systems fell well below the 0.6ha (6,000m^2) threshold, with most scenarios only requiring tens or hundreds of square meters per hectare of production. NbS systems in the 7 PLm^{-2} and 20 PLm^{-2} scenarios required land area below the 0.6ha threshold but exceeded this threshold as production intensity and nutrient output increased. IMTA biofiltration, constructed mangrove treatment wetlands, and existing mangrove forests require 2.28ha, 1.34ha, and 1.27ha, respectively, per 1ha of production pond in the 120 PLm^{-2} scenario. Silvoaquaculture exceeds the 0.6ha threshold in the 7 PLm^{-2} scenario where there is a fixed ratio of 60% production pond to 40% mangrove area (i.e., 16,670m^2 total area results in 10,000m^2 production area and 6,670m^2 treatment area).

Ecosystem services

NbS systems, by definition, deliver more ecosystem services than conventional systems by integrating nature and natural processes into effluent treatment. Commercial filters and woodchip bioreactors are not ecosystems and, thus, do not provide additional ecosystem services but are systems that convey environmental benefits through their water quality improvements. Meanwhile, settling ponds have been categorized as types of "human-made wetlands" that are able to provide some ecosystem services [105]. Existing mangrove forests received the highest cumulative score (35 out of 54) with silvoaquaculture and constructed mangrove wetlands at 31 and 30, respectively. The degree to which specific treatment systems can deliver on ecosystems service sub-categories would be context dependent with larger, contiguous parcels likely provided improved service provision over smaller, fragmented parcels.

Normalization and aggregation of performance indicators

While identifying whether specific performance criteria are above or below a given threshold is useful when trying to evaluate specific attributes, it does little to provide a comparison of treatment systems as a whole. Normalizing values across a vector between 0 and 1.00 within each performance indicator allows for relative comparisons when units vary across performance indicators. An aggregated indicator value, using the normalized criterion value and equal weighting for all indicators, provides a single output that is used to rank the conventional and NbS treatment systems for production intensities of 7 PLm^{-2} (Table 7), 20 PLm^{-2} (Table 8), 75 PLm^{-2} (Table 9), and 120 PLm^{-2} (Table 10). Although values within specific

Table 7. Normalization and aggregation of treatment system indicators at a stocking density of 7 PL m^{-2}.

Treatment Systems	Treatment System Type	Mean Value	TN	TP	TSS	EAC	m^2	Ecosystem Services
Existing Mangrove Forests	NbS	0.73	0.45	0.29	1.00	1.00	0.97	0.66
Constructed (Mangrove) Wetlands	NbS	0.71	0.48	0.63	0.74	0.89	0.88	0.66
Threshold		0.65	0.42	0.42	0.74	0.92	0.74	0.66
Settling Ponds	Conventional	0.55	0.21	0.23	0.63	0.88	1.00	0.33
Woodchip Bioreactors	Conventional	0.53	0.52	0.16	0.67	0.85	1.00	0.00
Silvoaquaculture	NbS	0.37	0.02	0.00	0.00	0.82	0.71	0.66
Biofiltration (IMTA)	NbS	0.27	0.11	0.11	0.11	0.03	0.94	0.33
Commercial Filters	Conventional	NA	NA	NA	NA	NA	NA	NA

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Table 8. Normalization and aggregation of treatment system indicators at a stocking density of 20 PL m^{-2}.

Treatment Systems	Treatment System Type	Mean Value	TN	TP	TSS	EAC	m^2	Ecosystem Services
Existing Mangrove Forests	NbS	0.72	0.45	0.29	1.00	1.00	0.92	0.66
Constructed (Mangrove) Wetlands	NbS	0.70	0.48	0.63	0.74	0.92	0.76	0.66
Threshold		0.65	0.42	0.42	0.74	0.92	0.74	0.66
Settling Ponds	Conventional	0.56	0.21	0.23	0.63	0.95	0.99	0.33
Woodchip Bioreactors	Conventional	0.54	0.52	0.16	0.67	0.87	1.00	0.00
Biofiltration (IMTA)	NbS	0.26	0.11	0.11	0.11	0.08	0.84	0.33
Commercial Filters	Conventional	NA	NA	NA	NA	NA	NA	NA
Silvoaquaculture	NbS	NA	NA	NA	NA	NA	NA	NA

https://doi.org/10.1371/journal.pstr.0000076.t008

Table 9. Normalization and aggregation of treatment system indicators at a stocking density of 75 PL m^{-2}.

Treatment Systems	Treatment System Type	Mean Value	TN	TP	TSS	EAC	m^2	Ecosystem Services
Existing Mangrove Forests	NbS	0.68	0.45	0.29	1.00	1.00	0.65	0.66
Constructed (Mangrove) Wetlands	NbS	0.66	0.48	0.63	0.74	0.96	0.46	0.66
Threshold		0.65	0.42	0.42	0.74	0.92	0.74	0.66
Settling Ponds	Conventional	0.56	0.21	0.23	0.63	0.98	0.96	0.33
Woodchip Bioreactors	Conventional	0.54	0.52	0.16	0.67	0.89	1.00	0.00
Biofiltration (IMTA)	NbS	0.17	0.11	0.11	0.11	0.00	0.35	0.33
Commercial Filters	Conventional	NA	NA	NA	NA	NA	NA	NA
Silvoaquaculture	NbS	NA	NA	NA	NA	NA	NA	NA

https://doi.org/10.1371/journal.pstr.0000076.t009

Table 10. Normalization and aggregation of treatment system indicators at a stocking density of 120 PL m^{-2}.

Treatment Systems	Treatment System Type	Mean Value	TN	TP	TSS	EAC	m^2	Ecosystem Services
Threshold		0.65	0.42	0.42	0.74	0.92	0.74	0.66
Constructed (Mangrove) Wetlands	NbS	0.65	0.48	0.63	0.74	0.97	0.41	0.66
Existing Mangrove Forests	NbS	0.64	0.45	0.29	1.00	1.00	0.44	0.66
Settling Ponds	Conventional	0.55	0.21	0.23	0.63	0.98	0.94	0.33
Woodchip Bioreactors	Conventional	0.54	0.52	0.16	0.67	0.90	0.99	0.00
Commercial Filters	Conventional	0.52	0.45	0.16	0.97	0.55	1.00	0.00
Biofiltration (IMTA)	NbS	0.13	0.11	0.11	0.11	0.16	0.00	0.33
Silvoaquaculture	NbS	NA	NA	NA	NA	NA	NA	NA

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indicators may be above or below a given threshold, existing mangrove forests and constructed mangrove wetlands consistently ranked as the treatment systems with the top overall aggregated values. In all cases, except the 120 PLm^{-2} scenario, these types of treatment systems were ranked above the threshold value.

The ranking yields a single system with the highest aggregate value, however, the purpose of this study is to illicit a short-listed set of options that practitioners can use for further investigation [112]. Caution should be taken when comparing removal rates, economic factors, spatial requirements, and ecosystem service provision of different treatments due to the model's dependence on specific conditions and environmental characteristics drawn from literature, which will differ depending upon the characteristics of the site, species of study, and cultivation practices being employed within and across production systems.

Discussion

This analysis provides theoretical economic and environmental assessments of shrimp farm effluent treatment systems to compare their costs and benefits across a range of production scenarios. The conceptual approach of modelling these scenarios is intended to provide the foundation for further evaluation and would benefit from *in situ* farm data to better compare conventional and NbS treatment systems at commercial production levels.

Estimated nutrient loading

Nutrient effluents in shrimp aquaculture are commonly reported as a mass load or concentration, but either alone has limitations for assessing potential environmental impacts [36]. For example, mass loading (e.g., kg of nutrient ha^{-1} or t^{-1}) does not capture discrete time periods, such as pulses of effluent associated with complete pond drainage while harvesting shrimp. Conversely, while concentrations (e.g., mg/L or ppm) are important to understand nutrient solutes at a given point in time, they are less accurate in assessing total nutrients leaving the production system if samples are used to estimate average concentrations over the production cycle. Nutrient concentrations are also a function of farm-level water management, such that increased flushing or more frequent water exchange can decrease nutrient concentrations in effluent but could still be accounted for in total mass. In practice, water quality management should be informed by nutrient loads, nutrient concentrations, and water use throughout the production cycle and harvest for a comprehensive understanding of localized effluent management.

Reported nutrient loading for shrimp aquaculture in the literature tended to be from older studies and lower stocking densities. While our TN (62 kgN ha^{-1} yr^{-1}) and TP (19 kgP ha^{-1} yr^{-1}) loading for the 7PLm^{-2} scenario was plausible given the range of previously reported values at the same stocking density (78–216 kgN ha^{-1} yr^{-1} and 14–44 kgP ha^{-1} yr^{-1}) [26,109], both were towards the lower bounds. TN loading at the 20 PLm^{-2} scenario was also at the lower end of reported values (177–214 kgN ha^{-1} yr^{-1}) [110], but the modelled TP loading of 56 kgP ha^{-1} yr^{-1} was much higher than the 7.8–8.5 kgP ha^{-1} yr^{-1} found in a previous study [110]. However, TP loading of 50 kgP ha^{-1} yr^{-1} was reported for a stocking density of 13 PLm^{-2} [26], indicating a wide possible range under different production conditions. A stocking density of 71 PLm^{-2} with effluent loading of 935 kgN ha^{-1} yr^{-1} and 303 kgP ha^{-1} yr^{-1} [27] was the closest to compare to the 75 PLm^{-2} scenario, which was found to be slightly lower at 777 kgN ha^{-1} yr^{-1} and 246 kgP ha^{-1} yr^{-1}, respectively. Effluent loading approaching 120 PLm^{-2} was not found in published literature. While audit data from ASC certified farms are available and have been published [111], the reported values have already undergone some level of effluent treatment, and thus, wouldn't be appropriate for input values for the evaluated treatment systems.

Previous studies have utilized a nutrient mass balance approach to estimate total effluent loading [23,24,26,28,110], accounting for all nutrient inputs (e.g., feed, influent water, fertilizers, earthen ponds, stocked shrimp, etc.) and nutrient outputs (e.g., effluent water, seepage, denitrification, sediments, harvested shrimp, etc.). This approach is useful to gather empirical farm data but is limited in use to model scenarios given the change in input variables, pond chemistry, and resulting output values under a range of production intensities. Another approach, as used in this study, is to estimate nutrient waste loading using FCR and the difference between the decimal fraction of nitrogen or phosphorus in feed minus the decimal fraction of nitrogen or phosphorus in the live weight of the cultured animals [34]. This assumes that feed is the major nutrient input and that the nitrogen and phosphorus not utilized in shrimp biomass is discharged into the environment. Outputs from nutrient effluent modelling, regardless of the methodology, would be better supported with farm-level data and should be used cautiously given the diversity of geographies and production systems that shrimp aquaculture occupies.

Estimating a generic, or "average" nutrient composition of effluent is further complicated by a range of farm-level practices during the production cycle that can directly impact nutrient concentration, nutrient loading, and water use. Feed formulations and ingredient composition affect water quality [113], while macroaggregate formations of microorganisms (biofloc) enhance water quality by uptake of nitrogen compounds generated by microbial growth and competing for resources with pathogens [114]. Additives, like probiotics, protect shrimp from opportunistic pathogens [115] while having bioremediation benefits. These types of management approaches to maintain pond water quality during the production cycle vary widely from one farmer to another and, thus, were not included in the study. Settling ponds are commonly employed for effluent remediation, and have been found to be effective in reducing

TSS, but are less effective for reducing TN due to dissolved nitrogen compounds that do not settle–indicating that a more holistic approach to water quality management is necessary to mitigate nutrient discharge [25].

Water quality criteria

Identifying a universal metric for 'effective' nutrient removal is challenging and the reported removal rates assembled from literature are highly context dependent. While none of the evaluated treatment systems were found to remove 100% of nutrient pollutants, the degree to which effluent impacts waterways depends on the assimilative capacity of the receiving water to absorb excess nutrients, which can be a function of temperature, current speed, biotic factors, and other sources of nutrient inputs [33]. Further, the 'effective' amount of nutrient release into a water body for one farm is a function of the nutrient releases of all farms and other sources on that water body. Jurisdictional-level regulations and initiatives are critical for responsible intensification beyond the individual farm-level, given the high degree of surface water connectivity between most farms, which can affect water quality and disease prevalence [116,117]. Sustainable aquaculture practices must adopt effective and affordable wastewater treatment processes to address the full range of environmental risks as the sector grows and intensifies [118], taking into account physical, production, ecosystem, and social carrying capacities [119].

Removal rates collected through literature review for the evaluated treatment systems indicate that a 32% reduction in TN for the 120PLm^{-2} scenario to achieve ASC certification standards may be achievable for most treatment systems, but that a 67% reduction in TP is outside of the upper threshold reported in many studies. However, simply applying removal rates found in literature oversimplifies the comprehensive farm management practices necessary to reduce effluent loads. Farms would need to implement numerous water quality monitoring and management protocols to make meaningful improvements, and it may be that larger, more corporate farms are better equipped to achieve these targets and comply with international certification standards [111].

Equivalent annual costs per kg and financial considerations

Farm financial performance is highly variable based on production and market parameters, and assumptions used to generate revenue in this study should be adjusted for specific geographies and farm performance. For example, crop duration and number of production cycles per year vary across production systems and are dependent on management practices as well as disease prevalence. The impact of mortality on revenue and profit is a function of timing, and the assumption that mortality occurs at the beginning of the production cycle may slightly overestimate revenue, while the impact on profitability could be more significant.

Substantial profit margins are necessary for farmers to implement effluent treatments systems, which were found to be more difficult to obtain for low-intensity producers. Many of the treatment system EACs for the 7PLm^{-2} scenario were above the cost threshold, while increasing production intensity resulted in lower relative costs on a per kg shrimp basis. Profit margins fluctuate based on internal and external factors, such as farm efficiency, input costs, and market prices. Average profits have been found to range from about 16% EBIT in Indonesia to about 20% EBIT in Thailand, Vietnam, and India for *L. vannamei* [13,120–122]. The selected threshold of USD 0.18 kg shrimp^{-1}, or about 4% of EBIT, may be a significant diversion of revenue for average farmers to invest in effluent treatment, however, farm-level improvements, such as improved growth through better feed and the use of biofloc, can increase EBIT margins from 16% up to 23% and 21%, respectively, in Indonesia [13]. This suggests that the increased profit from improved farm management could cover the capital and operating costs necessary to remediate effluent outflow.

EAC calculations for constructed mangrove wetlands could be highly variable depending on localized restoration costs, which would require restoring production ponds or degraded areas back into mangrove habitat, incurring costs for design, labor, materials, and time. Depending on location, economic situation of the project host country, and initial condition of the restoration site, these costs can range from as high as USD 125,000 per ha in developed countries or as low as USD 100–1000 per ha in developing countries [123,124]. The effective removal rates and low operating costs of constructed mangrove wetlands make them viable solutions for developing countries in tropical climates [125], but project-specific costs need to be considered.

Several treatment systems presented in this study provide supplemental revenue sources that could offset some of the treatment costs. IMTA typically combines high-value target species (i.e., shrimp) with secondary extractive species, such as shellfish or seaweeds, which can be sold as feed, fertilizers, or for human consumption. Supplemental revenue would be especially important for IMTA since our results indicated that it is the treatment system with much higher overall costs. Additional fish and crustacean species can enter silvoaquaculture production areas providing up to 24% in additional yields and 14% in supplemental income [126]. Intact and restored mangrove forests could benefit from emerging blue carbon finance mechanisms, such as voluntary carbon credits. However, only a few mangrove conservation and restoration projects, including one recently led by Conservation International in the Bay of Cispatá, Colombia [127], have issued Verified Blue Carbon Units. Due to the variability across and within these systems to generate supplemental revenue, these benefits were not quantified in the study but could provide additional incentives.

Spatial requirements

The spatial requirements for NbS treatment options vary greatly, reflecting the variation of their removal efficiencies across, as well as within, systems [125]. Our results of 1.3ha of mangroves are slightly lower than previous studies that have suggested an area between 1.8ha and 21.7ha of existing mangrove forests needed for every one hectare of production pond [56,69,128]. The spatial requirements for existing mangroves are limited by their environmental capacity to mitigate effluent nutrients when considering clustered farm densities and local regulations on mangrove use [129]. The shared use of limited off-farm mangrove forests could create a scenario of over-pollution and over-saturation of nutrients without adequate and enforceable regulatory intervention. Spatial requirements for constructed mangrove wetlands used for wastewater treatment vary considerably as well. An area of just 0.086ha has been recommended for a stocking density of 100 PLm^{-2} [130] but ranges up to 12ha for intensive shrimp production [57], aligning with the 1.34ha findings for the 120 PLm^{-2} scenario of this study.

Opportunity costs of land dedicated to effluent treatment rather than production area represents a significant financial consideration for farmers. Land suitable for shrimp ponds, typically above tidal influence, are the most valuable to a farmer. Surplus land on a farm tends to be intertidal or otherwise unsuitable for production, and it is in these areas that NbS treatment options are most viable as low-cost, easy to implement risk buffers. However, the area of surplus land is often limited and not likely adequate to remediate effluent loads. Low-productivity ponds, or those that have been abandoned, can be viable options to implement NbS. Productivity tends to decrease over time for farms that are not properly managed, with typical lifespans of 10 to 13 years, but can be as low as one year, and after which they are likely to be abandoned [131]. Available parcels to implement NbS at scale remains a challenge, though sustainable intensification on a portion of a farm could sustain economic viability for farmers while creating sufficient area for NbS systems on the remaining portion of the farm [132].

Ecosystem services and benefits to the environment

Ecosystem services provide environmental benefits to individuals and communities but are often difficult to quantify financially as excludable goods. Attempts to standardize ecosystem service valuation often results in wide ranges of values across and within biomes [133], due to differing stakeholder perceptions and willingness to pay for such services [134]. Classifying ecosystem service benefits by beneficiary (i.e., individual, group, community, etc.) addresses some heterogeneity [135], but the monetary value that an individual farm may gain from ecosystem services is challenging to determine, making it an unconvincing value proposition for farmers to invest in. Additional financial structures and mechanisms are needed that can capture the value of ecosystem services, for the individual and community, to incentivize farmers towards supporting such initiatives.

Conventional, or gray, infrastructure, while effective for their designed purpose, does not provide additional ecosystem services and can have negative environmental impacts as a result of degraded surrounding natural ecosystems, resource and energy use, or other waste outputs [7]. Both commercial filters and woodchip bioreactors were found to have a small footprint to be effective at any given production intensity, relative to other treatment systems. However, commercial filters have high energy requirements to pump water and have resource-intensive parts with finite lifespans that will need to be replaced. Woodchip bioreactors require a complete replacement of woodchip media at least once every 5 years [55]. Although settling ponds can include building materials when lined with plastic or cement, many are earthen basins that can be defined as human-made wetlands with the potential for minimal ecosystem service provision [105].

Approximately 52% of mangroves across Asia and South America have been deforested since 1970, with shrimp aquaculture expansion accounting for 28% of that loss [19, 136], and up to 63% to 76% of loss at a provincial level [19,20]. Although the rates of mangrove deforestation have decreased from as high as 3.6% per year to 0.05–0.7% per year over the last two decades [19,137,138], the multitude of benefits provided by intact mangrove forests remain absent. Silvoaquaculture may achieve additional mangrove area, but the widespread implementation in areas such as the Mekong Delta has led to increased mangrove forest fragmentation [139] with limited benefits to overall ecosystem services. Over-reliance on the natural remediation capacity of existing mangroves forests comes with significant risks, potentially having adverse effects for the ecosystems services they provide. Recent attention has been directed towards the impacts of effluent on soil composition and carbon stocks, where continual aquaculture effluent has driven cumulative increases in soil nutrients, increased emissions, and contributed to localized eutrophication potential [140–144]. Cumulative impacts, including effects to aquatic food webs, from high-production areas as seen in Asia, likely have significantly larger influence on natural nitrogen cycling than previously anticipated [35]. The use of existing mangrove forests to abate large volumes of unregulated aquaculture effluent has limitations and the adverse impacts to ecosystem service delivery warrants additional attention. However, many of these complications can be addressed if mangrove ecosystems can be appropriately designed to mitigate effluents using constructed treatment wetlands. For example, treatment wetlands have been proposed to be built within existing ponds when space is limited by compacting the bottom and creating new berms, therein providing deep basins to extend hydraulic residence time and increase contact surface area between the effluent, soil matrix, and root zone [130].

Widespread adoption of NbS in the aquaculture sector depends upon building the evidence that validates such an approach and growing the experience of engineers, contractors, and governments to finance and implement NbS projects. While concepts that integrate NbS, such as hybrid green-gray infrastructure, are emerging they are not yet in common use [145]. However, early examples show promise, such as converting cultivated land back into mangrove ecosystems to help manage coastal retreat and erosion [146] as well as shrimp farming that supports Climate Compatible Development [147] where "mixed production systems" of aquaculture and mangrove restoration take place simultaneously [148]. One obstacle to widespread adoption is the lack of information that decision makers have to robustly and efficiently compare the performance of conventional and NbS alternatives to make informed investment choices. Transparent and honest communication within and across sectors on the efficacy of NbS pilots, innovations, and applications should be encouraged to help facilitate the use of conventional and NbS approaches where appropriate.

Areas for further research

In general, there is a lack of robust data on shrimp farm effluent loads and treatment systems. Effluent waste loads in this analysis were calculated based on production assumptions that will vary across producers, geographies, and time. Findings from this study allude to the potential costs and benefits of conventional and NbS treatment systems but would benefit from further validation through actual farm-level effluent data and side-by-side treatment comparisons. Specifically for constructed mangrove wetlands, where characteristics of the wetland changes as mangroves mature, additional data is needed to understand the remediation potential and ecosystem service provision over the lifetime of the system.

Existing analyses evaluating commercial applications of NbS for effluent treatment and environmental benefits are sparse, however, initial results are encouraging. A 10-year pilot constructed wetland system in the Pearl River Delta of China designed to treat municipal wastewater recently observed that two mangrove species maintained a steady and efficient treatment performance and did not require additional harvesting, replanting, or maintenance, whereas herbaceous vascular plants may incur addition maintenance costs and variable removal efficiencies [91,149]. In another case, 120ha of mangrove wetland used to treat effluent from 286ha of shrimp farms in Colombia found that the system was effective in eliminating suspended solids and that a large and resident bird population had developed [150]. Continuous aquaculture effluent has been found to cause elevated soil organic carbon, TN, and TP in proximity to discharge point sources, but these concentrations rapidly decreased through 300m of mangrove stands [140]. However, the study warned that carbon sequestration potential could be impacted from long-term nutrient inputs if mangrove forests were too small to fully treat effluent. Conversely, accumulation of TN and TP by *Sonneratia apetala Buch-Ham* indicated an increase of 50% in biomass with higher wastewater concentration and demonstrated a linear correlation between mangrove biomass and nutrient inputs [151]. Additional, long-term research on the bioremediation capacity of mangroves to mitigate nutrient effluent from commercial aquaculture is needed to document the potential benefits and trade-offs associated with constructed mangrove treatment wetlands.

Future studies of aquaculture effluent management should consider different combinations of hybrid treatments for commercial application. While modelled effluent treatments showed promise individually, our approach does not allow for the use of hybridizing treatments in sequence or in parallel due to the increasing complexity of biogeochemical processes interacting with the effluent from one treatment to the next. Although we would anticipate that hybrid treatment methods, like using settling ponds before constructed mangrove wetlands, would confer ways of balancing tradeoffs and constraints of either treatment individually, a lack of water quality data inhibits the assessment of such an approach.

Conclusions

As shrimp aquaculture production intensifies, treatment of effluent is critical to minimize and avoid negative impacts to surrounding aquatic environments. Constructed mangrove wetlands are an NbS approach that can meet the intensification needs of aquaculture producers and reduce negative impacts from effluent at competitive costs, while also providing ecosystem service co-benefits. However, limited availability of parcels to implement NbS remain a major obstacle and alternative incentives for farms, such as sustainably intensifying in a smaller farming footprint and restoring mangroves on unused parcels, will need to be explored for shrimp farmers to realize potential benefits and utilize such systems. Additional studies and pilots are needed to inform practitioners and policymakers on the diversity of NbS applications and to gain larger stakeholder acceptance and support.

Supporting information

S1 Text. Supplementary methods.
(DOCX)

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