



# **Ocean Deacidification Technologies for Marine Aquaculture**

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Abstract: The increase in partial pressure of  $CO_2$  in the oceans directly affects the productivity and survival of coastal industries and ecosystems. For marine aquaculture, the decreased alkalinity of seawater results in reduced availability of carbonates for marine organisms to build their shells, leading to decreased aquaculture quality and productivity. The industry has been implementing recirculating aquaculture systems (RASs) to reduce  $CO_2$  in feedwaters, but recent interest in ocean-based  $CO_2$  capture has led to additional strategies that may be relevant. The new methods in addition to  $CO_2$  removal offer capture options for enhanced aquaculture sustainability. Here, we review and compare early-stage and commercially available technologies for deacidification of seawater and their suitability for aquaculture. Most methods considered rely on a voltage-induced pH swing to shift the carbonate/bicarbonate equilibrium toward the release of  $CO_2$ , with subsequent capture of the released  $CO_2$  as a gas or as solid mineral carbonates. The modular design and distributed deployment potential of these systems offers promise, but current demonstrations are limited to bench scale, highlighting the need for sustained research and development before they can be implemented for marine aquaculture.

Keywords: CO2 removal; aquaculture; mCDR

# 1. Introduction

Anthropogenic greenhouse gas (GHG) emissions continue to alter the global climate. As it stands,  $CO_2$  contributes the most toward the rise in global temperature when compared to any other GHG. Since the end of the preindustrial age (1750), levels of  $CO_2$  in the atmosphere have risen by 47% [1], and, even within just the last decade, levels have risen by 1.4% per year on average [2].Unfortunately, much of the global society is still reliant on carbon-based fossil fuels, and, to keep the rise in global temperature under the 1.5 °C goal set by the Paris Climate Agreement, a transition to carbon-neutral energy sources must be made quickly, but meeting this audacious goal will also require the simultaneous development of negative-emission technologies (NETs) and the implementation of progressive policies, regulations, and incentive structures for rapid technology adoption [3].

Significant attention and resources have been devoted to terrestrial NETs, with several strategies developed and demonstrated at varying scales, ranging from age-old afforestation/reforestation [4,5] to hybrid bioenergy with carbon capture and storage (BECCS) [6,7] and modular direct air capture (DAC) technologies [8,9]. While the widespread adoption of afforestation and BECCS is limited by competing land-use challenges, the use of DAC is limited due to high cost. Most importantly, while these strategies address CO<sub>2</sub> emissions, they fail to tap into the planet's largest CO<sub>2</sub> sink—the oceans.

# 1.1. $CO_2$ in the Ocean

The oceans absorb about 33% of all anthropogenic  $CO_2$  emissions. When compared in equivalent units, bulk  $CO_2$  in air and ocean environments is about 720 Gt and 38,400 Gt,



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**Copyright:** © 2022 by the authors. Licensee MDPI, Basel, Switzerland. This article is an open access article distributed under the terms and conditions of the Creative Commons Attribution (CC BY) license (https:// creativecommons.org/licenses/by/ 4.0/). respectively [10]. The higher average  $CO_2$  concentration and significant  $CO_2$  reserves in global oceans offer an opportunity for ocean-based climate solutions.  $CO_2$  absorbed by the oceans is converted to carbonic acid and subsequently  $H^+$  and bicarbonate, which decreases the saturation state of calcium carbonate and results in reduced availability of carbonates for marine organisms to build their shells, leading to decreased productivity and quality of aquaculture [11]. Ocean acidification also has direct impacts on coastal ecosystem, communities, and marine-based economies [12–14]. Hence, ocean-based carbon-capture solutions not only offer opportunities to utilize a concentrated  $CO_2$  source for high-impact NETs, but are also a means to address anthropogenic ocean acidification [15].

Once absorbed by the ocean,  $CO_2$  changes forms, as seen in Equations (1)–(3).

$$CO_2(g) + H_2O \leftrightarrow H_2CO_3$$
 (1)

$$H_2CO_3 \leftrightarrow HCO_3^- + H^+$$
 (2)

$$HCO_3^- \leftrightarrow CO_3^{2-} + H^+$$
 (3)

All the forms of CO<sub>2</sub> including HCO<sub>3</sub><sup>-</sup> and CO<sub>3</sub><sup>2-</sup> are in equilibrium with each other; however, slight changes in pH, temperature, and salinity affect the relative concentrations of those carbon species. At the pH of seawater (pH 8), approximately 0.5% of the total carbon dioxide,  $[CO_2]_T$ , exists as dissolved CO<sub>2</sub> gas, while 88.5% exists as HCO<sub>3</sub><sup>-</sup> and 11% exists as CO<sub>3</sub><sup>2-</sup> [16]. It is known that lowering the pH (increasing [H<sup>+</sup>]) to a value less than 6 changes equilibrium conditions to favor the production of CO<sub>2</sub> gas [17], while raising the pH greater than 8.5 favors the formation of CO<sub>3</sub><sup>2-</sup> [18]. Manipulating this carbonate/bicarbonate equilibrium is central to the operation of many ocean CO<sub>2</sub> capture technologies.

#### 1.2. Ocean Acidification Impacts on Marine Aquaculture

In addition to the broader impacts of ocean-based  $CO_2$  removal, there are direct ecosystem and economic benefits from ocean deacidification. Coastal industries, particularly marine aquaculture, are highly sensitive to ocean  $CO_2$  levels and pH changes, which can lower the saturation state of aragonite, a crystalline form of calcium carbonate that many organisms use to build their shells [19,20]. Marine aquaculture production largely consists of bivalve mollusks such as oysters, clams, and mussels, as well as salmon and shrimp [21]. Shellfish, a general term used to describe aquatic shelled mollusks and crustaceans used for food, are directly impacted by the reduced availability of aragonite to build shells, leading to decreased aquaculture quality and productivity.

In 2007, Whiskey Creek Shellfish Hatchery near Tillamook, Oregon experienced a large die-off of oyster larvae after exposure to low-pH seawater [22]. Similar events have cost the US shellfish aquaculture industry approximately 110 million USD annually [23–25]. Beyond the aquaculture industry, ocean acidification also impacts the income of coastal communities that rely on tourism surrounding coral reefs [21,26–30]. On a global scale, an annual economic loss of 97–301 billion USD (in 2014) is estimated to result from ocean acidification [31].

Although localized parameters such as ocean current, water temperature, and alkalinity play a role in the rate and extent of acidification, the overall trend of decreasing pH has been seen almost everywhere in the world [32–36]. Studies have clearly demonstrated the link between CO<sub>2</sub> levels and aquaculture quality and productivity, as shown in Figure 1 [37]. As a result, hatcheries have developed mitigation strategies to minimize uncertainty and to better protect their assets. With consistent monitoring and early detection, localized buffering, site and breeding selectivity, and nutritional enhancements, the aquaculture industry has been able to curtail some of the effects of ocean acidification [38]; however, those methods only offer temporary relief. Capturing CO<sub>2</sub> from the ocean, often referred to as marine carbon dioxide removal (mCDR) could allow for ocean-based climate solutions while simultaneously addressing ocean acidification. Furthermore, when mCDR methods are applied in existing coastal industries such as algal cultivation [39,40] or coastal infras-



tructure [41,42], a clear economic incentive to deploy ocean-based climate solutions can be established.

**Figure 1.** Impacts of increasing ocean  $CO_2$  levels on clams. Figure reprinted from [37] with permission from the Proceedings of the National Academy of Sciences of the United States of America. (A) Image of full individual larvae under specified  $CO_2$  level. (B) A magnification of the outermost shell of individuals under specified  $CO_2$  level.

#### 2. Marine Carbon Dioxide Removal Methods

Various mCDR strategies (Figure 2) have been considered, ranging from artificial upwelling and ocean alkalinity enhancement to seaweed cultivation and electrochemical CO<sub>2</sub> stripping [43]. However, given that all of the approaches are in the early stages of development and the scarcity field demonstrations to date, their estimated potential for impact is highly variable, and the risks associated with large-scale implementation are not known [43,44]. The benefits and suitability of these techniques also depend on the local industry (onshore vs. offshore), ecosystem, and competing resource needs.

Given considerations of fish health and food safety, chemical alkalinity enhancement of feedwaters resulting in CaCO<sub>3</sub> precipitation (i.e., calcium depletion) seems nonideal. On the other hand, for seaweed cultivation, despite being a chemical input-free carbon capture strategy, the spatiotemporal pH changes generated are not known. Furthermore, aquaculture facilities may need additional resources to grow and manage seaweed, making it less attractive. Hence, several promising mCDR strategies may not be suitable for deacidification of aquaculture feedwaters.

Specific to marine aquaculture, which is expected to significantly grow by 2050 [45], coupling with modular mCDR systems that can be easily added to existing seawater intake at the facilities would be preferrable. Additionally, systems that can be operated using renewable energy and with minimal chemical inputs would allow for sustainable seafood production. Considering these factors, here, we review technologies suitable for deacidifying feedwaters for coastal aquaculture.



**Figure 2.** Strategies for ocean CO<sub>2</sub> removal. Credit: National Academies of Sciences, Engineering, and Medicine. 2021. A Research Strategy for Ocean-Based Carbon Dioxide Removal and Sequestration. https://doi.org/10.17226/26278 (accessed on 9 February 2022). Reproduced with permission from the National Academy of Sciences, courtesy of the National Academies Press, Washington, DC, USA.

## 3. Technologies for CO<sub>2</sub> Removal from Aquaculture Feedwaters

 $CO_2$  removal methods suitable for deacidification of marine aquaculture feedwaters, both commercially available technologies and methods in early stages of development, are described below. These methods can be broadly classified into physical (trickling filter), chemical (ion exchange), and electrochemical (bipolar membrane electrodialysis and electrolytic cation exchange) approaches, and we include a comparison of the methods in terms of cost, efficiency, and ease of implementation later in the review.

# 3.1. Trickling Filter and Recirculation Aquaculture Systems

Recirculating aquaculture systems (RASs) were first developed in the 1950s [46,47] and have been in use for the past 40 years [48]. Not only are they a viable option for the onshore production of fish such as salmon, seabass, cod, catfish, eel, and perch [49,50], but they also represent a commercially proven method for removing  $CO_2$  from water under certain conditions. In general, RAS is a method in which fish are grown in a closed-loop set of tanks where water is recycled, and the addition/removal of nutrients/toxins is heavily controlled by different types of filters. As a result, oxygen concentration in the water decreases over time due to respiration, while CO<sub>2</sub> and nitrogen concentrations increase. Elevated levels of CO<sub>2</sub> in the water are highly toxic to fish; thus, its cost-effective removal is very important [51,52]. In an open system such as a net-pen or flow-through, the depletion of  $O_2$  and buildup of  $O_2/N$  are nonissues because the water is constantly being replaced by the surrounding environment. Within RASs, the trickling filter (TF) is designed to provide aeration and some  $CO_2$  removal. In a TF, water is pumped to the top of the filter, where gravity pulls the water down through the filter medium (Figure 3) [53]. The medium can be unstructured/structured polyethylene [54], wood wool/sawdust [55], constructed wetland [56], or other materials that break up the flow of water. Aeration of the water in

combination with an air stream in the direction opposite to the flow of water (caused by a fan) allows  $CO_2$  to diffuse from the water to the surrounding air. It is then usually pumped outside away from the facility. In 2020, Karimi et al. tested the ability of a Bionet-filled trickling filter to remove  $CO_2$  under different gas-to-liquid ratios (GLRs) [54]. They found that a GLR of 5:1 was the most cost-effective setup for their RAS, decreasing the [CO<sub>2</sub>] from 7.2 to 5.1 mg/L. A similar result was reported by Summerfelt et al. in 2003, where they stated that a GLR over 5:1 only led to marginally better  $CO_2$  removal [57]. While RASs offer a relatively simple physical method for the reduction of dissolved  $CO_2$  in water, they only remove  $CO_2$  from water when the relative concentration is higher in the water than the surrounding air. Albeit simple and commercially available, the implementation of large-scale TF and RAS to remove  $CO_2$  from ocean water will also require subsequent  $CO_2$  capture strategies in order to be environmentally sustainable.





#### 3.2. Ion-Exchange Resins

Ion-exchange resins (IER) are widely used in water treatment and in industrial processes. The resins are sold as anion- and cation-exchange types. While anion-exchange resins can be used to directly capture  $CO_2$  as bicarbonate ions, cation-exchange resins can be used to indirectly capture  $CO_2$  by altering the seawater pH and altering the carbonate/bicarbonate equilibrium [58]. Both resin types need to be regenerated upon saturation using chemical solutions (acid, base, salt solution). In case of the anion-exchange resins, the significantly higher concentrations of chloride ions  $(240 \times)$  in seawater compared to bicarbonate make them impractical to use. The measured CO<sub>2</sub> capture capacity of an anionexchange resin in Key West seawater was just 0.07 mg/g of resin, making it impractical for large-scale applications [59]. Although IERs may be suitable for land-based, near-coastal applications, the need for a strong base (NaOH) for regeneration makes them less practical. Similarly, the use of cation-exchange resins to acidify seawater showed a low capacity of  $\sim$ 0.32 mg of CO<sub>2</sub>/g of resin [60], and they require either deionized water or strong acid (HCl) for regeneration, making use at scale less economical. However, IERs for carbon capture may be an option in the future if electrical resin regeneration either directly via water splitting or indirectly via electrochemical onsite acid/base generation from seawater and renewable energy is used. Given the limited promise of IER at this stage, the method is not discussed further in this review.

#### 3.3. Bipolar Membrane Electrodialysis

A bipolar membrane electrodialysis (BPMED) unit is made up of multiple membrane layers stacked between an anode and cathode. When adequate energy is supplied to the unit, water at the surface of the bipolar membrane (an ion-exchange membrane made up of a cation- and an anion-exchange layer) dissociates into H<sup>+</sup> and OH<sup>-</sup> ions. The movement of the ions is dictated by the applied potential, causing cations to move in the direction of the cathode and anions to move toward the anode. This and the order of the cation-exchange membranes (CEMs) and anion-exchange membranes (AEMs) create alternating streams of acidic and basic solutions (Figure 4a). Within the acidified water (pH 4), the low pH shifts the  $CO_2$ -carbonate equilibrium to favor the formation of  $CO_2$ . Membrane contactors after the BPMED unit then strip the dissolved  $CO_2$  from the water (Figure 4b) before the basified and acidified streams are eventually recombined for disposal. Waste stream neutralization is critical to ensure that water is safe for local ecosystems in terms of pH and alkalinity before being discharged back into the ocean. Early prototypes demonstrated low efficiency due to the formation of  $CO_2$  bubbles on the surfaces of the membranes leading to reduced effective surface area and, consequently, lower ion transport. However, the issue has since been overcome by placing the unit under higher pressure to minimize the formation of gas bubbles [61]. Higher operating pressures (6 atm vs. 1.5 atm) resulted in a 29% decrease in CO<sub>2</sub> desorption energy (471 kJ/mol CO<sub>2</sub> to 333 kJ/mol  $CO_2$ ) from a carbonate solution. However, it is important to note that this configuration relies heavily on the movement of  $HCO_3^-$  and  $CO_3^{2-}$  across AEMs in the membrane stack. Relative to dissolved inorganic carbon (DIC) concentrations in natural seawater, Clconcentrations are significant ( $\sim 200 \times$ ). Thus, if high-pressure BPMED is used with natural seawater, the efficiency is expected to drop considerably as most of the energy would be used to carry  $Cl^{-}$  across the membranes instead of  $CO_2$  carrying  $HCO_3^{-}$  or  $CO_3^{2-}$ . As a comparison, conventional BPMED technology relies on the movement of H<sup>+</sup> and OH<sup>-</sup> across the membrane stack to lower or raise the pH of natural seawater.



**Figure 4.** Figure reprinted from [62]. Copyright (2020) Nature Communications. Licensed under Creative Commons Attribution 4.0 International Public License. (**a**) Schematic drawing of seawater in a BPMED cell. (**b**) Process flow diagram of experimental setup for CO<sub>2</sub> extraction from seawater.

CO<sub>2</sub> capture by BPMED via acidification allows for the extraction of gaseous CO<sub>2</sub>, which can be concentrated and permanently captured when used as a strengthener in cement [41], or recycled when used in the production of chemicals (urea, methanol, formaldehyde) [63], or as a feedstock for algae cultivation [64]. Various BPMED prototypes have been designed and tested using synthetic seawater, which operate on the same fundamental principles, but differ in membrane configuration and electrolyte solutions, leading to different energy efficiencies for CO<sub>2</sub> removal, ranging from 155.4 kJ/mol CO<sub>2</sub> [62] to 242 kJ/mol CO<sub>2</sub> [18]. Alternatively, CO<sub>2</sub> capture by BPMED via basification relies on precipitating CaCO<sub>3</sub> minerals, which have various uses as a cement precursor, filler in plastics, and ingredient in dietary supplements [65–67]. Zhao et al. exploited this phenomenon, utilizing BPMED and induced crystallization to perform mineralization and CO<sub>2</sub> adsorption from synthetic seawater [68]. Similarly, de Lannoy et al. studied CaCO<sub>3</sub> precipitation facilitated through their BPMED unit [18]. They found, however, that the pH must be held within the narrow range of 9.3 to 9.6 to avoid precipitation of Mg(OH)<sub>2</sub> that dominates when pH

is greater than 9.6, rendering the process useless in its ability to capture carbon. When compared to  $CO_2$  extraction driven by acidification, the process of basification would not require vacuum pumps or membrane contactors to capture  $CO_2$ , allowing for lower overall capital, operational, and maintenance costs [69]. Although BPMED (both acidification and basification) offers an elegant solution to ocean deacidification, the technology is in its very early stages. The pathway has not yet been tested in an ocean environment; instead, only synthetic seawater and brine solutions in the lab have been used. Thus, the technology performance in harsh marine conditions is unknown. Both methods rely heavily on membranes which are expensive and highly susceptible to fouling/scaling [70]. Specific to basification, there is limited research beyond preliminary bench-scale experiments, and the impact of replacing Ca<sup>2+</sup> by Na<sup>+</sup> in the waste returned to the coastal environment is not known. Most importantly, carbonate mineral precipitation, forming CaCO<sub>3</sub>, diminishes the levels of free Ca<sup>2+</sup> that are locally available for marine calcifiers to build strong shells [71] and, hence, not suitable for marine aquaculture applications.

## 3.4. Electrolytic Cation Exchange

The electrolytic cation exchange module (E-CEM) was first reported in 2014 by Willauer et al. [58,72–74]. The technology was designed primarily to generate fuels from seawater for naval applications. The E-CEM itself is a three-compartment cell that sits between an anode and cathode made of platinum-coated titanium (Figure 5). Between each compartment sits a cation-exchange membrane (CEM) designed to facilitate proton and sodium ion transport for pH gradient generation. To begin the process, filtered seawater and deionized (DI) water generated using a reverse-osmosis system are pumped into the E-CEM, with filtered seawater into the center compartment and DI water into the two electrode compartments. On the surface of the anode, DI water is oxidized to generate  $H^+$  and  $O_2$ , whereas water is reduced to  $OH^-$  and  $H_2$  at the cathode. Driven by the applied potential, the  $H^+$  ions move across the first CEM into the seawater stream where they interact with the  $CO_3^{2-}$ and  $HCO_3^-$  to form  $H_2CO_3$  and, subsequently, dissolved  $CO_2$ . At the same time, Na<sup>+</sup> from the seawater in the center compartment moves through the second CEM and mixes with the OH<sup>-</sup> to form NaOH. The acidified seawater and electrolyte solutions then go through separate membrane contactors/vacuum strippers to remove the  $CO_2$  and  $H_2$  gases before the solutions are discharged. E-CEM has been tested in different configurations involving the presence/type of ion exchange resin in the compartment, the electrode compartment volume, and the endplate material. For example, removing the cation exchange resin from the compartments improved process times to achieve necessary pH for CO<sub>2</sub> release, while decreasing the electrode compartment volume lowered module resistances by about 33%. By design, E-CEM minimizes inorganic membrane scaling by alternating pH swings using symmetric electrodes that allow for polarity reversal. However, there are several other challenges that remain, especially in terms of energy use and cost of E-CEM for deacidification. The energy for deionization of seawater can be significant and require considerable pretreatment that might add to overall process chemical use and disposal; further membrane contactors are expensive, and E-CEM requires not one but two contactors (for both  $CO_2$  and  $H_2$ ). Not using appropriately deionized water runs the risk of E-CEM generating Cl<sub>2</sub> gas and related oxidative byproducts harmful to fish while not reliably capturing the combustible H<sub>2</sub>, which is a safety concern. Redesigning E-CEM for operation with a closed-loop anolyte and catholyte with favorable redox voltages and easy-to-handle byproducts may make it better suited for marine aquaculture applications. E-CEM technology in its current configuration is not optimized for  $CO_2$  capture from seawater, and if  $CO_2$  generation is the ultimate goal, decoupling the process from  $H_2$  and O<sub>2</sub> production may improve overall process efficiency.



**Figure 5.** Reprinted (adapted) with permission from [72]. Copyright (2017) American Chemical Society.

# 4. Comparison of Technologies

Technologies for  $CO_2$  removal from seawater are in their early stages of development, with only one technology being successfully demonstrated in field settings. As a result, it is important to note the benefits and drawbacks of the different approaches to recognize how they can be tailored for marine aquaculture (Table 1).

Table 1. Summary of technologies for seawater deacidification by CO<sub>2</sub> removal.

CO <sub>2</sub> Removal Method	Feedwater Used	Flow Rate	CO <sub>2</sub> Capture Efficiency	Product	Energy Consumption	Key Reference
TF in RAS	Freshwater, seawater	$100  \mathrm{L} \cdot \mathrm{min}^{-1}$	CO not captured	CO <sub>2</sub> (g)	26,000 kJ·kg <sup>−1</sup> CO <sub>2</sub> 1100 kJ·mol <sup>−1</sup> CO <sub>2</sub> <sup>a</sup>	[54]
BPMED (via acidification)	Synthetic seawater, Brine	$2100 \text{ mL} \cdot \text{min}^{-1}$	80.8%	CO <sub>2</sub> (g)	390–640 kJ·mol <sup>−1</sup> CO <sub>2</sub>	[75]
BPMED (via basification)	Synthetic seawater	Not reported <sup>b</sup>	Not reported	CaCO <sub>3</sub> (s)	546–896 kJ·mol <sup>−1</sup> CO <sub>2</sub> ¢	[18]
BPMED (via acidification)	Synthetic seawater	$37 \text{ mL} \cdot \text{min}^{-1}$	71%	CO <sub>2</sub> (g)	0.98–5.82 kWh·kg <sup>-1</sup> CO <sub>2</sub> <sup>d</sup> 155.4 kJ·mol <sup>-1</sup> CO <sub>2</sub>	[62]
BPMED (with reactive crystallization)	Synthetic seawater	$667 \mathrm{mL}\cdot\mathrm{min}^{-1}$	50%	CaCO <sub>3</sub> (s)	0.89 kWh⋅kg <sup>-1</sup> CO <sub>2</sub> 140 kJ⋅mol <sup>-1</sup> CO <sub>2</sub>	[68]
E-CEM	Natural seawater	1900 mL $\cdot$ min <sup>-1</sup>	92%	CO <sub>2</sub> (g), H <sub>2</sub> (g)	22 kWh·kg <sup>−1</sup> CO <sub>2</sub> 3500 kJ·mol <sup>−1</sup> CO <sub>2</sub> <sup>e</sup>	[72]

<sup>a</sup> Value comes from heat loss of the system caused by water descending through trickling filter, causing release of CO<sub>2</sub>, not including capture or system operation [54]. <sup>b</sup> No flow rate data available because testing was not done within BPMED unit. <sup>c</sup> 40% more energy than acid process from [18]. <sup>d</sup> Range of values arise from colocation with desalination plant vs. standalone setup at industrial scales. <sup>e</sup> Calculations performed for lowest-energy configuration in [72]: average power = 265 W per 45 min cycle, 1 mol CO<sub>2</sub> generated per mol H<sub>2</sub>, and a gas generation rate of 0.006 mol·min<sup>-1</sup> at 20 A·kJ·mol<sup>-1</sup> calculation assumes a 1 h run time.

#### 5. Summary and Future Outlook

RASs operate on the principle of high  $pCO_2$  in the water relative to the surrounding air. Passage of the saturated water through an aerated trickling filter allows for the gas to be continually removed from the water stream. RAS technology has already demonstrated CO<sub>2</sub> removal from fresh and seawater systems, and it is the most cost-effective deacidification approach for aquaculture; however, it is not designed to capture CO<sub>2</sub>. As seen in Table 1, the energy consumption by RASs is high relative to other methods, which is most likely caused by the need for temperature equilibration after evaporative cooling within the trickling filter. Adding a membrane contactor would enable CO<sub>2</sub> capture and make RASs more sustainable, but membrane contactors are expensive and add energy and maintenance costs [76]. Here, we note that the other methods included in Table 1 either do not integrate pumping costs into the overall energy calculations or assume colocation with a desalination plant to minimize pumping-related capital and operating expenses. Hence, a direct comparison of RAS energy use to that of the other (emerging) technologies may not be very meaningful.

Although achieved in different ways, electrochemical methods rely on a pH swing ( $\Delta pH$  can be 2–6 units) to release CO<sub>2</sub> by altering the carbonate/bicarbonate equilibrium. BPMED creates  $H^+$  and  $OH^-$  on the surface of the bipolar membrane due to an applied voltage across the membrane stack. These ions then move through cation- and anionexchange membranes to create streams of acidic and basic seawater or brine from which the  $CO_2$  can be extracted. E-CEM uses oxidation and reduction of water to generate  $H^+$ , which acidifies the input seawater while simultaneously producing H<sub>2</sub> gas, which can also be used as a fuel source. However, the process requires deionized water and multiple membrane contactors, making it less practical compared to BPMED. If operated using renewable energy, BPMED can offer a sustainable solution to deacidification with  $CO_2$ capture, by producing either gaseous  $CO_2$  (acidified approach) or solid CaCO<sub>3</sub> (basified approach). The  $CO_2$  produced can be used as a precursor for chemicals or as a feedstock for algae growth. While capture via  $CaCO_3$  precipitation produces an easy-to-handle solid product and eliminates the need for expensive membrane contactors, the process decreases calcium availability in feedwater, thus negatively impacting aquaculture quality and productivity [29,77,78].

In summary, current mCDR technologies that are suitable for deacidification of aquaculture feedwaters are in their very early stages of development and must improve energy efficiency, enhance material durability in harsh marine conditions, and decrease component costs to be viable for aquaculture applications. The lack of specialized infrastructure, and the difficulty in accessing reliable and inexpensive sources of carbon-neutral energy are additional considerations. A recent technoeconomic analysis by Eisaman et al. [69] compared the cost of the BPMED-based method as a standalone operation vs. when it is coupled to a desalination plant or a seawater-cooled powerplant, concluding that colocation with a desalination facility to leverage existing infrastructure is key to making BPMED cost-competitive. Although the technologies are in their early stages of development, their modular design and distributed deployment potential offer a promising outlook. However, significant research and development are required to take these technologies from lab to market, and given the pressing need for sustainable deacidification solutions, making the systems cost-effective in the near term may depend on the availability of government subsidies or tax credits.

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