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Review

Bio-mitigation based on integrated multi-trophic aquaculture in temperate coastal waters: practice, assessment, and challenges

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ABSTRACT. In general, aquaculture wastes from traditional aquatic organism cultivation rapidly deteriorate the water quality of the surrounding ecosystems, endangering animals living in the area. The integrated multi-trophic aquaculture (IMTA) system is a bio-mitigation strategy to alleviate the adverse impacts caused by aquafarming pollutants on the environment and aquatic species. This study provides an overview of the IMTA system, explains the interactive processes among the different trophic levels, summarizes the major practices being followed around the temperate coastal waters with a field case study in Japan, and discusses the assessment of IMTA bio-mitigation efficiency through experimental greatly dependent on the customs and market values in the local IMTA practice. Bio-mitigation efficiency acquired in a land-based experiment exhibits its limitation in approach and conducts a comprehensive analysis on the possibility of applying numerical models to evaluate IMTA effectiveness. The selection of a suitable candidate organism is estimating that in the same or different culture conditions with various biomasses of extractive species. However, in open water experiments, it is difficult to evaluate the bio-mitigation effect of extractive species because the initial biomass ratio (IBR) of the extractive to target species is too small. Alternatively, the possibility of applying existing numerical models to assess IMTA is relatively low. In conclusion, an optimally designed large-scale IMTA experiment is required, in which the IBR of the extractive to target species is adequately considered, and a full-scale IMTA model should be further improved with a database of individual-based submodels for IMTA candidate organisms.

Keywords: aquaculture wastes; bio-mitigation efficiency; integrated multi-trophic aquaculture; IMTA; numerical model

INTRODUCTION

Aquaculture, the fastest-growing food production sector worldwide, is currently facing several problems such as self-pollution concerning sustainable development (Troell *et al*., 2009; Alexander & Hughes, 2017). The harmful effect of wastes derived from aquaculture operations on aquatic environment is often questioned and studied (Yokoyama, 2013; Park *et al*., 2015). Less than 30% nutrients from the feed can be utilized by fish for growth, whereas the remaining amounts of particulate organic matter (unconsumed feed and fish feces) and inorganic nutrients (fish excretions) are released into water, causing eutrophication, harmful

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algal blooms, and deoxygenation of the bottom water (Yokoyama, 2013; Irisarri *et al*., 2015; Martínez-Espiñeira *et al*., 2015). Rapid water quality deterioration endangers animals living in the aquacul-ture area, often leading to fish diseases and death.

Much attention has been focused on implementing possible solutions such as aeration, controlling aquaculture density, offshore aquaculture, and co-culture to reduce the negative impacts caused by traditional intensive aquaculture. Aeration can effectively increase the oxygen saturation in the aquatic environment and, consequently, mitigate the negative effects of aquaculture wastes (Boyd *et al*., 2018; Wang *et al*., 2018); however, this method is not cost-effective, constraining its application in practical operations. Offshore aquaculture offers a better environment for aquaculture wastes diffusion (Van den Burg *et al*., 2017; Lester *et al*., 2018), but robust fish cage systems are required in the offshore environment due to severe conditions (*e.g.*, strong water flow, high wave). Integrated multi-trophic aquaculture (IMTA), a co-culture system, is an effective environmental bio-mitigation system (Fig. 1). The IMTA system achieves sustainable aquaculture development by recycling aquaculture wastes such as food resources through co-cultivating the targeted species with others having different feeding habits at different trophic levels (Chopin *et al*., 2013; Neori *et al*., 2017). The wastes caused by the targeted species (*e.g.*, fish) could become food for other species. For example, small organic particles from fish cage or resuspended from the sediment can be used by suspension feeders, while larger particles sinking at the bottom of the sea can be consumed by deposit feeders; seaweeds can absorb the dissolved nutrients generated by the cultured species or released from sediment through water flow exchange (Yokoyama & Ishihi, 2010; Yokoyama, 2013). The bio-mitigation effect of the IMTA system can assimilate the wastes and various potential pollutants, thus alleviating the adverse effects caused by the accumulation of such materials.

In the last few decades, IMTA has been developed and examined by many countries. Several studies on the possible applications of IMTA for land-based, inshore, and offshore systems with special interest in seaweed and shellfish have been examined (Buschmann *et al*., 2008; Fang *et al*., 2016; Perdikaris *et al*., 2016; Neori *et al*., 2017). The recent research includes a concern on assessing the bio-mitigation efficiency of IMTA based on experiments (Li *et al*., 2014; Irisarri *et al*., 2015; Martínez-Espiñeira *et al*., 2015; Milhazes-Cunha & Otero, 2017; Laramore *et al*., 2018). Nevertheless, due to the complex interactive processes involved in the IMTA, it is questionable whether a relatively accurate assessment of IMTA bio-mitigation effectiveness can be fully achieved through partially balanced experiments (Troell *et al*., 2009). The numerical models developed and validated based on small-scale experiments, as an alternative, would be a useful approach to evaluate the overall bio-mitigation efficiency of an IMTA system (Chopin *et al*., 2013; Cubillo *et al*., 2016; Zhang & Kitazawa, 2016; Silfiana *et al*., 2018). However, the research regarding the feasibility of applying existing numerical models to assess IMTA bio-mitigation effectiveness has not been fully conducted (Granada *et al*., 2016).

With a focus on the status and challenge of IMTA assessment methods, this study was primarily intended to contribute to the practice of IMTA and assessment of its bio-mitigation efficiency in temperate coastal ecosystems.

Data collection and analysis

Data in this study were collected from two sources. One source was the field investigation conducted at an IMTA site located at Kagoshima Prefecture, Japan. The growth data of seaweed *Laminaria japonica* cultured around fish cages for three months were collected. The other source was peer-reviewed literature gathered based on subjects related to the experiment and/or numerical model for IMTA.

In the case where the specific information was missing, conversion factors were used: $1\ \mu\text{mol L}^{-1} = \frac{1\ \text{mg L}^{-1}}{\text{Molecular weight}} \times 1000$; $1\ \text{mg g}^{-1} = 0.10\%$. For calculating the specific growth rate (SGR) in terms of wet weight, the following equation was adopted:

$$\text{SGR}(\%\ d^{-1}) = \frac{(\text{In FWW} - \text{In IWW})}{\text{D}} \tag{1}$$

where FWW is the final wet weights of studied species (g), IWW is the initial wet weights of studied species (g), and D is the duration between two weights (day).

To assess the difference in initial biomass between the target species and extractive species, the initial biomass ratio (IBR) was used:

$$\text{IBR}(-) = \frac{\text{IBTS}}{\text{IBES}} \tag{2}$$

where IBTS and IBES are the initial biomasses of the target and extractive species, respectively. Since the unit of biomass varies with study area and species, the biomass measured under laboratory conditions is recalculated as g m^{-2} or g m^{-3} while it is kg $farm^{-1}$ or kg $cage^{-1}$ under open water conditions in the present analysis. When calculating IBR, the unit of initial biomass should be consistent, *e.g.*, kg $cage^{-1}$ for both target and extractive species.

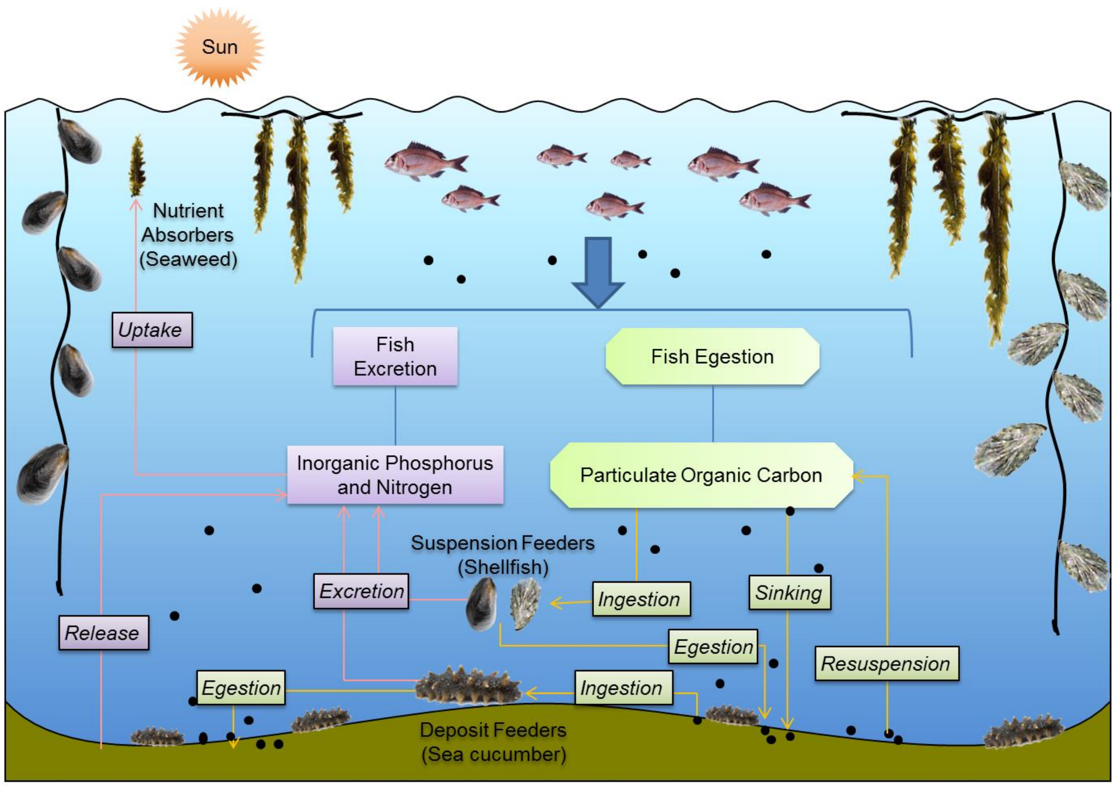

Figure 1. Conceptual diagram of the Integrated multi-trophic aquaculture (IMTA) system.

Bio-mitigation strategy of self-pollution: IMTA

Traditional aquatic organism cultivation self-pollute the surrounding ecosystems by deoxygenating the bottom water with the organic wastes and possibly promoting the growth of harmful algal blooms due to the dissolved nutrients, negatively impacting the animals inhabiting the aquaculture area. Therefore, the IMTA system aims to achieve sustainable aquaculture development, by recycling aquaculture wastes such as food resources through co-cultivating the targeted species with others having different feeding habits in different trophic levels, and improving the efficiency and productivity of intensive monoculture systems.

Various species of nutrient absorbers, suspension feeders, deposit feeders, and other organic-extractive organisms (Table 1) are candidate organisms that can be co-cultured with the targeted species (finfish, *e.g.*, red sea bream F5, Atlantic salmon F7) in an IMTA system (Zhou *et al.*, 2006; Aveytua-Alcazar *et al.*, 2008; Abreu *et al.*, 2009; Mao *et al.*, 2009; Shi *et al.*, 2011; Yokoyama, 2013; Brito *et al.*, 2014; Yokoyama *et al.*, 2015; Cubillo *et al.*, 2016; Fang *et al.*, 2016; Alexander & Hughes, 2017; Shpigel *et al.*, 2017; Laramore *et al.*, 2018; Zamora *et al.*, 2018). Wastes released from the fish farm can be used as food sources for inorganic and organic nutrient-extractive species. For instance (Fig. 1), particulate organic wastes (*e.g.*, fish fecal matter, waste fish feed) primarily sink down to the bottom of the sea, which can be ingested by the deposit feeders (*e.g.*, Japanese sea cucumber D1, giant California sea cucumber D5) as food, consequently mitigating the problem of hypoxic bottom water due to increased oxygen consumption during bacterial decomposition of excessive organic matter. Dissolved nutrients (*e.g.*, phosphorus, nitrogen), another form of aquaculture wastes, cause eutrophication, which increases the risk of harmful algal blooms. Planting nutrient absorbers (*e.g.*, seaweeds N3, N7, N12) can minimize this risk through their competition with phytoplankton for resources, and periodic harvesting of microalgae will speed up the removal of dissolved nutrients. Suspension feeders (*e.g.*, mussels S5, S6, scallops S1, S2, S8, oysters S3, ark shells S9) and other organic-extractive species (*e.g.*, prawns O2, O4, abalone O3, sea urchin O1, jellyfish O5) are capable of further filtering the phytoplankton and disperse small particle organic materials from both fish food and feces in the water column. Thus, effective utilization of aquaculture wastes through the IMTA system can mitigate and alleviate the adverse impacts on the environment and aquatic animals.

Table 1. A part of representative targeted fish species and inorganic and organic nutrients extractive species in IMTA. *The initial capital letter of items combined with a number is used to identify the corresponding species, *e.g.*, F1 denotes the finfish *Anoplopoma fimbria*, N13 represents nutrient absorber *Zostera marina*, etc.

Number*	Finfish (F)	Nutrient absorber (N)	Suspension feeder (S)	Deposit feeder (D)	Others (O)
1	*Anoplopoma fimbria*	*Alaria esculenta*	*Argopecten irradians*	*Apostichopus japonicus*	*Anthocidaris crassispina*
2	*Oncorhynchus tshawytscha*	*Ecklonia radiata*	*Chlamys farreri*	*Australostichopus mollis*	*Fenneropenaeus chinensis*
3	*O. mykiss*	*Gracilaria chilensis*	*Crassostrea gigas*	*Cucumaria frondosa*	*Haliotis discus hannai*
4	*O. kisutch*	*G. birdiae*	*C. virginica*	*Holothuria pervicax*	*Pandalus platyceros*
5	*Pagrus major*	*G. lemaneiformis*	*Mytilus edulis*	*Parastichopus californicus*	*Rhopilema esculenta*
6	*Pseudocyanea crocea*	*G. verrucosa*	*M. trossulus*		
7	*Salmo salar*	*Laminaria japonica*	*Perna canaliculus*		
8	*Seriola quinqueradiata*	*Macrocystis pyrifera*	*Patinopecten yessoensis*		
9	*Sparus aurata*	*Porphyra umbilicalis*	*Scapharca broughtonii*		
10	*Takifugu rubripes*	*Saccharina latissima*			
11	*Thunnus orientalis*	*Ulva lactuca*			
12		*U. ohnoi*			
13		*Zostera marina*			

Practice of IMTA

The rudiment of IMTA in fresh water has been found in China centuries ago, which was applied in marine aquaculture and arose the interest of many other countries in Asia (Bangladesh, Japan, Korea), North America (Canada, United States), Europe (France, United Kingdom, Spain), Oceania (Australia, New Zealand), and South America (Chile) in the last few decades (Yang *et al*., 2004; Zhou *et al*., 2006; Buschmann *et al*., 2008; Abreu *et al*., 2009; Ren *et al*., 2012; Yokoyama, 2013; Irisarri *et al*., 2015; Park *et al*., 2015; Fang *et al*., 2016; Shpigel *et al*., 2017). The selection of suitable candidate organisms varies with locations (Table 2); for example, the blue mussel (*e.g*., D5) is Canada's top shellfish aquaculture product and was initially chosen as the organic-extractive species in the Bay of Fundy. However, this species is rarely considered in Asian countries, except for the Sungo Bay of China, where only a small portion (*ca.* ratio of mussel to scallop is 0.14:1) is cultured. Their contrasting food preferences can explain this difference between Canada and Asian countries. The deposit feeder D3, distributed from the Gulf of Alaska to southern California, is considered on the east coast of Canada. At the IMTA sites of China and Japan, D1 is selected as the candidate organism, which is distributed in the northern regions of the Pacific coastal waters. Although both of the candidate organisms have commercial value and the potential ability for extracting relative large organic particles, the local species D1 has a higher market value, with increasing demand in Asia, especially in China. A similar situation can be seen for the inorganic-extractive components. Nutrient absorbers N7 and N10 are mainly concerned by Asia and Europe, respectively. In general, the selection of suitable candidate organisms is decided mainly by the different customs and market value in local areas, indicating that priority will be given to the local species with a high commercial value.

As a productive and environmentally friendly system, the benefit of IMTA practice is summarized and included in Table 2. At several IMTA sites in Kyuquot Sound (Canada), growth trials showed that the blade length of N10 increased from 0.1 to 0.4 m (*ca.* 3.8 times) during a period from April 9th to July 14th. In the Sishili Bay (China), sea cucumber D1 in hanging scallop lantern nets can survive and grow well by ingesting bivalve wastes; the recovery rate of D1 is 114.8% higher than that observed in monoculture. In the western region of Japan, the growth of seaweed N7 around Pacific bluefin tuna F11 cages at IMTA sites was investigated in this study. The blade length of N7 grew prosperously from 0.3 to 2.5 m after 67 days under eutrophicated water environment. In Gokasho Bay (Japan), the species N12 and D1 are co-cultured around and below fish cages. Growth rates of these two species cultured in proximity to finfish cages are 62 and 58% higher than those grown far from finfish cages (control sites), respectively. After 238 days, the wet weight of D1 increased from 0.18 to 160 g, representing that D1 shows its capacity to reduce the aquaculture-derived organic wastes loading, which consequently alleviates the negative impacts of aquaculture operation, and to change harmful debris into a marketable product-yielding extra income without additional inputs. The benefits described above reflect that the extractive organisms in the IMTA system could grow faster than those under control sites, resulting in the reduction of inorganic nutrients and organic particles released from target species loading on the aquatic environment.

Table 2. Summary of IMTA practices in Canada, China, and Japan. [a]Seaweed in dry weight and bivalves with shells. [b]Currently, sea cucumber D4 was decided for co-culture under fish cages but has not been conducted yet.

Country	Location	Candidate organisms					Benefit	Reference
		F	N	S	D	O		
Canada	Kyuquot Sound	F1	N10	S3 S8	D5		Blade length of N10 increased to 3.8 times after 67 days	Blasco (2012)
Canada	Bay of Fundy	F7	N1 N10	S5	D5		Growth rates are 46% (N1, N10) and 50% (S5) higher	Troell *et al.* (2009); Chopin *et al.* (2013)
China	Sungo Bay	-	N7	S2 S5	-	O3	Annual production: 8.0×10^4 t (N7); 1.2×10^5 t (S2, S5, O3) [a]	Shi *et al.* (2011)
China	Sishili Bay	-	-	S2 S1 S3	D1	-	Recovery rate is 114.8% (D1) higher	Zhou *et al.* (2006)
China	Cofferdam in Rongcheng	-	-		D1	O2 O5	Biomass increased after 13 months: 1.3×10^4 (D1), 1.0×10^5 (O5) kg km^2	Li *et al.* (2014)
China	Zhangzidao Island	-	N7	S9 S8	D1	O1 O3	Total production in 2005: 28,000 t, and a net profit of US $18 million	Troell *et al.* (2009)
Japan	Gokasho Bay	F5	N12	-	D1		Growth rates are 62% (N12) and 58% (D1) higher	Yokoyama & Ishihi (2010); Yokoyama (2013)
Japan	Goshoura Island	F5 F10	N7	-	D1	O3	Seaweed cultivation (N7) would be effective for supplying oxygen to water in fish farms at upper layers	Kadowaki & Kitadai (2017)
Japan	West coast of Japan	F11	N7	-	D4[b]	-	The blade length of seaweed (N7) growth rate 3.28 cm d^{-1} (Fig. 2)	This study

Figure 2. The growth of seaweed *L. japonica* at the IMTA sites in the western region of Japan.

Experimental approach to bio-mitigation assessment

When the nutrients released from aquaculture operations are fully balanced by the harvest of the extractive components such as seaweed, mussel, and sea cucumber, the IMTA system can create the largest environmental and economic benefits. Matter and energy fluxes within IMTA and between IMTA and its surrounding environment need to be qualified and quantified to assess the optimal design and bio-mitigation efficiency for achieving sustainability (Reid *et al*., 2009; Chopin *et al*., 2013; Wartenberg *et al*., 2017). An experimental approach can be used as an assessment method.

The recent studies on bio-mitigation assessment through land-based experiments have been summarized in Table 3. The absorption capacity of ammonium (NH_4-N) tends to decrease with increased biomass of seaweed N5 in Case LB1, while such tendency was not observed in Case LB2, where the nutrient absorber was N4 that belongs to the same genus as N5. In terms of phosphate (PO_4-P), a positive relationship between the absorption capacity and initial biomass was demonstra-

Table 3. The assessment of the bio-mitigation effect of IMTA in the land-based experiment. [a]ES means the extractive species used in each study, [b]initial biomass units are g m^{-2} for deposit feeder (*e.g.*, D1, D2) and g m^{-3} for nutrient absorber (*e.g.*, N4, N5), respectively. The bio-mitigation capacity is the difference of each item value between IMTA and control trials at the end of the experiment. [d]TOC and TON represent the total organic carbon and nitrogen in sediment, respectively.

Case	ES[a]	Scale	Initial biomass (g m^{-2} or g m^{-3})[b]	IBR (-)	Bio-mitigation[c]						Reference
					NH_4^+N	NO_2^-N	NO_3^-N	PO_4^{-3}-P	TOC[d]	TON[d]	
					(μmol L^{-1})				(mg g^{-1})		
LB1	N5	Tank (3 m^3)	69.3	0.16	2	-		2.7			Mao *et al.* (2009)
			139.1	0.33	6.7			3.4	-		
			263.5	0.61	6.3			3.8			
			347.6	0.80	5.6			4.1			
LB2	N4	Tank (0.04 m^3)	2,500	14.7	10	-1.1	12.4	-0.1			
			5,000	29.4	6.1	-3.0	24.7	-2.3	-		Brito *et al.* (2014)
			7,500	44.1	13.3	-2.2	23.7	-2.4			
LB3	D1	Tank (20 m^2)	34.6	0.42					3.2	0.3	Zhou *et al.* (2006)
LB4	D1	Net enclosure in Pond (64 m^2)	75.3	4.3		-			5.3	0.53	Ren *et al.* (2012)
LB5	D2	Tank (0.2 m^2)	587.5	-					0.4	0.0	Slater & Carton (2009)

ted in Case LB1, while a negative relationship occurred in Case LB2. From the co-cultured sea cucumber in tanks or ponds, the species D2 showed a relatively low ability to remove TOC and TON in sediments (Cases LB5), although the biomass of D2 is larger than that of D1 (Cases LB3 and LB4). For both seaweed (Cases LB1 and LB2) and sea cucumber (Cases LB3 to LB5), no linear relationship was found between the measured waste removal efficiency by extractive species and its initial biomass, and it corresponds with the results of the study conducted by Troell *et al.* (2009), which emphasized that the removal efficiency is nonlinear with a biomass of extractive species. Moreover, different culture environments (*e.g*., light intensity, water temperature) affect the growth of extractive species, leading to various bio-mitigation capacities. Hence, the waste removal efficiency acquired in a land-based experiment is not suitable in estimating the bio-mitigation efficiency in same or different culture conditions with different biomasses of extractive species. As most of the land-based experiments are conducted under controlled conditions, interactions between co-cultured species and their natural physical and ecological environments cannot be well represented. For example, water flow in a field exerts a great influence on the IMTA system due to the current direction and velocity. Troell *et al.* (2009) mentioned that the attachment of periphyton is influenced by the current velocity in the offshore environment.

A strong current velocity causes a problem for mussels and seaweeds, which need to attach to the ropes or fish cage nets, consequently leading to biomass losses in the IMTA systems (Halling *et al*., 2005). In terms of the current water direction, its importance has already been observed in an integrated kelp and scallop co-culture system. When the direction of long-line ropes is not parallel to that of the water current, the kelps (2 or 3 m in length) may be wrapped up in the scallop nets, resulting in the reduction of nutrients and water flow through the nets. Subsequently, the scallop biomass is reduced due to insufficient food supplies. Also, the diffusion of organic particles and dissolved nutrient concentrations vary as water flow changes (Keeley *et al*., 2013).

On the other hand, the impact of aquaculture facilities on the hydrodynamic environment (*e.g*., water current reduction) has been reported (Shi *et al*., 2011). Thus, the experiments conducted in the real sea condition that may reflect such interactions are summarized and included in Table 4. The growth rate of seaweed N3 placed 10 m east of fish cages was higher than that placed in other directions (specific growth rate SGR is 8.6% d^{-1}, Case OP2), while in Case OP3, SGR of N3 placed 800 m north of salmon cages was slightly higher than that placed 100 m in the opposite direction, showing better performance both in productivity and nitrogen uptake. Under different fish cages (Cases OP4 and OP5), SGRs of sea cucumber D1 are 1.3 and 2.2, which can be explained by diffusion of particulate organic and dissolved inorganic wastes due to water flow; the water flow unevenly distributes the waste in the water column and on the sea bottom, respectively. Keeley *et al*. (2013) found that the observed distribution range of organic wastes at a high-flow site (mean current velocity 19.94 cm s^{-1}) was more extensive and more diffused than that at a low-flow site (mean current velocity 3.58 cm s^{-1}) in Marlborough Sounds, New Zealand. At a high-flow site, the organic waste extends farther away from the center of fish farm to the south and west; it is mainly accumulated around the fish cages towards the southeast direction. Open water provides a well-balanced experimental condition,

Table 4. The assessment of the bio-mitigation effect of IMTA in open sea experiment. [a]TS and ES denote the target and extractive species used in the study, respectively. [b]SGR of extractive species is presented. '10 m, East, 8.6' means the SGR is 8.6 when extractive species is placed at 10 m east of fish cages.

Case	TS/ES[a]	Initial biomass of target/extractive species (kg farm^{-1} or cage^{-1})	IBR (-)	SGR (% d^{-1})[b]	Bio-mitigation (μmol L^{-1})	Reference
OP1	F6/N6	373.5/1.2 (kg cage^{-1})	0.003	8.3	NO_2^--N 0.3 NO_3^--N 3.8 NH_4^+-N 11.7 PO_4^{-3}-P 0.3	Huo *et al.* (2012)
OP2	F3, F4/N3	30/0.4 (MT/kg farm^{-1})	1.4×10^{-5}	10 m, East, 8.6 10 m, North, 7.9 10 m, South, 6.7 10 m, West, 5.6	-	Troell *et al.* (1997)
		40/0.3 (MT/kg farm^{-1})	6.9×10^{-6}	10 m, West, 6.5 150 m, West, 4.3		
OP3	F7/N3	-/2.3 (-/kg farm^{-1})	-	100 m, North, 4.0 800 m, South, 4.2	-	Abreu *et al.* (2009)
OP4	F5/D1	3/0.006 (MT/kg cage^{-1})	1.9×10^{-6}	1.3	-	Yokoyama (2013)
OP5	F8/D1	-/0.003 (-/kg farm^{-1})	-	2.2	-	Yokoyama *et al.* (2015)

which facilitates a proper examination of the complex interactive processes that connect the biomass, nutrient uptake, and nutrient concentration of a balanced IMTA system (Troell *et al*., 2009). However, the measurement of bio-mitigation efficiency by co-cultured aquatic species is currently restricted (Table 4).

The initial biomass ratio (IBR) values in open-water experiments are lower than that observed in land-based experiments (Tables 3-4) because the biomass of extractive species is quite smaller compared with target ones in open-water conditions. Bio-mitigation efficiency may not be easily measured, especially when the IBR value is <0.003. Thus, to evaluate the overall bio-mitigation effect, the IMTA experiment with an optimal design is required, in which the IBR of extractive to target species and the location for co-culture species are adequately considered, to mitigate aquaculture wastes produced around the fish farm at the maximum.

Numerical approach

A mathematical model is a potential tool to understand the high complexity interactions among physical, biochemical, and hydrodynamic characteristics in an IMTA system, which is difficult to achieve by the traditional small-scale experiments (Troell *et al.*, 2009; Ren *et al.*, 2012). Therefore, to some degree, a mathematical model developed based on the evidence from laboratory experiments under controlled conditions and validated by the small-scale field observation data at IMTA sites, constitute a useful approach to determine the overall bio-mitigation efficiency of the IMTA system. The validated model could be further used to understand the local physical, biochemical, and hydrodynamic characteristics and the physiological and metabolic features of co-cultured organisms for optimizing the IMTA design (Chopin *et al.*, 2013).

The mathematical model of IMTA aims to describe the interactions among co-cultured species and the interplay between these aquatic species and their surrounding water environment. The model should consist of a hydrodynamic, ecosystem, and individual-based components. In the last few years, many numerical models have been developed for simulating the IMTA system (Table 5). A 3D physical-biological coupled aquaculture model (NM4) was introduced and applied at the Sungo Bay to study the carrying capacity of kelp under bivalve rafts. The carrying capacity indicates the maximum amount of a species that the IMTA system can sustain, which is an important indicator for the IMTA design. Model NM6 represented a generic IMTA ecosystem model using 2D finite element hydrodynamic component, providing a research tool to fine-tune the design of field trials to optimize yields from each trophic level. This model (NM6) focuses on finfish-shellfish-detritivore primary producer systems, including salmon, mussels, sea cucumbers, seaweed, and material cycling in their ambient environment. These models presented in Table 5 focuses on different aspects of IMTA for the variety of research purposes. Consequently, various required details for IMTA model are not presented.

In the hydrodynamic field, many 3D ocean tidal and current models have been developed to simulate the essential conditions such as tide, water flow, water temperature, salinity, etc. (Kitazawa & Yang, 2012; Chen *et al*., 2014). In IMTA models (Table 5), except for 1D model and the model without consideration of the physical environment, most studies have adopted

Table 5. Potential numerical models for the bio-mitigation assessment of IMTA.

Numerical model	Hydrodynamic component		Ecosystem component			Individual-based component	Reference
	Hydrodynamic	Aquaculture facility	Pelagic	Benthic	Aquaculture wastes		
NM1	2D	-	✓	-	-	2	Duarte *et al*. (2003)
NM2	1D	-	✓	✓	-	2	Aveytua-Alcázar *et al*. (2008)
NM3	2D	-	✓	Partly	-	1	Grangeré *et al*. (2010)
NM4	3D	✓	✓	-	-	1	Shi *et al*. (2011)
NM5	-	-	-	Partly	✓	2	Ferreira *et al*. (2012)
NM6	2D	-	✓	Partly	-	5	Ren *et al*. (2012)
NM7	3D	-	✓	Partly	-	1	Broch *et al*. (2013)
NM8	-	-	-	-	Partly	3	Hadley *et al*. (2015)
NM9	2D	-	✓	-	-	1	Filgueira *et al*. (2014)
NM10	3D	✓	✓	Partly	✓	2	Zhang & Kitazawa (2016)
NM11	-	-	-	✓	✓	4	Cubillo *et al*. (2016)

2D (NM1, NM3, NM6, and NM9) or 3D (NM4, NM7, and NM10) version. However, the stratification of water temperature, salinity, and dissolved oxygen during warm seasons may not be well represented in 2D models when the average depth is used for vertical direction. As a significant effect factor of the hydrodynamic environment, aquaculture facilities suspended in the IMTA area have shown that the water flow tends to slow down due to the extra drags caused by facilities such as rafts and/or ropes of mussels, scallop, or kelp (Shi *et al*., 2011; O'Donncha *et al*., 2013). The water exchange rate is reduced to approximately 59% due to the increased bottom friction in suspended aquaculture of marine bivalves' area (Grant & Bacher, 2001), indicating that if the drag of aquaculture facilities is ignored, the water exchange rate and renewal of nutrients and food supply will be overestimated (Shi *et al*., 2011; Cranford *et al*., 2014). Some studies have presented the modeling of the drag caused by long-line or raft structures (Plew, 2011; O'Donncha *et al*., 2013). The drag caused by fish cages has been studied in water tank conditions, and the simplified form of drag force has been applied to a 2D model based on a quadratic drag law (Kristiansen & Faltinsen, 2012; Shimizu *et al*., 2018). However, the impact of an aquaculture facility on the physical environment is rarely considered in most IMTA models (Table 5). Drags due to raft structures at the surface with a blade length of kelp in the water column and cylinder-shaped fish cages with square mesh nets are simulated in models NM4 and NM10, respectively. A relatively simplified equation is used to estimate drag force in these two models. In the field, the facilities of fish farms have different forms and sizes. In the case of fish cages, drag force changes with its shape (*e.g*., cylinder, hexagon, rectangle, etc. with rhombus square mesh nets) and size as the project area of the fish cage are varied. Further-more, some effect factors such as deformation of nets, sessile organisms on nets, the wake of nets, and swimming of finfish cause uncertainty in the drag force estimation in the real sea, but these factors cannot be easily considered in numerical models.

Marine ecosystem models describing the biogeochemical fluxes and material cycling have been developed and applied to assess environmental impact and carrying capacity of aquatic farming systems during the last several decades (Perrot *et al*., 2014). Since aquaculture wastes are transported within the pelagic ecosystem and deposited at the sea bottom, a significantly negative impact on the local environment was observed, especially on the benthic ecosystem. Yokoyama *et al*. (2006) concluded that as the volume of aquaculture wastes increases, the concentration of dissolved oxygen decreases logarithmically and found that the water was nearly anoxic at stations where the content of aquaculture-derived nitrogen in bottom sediment reached 2 mg g^{-1} in the inshore area, where the water depth is approximately 18.5 m. Some aquaculture wastes models have been developed as predictive or explanative tools that seek to predict the ecological impact associated with fluxes of waste materials from aquaculture area (Reid *et al*., 2009; Keeley *et al*., 2013). Benthic and aquaculture wastes are incorporated or partly considered into some models (Table 5), although the pelagic ecosystem is included in the majority of the IMTA models.

As the core part of the IMTA system, co-cultured species should be well simulated to understand the interactions between them under natural water environments. Various species from multiple trophic levels such as inorganic nutrient absorber and organic-extractive organisms have been considered as the candidate organisms to be co-cultured with targeted species (Table 1). However, only one or two species are considered in the existing IMTA models, except for NM6 and NM8 models, in which five and three species are included, respectively (Table 5). In general, three species from different trophic levels (targeted organism, inorganic, and organic extractive organisms) should be taken into account for balancing the material cycling in

Table 6. Consideration of the IMTA candidate organisms in numerical models.

Numerical model	Individual-based component				
	Finfish (F)	Nutrient absorber (N)	Suspension feeder (S)	Deposit feeder (D)	Others (O)
NM1	-	N7	S2, S3	-	-
NM2	-	N11, N13	-	-	-
NM3	-	-	S3	-	-
NM4	-	N7	-	-	-
NM5	F9	-	S3	-	-
NM6	F2	N2	S7	D1	O5
NM7	-	N10	-	-	-
NM8	-	N8, N9, N11	-	-	-
NM9	-	-	S4	-	-
NM10	-	N12	-	D1	-
NM11	F7	N1	S3	D5	

the IMTA system. However, the same trophic level species are included in models NM2 and NM8, as well as in models NM3, NM4, NM7, and NM9, which could be regarded as only a single level in the model. A comprehensive consideration of trophic levels was only found in the NM6 model. The common individual-based model relies on the Scope for Growth (SFG) concept (Bayne & Newell, 1983) or the Dynamic Energy Budget (DEB) theory (Kooijman, 2010). Both SFG and DEB have been fully developed for various species and successfully applied to many growth models, *e.g.*, SFG for S5 (Kitazawa *et al.*, 2008) and DEB for F11 (Jusup *et al.*, 2011), but they are not integrated into the IMTA model yet (Table 6). With further insight into the candidate organisms used in practice, the individual-based models for these species are still not sufficient. Therefore, a database is required for an individual-based model of IMTA candidate organisms.

Since a relatively identical mathematical model has not been formed for the fully balanced IMTA system yet, it is difficult to achieve a highly accurate assessment of IMTA bio-mitigation efficiency based on the existing IMTA models. Further investigations are required to establish a model framework that fully considers natural biogeochemical fluxes within IMTA and between IMTA and its surrounding environment.

CONCLUSIONS

In conclusion, the selection of the suitable candidate organism mostly depends on the customs and market values in the local IMTA practice, and priority should be given to the local species with high commercial values. To evaluate the overall bio-mitigation effect based on the experimental approach, an optimally designed large-scale IMTA experiment is required, in which the IBR of extractive to target species and location for co-culture species are adequately considered. It is difficult to achieve a highly accurate assessment of IMTA bio-mitigation efficiency based on the existing IMTA models. Therefore, to develop an efficient bio-mitigation strategy for global environmental impacts caused by aquaculture wastes, a full-scale IMTA model should be further improved by creating: 1) a model framework that considers natural biogeochemical fluxes within IMTA and between IMTA and its surrounding environment and 2) a database of individual-based sub-models for IMTA candidate organisms.

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