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The sanitation service of seagrasses – Dependencies and implications for the estimation of avoided costs

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ABSTRACT

Seagrasses are capable of sanitizing coastal seawaters polluted by fecal bacteria. In this work, the reduction of *Enterococci* concentration in the presence of a seagrasses' assemblage (Pacific Ocean) was related to the decrease in the probability of gastroenteritis. A linear model fitted to data extracted from the literature showed a 20% reduction of this probability in the presence of these plants. Seagrass sanitation effect was estimated to allow avoiding ca. 24 million gastroenteritis cases/year, globally. Considering a global cost of gastroenteritis of ca. US$ 372 million/year, the global avoided cost, assuming that the sanitation service was always effective, was estimated to be ca. US$ 74 million/year (2020 US$). The seagrass sanitation effect appears genera/geographic dependent, and the targeted pathogen may change as well. Thus, the global estimates were roughly adjusted, obtaining conservative figures of ca. 8 million avoided cases/year and ca. US$ 24 million/year of avoided cost. Considering the importance of this Ecosystem Service (ES) for public health and the potential global spreading of diseases driven by climate change, further research is needed to ascertain the scope of this seagrass ES worldwide.

1. Introduction

Marine seagrasses have many ecological functions and provide several Ecosystem Services (ESs; Cullen-Unsworth et al., 2014; Nordlund et al., 2016; Ugarelli et al., 2017; Nordlund et al., 2018). The most recent observed function is a natural biocide action that generates the ability to remove microbiological contamination. Such an ability was demonstrated recently by associating the presence of seagrasses with an effective reduction of the presence of certain fecal bacteria (i.e., *Enterococci* CFU/100 ml[1]) in coastal waters (Lamb et al., 2017). These bacteria are a proxy for more dangerous pathogen species responsible for diseases in humans, fish, and invertebrates. In addition, *in vitro* studies have demonstrated the antibacterial potential of many seagrasses against human pathogens (Alam et al., 1994; Choi et al., 2009; Mayavu et al., 2009; Kannan et al., 2010). These findings are exactly the opposite of the evidence found by Grant et al. (2001), showing that saltwater marsh sediments and vegetation (mostly *Salicornia virginica*) may be relevant sources of *Enterococci* bacteria that then leak into the closest surf zone. The type of vegetation and the geomorphological, chemical, and physical characteristics of a coastal site does matter in the sanitation role that aquatic plants may play (Palazon et al., 2018). In contrast to the observations by Lamb et al. (2017), Webb et al. (2019) did not find conclusive evidence of microbial contamination reduction by the seagrass *Zostera* spp. in the coastal waters of San Diego (CA, USA). Interestingly, Palazon et al. (2018) found, in a microbiological screening of the coastal waters of 270 Spanish beaches, a not statistically significant reduction of *Enterococci* concentration (CFU/100 ml) in the presence of *Posidonia oceanica*, although the presence of *Posidonia* significantly reduced the concentration of *Escherichia choli*.

Thus, scanning the current literature, conclusions about the role of

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[1] CFU/ml: Colony Forming Units per milliliter is a unit used in microbiology. It gives an estimate of the viable microorganisms (e.g., bacteria) present per milliliter of a sample and thus in the environment form which the sample was taken. It refers to the number of colonies (i.e., populations of cells visible to the naked eye) of any microorganism that grow on a plate (usually a Petri dish) of media.

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aquatic vegetation in sanitizing local conditions are controversial. Such controversial conclusions suggest that there might be some limits to the anti-pathogen action played by seagrasses, and qualitative differences may be linked to the pathogen type and seagrass genera and/or species. It is likely that, given a specific pathogen organism, there is a range of bacteria concentration within which some plant genera provide an active sanitation service. Moreover, above a certain threshold, seagrasses may cease to affect pathogens. Instead, they may start suffering a population depletion due to the high concentrations of bacteria in water (e.g., Elliott et al., 2006; Jones et al., 2018), and this, in turn, would further reduce the potential sanitation effect these plants may ensure (Liu et al., 2018).

The seagrass assemblages capable of exerting a sanitation effect included mainly *Enhalus acoroides*, *Thalassia hemprichii*, and *Cymodocea rotundata* (Lamb et al., 2017, in their Supplementary Material), i.e., the typical seagrass assemblages of the Indo-Pacific Area, according to Short et al. (2007). In contrast, Webb et al. (2019) nonconclusive evidence of a seagrass anti-pathogen effect occurred in a different geographical area (San Diego, California, USA) where *Zostera* spp. seagrasses were the dominant genus/species (i.e., the Temperate North Pacific seagrasses' biogeographic region, according to Short et al., 2007) while *P. oceanica* was capable of significantly reduce *Escherichia coli* (but not *Enterococci*) concentration in the Mediterranean Sea (Palazon et al., 2018). Thus, the seagrass sanitation-effect ES seems to be, as well as many other seagrass ESs, genera/geographic area dependent (Nordlund et al., 2016), and it seems to occur, at least given the (limited) search effort to date, only in two of the six seagrasses' biogeographic regions considered by Short et al. (2007). The work of Lamb et al. (2017) was entirely devoted to the biological aspects of the anti-pathogen ES that marine seagrasses may provide to the advantage of several species, humans included. Although those authors emphasized the importance of this ES for human well-being, they did not estimate the potential reduction of the known global economic burden of thalassogenic diseases (Shuval, 2003).

It has been estimated that 380 billion m^3 (380 trillion l) of municipal wastewaters are produced globally every year (i.e., five times the volume of Niagara Falls each year, Qadir et al. 2020). 40–80% of these waters are discharged untreated (Hernández-Sancho et al. 2015) and carry (directly or indirectly) an enormous flux of pathogens into the sea (Tuholske et al., 2021), often even from the same wastewater plants meant at abating them where they instead survive and thrive (Newton and McClary 2019). Polluted seawaters have become thus a source of serious global public health concern (Pougnet et al. 2018; Weiskerger et al., 2019; Tuholske et al., 2021). For instance, a recent meta-analysis of 19 papers has shown that, in high-income countries, the immersion (e.g., bathing) in polluted seawaters may cause an increased risk of experiencing symptoms of various illnesses including respiratory, ear, eye, skin, and gastrointestinal infections (Leonard et al., 2018).

Sea water-borne diseases may affect both recreational and occupational users of marine waters and may target only a few people occasionally or explode yearly in severe larger out-breaks heavily impacting the health of many people (Henrickson et al., 2001). Infant and children may be particularly affected by seawater-borne pathogens (Arnold et al., 2016; DeFlorio-Barker et al., 2018; Verhougstraete et al., 2020, and Supplementary Materials, SM1). In high-income countries, a large part of thalassogenic diesases cost is attributable to sanitary treatment and work absenteeism while social costs and death toll are relatively low. Conversely, in low- and middle-income countries, social costs and death toll, especially among children, are the major burden (Bartsh et al., 2016; Jamison et al., 2018; UNO-DESA, 2020). Seagrasses' sanitation capability could play a role in mitigating this (these) cost(s) and thus it might be a valuable ES.

As for the other seagrasses' ESs, although many gaps still affect their economic valuation (Dewsbury et al., 2016; Himes-Cornell et al., 2018), several estimates can be found in an extensive literature, namely: i) the seagrass contribution to fishery estimates span from a minimum (min) of ca. US$ 33.8/ha/year to a maximum (Max) of US$ 47,232/ha/year (SM2) while for the whole Mediterranean, estimates based on residency of species in the seagrasses gave € 58–91 million/year for commercial fishery, and € 112 million/year for recreational fishery (Jackson et al. 2015); ii) seagrass carbon sequestration was estimated to provide, in various areas of the world, a min US$ 414,159 and a Max US$ 7 million (SM2) while carbon stocked in the currently existing European seagrass beds was estimated in US$168,749,727 (EU Allowances price 2012) (Luisetti et al., 2013), and US$ 19 million of extra-local avoided SCC (Social Carbon Costs as avoided damages of a unit reduction of CO_2 or its equivalent emissions) was the estimated value for the carbon stored in Gazy Bay, Kenya, with China, Europe, and the USA as the main beneficiaries (De los Santos et al., 2020); iii) Campagne et al. (2015) estimated that *Posidonia oceanica* coastal protection from erosion service provides € 188/ha/year while in the Caribbean (Martinique) the same service was estimated in € 12,100/ha/year (Failler et al., 2015); iv) wastewater treatment value was estimated to be € 60/ha/year in the Mediterranean Sea (Campagne et al., 2015) and € 1,100/ha/year in the Caribbean (Martinique) (Failler et al., 2015); v) the contribution to touristic activities was valued in € 7,800/ha/year in the Caribbean (Martinique) (Failler et al., 2015); vi) cultural service (such as research/knowledge contribution) estimates were € 0.33/ha/year, in the Mediterranean Sea, (Campagne et al., 2015) and € 210,000/year, in the Caribbean (Failler et al., 2015).

However, contrary to the above seagrasses' ESs, sanitation power has not only received much less attention as an ES *per se* (as far as we know, only three papers have addressed this ES), but it is also clear that the estimates of the health and economic benefits this peculiar ES may provide to human beings are lacking. While there is a need for more research to disentangle the actual role seagrasses may play in facilitating sea sanitation, significant details about the potential reduction of health and economic damages are necessary too. The present work thus primarly aims to make a first step toward the filling of this gap.

There is, moreover, another important aspect that motivated the present study. The recent (and ongoing) COVID-19 (SARS-CoV-2) pandemic indicates that damaged and stressed unhealthy ecosystems may favor both the spillover of pathogens from animals to humans and the spreading of infectious diseases in human populations (Dobson et al., 2020; Guégan et al., 2020). Healthy ecosystems may protect against disasters often labeled as "natural" while, in reality, triggered by human-induced environmental dyscrasias. This misconception underlines the importance of a unitary approach to public health policy, the so-called one-health approach, i.e., healthy humans in a healthy environment (Schmiege et al., 2020, and references therein; Gillespie et al., 2021). Such an approach sees natural systems' conservation and restoration as strategic tools useful to control pathogens in natural populations and prevent their spillover and/or spreading in humans.

However, nature conservation may often appear a useless constraint to economic development, and restoration is frequently considered too expensive. They are believed, thus, not worth being pursued. This happens because we still have, in many cases, only poor or even none estimates of the economic benefits healthy ecosystems may provide, although many valuable efforts have been made to overcome this lack of knowledge (e.g., Millennium Ecosystem Assessment 2005 https://www.millenniumassessment.org/en/index.html, TEEB Home – The Economics of Ecosystems and Biodiversity http://teebweb.org/; IPBES https://www.ipbes.net/; Ecosystem Services Partnership https://www.

es-partnership.org/). Well-functioning ecosystems may allow avoiding the much greater costs we have to face when these natural systems are damaged, stressed, and ill-functioning (e.g., TRUECOST Report, 2013; ISO14008, 2019). In the Covid-19 pandemic case, for example, it turned out that the present value of the cost of the measures that could have helped to prevent the pandemic (and may still help to prevent the occurrence of further analogous events) would be (is), in ten years, only 2% of the huge economic burden humanity is facing because of the virus spreading (Dobson et al., 2020).

In this context, the effort we pursued here to estimate the sanitary costs seagrasses may help to avoid meant at providing information that can be essential (along with that concerning other benefits provided by seagrasses' ESs) in costs/benefits analysis (e.g., Markandya, 2016) aimed at valuing the convenience of conservation and restoration policy of the seagrasses viewed as a "public health device" capable of curbing pathogens concentration in the seawater reducing thus the number of cases and the consequential sanitary and welfare/social costs.

Thus, here we provide an attempt "first-order" estimate of such a putative potential reduction of cases and economic damages through estimates of the avoided cases and cost that the seagrass "water-sanitation" ES may ensure for human well-being. Specifically, we determined the costs that would have been incurred in the absence of seagrasses as a natural anti-pathogen provider (according to TEEB, 2010). Our specific aims were: i) to estimate the reduction of the risk of catching a specific seawater-borne disease when a given seagrass assemblage is present, ii) to estimate the reduction in the number of the cases per year the seagrass sanitation effect may ensure globally, iii) to attempt a rough estimate at a global level of the avoided cost ensured by this ES, and finally, iv) to adjust this estimate taking into account as much as possible the influence of seagrass taxonomy, geographic distribution, and ecosystem/biodiversity integrity on their sanitation properties, worldwide.

2. Material and methods

2.1. Rationale and approach key assumptions

In Fig. 1, we presented the various pathways of potential seagrass-mediated cost reductions, with the ones considered in this work shown in red/orange, i.e., we focused here on the seagrass-mediated reduced cost of Gastroenteritis Ilness (GI, SM3) which is a portion of this larger framework characterized by different pathways that seawater-borne pathogens follow from the source (point or diffuse) to the other living organism, humans, and animals, they may infect.

Untreated and even treated wastewaters are the major sources of pathogens worldwide (far-left box, Fig. 1, Hernández-Sancho et al., 2015; Newton and McClary, 2019). Once entered seawaters, pathogens may resist in inactive states and deposit into the sediments (Pandey et al., 2014; Weiskerger et al., 2019) or, as it has been shown recently, attach to plastic macro and micro debris (Bowley et al., 2021). In this state, they may wait for good environmental conditions to happen and then thrive, reaching high concentrations at which they may become very infective for both humans and other vertebrate and invertebrate species. They may also concentrate within edible organisms (fish, shellfish, and crustaceans). The larger left box – "Bacteria/viruses diseases" box – including the three smaller boxes "Edible fish, shellfish, crustaceans, etc."; "Enterococci CFU/100 ml"; and "Corals & other invertebrates, fish, diseases", represents the mentioned above conditions (Fig. 1). *Enterococci* concentration is a proxy of the much more aggressive pathogens capable of affecting people (Kay et al., 1994; Kay et al., 2004).

From the three seawater sources of pathogens, pathways through which human's health and wealth are affected both directly and indirectly branch off (the three boxes at right in Fig. 1), i.e.: i) the contaminated seafood, ii) the immersion (bathing, surfing, canoeing, etc.) into and the direct ingestion of polluted waters where infective bacteria and viruses are thriving, iii) the coral bleaching and other invertebrates and fish mortality causing ecosystem damages. Both the ingestion of contaminated seafood and the increased risk of getting sick (specifically of getting GI) by water immersion and ingestion imply healthcare and welfare costs estimated to be locally and globally relevant. Moreover, the increased risk of contracting a disease and the ecosystem damages may lower the appeal of previously attractive beaches and coastal recreational areas (small, rounded rectangle in Fig. 1).

The result (far-right box in red, Fig. 1) is an increase of damages to human well-being which translates into a burden of economic and social costs including, sometimes, a heavy death toll, depending on the different income and socio-economic conditions of the different areas of the world. According to Lamb et al. (2017) results, seagrasses' sanitation effect (rounded rectangle in the uppermost of Fig. 1) may play an important role in reducing the concentration of the pathogen (represented by *Enterococci* CFU/100 ml) thereby reducing each of the three sources of damages shown in Fig. 1 (this depressive effect is represented by three rounded-head arrows interfering with the arrows stemming from the three different pathogens boxes, Fig. 1).

We focused here on the specific effect that *Enterococci* as a proxy of other pathogens may have in increasing the risk of getting GI via immersion in or ingestion of seawater polluted by these organisms (red letters and red arrow in Fig. 1). We extracted from the literature both the global GI economic cost and the number of GI cases per year occurring globally (far-right red box in Fig. 1). Then we estimated the seagrass-mediated avoided GI cases and costs, i.e., the avoided cases and costs seagrasses may ensure through their depressing effect on *Enterococci* and thus on pathogens concentration (red rounded-head arrow stemming from a rounded uppermost rectangle in Fig. 1).

We tackled this specific subset of the broader problem sketched in Fig. 1 because reliable data and information that can be used to make the combination of bio-ecological, epidemiological, and economic computations needed to scale the seagrasses-mediated avoided clinical cases and cost estimates up to a global level are readily available, as far as we know, only for *Enterococcus* and GI.

2.2. Lower probability of contracting gastroenteritis due to the presence of seagrasses

To estimate the reduction of the probability of catching GI when seagrasses are present, we relied on the best available knowledge by combining two data sets obtained from the literature. Namely, the UK epidemiological data of *Enterococci* concentration (CFU/100 ml) vs. the Probability of GI (PGI, derived from the works of Kay et al., 1994; and Fleisher et al., 1996, and reported in a paper by Ashbolt and Bruno, 2003), and the average *Enterococci* (CFU/100 ml) concentrations that Lamb et al. (2017) found in a tropical sand flat where a specific seagrass assemblage was either present or absent.

The following data have been extracted from literature (by Using Plotdigitizer[2]): i) from figure 1 in the paper by Ashbolt and Bruno (2003), the UK epidemiological data, i.e., 17 couples of points (x, y), where $x =$ *Enterococci* concentration CFU/100 ml, and $y =$ probability of GI, and 17 (x, y) data-points of a dose–response relationship, namely the Max Risk Model (MRM), that Ashbolt and Bruno fitted to these same data set (which is more reliable for estimating a dose–response relationship than other data sets for the reasons specified in Kay et al., 2004); ii) from figure 1 of Lamb's et al. (2017) work, 2 average *Enterococci* concentrations (CFU/100 ml) when seagrasses were either present or absent (in a sea flat).

Following the footsteps of Kay et al. (1994), we fitted a linear equation to the *Enterococci* concentration (CFU/100 ml) vs. the

[2] Plotdigitizer is a computer program that allows to digitize data points off scanned images of plots (in GIF, JPEG, or PNG format). Just by clicking on each data point, one can acquire the desired data that then get stored in a text file.

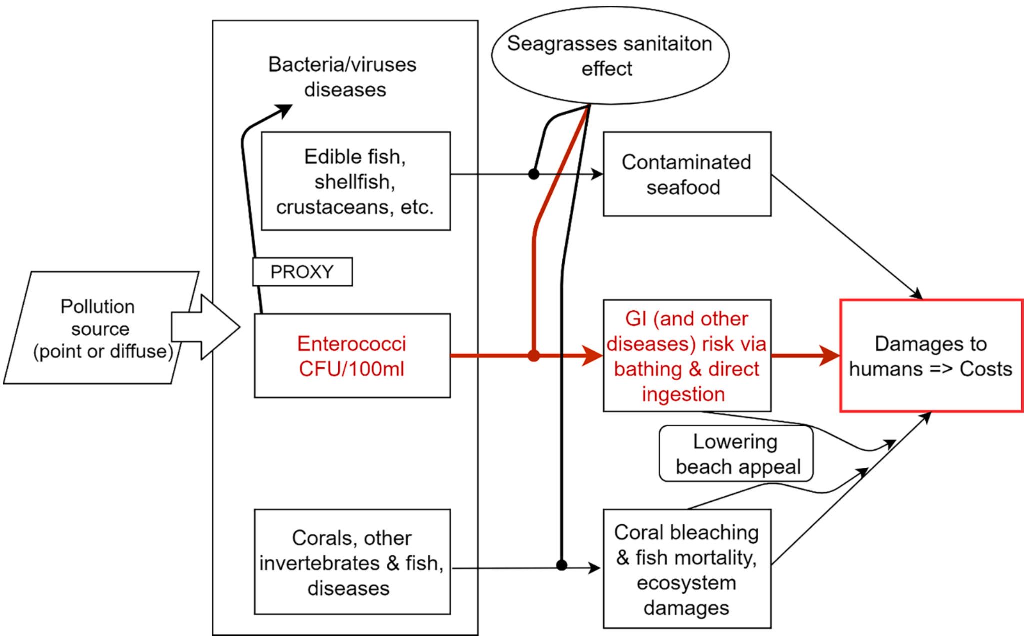

Fig. 1. Pathways of infection of humans and other organisms, and a cascade of direct and indirect damages to human health and the economy, i.e., costs (according to the findings of Lamb et al., 2017). The direct cost reduction (i.e., avoided cost) estimated in this work is presented in red–orange. The horizontal arrows indicate the pathways and their direction along with the negative effect on human health and economic earnings. The arrows stemming from the seagrass sanitation effect box represent the depressing antibiotic action that seagrasses may exert on either human or non-human pathogens. This action implies a reduction of the adverse effects on human health and economic earnings and, therefore, a decrease in economic costs, i.e., an increase of avoided costs (realized through Diagrams.net).

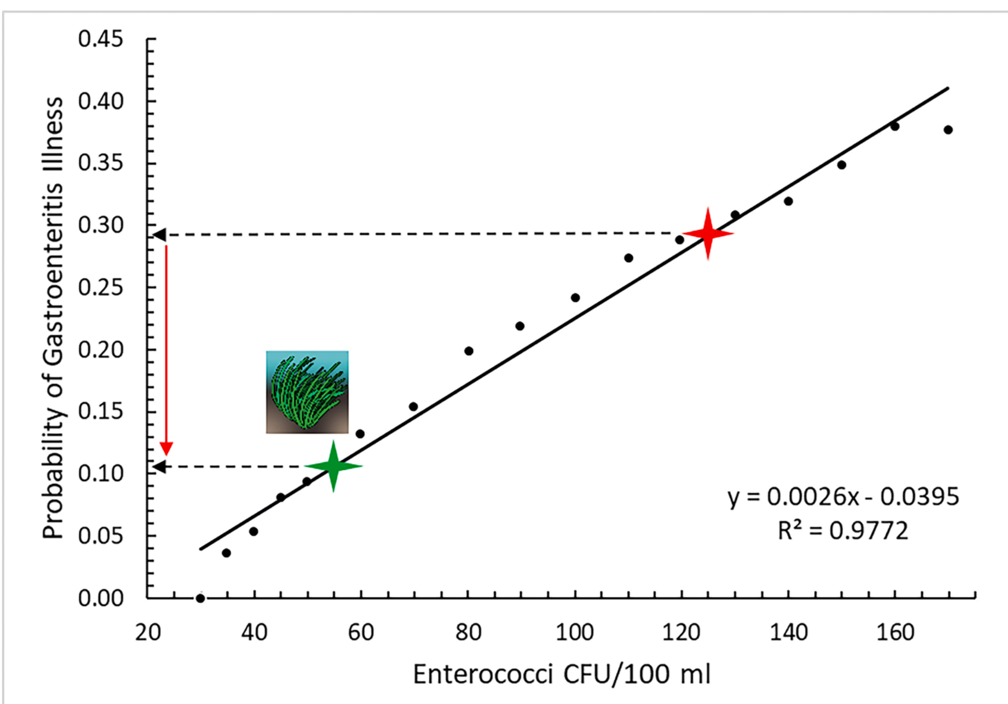

Fig. 2. The dose–response linear fit of UK epidemiological data (black dots) for Gastrointestinal Illness (GI) (data plot redrawn from Ashbolt and Bruno 2003 after Kay et al. 1994). Plotted are also the Lamb's et al. (2017) averages data for *Enterococci* concentration (CFU/100 ml) in the absence – a red star – and in presence – a green star and a seagrass icon – of seagrasses. The red arrow indicates three folds drop in the probability of GI in the presence of a seagrasses' assemblage.

probability of GI data and obtained a simple dose–response relationship, and then plotted it in Fig. 2 (an approach like that of Cabelli et al., 1982). In Fig. S4, we plotted the MRM data-fit we had extracted from the Ashbolt and Bruno (2003) paper (SM4). Then, we plotted the *Enterococci* concentration averages that we had extracted from Lamb et al. (2017) on these two graphs (i.e., Fig. 2 and S4) neglecting the confidence intervals (CI) because CIs are not the actual minimum or maximum values of the data set.

Thus, we found the Probability of catching GI (PGI) in the absence or the presence of the seagrass assemblage occurring in the tropical area where Lamb et al. (2017) carried out their research. That is to say, an estimate of the probability of getting sick (through immersion) when these plants are absent, PGI_{Sa}, or present, PGI_{Sp}.

This probability was estimated using a simple linear model fitted to the "crude" probability of getting GI (values derived from UK epidemiological data set) and this might lead to underestimating the actual risk of catching the disease (Pond, 2005). However, Ashbolt and Bruno (2003) fitted the same set of data using an appropriate risk model, i.e., the Maximum Risk Model, obtaining a particularly good agreement between the data and the model for higher doses (higher *Enterococci* CFU/100 ml) but not for lower ones (Fig. S4).

MRM is a (single-hit) risk model of the form $P_{inf} = (1\text{-}e^{-rD})$, i.e., an exponential model where P_{inf} represents the probability of infection of a host, r is the probability, assumed as constant, that a pathogen will survive all host barriers and colonize it, and D is the mean ingested dose of pathogens the host is exposed to (FAO/WHO, 2003). It gives the upper confidence level of a dose–response relationship taking adequately into account the fact that the risk cannot exceed the probability of exposure (Teunis and Havelaar, 2000). Ashbolt and Bruno (2003) MRM good fit was obtained (for higher doses) assuming r constant and equal to 1, D = 50 ml, an *Enterococci*: viruses ratio equal 1:175, and the number of people getting ill equal to half (50%) of those infected (not all exposed people that get infected also get ill).

The fact that MRM fits well the UK epidemiological data for the higher *Enterococci* doses but overestimates the probability of getting GI for lower ones (Ashbolt and Bruno, 2003) implies that the probability of getting GI should be considered reliably estimated only for higher doses. A different risk model, such as a threshold one, might perform better since it also account for the observed variation of the probability of getting GI at lower *Enterococci* doses, as stressed by Ashbolt and Bruno (2003). However, our linear dose–response model is a simple and reasonable compromise that allowed us to estimate the probability of getting GI at both lower and higher doses.

2.3. Gastroenteritis illness global cost

Shuval (2003) estimated the number of Global thalassogenic GI Cases/year (GGICs/year) and then the Global (thalassogenic) GI Cost/year (GGIC/year) through a DALY approach i.e., the Disability Adjusted Life Years method. Such an approach gives the economic estimate of global GI cost/year (GGIC/year) as the number of DALYs/year imputable to GI times the estimated cost of each DALY.

DALY is defined as follows: "One DALY represents the loss of the equivalent of one year of full health. DALYs for a disease or health condition are the sum of the years of life lost due to premature mortality (YLLs) and the years lived with a disability (YLDs) due to prevalent cases of the disease or health condition in a population" (WHO https://www.who.int/data/gho/indicator-metadata-registry/imr-details/158). Details on DALYs estimation procedure can be found in Devleesschauwer et al. (2014).

We relied on these Shuval's (2003) global estimates, although we were aware that they are only rough "first-order", likely underestimated, values (as Shuval himself stated in his 2003 paper). Thus, in our successive computations, we used the following Shuval's estimated quantities: i) the number of Global thalassogenic GI Cases/year (GGICs/year), ii) the number of DALYs/year imputable to GI (DALYs/year), iii) the cost per DALY in 2003 US$, iv) the Global (thalassogenic) GI Cost/year (GGIC/year). We adjusted these values for inflation and gave them in US $ of 2020.

The above estimates implicitly include the sanitation effect of the presence of seagrasses. The epidemiological phenomena Shuval (2003) scrutinized (the Global GI Cases/year) were observed in coastal areas around the world where seagrasses are present and may (or may not) provide a sanitation service.

2.4. Estimates of global avoided cases and cost

To estimate the Global GI Avoided Cost/year (GI GAC/year in 2020 US$ dollars) delivered by the seagrass sanitation effect, we assumed that, if seagrasses were absent at global scale, the number of Global GI Cases, GGICs/year, would be subject to a net increase **Δ**GGICs/year given by the following equation:

$$\boldsymbol{\Delta} GGICs/year = (GGICs/year) \times \boldsymbol{\Delta} PGI \tag{Eq. 1}$$

Where **Δ**PGI = (PGI_{Sa} – PGI_{Sp}), and PGI_{Sa} (the probability of getting sick when seagrasses are absent) and, PGI_{Sp} (the probability of getting sick when seagrasses are present) were those estimated in Section 2.2. This increase in the number of global GI cases (**Δ**GGICs/year) would produce an increase in the number of DALYs/year (**Δ**DALYs/year) that is given by the following equation (assuming a simple proportional increase of the DALYs/year):

$$\boldsymbol{\Delta} DALYs/year = (DALYs/year \times \boldsymbol{\Delta} GGICs/year)/GGICs/year \tag{Eq. 2}$$

Where DALYs/year and GGICs/year correspond to the Shuval's (2003) estimates while **Δ**GGICs/year was computed according to the previous Eq. (1).

Considering that **Δ**GGICs/year = (GGICs/year) × **Δ**PGI, Eq. (2) simplifies as follows:

$$\boldsymbol{\Delta} DALYs/year = DALYs/year \times \boldsymbol{\Delta} PGI \tag{Eq. 3}$$

The increase of the Global GI Cost/year (**Δ**GGIC/year, in 2020 US$), if seagrasses were absent at global scale, was then given by:

$$\boldsymbol{\Delta} GGIC/year = (\boldsymbol{\Delta} DALYs/year) \times (US\$ \ per \ DALY) \tag{Eq. 4}$$

Where US$ per DALY was given in 2020 US$ (i.e., inflation-adjusted values according to the previous Section).

This increase in the global GI cases and cost per year would sum up to the previously Shuval's estimates of GGICs/year and GGIC/year if seagrasses were absent at a global scale. Since they are present, they should allow avoiding these extra yearly cases and costs. Therefore, **Δ**GGICs/year and **Δ**GGIC/year are also an estimate of the Global GI Avoided Cases (GI GACs/year) and the Global GI Avoided Cost (GI GAC/year) that seagrasses may ensure yearly.

2.5. Conservative estimate of global avoided cases and cost

We assumed that most seagrass assemblages or single species seagrass meadows always have a sanitation effect worldwide. However, as we stressed in the Introduction, seagrasses' sanitation effect does not seem to occur everywhere and be always effective on the same pathogen. At present, seagrasses seem to provide their sanitization effect only in two of the six seagrasses' biogeographic regions considered by Short et al. (2007).

We assumed that a 1/3 reduction of the above estimated GI GACs/year and GI GAC/year (previous Section) roughly adjust for this geographically limited occurrence of the seagrasses' sanitation power.

3. Results

3.1. Reduction of the probability of gastroenteritis illness (PGI) in the presence of seagrasses

In Fig. 2, the *Enterococci* CFU/100 ml concentrations vs. the probability of getting GI, i.e., the UK epidemiological data points (from Ashbolt and Bruno 2003, after Kay et al., 1994), the linear fit of these actual data along with the fitted linear equation (i.e., the dose–response relationship), and R^2 were reported. On the same graph, also the average *Enterococci* CFU/100 ml concentrations Lamb et al. (2017) found when a seagrass assemblage was either present (green star and seagrass icon) or absent (red star) are shown. The linear fit explained 97% of the variance ($R^2 = 0.977$) and for the slope $F_{1,15} = 594.5$ ($P < 0.0001$). Cabelli et al. (1982) used a similar approach but regressed the mean Enterococcus density/100 ml vs. the swimming associated rate for GI symptoms/1000 persons linearly while here we followed the same method used by Kay et al. (1994).

The average concentration of *Enterococci* was 125 CFU/100 ml when seagrasses were absent and dropped to 55 CFU/100 ml when seagrasses were present (extracted data from Lamb et al., 2017, presented in Fig. 2). From the dose–response linear equation that we fitted to the UK epidemiological data (i.e., *Enterococci* CFU/100 ml vs. Probability of GI data extracted from Ashbolt and Bruno, 2003), we found that the Probability of GI (PGI) decreased accordingly (Fig. 2). PGI dropped from about 0.30 when seagrasses were absent (*Enterococci* average = 125 CFU/100 ml, a red star in Fig. 2) to about 0.10 when these plants did grow and thrive (*Enterococci* average = 55 CFU/100 ml, a green star in Fig. 2). Thus, from our linear model $PGI_{Sa} = 0.30$ (seagrasses absent) while $PGI_{Sp} = 0.10$ (seagrasses present).

3.2. Gastroenteritis illness global cost

The thalassogenic GI global cases/year given in Shuval (2003) was equal to 120,300,000, i.e., the total number of actual yearly cases of GI resulting from the immersion (e.g., swimming/bathing) in the wastewater-polluted coastal seawaters of the world (GGICs/year). Shuval (2003) estimated, through the DALY method, that these 120,300,000 Global GI Cases/year (GGICs/year) would generate a global GI Cost/year (GGIC/year) of US$ 264 million/year, considering a cost per DALY of US$ 4,000 in US$ of 2003. The following table summarizes Shuval's (2003) estimates (Table 1).

Table 1
Shuval's (2003) estimates of global GI cases, related DALYs, and cost per year.

Shuval's (2003) estimated values	
# Global GI cases/year (GGICS/year)	120,300,000
# DALYs/year	66,000
US$ per DALY	4,000
Global GI cost/year (GGIC/year) in US$	264,000,000

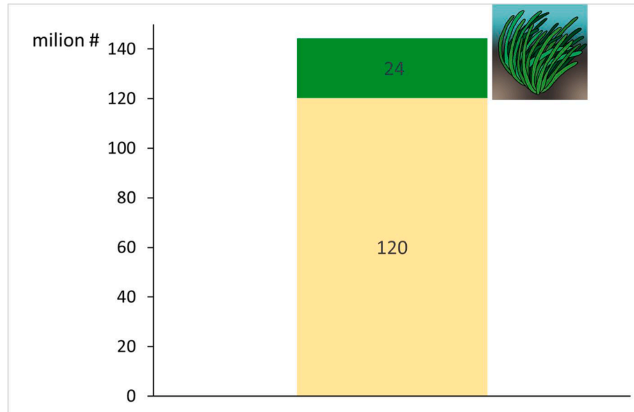

Fig. 3a. Estimate of the Global GI Avoided Cases/year provided by seagrasses – green – compared to the Global GI Cases/year estimate according to Shuval (2003) – yellow.

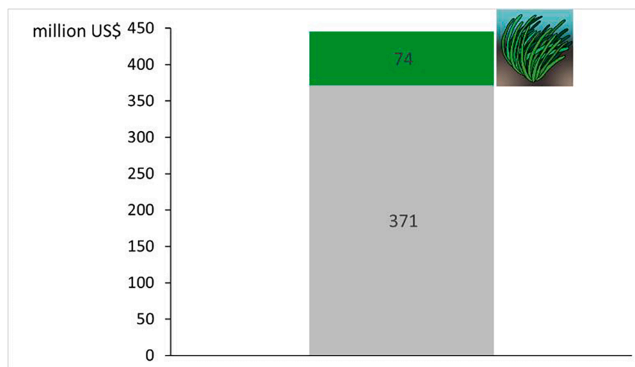

Fig. 3b. Estimate of the Global GI Avoided Cost/year provided by seagrasses – green – compared to the Global GI Cost/year estimate according to Shuval (2003) – grey – (values are in 2020 US$).

When adjusted for inflation, US$ 4,000 per DALY and the US$ 264,000,000/year GGIC (in 2003 US$) were equal to US$ 5,626 per DALY and US$ 371,316,000/year in 2020 US$, respectively. These quantities were the input of our further computations.

3.3. Estimates of global avoided cases and cost

Assuming that seagrasses were absent globally, the number of Global GI Cases, GGICs/year, would be subject to a net increase **Δ**GGICs/year that we computed according to equation 1 (Section 2.4) and the PGIs values previously estimated (Section 3.1).

$$\Delta GGICs/year = 120,300,000 \times (0.30 - 0.10) = 120,300,000 \times 0.20 = 24,060,000$$

Therefore, we obtained, under the assumption of the absence of seagrasses worldwide, a 20% increase of the global 120,300,000 GI cases/year (GGICs/year), i.e., 24,060,000 GI cases more per year (**Δ**GGICs/year). Since seagrasses are present, they would avoid this further burden of yearly GI cases worldwide.

According to the equations (3) (Section 2.4), the increase in the number of DALYs, **Δ**DALYs/year, assuming seagrasses were globally absent, would be:

$$\Delta DALYs/year = 66,000 \times 0.20 = 13,200$$

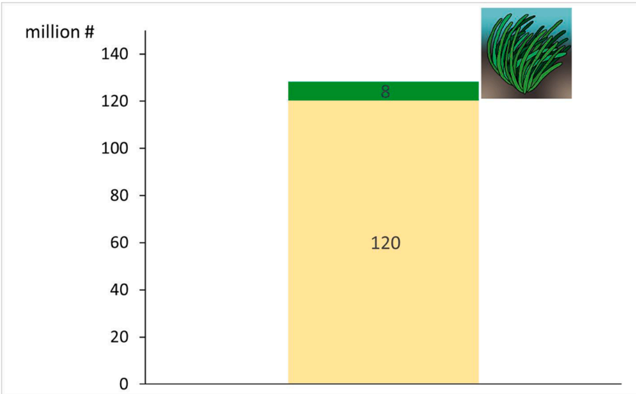

Fig. 4a. Conservative estimate of the Global GI Avoided Cases/year provided by seagrasses – green – compared to the Global GI Cases/year estimate according to Shuval (2003) – yellow.

Thus, according to equation (4) (Section 2.4) the global GI cost/year increase, **Δ**GGIC/year, would be equal to (in 2020 US$):

$$\Delta GGIC/year = 13,200 \times US\$\ 5,626$$
$$= US\$\ 74,263,200/year \qquad (2020\ US\$)$$

If seagrasses were absent globally, this value (ca. 74 million dollars/year) would be the GI global cost/year increase (**Δ**GGIC/year). Thus, it is the avoided cost that seagrasses would ensure globally per year (GI GAC/year) since these plants are present (at least for GI).

We plotted these estimates along with the Global GI Cases/year (GGICs/year) and Global GI Costs/year (GGIC/year) in Fig.3a and 3b.

3.4. *Conservative estimates of global avoided cases and cost*

When (roughly) adjusted for the limited geographic occurrence of seagrasses' anti-pathogen effect (according to Section 2.5 criterion, i.e., a 1/3 reduction), the above estimates gave:

$$Cons - est.\ GI\ GACs/year = 1/3\ \times\ 24,060,000/year = 8,020,000/year$$

$$Cons - est.\ GI\ GAC/year = 1/3 \times US\$\ 74,263,200/year$$
$$= US\$\ 24,754,400/year\ (2020\ US\$)$$

We plotted the estimated conservative values along with Global GI Cases/year (GGICs/year) and Global Costs/year (GGIC/year) in Fig. 4a and Fig. 4b.

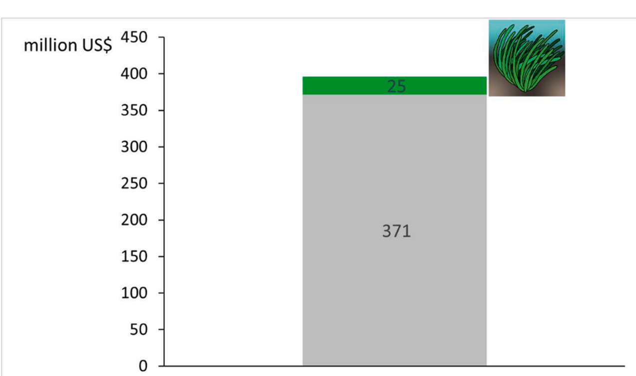

Fig. 4b. Conservative estimate of the Global GI Avoided Cost/year provided by seagrasses – green – compared to the Global GI Cost/year estimate according to Shuval (2003) – grey – (values are in 2020 US$).

4. Discussion

4.1. *The lower probability of contracting GI thanks to the presence of seagrasses*

The linear fitting of the '*Enterococci* concentration (CFU/100 ml) vs. the probability of getting GI (UK) data was particularly good (Section 3.1, Fig. 2) and it is on this dose–response relation that we based our inference about the lowering of the probability of GI due to the presence of seagrasses (Fig. 2). This linear dose–response relation was not too much different from MRM for higher doses, and this gave us confidence about the reliability of our probability GI estimates for the higher levels of *Enterococci* concentration. However, for lower doses, the probability of GI estimates could be less reliable because the Ashbolt and Bruno (2003) MRM fails to encapsulate that part of the dose–response curve, and therefore the risk of getting GI could be not adequately estimated neither by MRM nor by our linear model.

Nonetheless, our goal here was to use the approximate estimates of the probability of getting GI for an objective that is quite different from precise epidemiological scrutiny aimed, for example, at setting water quality criteria (as was the case in the original work of Kay et al., 1994; Kay et al., 2004). We wanted to make global economic estimates that do not require great precision; rather, they were meant at giving a reasonable order of magnitude of both avoided cases and cost that seagrasses may ensure through their effect on pathogens. Therefore, we were sufficiently confident that the estimates we got through a linear dose–response model fitted to UK epidemiological data were good enough for our purposes. Thus, we were able to show that the presence of the Indo-Pacific seagrass assemblage considered by Lamb et al. (2017) can significantly reduce a beachgoer's probability of contracting GI, i.e., a 20% reduction in PGI.

However, the apparent limited genera/geographical area occurrence of the seagrasses' sanitation power implies that the extrapolation we have done to estimate this economically valuable seagrasses' ES at a global scale, using the result mentioned above on PGIs reduction, must be considered with some caution. In fact, because of minimal wastewater treatment prevailing in tropical areas (i.e., at least part of the Indo-Pacific seagrass area), *Enterococci* concentrations may increase well above the highest level included in the UK epidemiological data and, consequently, PGI may change, although it does not seem to significantly increase (at least in the tropical Brazilian waters scrutinized by Verhougstraete et al. (2020)). Nonetheless, the Lamb et al. (2017) average *Enterococci* concentrations (CFU/100 ml) were within the limits of the UK epidemiological data that we used (Fig. 2), and this allowed to make reliable estimates. Sun exposure and temperatures higher than those usually occurring in the UK characterized the geographic area where Lamb et al. (2017) found the seagrass anti-pathogen effect, though. Therefore, our infection risk estimates might be biased.

However, higher temperatures may have opposite effects on fecal bacteria. On the one hand, they cause more virulent transmissibility of infectious diseases and an increase in the duration of the annual period during which pathogens cause a problem (Harvell et al., 2002; Altizer et al., 2013). On the other hand, they contribute to fecal bacteria decay/inactivation but in a far less effective way than longer sun exposure (e.g., Sinton et al., 2002; Sagarduy et al., 2019). The two opposite effects (i.e., large positive temperature effect vs. smaller negative temperature and large negative sun exposure effect) most likely compensate each other, implying that our PGI estimates based on UK epidemiological data and tropical *Enterococci* concentrations may provide sufficiently reliable figures of the actual lower probability of infection.

4.2. *Global avoided cases and cost*

The 20% reduction in the PGI implies that, in the absence of seagrasses, the number of GI cases per year (GGICs/year) would be subject to the same 20% increase, or equivalently that the seagrasses presence may ensure 20% less GGICs/year. Considering the geographic area/genera limitation of seagrass sanitation power, the conservative 1/3 reduction of this percentage would suggest that at least a 7% less GGICs/year would be ensured by the seagrasses' presence. This result is our main finding and has both local and global implications. In specific local cases, the presence of seagrasses meadows seems to imply a lower (20% less) public health impact of the GI due to seawater pollution. This information may be essential for public health and conservation – one-health – policy strategic choices. Globally, we found that considering Shuval's (2003) estimate of GGICs/year (i.e., 120,300,000), seagrasses may avoid that 24,060,000 or, conservatively 8,030,000 GI cases per year would sum up to the current GGICs/year (i.e., **Δ**GGICs/year and its conservative estimate).

As for global GI avoided cost (GI GAC/year), we found that relying on Shuval's global GI cost per year (GGIC/year) figure, i.e., US$ 371,316,000/year (2020 US$), the presence of seagrasses would ensure to avoid ca. US$ 74 million or, conservatively, ca. US$ 24 million per year of extra cost (in 2020 US$). The magnitude of these estimates depends on the original GGIC/year estimate because GI GAC/year estimates are a fixed percentage (20 or 7%, respectively) of GGIC.

It is likely that Shuval's GGIC/year is an underestimate of the actual global GI cost/year value, considering both local (coastal stretches and beaches, e.g., Dwight et al. (2005); Given, Pendleton, and Boehm 2006) and nationwide estimates of GI costs (DeFlorio-Barker et al., 2017; Barker et al., 2018; DeFlorio-Barker et al., 2018; see SM5 for a thorough discussion). The global economic burden of Norovirus GI was estimated, through a simulation model, by Bartsh et al. (2016). They heve shown that the direct sanitary costs may result in a total of US$ 4.2 billion per year (central estimate), 62% of which were attributable to health expenses in high-income countries, while social costs (mostly missing productive days) that may amount to US$ 60.3 billion per year (central estimate), seem to heavily impact low- middle-income countries in terms of household's income spending and death toll. Overall, the poorer countries seem to have a greater cumulative Norovirus disease burden, i.e., 82% of total global illness and 97% of global deaths, compared to high-income countries. Norovirus GIs are one-fifth of all gastroenteritis illnesses, which include all sorts of gastroenteritis (i.e., food and drinking water-borne GIs) and not only the thalassogenic ones that are, in turn, only a fraction of all GI. Therefore, the Bartsh et al. (2016) results are not fully comparable to Shuval's ones. Those figures suggest, though, that it is highly likely that GGIC/year is one order of magnitude higher than Shuval's (2003) estimate.

One order of magnitude higher GGIC/year would imply a consequent increase in the order of magnitude (i.e., ca. a billion or tens of billions) of the seagrass-ensured GI GAC/year. Thus, our GI GAC/year values should be considered highly underestimated. Nonetheless, our findings are useful because as soon as such a GGIC/year better estimate will be available the corresponding global avoided cost ensured by the seagrass sanitation effect could be roughly estimated computing a 20 (non-conservative) – 7% (conservative) increase of GGIC/year, assuming that seagrasses were globally absent (and, thus, the global GI Global Avoided Cost/year since they are present). The same would also hold for the GI Global Avoided Cases per year (GI GACs/year) in the case a new assessment of the number of Global GI Cases per year (GGICs/year) will be available in the future.

4.3. Equitable conservation of seagrass sanitation service

If the Lamb et al. (2017) findings are confirmed for the whole Asian-Pacific Ocean region, the conservation of seagrass meadows will prove to be extremely important for the coastal areas of that region of the world where the poorest may be heavily exposed to seawater pathogens and cannot get or afford good sanitary treatment (Jamison et al., 2018). Seagrasses sanitation ES may not reduce poverty directly, but it may well reduce the high vulnerability of the poorest to thalassogenic diseases, specifically GI (Suich et al., 2015). Keeping seawater pathogens at a lower concentration, seagrass sanitation ES may help the poorest run a lower risk of getting sick, allowing them a healthier life and a lower probability of further impoverishment. In these coastal areas of the world, the 20% seagrass-mediated reduction of the thalassogenic GI incidence (i.e., cases) and economic cost we have found here may mean millions of people alive, healthier, less poor, and fewer economic resources needed for building/improving health care system, every year. The damage of seagrasses in these areas would imply thus a larger humanitarian burden and not simply an increase in sanitary costs as it is likely to be for high-income countries (SM6). It would seem tropical "seagrass forests" are capable, in the marine biome, of a sort of "pathogens-containment" analogous to that tropical forests do in the terrestrial biome (Olivero et al., 2017; Guégan et al., 2020).

5. Limitations

5.1. General limitations

Our work focused exclusively on GI cost, which is only a part of a broader framework of many other sources of cost (see Section 2.1, Fig. 1). In our analysis, we did not include the indirect costs due to the reduction in the appeal of polluted coastal recreational areas (Fig. 1 right, lower, rounded rectangle). Moreover, our estimates did not include (Fig. 1): i) the positive (i.e., cost reduction) indirect effect of a lower incidence of GI due to the ingestion of less contaminated seafood (e.g., clams and fish) by humans; ii) the other water-borne diseases such as ear, eye, and respiratory system illnesses; iii) the beneficial effect on the entire ecosystem's health status that seagrasses favor and that humans enjoy (Lamb et al., 2017). Therefore, our estimates are, from this larger perspective, very conservative underestimates of the likely much higher avoided cost seagrasses may ensure via their total anti-pathogen action (whenever they do provide this ES).

5.2. Specific limitations

Many factors may weaken the reliability of our PGIs estimates, such as flooding events, which increase the risk of infection (e.g., de Man et al., 2014), and the exposed people typology. Local coastal community members and beachgoers may exhibit an acquired immunity that foreign (tourists) beachgoers do not possess (e.g., Prieto et al., 2001). Furthermore, the bathing pattern, i.e., beach attendance as the number of beach visitors/dwellers through time and the excess illness due to swimming linked to the fraction of swimmers per day should be considered (see Turbow, Osgood, and Jiang, 2003; Given, Pendleton, and Boehm, 2006). Bathing pattern data may be essential because Papastergiou et al. (2012) found that bathers may run an elevated risk of suffering from GI (and other diseases) symptoms even when bathing water quality complies with legal standards (both EU and EPA). However, they also found a relationship between bather density and gastrointestinal (and respiratory) illness. Therefore, it is possible that the infection is transmitted from bather to bather through the water (see also Fattal et al., 1991). Changes in the number and behavior of beachgoers/coastal dwellers through time (i.e., bathing pattern) may significantly influence the spreading of infection thus, and the presence/absence of sanitation seagrass assemblages/species may matter in this case.

Moreover, a higher level of pollution, i.e., an *Enterococci* concentration (CFU/100 ml) greater than the max concentration considered here (i.e., 125 CFU/100 ml, according to Lamb et al. 2017), may lead to different local PGIs values and therefore to a different percentage decrease/increase in presence/absence of seagrasses. Different pollution conditions occurring in the diverse beaches of the world will give thus an unequal contribution to the global decrease in PGIs whenever seagrasses are present. Consequently, our generalization should be taken with some caution. We are also assuming that the halving effect seagrasses have on

Enterococci concentration (as found by Lamb et al. 2017) also holds for pollution levels higher than those Lamb et al. (2017) found in the place where they carried out their research. This assumption however, might not be valid and PGIs values may not decrease/increase globally as much as we have assumed here.

Global GI Avoided Cases and Cost (per year) estimates seem to be subject to several limitations: taxonomic/ecological, epidemiological, and economic ones. The first limitation concerns the Global GI Cases and Cost/year estimates which constrains all the successive computations of GI GACs/year and GAC/year. Both GGICs/year and GGIC/year Shuval's estimates are likely underestimated, as we stressed above, and therefore our avoided cases and avoided cost estimates are probably underestimated as well.

The second limitation relates to the generalizability of the seagrass sanitation effect. As we stressed above, seagrass sanitation is genera/geographic area dependent. It is well-documented that it is indeed effective in an Indo-Pacific archipelago through the seagrass assemblage occurring there (Lamb et al., 2017). However, it cannot be arbitrarily extended worldwide *sic et simpliciter*. In other words, the presence of a seagrass meadow does not always guarantee an effective anti-pathogen action. However, the Palazon et al. (2018) finding of a significant decrease of *Escherichia coli* concentration in Mediterranean coastal waters where *Posidonia oceanica* was present suggests that the seagrass sanitization ES could be extended to the Mediterranean latitudinal areas (*sensu* Short et al., 2007).

Palazon et al. (2018) also found a not statistically significant reduction of *Enterococci* concentration in the presence of *Posidonia*. This finding implies that the seagrass sanitation effect not only changes according to the seagrass genera/geographic distribution area but may also change according to the targeted pathogen-type/s. The latter represents a further complication for the estimation of a global effect from which to derive a meaningful global avoided cost estimate. This complication prevented us from utilizing the dose–response models fitted to UK epidemiological data to estimate *Posidonia*'s sanitation effect in Mediterranean waters since the UK data consider *Enterococci* and not *Escherichia coli* (CFU/100 ml) as an independent variable of the dose–response relationship.

A third drawback that might lead to overestimated GI GAC/year figures is that not all the coastal stretches where seagrasses are present are also used as tourist-bathing beaches or dwelling and working (e.g., fishing) sites. Assuming that seagrasses were globally absent, without considering the actual beach use, our **Δ**GGICs/year is an overestimate. In fact, along some of the coastal stretches of the world, almost no one (or very few) goes swimming or dwell and work. Therefore, almost no one runs the risk of getting GI, irrespective of the presence/absence of seagrasses. Moreover, not all the beaches used for recreation (bathing-swimming) or for dwelling and working host seagrass meadows. Therefore, seagrasses do not play their potential sanitation role everywhere along the coastal stretches of the world where people go swimming, dwell, and work. We tentatively and roughly tryed to adjust our GI GAC/year original estimates through a 1/3 reduction. However, such a criterion is only a rule of thumb approach that gives a very conservative estimate.

6. Open questions and further research needs

It is important to emphasize that the seagrass sanitation effect ES may directly affect public health. Despite this, as far as we know, only three papers have addressed this peculiar ES in a specific way (i.e., Lamb et al., 2017; Palazon et al., 2018; Webb et al., 2019). Therefore, further research on the role and limits globally exhibited by seagrasses as living sanitation providers, along with *ad hoc* planned epidemiological surveys in the presence/absence of these plants, are needed to ascertain better the environmental, epidemiological, and economic scope of this ES.

The other sources of cost (other than GI, Fig. 1, Section 2.1 and 4.2.1) and the role seagrasses may play in mitigating them need to be estimated in further studies. For instance, Shuval (2003) estimated that the cost of illnesses due to food sources contaminated by polluted coastal waters (see the relative path in Fig. 1) is of the order of 12 billion dollars/year. We do not know how much seagrasses mitigate this relevant economic burden through their sanitation effect (at least in the regions where they do play such a role). To estimate this contribution, one would need to know the magnitude of the seagrass sanitation effect on food source contamination. This latter information is not readily available, as far as we know. The seagrass sanitation effect ES may also avoid/prevent beach closures due to legal bacterial pollution limits overshooting. Thus, it may also contribute indirectly to the provision of recreational beach ESs and the related economic benefits (Fig. 1).

The local seagrasses-mediated reduction of the PGIs will need a further assessment because the magnitude of *Enterococci*'s concentration abatement in the presence of seagrasses, whenever it does occur, may widely depend on local conditions, such as seawater pollution level (*Enterococci* CFU/100 ml), seagrass meadows extension, integrity, and species identity, climatic environment, and bather's typology (local people vs. foreigners). Moreover, updated estimates of the global GI cases and cost per year (GGIC/year) are needed too because the magnitude of avoided cases and cost depends on these global estimates.

The sanitation effect of seagrasses may be directly imputable to the plants (Alam et al., 1994; Mayavu et al., 2009; Kannan et al., 2010; Choi et al., 2009) or else to the microbiota developed within their ecosystem (e.g., the fungal community, especially *Eurotiales*, genus *Penicillium*, Ugarelli et al., 2017). This latter point would deserve further attention because it cannot be taken for granted that the antibiotic activity is a direct effect of the seagrasses themselves. The seagrass sanitation effect may depend on the overall integrity and biodiversity of the seagrass ecosystem and not only on the seagrass genera/species *per se* (Nordlund et al., 2018). Interestingly, the Indo-Pacific Area, where the seagrass sanitation effect seems most evident (Lamb et al., 2017), is also characterized by the highest seagrass biodiversity (Short et al., 2007).

The need for further studies on the seagrass sanitation ES is particularly relevant nowadays because human's environmental-health misbehaviour, threatening the dynamical equilibria governing the seagrasses' sanitation power, may favor the pathogen diffusion and their spill over and impact on humans and other living beings (Fig. 1). For instance, we speculate that because of the spreading of antibiotic-resistant bacteria (Baker et al., 2017; SM7) seagrasses' antibiotic power might have been disabled in some regions of the ocean (e.g., the Temperate Pacific Ocean where Webb et al. 2019 did not find evidence supporting the seagrass sanitation effect), or it might be disabled in the future. This problem deserves extreme attention because through this anthropogenic-induced microorganisms' antibiotic-resistance to seagrasses' antibiotic action, we might run the risk of favoring a worldwide upsurge of thalassogenic antibiotic-resistant diseases (e.g., 7–20% more global GICCs/year, according to our estimates, caused furthermore by antibiotic-resistant pathogens).

Moreover, the planet is being threatened by the possible escalation of disease spreading due to the increase of global temperature and extreme events driven by climate change (Morens, Folkers & Fauci, 2004; Costello et al., 2009; de Roda Husman and Schets, 2010; Cann et al., 2013; Altizer et al., 2013), such as floods (which may significantly affect the risk of infection, e.g., de Man et al., 2014). Seagrasses may help to mitigate a climate-change-induced disease surge through their sanitation-power ES (unless we do disable it through the mechanism mentioned above). Besides, these plants provide many other ESs that can compensate/mitigate several detrimental effects of climate change (Cullen-Unsworth et al., 2014; Nordlund et al., 2016; Ugarelli et al., 2017; Nordlund et al., 2018). Seagrasses are, thus, a valuable source of present and future benefits for human beings, especially, as we have stressed above, for those populations of poorer people who run the risk of being afflicted by climate change the harder way (UNO-DESA, 2020) since they have almost exclusively relied (at least till now and often unawarely) on these ESs.

7. Conclusions

Considering the above findings, seagrass ecosystems' protection and rehabilitation (when damaged) should be a priority for any sensible and wise political action willing to safeguard global human health and wealth. Unfortunately, the global trend seems to go in the opposite direction, and increasing temperature along with the direct human pressures/impacts have already destroyed 29% of the known area of these plants since seagrass populations were initially recorded in 1879 and are continuing to destroy them at an ever-increasing rate (Waycott et al., 2009). This trend would imply, in turn, a combined effect of the temperature increase and a consequent increase in disease transmissibility/annual duration of the period during which pathogens are a problem and an increase in the probability of getting sick due to the reduction of those seagrass populations providing the sanitation ES. This combination would magnify (probably also in a non-linear way, Koch et al., 2009) both the sanitary risks/costs and the other economic damages (linked to the other seagrass ESs), thus increasing the global number of cases and costs and threatening the health and wellbeing of the large portion of human beings living in the coastal areas of the world (Kummu et al., 2016).

However, recent findings (De los Santos et al., 2019) show a positive reversal of the previously negative trend for European seagrass populations. This reversal is very likely related to better environmental conditions along European coastlines and suggests that appropriate management and conservation efforts may effectively mitigate and repair previous seagrass damage. Restored seagrass ecosystems can favor a return of the ESs provided by these plants (Orth et al., 2020), hopefully including the epidemiologically and economically relevant sanitation effect (whenever it does occur) that we considered in this paper.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Author contributions

FAA conceived this study, whose design was developed with GS and MCM. FAA wrote the first draft of the manuscript while GS, MCM, and CM reviewed and commented on the final version; GS provided research funds and facilities.

Competing financial interests

The authors declare no competing financial interests.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2022.101418.

References

Alam, K., et al. 1994 Preliminary screening of seaweeds, seagrass and lemongrass oil from Papua New Guinea for antimicrobial and antifungal activity. Int. J. Pharmacogn. Phytochem. Res., 32 (4), 396-399. doi/abs/10.3109/13880209409083022.

Altizer, S., et al., 2013. Climate change and infectious diseases: From evidence to a predictive framework. Science 341, 514–519. https://doi.org/10.1126/science.1239401.

Arnold, B.F., Wade, T.J., Benjamin-Chung, J., Schiff, K.C., Griffith, J.F., Dufour, A.P., Weisberg, S.B., Colford, J.M., 2016. Acute gastroenteritis and recreational water : Highest burden among young US children. Am. J. Public Health 106 (9), 1690–1697. https://doi.org/10.2105/AJPH.2016.303279.

Ashbolt, N.J., Bruno, M., 2003. Application and refinement of the WHO risk framework for recreational waters in Sydney, Australia. J. Water Health 01 (3), 125–131. https://doi.org/10.2166/wh.2003.0015.

Baker, S., Thomson, N., Weill, F.-X., Holt, K.E., 2017. Genomic insights into the emergence and spread of antimicrobial-resistant bacterial pathogens. Science 360 (6390), 733–738. https://doi.org/10.1126/science:aar3777.

Barker, S.F., Zomer, E., O'Toole, J., Sinclair, M., Gibney, K., Liew, D., Leder, K., Kirk, M., 2018. Cost of gastroenteritis in Australia: A healthcare perspective. PLoS ONE 13 (4), e0195759. https://doi.org/10.1371/journal.pone.0195759.

Bartsh, S.M., Lopman, B.A., Ozawa, S., Hall, A.J., Lee, B.Y., Olson, D.R., 2016. Global economic burden of Norovirus gastroenteritis. PLoS ONE 11 (4), e0151219. https://doi.org/10.1371/journal.pone.0151219.

Bowley, J., Baker-Austin, C., Porter, A., Hartnell, R., Lewis, C., 2021. Oceanic hitchhikers – Assessing pathogen risks from marine microplastic. Trends Microbiol. 29 (2), 107–116. https://doi.org/10.1016/j.tim.2020.06.011.

Cabelli, V.J., et al., 1982. Swimming-associated Gastroenteritis and water quality. Am. J. Epidemiol. 115 (4), 606–616.

Campagne, C.S., Salles, J.-M., Boissery, P., Deter, J., 2015. The seagrass *Posidonia oceanica*: Ecosystem services identification and economic evaluation of goods and benefits. Mar. Poll. Bull. 97 (1-2), 391–400. https://doi.org/10.1016/j.marpolbul.2015.05.061.

Cann, K.F., Thomas, D.R., Salmon, R.L., Wyn-Jones, A.P., Kay, D., 2013. Extreme water-related weather events and waterborne disease. Epidemiol. Infect. 141 (4), 671–686. https://doi.org/10.1017/S0950268812001653.

Choi, H.-G., Lee, J.-H., Park, H.-H., Sayegh, F.A.Q., 2009. Antioxidant and antimicrobial activity of *Zostera marina L.* extract. Algae 24 (3), 179–184. https://doi.org/10.4490/ALGAE.2009.24.3.179.

Costello, A., Abbas, M., Allen, A., Ball, S., Bell, S., Bellamy, R., Friel, S., Groce, N., Johnson, A., Kett, M., Lee, M., Levy, C., Maslin, M., McCoy, D., McGuire, B., Montgomery, H., Napier, D., Pagel, C., Patel, J., de Oliveira, J.A.P., Redclift, N., Rees, H., Rogger, D., Scott, J., Stephenson, J., Twigg, J., Wolff, J., Patterson, C., 2009. Managing the health effects of climate change. Lancet 373 (9676), 1693–1733. https://doi.org/10.1016/S0140-6736(09)60935-1.

Cullen-Unsworth, L.C., et al., 2014. Seagrass meadows globally as a coupled social–ecological system: implications for human wellbeing. Mar. Poll. Bull. Seagrass Meadows in a Globally Changing Environment 83, 387–397. https://doi.org/10.1016/j.marpolbul.2013.06.001.

DeFlorio-Barker, S., Wade, T.J., Jones, R.M., Friedman, L.S., Wing, C., Dorevitch, S., 2017. Estimated costs of sporadic Gastrointestinal Illness associated with surface water recreation: A combined analysis of data from NEEAR and CHEERS studies. Environ. Health Pers. 125 (2), 215–222. https://doi.org/10.1289/EHP130.

DeFlorio-Barker, S., Wing, C., Jones, R.M., Dorevitch, S., 2018. Estimate of incidence and cost of recreational waterborne illness on United States surface waters. Environ. Health 17 (1). https://doi.org/10.1186/s12940-017-0347-9.

De los Santos, C. B, et al., 2019. Recent trend reversal for declining European seagrass meadows. Nature Comm. 10, 3356. https://doi.org/10.1038/s41467-019-11340-4.

De los Santos, C. B., et al. 2020 Seagrass ecosystem services: Assessment and scale of benefits. In: Out of blue: The value of seagrasses to the environment and the people. UNEP, Nairobi, pp. 21-35. ISBN: 978-92-807-3780-6.

de Man, H., van den Berg, H.H.J.L., Leenen, E.J.T.M., Schijven, J.F., Schets, F.M., van der Vliet, J.C., van Knapen, F., de Roda Husman, A.M., 2014. Quantitative assessment of infection risk from exposure to waterborne pathogens in urban floodwater. Water Res. 48, 90–99. https://doi.org/10.1016/j.watres.2013.09.022.

de Roda Husman, A.M., Schets, F.M., 2010. Climate change and recreational water-related infectious diseases. RIVM Report 330400002 (2010), 46.

Devleesschauwer, B., et al., 2014. Calculating disability-adjusted life years to quantify burden of disease. Int. J. Public Health 59, 565–569. https://doi.org/10.1007/s00038-014-0552-z.

Dewsbury, B.M., Bhat, M., Fourqureanet, J.W., 2016. A review of seagrass economic valuations: Gaps and progress in valuation approaches. Ecosystem Ser. 18, 68–77. https://doi.org/10.1016/j.ecoser.2016.02.010.

Dobson, A.P., et al., 2020. Ecology and economics for pandemic prevention. Science 369, 379–381. https://doi.org/10.1126/scien ce.abc3189.

Dwight, R.H., Fernandez, L.M., Baker, D.B., Semenza, J.C., Olson, B.H., 2005. Estimating the economic burden from illnesses associated with recreational coastal water pollution - a case study in Orange County, California. J. Environ. Manag. 76 (2), 95–103. https://doi.org/10.1016/j.jenvman.2004.11.017.

Elliott, J.K., Spear, E., Wyllie-Echeverria, S., 2006. Mats of *Beggiatoa* bacteria reveal that organic pollution from lumber mills inhibits growth of *Zostera* marina. Mar. Ecol. 27 (4), 372–380. https://doi.org/10.1111/j.1439-0485.2006.00100.x.

Failler, P., Pètre, É., Binet, T., Maréchal, J.-P., 2015. Valuation of marine and coastal ecosystem services as a tool for conservation: The case of Martinique in the Caribbean. Ecosystem Serv. 11, 67–75. https://doi.org/10.1016/j.ecoser.2014.10.011.

FAO/WHO 2003. Hazard characterization for pathogens in food and water: Guidelines. Microbiological risk assessment series, No. 3, FAO/WHO, Rome-Geneve p. 61, ISSN 1726-5274.

Fattal, B., Peleg-Olevsky, E., Cabelli, V.J., 1991. Bathers as a possible source of contamination for swimming-associated illness at marine bathing beaches. Int. J. Env. Health Res. 1 (4), 204–214.

Fleisher, J.M., et al. 1996 Non-enteric illnesses associated with bather exposure to marine waters contaminated with domestic sewage: the results of a series of four intervention follow-up studies. Am. J. Public Health 86, 1228–34. doi.org/10.2105/ajph.86.9.1228.

Gillespie, T.R., Jones, K.E., Dobson, A.P., Clennon, J.A., Pascual, M., 2021. COVID-Clarity demands unification of health and environmental policy. Global Change Biol. 27 (7), 1319–1321. https://doi.org/10.1111/gcb.v27.710.1111/gcb.15508.

Given, S., Pendleton, L.H., Boehm, A.B., 2006. Regional public health cost estimates of contaminated coastal waters: a case study of gastroenteritis at Southern California beaches. Environ. Sci. Technol. 40, 4851–4858. https://doi.org/10.1021/es060679s.

Grant, S.B., Sanders, B.F., Boehm, A.B., Redman, J.A., Kim, J.H., Mrše, R.D., Chu, A.K., Gouldin, M., McGee, C.D., Gardiner, N.A., Jones, B.H., Svejkovsky, J., Leipzig, G.V., Brown, A., 2001. Generation of Enterococci bacteria in a coastal saltwater marsh and its impact on surf zone water quality. Environ. Sci. & Tech. 35 (12), 2407–2416. https://doi.org/10.1021/es0018163.

Guégan, J.-F., Ayouba, A., Cappelle, J., de Thoisy, B., 2020. Forests and emerging infectious diseases: unleashing the beast within Environ. Res. Lett. 15 (8), 083007. https://doi.org/10.1088/1748-9326/ab8dd7.

Harvell, C.D., Mitchell, C.E., Ward, J.R., Altizer, S., Dobson, A.P., Ostfeld, R.S., Samuel, M.D., 2002. Climate warming and disease risks for terrestrial and marine biota. Science 296 (5576), 2158–2162. https://doi.org/10.1126/science:1063699.

Hernández-Sancho, F., et al. 2015 Economic valuation of wastewaters. The cost of action and the cost of no action. UNEP, 1-70, ISBN: 978-92-807-3474-4.

Henrickson, S.E., Wong, T., Allen, P., Ford, T., Epstein, P.R., 2001. Marine swimming–related illness: Implications for monitoring and environmental policy. Env. Health Perspect. 109 (7), 645–650. https://doi.org/10.1289/ehp.01109645.

Himes-Cornell, A., Pendleton, L., Atiyah, P., 2018. Valuing ecosystem services from blue forests: A systematic review of the valuation of salt marshes, sea grass beds and mangrove forests. Ecosystem Serv. 30, 36–48. https://doi.org/10.1016/j.ecoser.2018.01.006.

ISO 14008:2019 Monetary valuation of environmental impacts and related environmental aspects. www.iso.org/standard/43243.html.

Jackson, E.L., Rees, S.E., Wilding, C., Attrill, M.J., 2015. Use of a seagrass residency index to apportion commercial fishery landing values and recreation fisheries expenditure to seagrass habitat service. Cons. Biol. 29 (3), 899–909. https://doi.org/10.1111/cobi.12436.

Jamison, D.T., et al. (Eds) 2018 Disease control priorities: Improving health and reducing poverty. (Third edition), Volume 9, Washington, DC: World Bank. doi:10.1596/978-1-4648-0527-1.

Jones, B.L., Cullen-Unsworth, L.C., Unsworth, R.K.F., 2018. Tracking Nitrogen source using δ15N reveals human and agricultural drivers of seagrass degradation across the British Isles. Front. Plant Sci. 9, 133. https://doi.org/10.3389/fpls.2018.00133.

Kannan, R.R.R., Arumugam, R., Anantharaman, P., 2010. Antibacterial potential of three seagrasses against human pathogens. Asian Pacific J. Trop. Med. 3 (11), 890–893. https://doi.org/10.1016/S1995-7645(10)60214-3.

Kay, D., et al., 1994. Predicting the likelihood of gastroenteritis from sea bathing: results from a randomized exposure. Lancet 344, 905–909. https://doi.org/10.1016/s0140-6736(94)92267-5.

Kay, D., Bartram, J., Prüss, A., Ashbolt, N., Wyer, M.D., Fleisher, J.M., Fewtrell, L., Rogers, A., Rees, G., 2004. Derivation of numerical values for the World Health Organization guidelines for recreational waters. Water Res. 38 (5), 1296–1304. https://doi.org/10.1016/j.watres.2003.11.032.

Koch, E.W., et al., 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Front. Ecol. Environ. 7 (1), 29–37. https://doi.org/10.1890/080126.

Kummu, M., de Moel, H., Salvucci, G., Viviroli, D., Ward, P.J., Varis, O., 2016. Over the hills and further away from coast: global geospatial patterns of human and environment over the 20th–21stcenturies. Environ. Res. Lett. 11 (3), 034010. https://doi.org/10.1088/1748-9326/11/3/034010.

Lamb, J.B., et al., 2017. Seagrass ecosystems reduce exposure to bacterial pathogens of humans, fishes, and invertebrates. Science 355 (6326), 731–733. https://doi.org/10.1126/science.aal1956.

Leonard, A.F.C., et al. 2018 Is it safe to go back into the water? A systematic review and meta-analysis of the risk of acquiring infections from recreational exposure to seawater. Int. J. Epidemiology, 2018, 572–586 doi: 10.1093/ije/dyx281.

Liu, S., Jiang, Z., Deng, Y., Wu, Y., Zhang, J., Zhao, C., Huang, D., Huang, X., Trevathan-Tackett, S.M., 2018. Effects of nutrient loading on sediment bacterial and pathogen communities within seagrass meadows. MicrobiologyOpen 7 (5), e00600. https://doi.org/10.1002/mbo3.2018.7.issue-510.1002/mbo3.600.

Luisetti, T., Jackson, E.L., Turner, R.K., 2013. Valuing the European 'coastal blue carbon' storage benefit. Mar. Poll. Bull. 71 (1–2), 101–106. https://doi.org/10.1016/j.marpolbul.2013.03.029.

Markandya, A., 2016. Cost benefit analysis and the environment: How to best cover impacts on biodiversity and ecosystem services. OECD Environment Working Papers No. 101.

Mayavu, P., Sugesh, S., Ravindran, V.J., 2009. Antibacterial activity of seagrass species against biofilm forming bacteria. Res. J. Microbiol. 4 (8), 314–319. https://doi.org/10.3923/jm.2009.314.319.

Morens, D.M., Folkers, D., Fauci, A.S., 2004. The challenge of emerging and re-emerging infectious diseases. Nature 430, 242–249. https://doi.org/10.1038/nature08554.

Newton, R.J., McClary, J.S., 2019. The flux and impact of wastewater infrastructure microorganisms on human and ecosystem health. Cur. Opinion Biotech. 57, 145–150. https://doi.org/10.1016/j.copbio.2019.03.015.

Nordlund, L.M., Koch, E.W., Barbier, E.B., Creed, J.C., Reinhart, K.O., 2016. Seagrass ecosystem services and their variability across genera and geographical regions. PLoS ONE 11 (10), e0163091. https://doi.org/10.1371/journal.pone.0163091.

Nordlund, L.M., Jackson, E.L., Nakaoka, M., Samper-Villarreal, J., Beca-Carretero, P., Creed, J.C., 2018. Seagrass ecosystem services – What's next? Mar. Pol. Bull. 134, 145–151. https://doi.org/10.1016/j.marpolbul.2017.09.014.

Olivero, J., et al., 2017. Recent loss of closed forests is associated with Ebola virus disease outbreaks. Sci. Rep. 7 (1429) https://doi.org/10.1038/s41598-017-14727-9.

Orth, R.J., et al., 2020. Restoration of seagrass habitat leads to rapid recovery of coastal ecosystem services. Science Adv 6, eabc6434. https://doi.org/10.1126/sciadv.abc6434.

Palazon, A., et al., 2018. Determination of the most influential factors in the concentration of bacteria in coastal waters. Int. J. Environ. Impacts 1 (1), 61–69. https://doi.org/10.2495/EI-V1-N1-61-69.

Pandey, P.K., Kass, P.H., Soupir, M.L., Biswas, S., Singh, V.P., 2014. Contamination of water resources by pathogenic bacteria. AMB Express 4 (1). https://doi.org/10.1186/s13568-014-0051-x.

Papastergiou, P., Mouchtouri, V., Pinaka, O., Katsiaflaka, A., Rachiotis, G., Hadjichristodoulou, C., 2012. Elevated bathing-associated disease risks despite certified water quality: A cohort study. Int. J. Environ. Res. Public Health 9 (5), 1548–1565. https://doi.org/10.3390/ijerph9051548.

Pougnet, R., et al. 2018 Maritime environment health risks related to pathogenic microorganisms in seawater. Int. Maritime Health, 69 (1), 35–45. DOI: 10.5603/IMH.2018.0006.

Pond, K., 2005. Water recreation and disease. Plausibility of associated infections: acute effects, sequelae and mortality. WHO - IWA Publishing, London, p. 239. ISBN13: 9781843390664 eISBN: 9781780405827.

Prieto, M.D., et al., 2001. Recreation in coastal waters: health risks associated with bathing in seawater. J. Epidemiol. Comm. Health 55, 442–447. https://doi.org/10.1136/jech.55.6.442.

Qadir, M., Drechsel, P., Jiménez Cisneros, B., Kim, Y., Pramanik, A., Mehta, P., Olaniyan, O., 2020. Global and regional potential of wastewater as a water, nutrient and energy source. Nat. Res. Forum. 44 (1), 40–51. https://doi.org/10.1111/narf.v44.110.1111/1477-8947.12187.

Sagarduy, M., Courtois, S., Del Campo, A., Garmendia, J.M., Petrau, A., 2019. Differential decay and prediction of persistence of *Enterococcus* spp. and *Escherichia coli* culturable cells and molecular markers in freshwater and seawater environments. Intern. J. Hygiene Environ. Health 222 (4), 695–704. https://doi.org/10.1016/j.ijheh.2019.04.011.

Sinton, L.W., Hall, C.H., Lynch, P.A., Davies-Colley, R.J., 2002. Sunlight inactivation of fecal indicator bacteria and bacteriophages from waste stabilization pond effluent in fresh and saline waters. Appl. Environ. Microbiol. 68 (3), 1122–1131. https://doi.org/10.1128/AEM.68.3.1122-1131.2002.

Schmiege, D., et al., 2020. One Health in the context of coronavirus outbreaks: A systematic literature review. One Health 10 (100170). https://doi.org/10.1016/j.onehlt.2020.100170.

Short, F., Carruthers, T., Dennison, W., Waycott, M., 2007. Global seagrass distribution and diversity: A bioregional model. J. Exp. Mar. Biol. Ecol. 350 (1-2), 3–20. https://doi.org/10.1016/j.jembe.2007.06.012.

Shuval, H., 2003. Estimating the global burden of thalassogenic diseases: human infectious diseases caused by wastewater pollution of the marine environment. J. Water and Health 1 (2), 53–62.

Suich, H., Howe, C., Mace, G., 2015. Ecosystem services and poverty alleviation: A review of the empirical links. Ecosystem Serv. 12, 137–147. https://doi.org/10.1016/j.ecoser.2015.02.005.

TEEB 2010 - The Economics of Ecosystems and Biodiversity. Ecological and economic foundations. Ed. Pushpam Kumar. Earthscan, London and Washington. www.teebweb.org/resources/glossary-of-terms/.

Teunis, P.F.M., Havelaar, A.H., 2000. The Beta Poisson dose-response model is not a single-hit model. Risk Analysis 20 (4), 513–520. https://doi.org/10.1111/risk.2000.20.issue-410.1111/0272-4332.204048.

Tuholske, C., et al., 2021. Mapping global inputs and impacts from of human sewage in coastal ecosystems. PLoS ONE 16 (11), e0258898. https://doi.org/10.1371/journal.pone.0258898.

Turbow, D.J., Osgood, N.D., Jiang, S.C., 2003. Evaluation of recreational health risk in coastal waters based on Enterococcus densities and bathing patterns. Environ. Health Perspectives 111 (4), 598–603. https://doi.org/10.1289/ehp.5563.

TRUCOST 2013 Natural capital at risk: The top 100 externalities of business TEEB-TRUCOST Report p. 1-82.

Ugarelli, K., et al. 2017 The seagrass holobiont and its microbiome. Microorganisms 5 (81,) 1-28. doi:10.3390/microorganisms5040081.

UNO-DESA 2020 World social report 2020. Inequality in a rapidly changing world. ST/ESA/372 United Nations publications, pp.198 ISBN 978-92-1-130392-6, eISBN 978-92-1-004367-0.

Verhougstraete, M.P., Pogreba-Brown, K., Reynolds, K.A., Lamparelli, C.C., Zanoli Sato, M.I., Wade, T.J., Eisenberg, J.N.S., 2020. A critical analysis of recreational water guidelines developed from temperate climate data and applied to the tropics. Water Res. 170, 115294. https://doi.org/10.1016/j.watres.2019.115294.

Waycott, M., et al. 2009 Accelerating loss of seagrasses across the globe threatens coastal ecosystems. PNAS 106 (30), 12377-12381. doi/10.1073/pnas.0905620106.

Webb, S.J., et al. 2019. Impacts of *Zostera* eelgrasses on microbial community structure in San Diego coastal waters. Elem. Sci. Anth. 7, 11. doi.org/10.1525/elementa.35.

Weiskerger, C.J., Brandão, J., Ahmed, W., Aslan, A., Avolio, L., Badgley, B.D., Boehm, A. B., Edge, T.A., Fleisher, J.M., Heaney, C.D., Jordao, L., Kinzelman, J.L., Klaus, J.S., Kleinheinz, G.T., Meriläinen, Päivi, Nshimyimana, J.P., Phanikumar, M.S., Piggot, A. M., Pitkänen, T., Robinson, C., Sadowsky, M.J., Staley, C., Staley, Z.R., Symonds, E. M., Vogel, L.J., Yamahara, K.M., Whitman, R.L., Solo-Gabriele, H.M., Harwood, V.J., 2019. Impacts of a changing Earth on microbial dynamics and human health risks in the continuum between beach water and sand. Water Res. 162, 456–470. https://doi.org/10.1016/j.watres.2019.07.006.