Estuarine, Coastal and Shelf Science 250 (2021) 107109

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: http://www.elsevier.com/locate/ecss

Local and meso-scale pressures in the eutrophication process of a coastal subtropical system: Challenges for effective management

Alessandra Larissa Fonseca [a,*], Alice Newton [b,c], Alex Cabral [d,e]

[a] Coordenadoria Especial em Oceanografia, Universidade Federal de Santa Catarina, 88040-900, Florianópolis, Brazil
[b] CIMA, FCT-Gambelas Campus, University of Algarve, 8005-139, Faro, Portugal
[c] NILU-IMPEC, Box 100, 2027, Kjeller, Norway
[d] Programa de Pós-Graduação em Oceanografia, Universidade Federal de Santa Catarina, 88040-900, Florianópolis, Brazil
[e] Department of Marine Sciences, University of Gothenburg, 413 19, Gothenburg, Sweden

ARTICLE INFO

Keywords:
Ecosystem services
DAPSI(W)R(M)
South Atlantic basins
Nutrient pollution
Plata plume water

ABSTRACT

Natural and anthropogenic pressures drive coastal eutrophication worldwide, depending on the system's physical and biogeochemical dynamics in multiple spatial and temporal scales. Understanding the complexity of this process is essential to support management efforts and sustainability. Nutrients load to the Bay of Santa Catarina Island (BSCI), an important area for mollusc aquaculture, fisheries and tourism in Brazil, were assessed to identify the pressures of the eutrophication process. An updated Driver-Pressure-State-Impact-Response framework was used to facilitate the understanding of the relationship between human activities and impacts on human welfare. Pressures from runoff and effluents from combined sources resulted in inputs of 1998 t N.year^{-1} and 155 t P.year^{-1} to the system. The watersheds were characterized as meso-active to eury-active for both N and P yields. In addition to the local anthropogenic pressures, meso-scale events, such as the seasonal influence of the Plata Plume River, act as an external source of nutrients, sometimes associated with harmful algae bloom events. The results show that eutrophication and its symptoms could impact 85% of the ecosystem services of the region. Management of eutrophication at BSCI requires integrated actions between the nine municipalities of the watershed, but there are obstacles in environmental legislation and political interest to promote it. This study provides the scientific basis for stakeholders and decision-makers to establish priorities and actions in coastal municipalities to minimize eutrophication.

1. Introduction

Cultural eutrophication has a global impact, affecting several types of water bodies from lakes to regional seas and generates significant losses in ecosystem services (Malone and Newton, 2020). Eutrophication - triggered by the entry of organic matter (OM) and nutrients, mainly nitrogen and phosphorus from the use of fertilizers in agriculture, and the lack of treatment of rural and urban wastewaters - promotes, among others, the disruption of the food chain, the loss of biodiversity and the formation of dead zones (Bricker et al., 2008; Diaz and Rosenberg, 2008; Smith and Schindler, 2009). To reverse this scenario, the 14th UN goal for sustainable development (SDG14) aims to reduce nutrient pollution in the biosphere (Griggs, 2013), establishing management responses that reduce the flux of nutrients and organic matter to the environment and promoting the conservation and restoration of the ecosystems of the land-ocean interface. Management strategies must also consider climate change scenarios, such as changes in precipitation, and the potential to intensify pressures on the environment (Boesch, 2019; Duarte and Krause-Jensen, 2018; Sinha et al., 2017).

Legislation and instruments to reverse the impacts of eutrophication, monitoring the environmental health status of coastal ecosystems, and periodically assessing current and future trends have been applied for decades in Europe, North America, Asia, and Australia (Bricker et al., 2008; Newton et al., 2014), with varying success (Boesch, 2019). As a result, the decrease of the trophic state has been registered in some ecosystems, which has led to a better understanding of the eutrophication sequels in the post-control of pressure sources, such as the irreversibility of biodiversity loss (Alexander et al., 2017). Brazil, country with ~8000 km of continental coastline, still faces difficulties to establish more participative coastal management (de Andrés et al.,

* Corresponding author.
E-mail address: alessandra.larissa@ufsc.br (A.L. Fonseca).

https://doi.org/10.1016/j.ecss.2020.107109
Received 14 April 2020; Received in revised form 29 October 2020; Accepted 15 November 2020
Available online 1 December 2020
0272-7714/

2018). Challenges include the definition of clearer public policies and environmental monitoring strategies in order to understand, for example, the effect of eutrophication on ecosystems coastal health (Andrade and Scherer, 2014; Turra et al., 2013). Appropriate environmental monitoring allows the assessment of environmental health status, to know its complexity and vulnerability to the anthropic actions, and to propose management mechanisms that promote sustainable development (Ferreira et al., 2011). In the absence of long-term data to quantify the pressures (e.g. nutrient loading) that trigger eutrophication in the ecosystem, nutrient inputs quantification, both by natural and anthropogenic sources, is an option that has been applied at the local (de Paula Filho et al., 2015; Teubner Junior et al., 2018), regional (Lacerda et al., 2008; Noriega and Araujo, 2009), and global (Seitzinger et al., 2010b) scales. Despite the uncertainties, studies indicate that nutrient runoff from anthropic activities surpassed natural inputs, mainly due to effluents without adequate treatment from agriculture, livestock, and domestic sources (Meybeck et al., 2006; Seitzinger et al., 2010b; Smith et al., 2003). Also, priority actions are necessary to reverse the impact of eutrophication on a global scale (Biermann et al., 2016).

There is a knowledge gap in the understanding of how anthropogenic pressures influence the eutrophication process of aquatic ecosystems of South America (Kroeze et al., 2012), which is relevant to the calibration of global models (Seitzinger et al., 2010b). The South Atlantic Hydrographic Basin (SAHB) is located in the subtropical region of Brazil, with a significant cluster of small and medium-size hydrographic basins, accounting for 2.2% of the Brazilian territory (ANA, 2015). These provide important ecosystem services, such as provisioning services and support important fisheries. The resident population is 13 million people, but the region is also important for tourism, so the visitor population swells this number during the high season (summer). The population is also growing because this area is undergoing economic development. Population density is three times bigger than the country average (ANA, 2015). These high anthropogenic pressures and the importance of fish stocks and biodiversity make the conservation of coastal and marine ecosystems a priority for the sustainability of this region (MMA, 2002). Periodic harmful algae blooms (HAB), a eutrophication symptom, are usually reported across the coast of South Brazil, forced by local pressure and meso-scale water intrusion (Plata Plume Water), causing significant economic losses (Alves et al., 2018). However, little is known about the anthropogenic pressures in the process of eutrophication in the land-sea interface of this region (Barletta et al., 2019; Van Der Struijk and Kroeze, 2010).

Relevant technical information and scientific knowledge can be used to inform stakeholders and decision-makers who can then participate in a proactive, adaptive management (Newton and Elliott, 2016). The DAPSI(W)R(M) framework (Elliott et al., 2017) aims to establish a common language among researchers, society and managers to facilitate the understanding of D-drivers (as demand) and human A-activities, their respective P-pressures and resulting change of S-state and the I-impact on human (W) welfare, as ecosystem services (ES). This cause-effect perspective allows defining R-responses (as Measures) to guarantee the conservation and recovery of ES. Cultural eutrophication is a change of the natural state (State Change) in the transition coastal ecosystems forced by human activities, as agriculture, urbanization and industry (Newton et al., 2014). It is endorsed by manageable endogenous pressures (EnMP), as the input of organic matter and nutrients from local activities, and by unmanaged exogenous pressure (ExUP), as the intrusion of oceanic eutrophic waters. Considering the complexity of the various uses (Drivers and Activities) and their impacts generated in a given ecosystem (Borja et al., 2013; Elliott et al., 2017), the detailed construction of the DAPSI(W)R(M) framework from of the State's definition of change (in this case, eutrophication) can establish a useful and objective line of reasoning for the implementation of coastal management strategies.

From a DAPSI(W)R(M) perspective, we test the hypothesis that the local pressure of anthropogenic nutrient loads, commonly seen in coastal systems of South America, acts in synergy with meso-scale events, i.e. the intrusion of nutrient-rich water masses, to produce elevated trophic conditions and loss of ecosystem services in a subtropical bay. The results of this study will provide a better understand of the eutrophication problem of coastal systems that receive nutrient inputs for both continental and oceanic sources, highlighting important management strategies to improve this scenario.

2. Methods

2.1. Study area

The Bay of SC Island (BSCI), located in the central region of SAHB and continental shelf of SC (48.56° W, 27.57° S), is set in the central and most populous region of Santa Catarina State (SC) in southern Brazil (Cabral et al., 2020). The system is shallow (average depth is 3.4 m) and covers an area of 430 km^2 (Garbossa et al., 2017). BSCI is formed by two water bodies, (North and South Bays), that are connected in the center by a narrow channel. Each bay is permanently connected to the ocean, in the north and south, respectively (Fig. 1). BISC is located in a mosaic of marine protected areas, such as the Right Whale Sanctuary and the Pirajubaé Extractive Area, the latter of which is used by traditional communities for the harvest of cockles, fish, and shrimp (Pagliosa et al., 2005).

The climate is subtropical, with a humid and hot summer, type Cfa according to the Köppen classification. Normal climate conditions, based on a mean of 30 years (1960–1991), are a temperature range from 16 °C to 25 °C and annual precipitation of 1760 mm (de Souza et al., 2018).

The continental shelf of SC is the most productive fishing region of the Brazilian coast, because of eutrophic water masses throughout the year (Bordin et al., 2019; Sabatini et al., 2018). Coastal upwelling associated with Ekman transport forced by northerly winds, promotes the intrusion of the nutrient-rich South Atlantic Central Water (SACW), mostly in the late-spring and summer seasons. The Plata Plume Water (PPW), influenced by Subantarctic Shelf Water, reaches the SC coast during winter forced by southerly winds, when maximum chlorophyll-*a* of 20 mg.m^{-3} has been observed in surface waters (Bordin et al., 2019; Braga et al., 2008).

2.1.1. Watersheds characteristic and anthropic activities

BSCI receives freshwater from many rivers that comprise a total watershed area of 1875 km^2. The main river is the Cubatão Sul (742 km^2), which drains to the South Bay (Fig. 1). Information about water quality is only available for 13 of the rivers (Table S1) that represent 94% of the total BSCI basin area. BSCI's watershed is formed by the rocky hills reaching an altitude of about 1000 m and by the coastal plain, where rivers show saline gradient and mangroves ecosystems. The region is the South limit distribution of mangrove forest in South America (Netto et al., 2018). Soils are mostly composed by Glei little humic, golic cambisol, podzolic and quartz sands (IBGE, 2011).

The population of the BSCI watershed is about 958 thousand people spread over 337 km^2 of urban area in 9 municipalities (Table S1). More than 55% of the population has septic tanks with only 16% linked to sewage treatment in primary and secondary wastewater treatment plants (WWTPs). Summer tourism increases the local population threefold and thus the pressures on water quality.

Gross Domestic Product (GDP) in 2015 by services, industry and harvesting activities in the Municipalities of BSCI was about USD 9.9 billion. This is attributed to industry, technology and informatics, footwear, non-metallic minerals and harvesting activities, each of these contributing about 20% of the GDP (FIESC, 2015). The agricultural area was about 7000 ha in 2015, with rice, sugar cane, cassava and corn as the main crops (IBGE, 2017). Within the animal sector, poultry (2727 thousands of heads) and cattle (70 thousands heads) were the most important activities, followed by goats and pork that sums 13000 heads

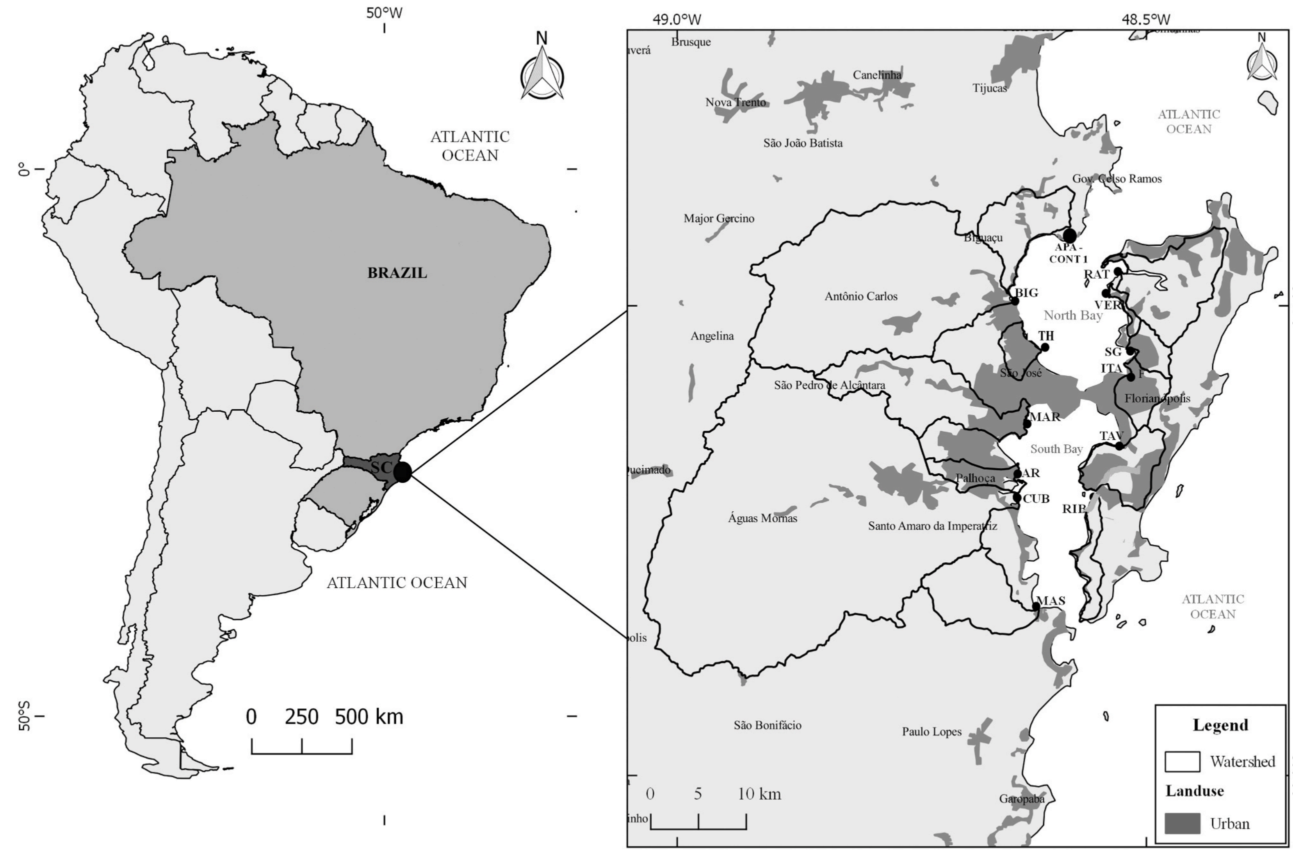

Fig. 1. Map of Bay of Santa Catarina Island (BSCI), South and North bays, and its watershed. Urbanization area for each Municipality is in dark gray. Rivers names are in bold: **AR**iriú, **MAR**uin, **BIG**uaçu, **CUB**atão, **ITA**corubi, **S**aco**G**rande, **RAT**ones, **T**. **H**enriques, **RIB**eirão, **APA**, **TAV**ares, **MAS**siambu. See Table S1 for river's description.

(IBGE, 2017). The production of seafood in BSCI is about 17000 t.year^{-1} of bivalve molluscs (oyster and mussel) from aquaculture farms, which represents 70% of the national production (Suplicy, 2018). Harmful algal blooms (HAB) occur periodically in the bay, causing losses to aquaculture and fisheries, as well as being a threat to human health (Alves et al., 2018; Garbossa et al., 2017).

2.2. Quantification of pressures from the watershed: load and yield of N and P

Estimating the emission of N and P for each activity or natural source is a useful tool in environments that are not regularly monitored, as in developing countries (de Paula Filho et al., 2015; Lacerda et al., 2008; Teubner Junior et al., 2018). We used an emission factor approach to identify the main anthropogenic and natural sources of N and P and their loads to the BSCI watershed. This is the tool used by environmental agencies in Europe (Crouzet et al., 1999) and North America (Schwarz et al., 2006). The anthropogenic pressures were characterized as rural (agriculture, livestock and atmospheric) and urban (sewage, runoff and atmospheric). These are the main anthropic inputs to natural waters in Brazil (Teubner Junior et al., 2018) and around the world (Seitzinger et al., 2010a). Equations (Table S2) and details of all the methodological procedures to estimate the emission of N and P are described according de Paula Filho et al. (2015) and Teubner Junior et al. (2018), who applied this methodology in a semi-arid and tropical Brazilian watershed, respectively. Details about variables applied in these equations are described below.

Atmospheric input (rainwater) of dissolved inorganic nitrogen (DIN) and phosphorus (DIP) in urban regions was estimated using the urbanized area of each municipality (Table S1), average annual precipitation (1825 mm.year^{-1}) from 2013 to 2015, and nutrient concentration (mg.m^{-3}) measured in the rainwater. Nutrients concentrations in rainwater of BSCI basins are scarce for nitrogen and non-existent for phosphorus. This study considered DIN concentration of 221 mgN.m^{-3}, estimated in rainwater from an urbanized site of Florianópolis city (Hoinaski et al., 2014) and DIP concentration of 11 mgP.m^{-3} (Mizerkowski et al., 2012) from Paranaguá city (population of ~134 thousand in 2010). Paranaguá watersheds have similar geomorphology, climate and anthropic development level as the basins of BSCI. The atmospheric inputs in rural and natural areas were calculated using the areas that are not urbanized (total area minus urbanized area) and the DIN (53 mgN.m^{-3}) and DIP (7 mgP.m^{-3}) concentrations estimated by Mizerkowski et al. (2012) for sites from Paranaguá Bay watershed without dense urbanization. Depending on the soil characteristics, 60–70% of the N and P that was deposited was retained in the soil and did not reach natural waters (Lacerda et al., 2008; de Paula Filho et al., 2015).

Nutrients load from domestic sewage in the urban areas were estimated from a total load of biochemical oxygen demand (BOD, t.year^{-1}) into natural waters for each municipality in 2013 (ANA, 2017), considering the sewage ratio BOD/DIN and BOD/DIP indicated in (von Sperling, 2007). For rural areas, the area of each crop (ha.year^{-1}) cultivated per municipality in 2015 was obtained from the Brazilian Institute of Geography and Statistics database (IBGE, 2017). Agricultural guidelines for the application of inorganic nitrogen and phosphate fertilizers (kg.ha^{-1}), loss rates (%) during the production and retention in the soil were considered for each crop (de Paula Filho et al., 2015; Eberhardt and Schiocchet, 2015). N and P emissions (t.year^{-1}) from livestock farming were estimated by the number of animals produced in 2015 in each Municipality from the IBGE database (IBGE, 2017). Daily production of manure (kg.animal^{-1}.day^{-1}), the concentration of inorganic N and P in manure produced (kg.animal^{-1}.day^{-1}), N–NH_3 emission (kg NH_3.animal^{-1}.year^{-1}) and soil retention factor were used as described in de Paula Filho et al. (2015).

The N and P loads by Municipality (t.year^{-1}) were extrapolated to loads for river considering the proportional area of each river watershed in each Municipality. We then considered the proportionality between areas (km^2) and applied the constant-share methods proposed by Smith et al. (2013). Those values were used to validate the nutrients load estimated, which was determined by the relationship between the nutrient loads estimated through emission factors and the loads measured *in situ*. Dissolved inorganic nitrogen and phosphorus concentrations from the thirteen rivers of BSCI watershed (N = 414, sampled from 2005 to 2010) where considered in this validation as *in situ* data, using the data set from Cabral et al. (2020). Nutrients loads for each river (t.year^{-1}) were calculated from DIN and DIP concentration (mg. m^{-3}) measured in each sampling day and the river discharge (m^3. month^{-1}), and then converted to t.year^{-1}. The river discharge (VQ, m^3. month^{-1}) of each basin was estimated (Table S2) considering rainfall, evaporation and the watershed area (Table S1), as described in Dupra et al. (2000). Precipitation (mm), evaporation (mm), and atmospheric temperature (^{o}C) were acquired from an automatic meteorological station (83,897), located in the South Bay, from INMET (National Institute of Meteorology, Brazil).

N and P yield estimated and observed per rivers (t.km^{-2}.year^{-1}) were normalized to a global yield average of N (0.355 t km^{-2}.year^{-1}) and P (0.095 t km^{-2}.year^{-1}), as suggested by Meybeck et al. (2006), with global reference values from Green et al. (2004) and Seitzinger et al. (2002), respectively. We also estimated the phosphorus and dissolved inorganic nitrogen yields (t.km^{-2}.year^{-1}) for each watershed area by regression equations (Table S2) proposed by Smith et al. (2003), which uses population number and river discharge (m^3.year^{-1}). Population number was estimated in each watershed from the population of each municipality (Table S1). The river discharge (V_Q, m^3.year^{-1}) of each basin was estimated as described before (Table S2).

2.3. *HAB occurrence and frequency as indicator of eutrophication*

HABs is a direct effect (Cloern, 2001) and a symptom of eutrophication. These have occurred periodically in the BSCI, affecting Ecosystem Services, the local economy, and Public Health. The National Program for Hygiene and Sanitary Control of Marine Bivalve Molluscs perform periodic analyses of harmful microalgae density and their toxins in order to prevent the commercialization of contaminated products by phycotoxins (as Diarrheic Shellfish Poisoning - DSP). Aquaculture farms of South (N =∼ 5) and North (N = ∼3) Bays were monitored monthly by the Agricultural State Agency (CIDASC). However, these data were not systematized and analyzed with respect to the eutrophication process. Results of this program were compiled to quantify the number of red tides alerts published from 2013 to 2018, except for 2015, when quantification of phytoplankton was not performed. According to Alves et al. (2018), alerts occurred when mouse bioassay were positive or when *Dinophysis* spp, *Pseudo-nitzschia* spp. or *Gymnodinium catenatum* density were higher than 500 cells.L^{-1}, 105 cells.L^{-1} and 100 cells.L^{-1}, respectively. Details about sampling and analysis are meticulously described in Alves et al. (2018). Numbers of alerts were relativized by the number of analyses carried out in each year per bays. We considered that each alert corresponds to a secondary symptom of eutrophication in the BSCI, as recommend by Bricker et al. (2008).

2.4. *Ecosystem services affected by eutrophication*

The BSCI has shown eutrophication symptoms in the water since the 1990's, according ASSETS analysis (Cabral et al., 2020). Sediment core data shows increasing CNP concentrations since the 1980's, concomitantly to demographic growth, supporting that eutrophication has been historical in the system (Lamego et al., 2017). The Common International Classification of Ecosystem Services (CICES V5.1) (Haines-Young et al., 2018) where used to list the Ecosystem Services (ES) from BSCI and from estuarine zone of its tributaries. A literature review was considered to identify the ES potentially impacted by eutrophication in the region.

2.5. *Data analysis*

Spearman correlation was performed to identify significant ($p < 0.05$) correlations between two variables. Only significant correlations are shown in the text. The Kruskal-Wallis test was applied to test significant ($p < 0.05$) differences between estimated and observed (*in situ*) nutrients loads (t.year^{-1}) and yield (t.km^{-2}.year^{-1}) to the BSCI. These statistical analyses were performed using RStudio 1.0 Open Source Software (R Core Team, 2015). Multidimensional Scaling ordination (MDS), integrated with Pearson ($p < 0.05$) correlation, were run to understand the similarity of the rivers that drain to the BSCI according nutrients loads (t.year^{-1}) and yields (t.km^{-2}.year^{-1}), estimated and observed, and water quality parameters (dissolved oxygen, chlorophyll-a, dissolved inorganic nitrogen and phosphate concentrations) acquired from the Cabral et al. (2020) data set.

3. Results

3.1. *Nutrients emission by activity*

Livestock production was 2.8 million heads in the BSCI watershed in 2015 (Table S3), mainly poultry (97%) and cattle (2.5%). There was three times more production in the South Bay basins (2082 × 10^3 heads) than in the North Bay (728 × 10^3 heads). Maximum production (1243 × 10^3 heads) was in the Cubatão watershed - Águas Mornas municipality. Agriculture occupied 3303 ha in the North Bay and 2898 ha in the South Bays basins (Table S3). There was 6201 ha of agriculture during 2015, 30% of it for rice, which represents the major production area in BSCI. Sugar cane (19%), cassava (13%), corn (11%) represented important cultures produced in the BSCI. The BSCI watershed received organic matter equivalent to 6761 tons of BOD in 2013, from domestic sewage (Table S3). The South Bay basins received 4591 tons, 68% of the total organic load to the BSCI basins.

Livestock, agriculture and sewage were the main sources of nitrogen and phosphorus to the BSCI watershed, that received 1998 t N and 155 t P in 2015 (Table 1). Rural sources represented two times more N than the urban sources (Table 1). Livestock represented 56% of the total N loaded into the bays, followed by sewage (30%) and agriculture (8%). The South Bay (1311 t N.year^{-1}) received two times more nitrogen than the North Bay (687 t N.year^{-1}).

Phosphorus loads were estimated at 62.1 and 92.5 t P.year^{-1} to the North and South Bays, respectively. Rural and urban sources showed similar P loads in the North Bay, mainly from sewage (50%) and agriculture (40%) (Table 2). However, in the South Bay, 70% of total P came from sewage (64 t P.year^{-1}), more than two times higher than the rural sources (27 t P.year^{-1}).

3.2. *Nutrients loads by rivers, estimated and observed data*

The mean river runoff was 941 × 10^6 m^3 year^{-1} (equivalent to 30 m s^{-1}). The Cubatão, Biguaçú and Maruin rivers represented 77% of the runoff (Table S6). The total watershed load average, calculated by nutrients *in situ* concentrations, was of 793 t N.year^{-1} and 109 t P.year^{-1} (Table 2). Those values represented 41% and 82% of the N and P load, respectively, estimated by the emission factor approach (Table 2) for these basin areas (1763 km^2): 1946 t N.year^{-1} and 132 t P.year^{-1}.

Maruin (32%), Cubatão (20%) and Biguaçu (14%) were the main sources of N to the BSCI (for absolute *in situ* load values see Table S6). Most of the N discharge occurred into the South Bay (529 t N.year^{-1}),

Table 1
Rural and urban nitrogen and phosphorus loads (t.year^{-1}) by municipalities localized in the basin of Bay of Santa Catarina Island (North and South Bays).

Sources	Rural			Urban		
	Agriculture	Livestock	Atmosphere	Sewage	Atmosphere	runoff
Nitrogen (tN.year^{-1})						
Bay of SC Island	140	1131	69	608	48	1
North Bay	71	377	27	195	16	0
South Bay	69	754	42	413	32	1
Phosphorus (tP.year^{-1})						
Bay of SC Island	47.2	2.9	8.5	94.6	1.2	0.2
North Bay	27.0	0.9	3.3	30.4	0.4	0.1
South Bay	20.2	2.0	5.2	64.3	0.8	0.1

Table 2
Load (t.year^{-1}) and local yield - global yield ratios (Yl/Yg[a]) of nitrogen and phosphorus measured *in situ* (*Is*) and estimated by emission factor (EF) and Smith's (Sm) approaches.

	tN.year^{-1}		YlN/Yg[a]			tP.year^{-1}		YlP/Yg[a]		
	I s	EF	*I s*	EF	Sm	*I s*	EF	*I s*	EF	Sm
BSCI	**793**	**1946**	**1.8**	**4.2**	**2.1**	**109**	**132**	**1.7**	**1.2**	**1.0**
North Bay	**265**	**771**	**1.8**	**4.4**	**2.3**	**86**	**58**	**3**	**1**	**1.1**
APA	8	51	0.4	2.4	1.5	2	3	0.4	0.5	0.7
Biguaçú	114	480	0.8	3.5	1.4	15	40	0.4	1.1	0.7
T. Henriques	25	69	1.4	4	2.7	25	4	5.4	0.9	1.3
Ratones	78	56	3.1	2.2	2.2	26	3	3.8	0.5	1.0
Veríssimo	3	12	0.9	3.3	2.6	1	1	1	0.7	1.2
Saco Grande	13	22	3.4	5.7	3.1	4	1	4	1.3	1.5
Itacorubi	23	81	2.8	9.9	2.8	13	5	5.9	2.3	1.3
South Bay	**529**	**1175**	**1.7**	**3.9**	**1.8**	**23**	**75**	**0.3**	**1.5**	**0.8**
Tavares	4	11	0.2	0.6	0.6	0	1	0.1	0.1	0.3
Ribeirão	7	20	0.9	2.8	2.4	1	1	0.5	0.6	1.1
Maruin	258	399	3.6	5.6	2.8	11	28	0.6	1.5	1.3
Ariru	97	258	4.7	12.6	3.7	2	35	0.4	6.4	1.7
Maciambu	7	3	0.3	0.1	0.7	3	0	0.3	0	0.3
Cubatão	160	495	0.6	1.9	0.7	6	10	0.1	0.1	0.3

[a] Ylocal/Yglobal classification Oligo-active: 0.2–0.1; Hypo– active: 0.5–0.2; Meso-active: 2–0.5; Eury-active: 5–2; Hyper-active: 10–5; Hot spots: >10.

since Maruin and Cubatão drain to this region (Fig. 1). Those rivers were also the most representative in the estimated N loads (Table 2). Although the estimated loads were two times greater than the *in situ* values, the Kruskal-Wallis test did not show a significant difference between these values (KW = 1.33, df = 1, p = 0.25). The rivers were classified as meso-active for N yield, estimated from *in situ* values and according to Smith's methodology (Table 2). However, rivers were classified as eury-active when estimated from N load by emission factor approach. None of the yields results showed statistical differences (KW = 3.51, df = 1, p-value > 0.05).

Ratones (24%), Três Henriques (23%), and Biguaçú (13%) were the main rivers contributing to the P flux into the BSCI, according *in situ*

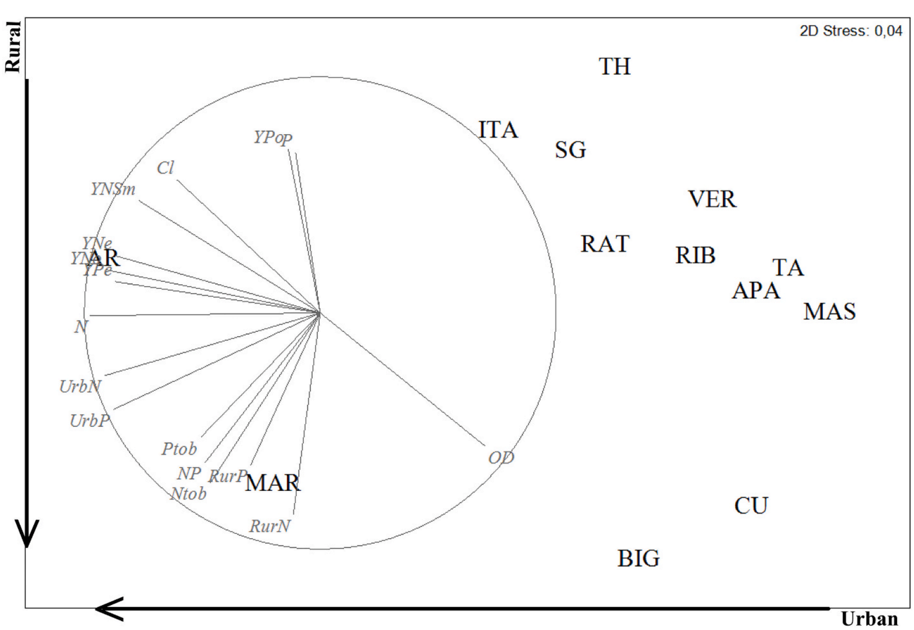

Fig. 2. MDS, with Pearson (p < 0.05) correlation, of **N** and **P** loads (t.year-1) estimated from **Urb**an and **Rur**al sources and total **ob**served *in situ*; of **N** and **P** Yields (t.km-2.year-1) estimated (**e**mission factor and **Sm**ith's) and **o**bserved; and water quality parameters (**D**issolved **O**xygen, **Ch**lorophyll-a, dissolved inorganic **N**itrogen and **P**hosphate concentrations) of rivers (**AR**iriú, **MAR**uin, **BIG**uaçu, **CUB**atão, **ITA**corubi, **S**aco**G**rande, **RAT**ones, **T. H**enriques, **RIB**eirão, **APA**, **TAV**ares, **MAS**siambu) that drain to BSCI.

measurements. The North Bay watershed was responsible for 80% (85.98 t $P.year^{-1}$) of this loading. However, when we considered the estimated loads, the South Bay basin is more important, with loading (74.71 t $P.year^{-1}$) mainly from the Aririú and Maruin rivers (Table 2). The Biguacú river that discharges in the North Bay was the most representative (30%) of the system. Rivers that drain to the South Bay were classified as hypo–active in relation to the P global trends, while the North Bay basin was eury-active and the BSCI basin was meso-active. In general, the *in situ* and estimated P loads did not differ (KW = 0.054, df = 1, p = 0.82) as well as the P yields (KW = 0.024, df = 1, p = 0.88).

N and P fluxes from the rivers (N = 13) showed a correlation ($p < 0.05$) with their estimated N (r = 0.92) and P (r = 0.55) loads. . Yields estimated by factor emission (r = 0.77 for N and r = 0.61 for P) and Smith's approaches (r = 0.77 for N and r = 0.71 for P) were significantly correlated ($p < 0.01$) with *in situ* DIN and DIP concentrations. The same was observed for the *in situ* yields (r = 0.87 for N and r = 0.74 for P).

The distribution of the main rivers of the BSCI according the MDS analysis (Fig. 2) was associated with urban (axis x) and rural (axis y) activity of its watershed. Urban descriptors, as total N (r = 0.91) and P (r = 0.87) estimated loads, correlated strongly with the first axis of the MDS. A gradient of urbanization was observed along this axis, with Aririú and Massiambu rivers in the opposite ordination. The total rural N (r = 0.85) and P (r = 0.65) estimated loads were associated with second axis and with Cubatão, Biguaçu and Maruin rivers.

3.3. HAB occurrence and frequency

A total of 134 alerts of HABs were issued from 2013 to 2018 (2015 was disregarded), which represents 10% of the total number of samples analyzed by CIDASC. There were significantly more alerts for the South Bay (KW = 17.14; df = 1; $p < 0.000$) than in the North Bay (Fig. 3). The alerts also did not always coincide in the two bays. Simultaneously alerts, both in the North and South bays, were observed in only 30% of the registers. The maximum of co-occurrence of the alerts emitted for both bays were observed in 2016, when alerts started in May and finished in July. Considering the seasonality, 32% of alerts occurred in the winter, following by spring (26%) and autumn (26%) seasons.

3.4. Ecosystem service of BISC

BSCI and its estuaries provide 49 classes of ES to society (Fig. 4), 41% as Regulation & Maintenance, 37% as Provision and 22% as Cultural. We identified 42 classes of ES that could be potentially impacted by eutrophication, mainly in a negative way, based on our results and on the available literature.

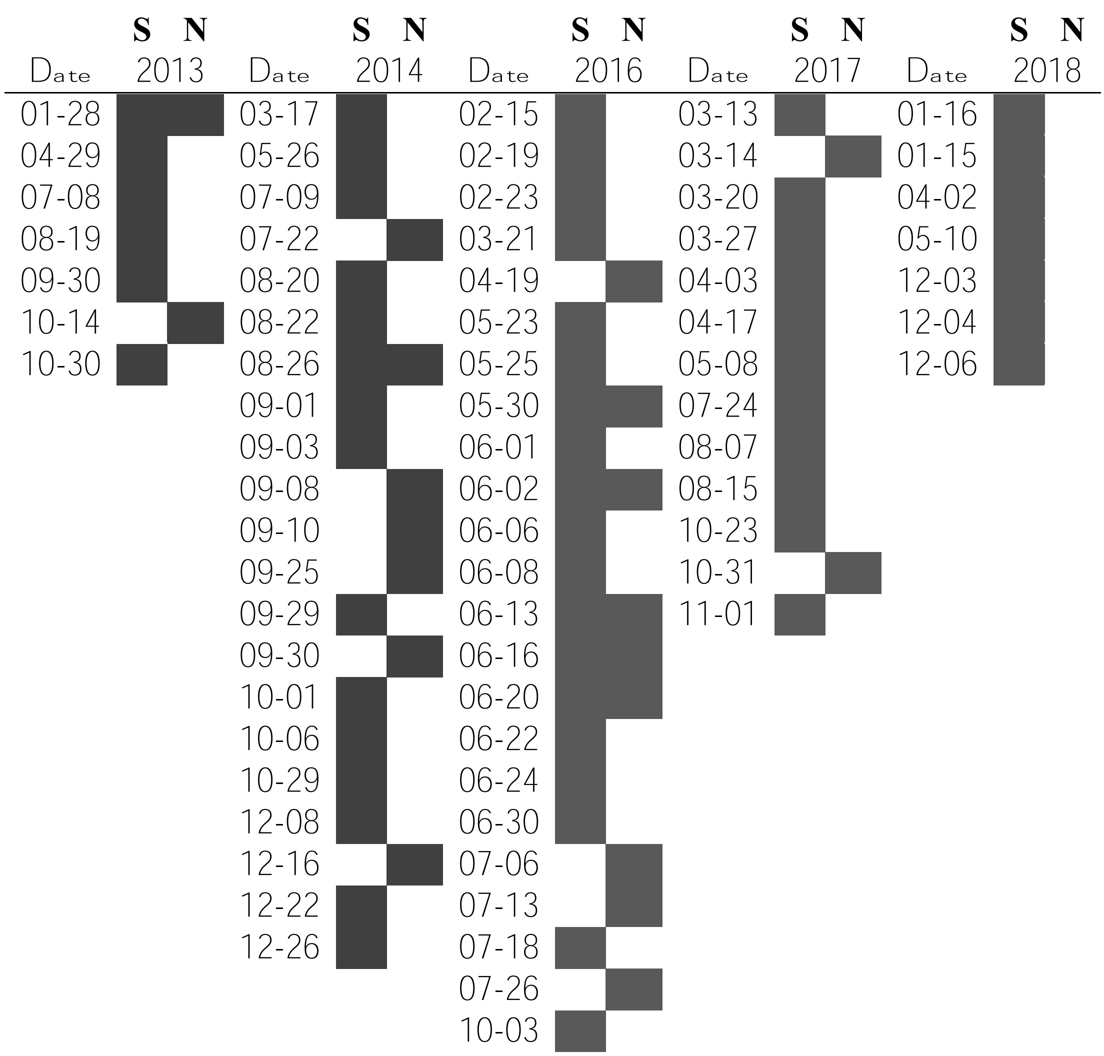

Fig. 3. Alerts of Harmful Algal Blooms (HABs) from 2013 to 2018 in the **S**outh and **N**orth bays. of BSCI.

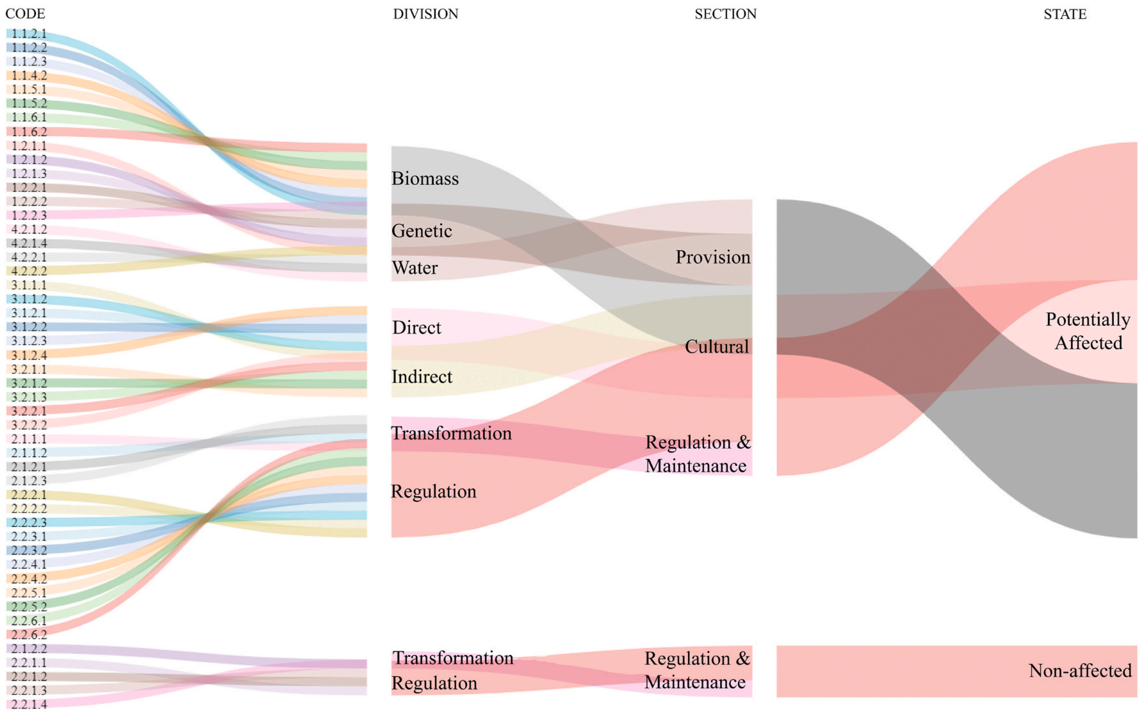

Fig. 4. Diagram (Sankey) of Ecosystem Services (section, division and code identification according CICES V5.1) from BSCI potentially affected and non-affected by eutrophication (State change). Description of each code, see Table S7.

4. Discussion

4.1. Eutrophication: its drivers, activities and pressures

Following the terminology proposed by Elliott et al. (2017), the Drivers - food production and occupation of space for society - have promoted livestock, agriculture, and urbanization. However, the lacks of proper management in those activities increase the flux of anthropogenic nutrients into the BSCI basins. One the these activities, urbanization, also increases pressures because of summer tourism (Driver), which triplicates the number of inhabitants of BSCI, amplifying N and P flux and, consequently, the trophic state of the local rivers (Cabral et al., 2020; Silva et al., 2016). These drivers are the main responsible for the nutrient loads entering the coastal zone in Brazil (Barletta et al., 2019; Teubner Junior et al., 2018) and worldwide (Meybeck et al., 2006; Seitzinger et al., 2010b; Smith et al., 2003).

The two methodologies used to quantify N and P input (de Paula Filho et al., 2015; Smith et al., 2003) allowed to calculate yield values observed (*in situ*) in the region, but only the application of the "emission factors approach" (de Paula Filho et al., 2015) identified the potential activities (livestock, agriculture, and urbanization) that change the state, as described above. The values estimated from the linear regression defined by Smith et al. (2003) characterize the state of the waters as meso-active, in agreement with the mean value observed *in situ* for the yield of N (Table 3), whereas the emission factor approach characterized the system as eury-active. However, when considering the variation of the loads observed *in situ* (Table S6), from the minimum to the maximum values, both methods are well adjusted to estimate the yield of N. Several authors (de Paula Filho et al., 2015; Molisani et al., 2013; Teubner Junior et al., 2018) give a margin of error up to three times for the estimated loads in relation to the observed *in situ*, due to the uncertainties of the methodology. Further, the estimates are based on the total form of N and P, while the *in situ* measurements are usually about inorganic dissolved compounds (e.g. DIN and DIP), as in this study. In relation to P values, both methods were satisfactorily and characterized the system as meso-active.

Table 3
Load (t.year^{-1}) and yield (t.km^{2}.year^{-1}) of N and P estimated to the Brazilian coastal basins. Classification was defined as ratio of local yield (Yl) normalized to global average yield (Yg) according Meybeck et al. (2006).

Systems	Lat°	Área (km^2)	Load (t.year^{-1})		Yield (t.km^2.year^{-1})		Yl/Yg
			N	P	N	P	Type N–P
Parnaíba River[a]	−2.7	42,810	14,517	8748	0.34	0.20	Meso-Eury
Ceará coastal[b]	−3.7	20,799	12,622	6704	1.65	0.62	Hyper
RGNorte coast[c]	−5.6	3300	5263	2762	5.08	1.59	Hot spots
Pernambuco[d]	−8.0	25,058	39,130	8914	1.56	0.36	Eury
Contas River[e]	−14	55,000	2299	1688	0.33	0.24	Meso-Eury
Vitória Bay[f]	−20	1925	10,784	5480	5.60	2.80	Hot spots
Macaé River[g]	−22	1765	1599	787	1.30	0.65	Eury-Hyper
Itajaí River[h]	−26	15,500	7889	641	0.51	0.04	Meso-Hypo
$BSCI_{estimat}$[i]	−27	1763	1957	133	1.10	0.08	Eury-Meso
$BSCI_{Smith}$[i]					0.67	0.08	Meso-Meso
$BSCI_{in\ situMean}$[i]			771	58	0.63	0.17	Meso-Meso
Mean Brazil		**18,736**	**10,074**	**4672**	**1.95**	**0.75**	

[a] De Paulo Filho et al. (2015).
[b] Lacerda et al. (2008), study in 15 basins.
[c] Lacerda et al. (2006), study in 7 basins.
[d] Noriega and Araujo (2009), study in 12 basins.
[e] De Paulo Filho et al. (2010).
[f] Teubner Junior et al. (2018).
[g] Molisani et al. (2013).
[h] Pereira Filho and Rorig (2016).
[i] This Study.

The Brazilian yields are, on average, 1.95 t $N.km^{-2}.year^{-1}$ and 0.75 t $P.km^{-2}.year^{-1}$, characterizing the coastal basins from hypo-active to hotspot (Table 3). Meybeck et al. (2006) characterized the South Atlantic basins (where BSCI is localized) in the global model COSCATs as eury-active for N, exceeding twice the values emitted in the global average. We also obtained, on average, a eury-active classification for N in the BSCI. However, most of the individual watershed values characterized the system as meso-active, which is in accordance to the global mean yield and equivalent to the values observed in the la Plata River (Argentina), Niger Delta (Niger), and Yellow River (China) (Meybeck et al., 2006).

The yields for N and P in the Vitória Bay (VB, SE Brazil), a yield hotspot (Table 3), were associated with high livestock contribution (up to 90%) when compared to other systems, including BSCI, where livestock contributed with 60% (Teubner Junior et al., 2018). In Rio Grande do Norte (RN, NE Brazil), a hotspot condition was also associated with the production of animal protein, in this case by marine shrimp aquaculture farms (De Lacerda et al., 2006). Although the relative importance of livestock was not so high for BSCI as for VB and RN, the contribution of this activity was significant higher for the N loads. In future scenarios, Seitzinger et al. (2010a) estimates an increase of N yields to the coastal zone by livestock contribution in Medium-Income Countries, such as Brazil, bringing the worst-case scenario to the coastal systems, such as BSCI. In response, the improvement of waste management in agriculture and livestock production is necessary in order to minimize nutrients inputs.

The N yield at BSCI correlated with urban characteristics, such as urbanized area and BOD, but not with the rural characteristics. However, the load of N correlated with both the urban and rural descriptors. Rural activities accounted for twice the N load estimated for BSCI, when compared to urban sources. Despite the importance of rural activities for N inputs, the basins size ($<$100 km^2) and the high population density ($>$1100 $inhab.km^{-2}$ in 40% of the basins), mainly in the shoreline of BSCI (de Andrés et al., 2018), indicate that urban sources generate strong pressures for the yield of N. In contrast, the load and the yield of P in the BSCI watersheds were related only with urban descriptors. Untreated urban sewage is the main source of P in Brazil and worldwide (Lacerda et al., 2008; Molisani et al., 2013; Noriega and Araujo, 2009; Pereira Filho and Rörig, 2016; Teubner Junior et al., 2018). Moreover, P loads are expected to increase in the urban centers of developing countries, where population is rising without effective wastewater treatment (Seitzinger et al., 2010a). Following the global classification of Smith et al. (2003), the BSCI basins have an average water flow and high population density (this study), conditions that decrease nutrient assimilation/exportation and elevate the susceptibility to long-term eutrophication.

4.2. State of eutrophication and its implication to ESs

Coastal eutrophication has increased in the last decades in a global scale, and is expected to continue to increase, especially where investment in mitigation responses, such as the treatment of urban and rural effluents, are scarce (Barletta et al., 2019; Bricker et al., 2008; (Seitzinger et al., 2010b). The risk of eutrophication for the BSCI rivers, as meso- and eury-active, corroborated with the poor and bad eutrophic status for BSCI indicated by the ASSETS model (Cabral et al., 2020). BSCI showed moderate-high (North Bay) to high (South Bay) susceptibility to eutrophication pressure according to these authors, that focus the eutrophication analysis by urban descriptors, especially in the shoreline (Garbossa et al., 2017). The present study brings the importance of rural activities for N loads, which dominate the innermost areas of the watershed and municipalities not bordering the bay. Nitrogen is the limiting nutrient to the primary production of the BISC (Cabral et al., 2020), reinforcing the potential of the N loads by rural activity to sustain the poor and bad scenarios for eutrophication characterized in this system. Among the ES classes identified here, those associated with energy production by physical forces (Provisioning), maintenance of the coastal physical structure (e.g. coastline), and thermal regulation (Regulation & Maintenance) are not affected by eutrophication, while all others might suffer some impact (Table S7).

Although there is no systematic monitoring of the trophic state of the rivers and the BSCI, the local benthic community has showed some impacts caused by eutrophication (Borja et al., 2010; Brauko et al., 2015; (Brauko et al., 2020b)). The loss of biodiversity and health of benthic fauna in the bay and the estuarine region of the tributaries has been associated with eutrophication indicators (OM, N, and P) (Brauko et al., 2020; Pagliosa and Barbosa, 2006; Souza, 2011; Weis et al., 2017). There has been no change in the response of the benthic communities to the impacts of eutrophication in the decade between those studies, indicating that the loss of the benthic community, as biodiversity and function, are chronic in the system, which could irreversibly damage the food chain and ES (Alexander et al., 2017; Bernardino et al., 2015). Biotic ES associated to benthic community in BSCI (De Lima et al., 2018; Weis et al., 2017), as provision of genetic material or as regulation and maintenance by weathering processes and their effect on sediment quality (Norkko et al., 2015), were negatively affected by eutrophication. However, a systematic monitoring program is necessary to evaluate the dimension of these impacts on the system, which could be intensified by climate change (Carneiro et al., 2020) and others anthropic intervention that suppress this community, as dredging to navigation or hydraulic landfill in the shoreline (de Melo et al., 2011).

The physical characteristics of a coastal ecosystem, as water renewal and dilution of contaminants, determine the levels of eutrophication (Barletta et al., 2019; Elliott and De Jonge, 2002). Water exchange between the BSCI and the continental shelf waters occurs through the extreme channels, while the local hydrodynamic favors water and particulate matter accumulation in the central region of the bay (de Souza et al., 2018). A standing wave is generated by the overlap of the tidal waves that converge at the center of the bay, most of the time to the central region of the South Bay (cSB) (de Souza et al., 2018). The pressure of nitrogen is also higher in cSB, due to the N loads of the Maruin, Aririú, and Cubatão rivers, which contribute to 64% of the entire N load to the system. The highest concentrations of N in the sediment and fecal coliforms in the water also occur in the cSB, because of the low hydrodynamics and the high urban density areas (Bonetti et al., 2007; de Souza et al., 2018; Vianna and Filho, 2018). The higher trophic state observed in the cSB in comparison with the rest of the system is due to the high N loads and the low hydrodynamics (Cabral et al., 2020). In other words, the symptoms of eutrophication may be more intensive in the cSB, as indicated by higher thermotolerant coliform concentration (Garbossa et al., 2017), and by lower diversity of macroalgae and macrobenthos communities (Brauko et al., 2020; Martins et al., 2012).

The three main secondary symptoms of eutrophication are harmful algal blooms (HABs), low dissolved oxygen levels and loss of submerged aquatic vegetation, a serious change of ecological state (Bricker et al., 2008). The BSCI has shown elevated dissolved oxygen, associated with physical processes, such as the turbulence of the water column by the winds and tidal currents (Cabral et al., 2020). However, recent hypoxic and anoxic events have been reported associated with eutrophication and heat waves (Brauko et al., 2020b). Futhermore, HABs events are periodically reported at BSCI, especially in the South Bay, corroborating with the results of the N loads. There is no information about the submerged aquatic vegetation of BSCI.

Blooms of *Dinophysis* spp and *Pseudo-nitzchia* sp. that produce toxins have been recorded in the Brazilian subtropical coastal ecosystems for decades (Alves et al., 2018), including BSCI. High densities of *Dinophysis* spp. (mostly *D. acuminata* complex), a mixotrophic organism, has been associated with local and meso-scale dynamics (Alves et al., 2018; Proença et al., 2017; Tibiriçá et al., 2015). Local scales dynamics can be considered as endogenous managed pressure whereas meso-scale dynamics are exogenous unmanaged (Elliott et al., 2017). In winter, south quadrant winds bring the Plata River Plume into the continental shelf of Santa Catarina, where BSCI is located (Braga et al., 2008). The eutrophic and nutrient-rich water mass sustain high phytoplankton biomass (Bordin et al., 2019) and boost the occurrence of *Dinophysis* spp. in the BSCI and nearby systems, such as occurred in May 2016 (Alves et al., 2018; Proença et al., 2017). Good conditions for the development of *Dinophysis* spp., such as water column stability and nutrients availability (mainly ammonium), is found in the enclosed coastal systems of SC (Alves et al., 2018; Hattenrath-Lehmann et al., 2015; Tibiriçá et al., 2015), reinforcing the local eutrophication pressure as a synergistic effect that increases the intensity and duration of HAB events.

The blooms of *Pseudo-nitzchia* sp. are more frequent in spring-summer (Tavares et al., 2009; Tibiriçá et al., 2015), as occurred on January 11th of 2018 in the BSCI. Rainfall volume was 145 mm on that day, after 170 mm accumulation in the previous week. This is equivalent to the normal precipitation for the whole January month, which can be classify as an anomalous event (Barcellos et al., 2020). The summer heavy rains fuel the algal population with nutrients, since both non-point and point source (e.g. sewage plants) pollution runoff increase. Furthermore, BSCI is located in a region where extreme events of precipitation and temperature have been increasing (Gouvêa et al., 2017), factors that might hinder the success of the strategies/responses to minimize the impacts of eutrophication (Duarte and Krause-Jensen, 2018).

Losses in aquaculture production (Provisioning) by HABs are very frequent in the BSCI and the coastal zone of SC State (Proença et al., 2017; Suplicy, 2018). An HAB event in 2016, the highest recorded between 2013 and 2018, was promoted by meso-scale pressure and generated the loss of 4 million USD on the commercialization of aquaculture production, compromising the livelihood of more than 500 families who work directly and indirectly in this activity (Proença et al., 2017). Food poisoning by red tides causes gastrointestinal diseases, vertigo and fever, and has been reported by the population of SC coast since the 1970s, mainly during the winter months (Schlemper, 2002). This author reinforces that the impact on human health is underestimated because many cases are unreported to the public health system.

Although exogenous pressures cannot be controlled by management responses (Dolbeth et al., 2016; Elliott et al., 2017; Newton et al., 2003), the control of endogenous sources might minimize the duration and intensity of HABs events. In future scenarios of eutrophication in the global coastal zone, nutrient loads from rivers are expected to increase the potential risk of HABs, and changes in the nutrient ratio further the threat (Bricker et al., 2008). Thus, to better understand the dimension of local pressures on these events, environmental monitoring that assesses the availability of nutrients (N, P and silicate) is crucial (Ferreira et al., 2011). We reinforce that the concentrations of metals and pesticides in water, sediment and biota (oysters and mussels) of BSCI are within the limits established by Brazilian legislation (de Souza et al., 2016; Sáenz et al., 2010), while nutrient concentrations are not (Cabral et al., 2020). Nevertheless, the frequency, intensity, and duration of primary and secondary eutrophication symptoms damage the diversity of niches and species (from the gene to the community) and the resilience capacity of the ecosystem (Alexander et al., 2017; Diaz and Rosenberg, 2008).

The lack of efficient wastewater treatment systems is observed in the region (Cabral et al., 2019) and it also affecting the trophic state and human health of BSCI, which is incompatible with the current tourism activities and seafood production. Two million tourists visited the BSCI each year, it generated approximately 250 million USD of revenue during the summer vacation (Cappelini et al., 2011). The impact of eutrophication on tourism is difficult to measure, since the loss of water quality also involves domestic effluent inputs, with enteric viruses, organic matter, and nutrients (Rigotto et al., 2010). The number of cases of waterborne diseases, such as gastroenteritis and dermatitis, was related to high trophic status in the Ratones river basin, which drains into the BSCI and has a protected environmental area (Silva and Fonseca, 2016). During the summer, the population triplicates in this basin, damaging the environmental quality of the river-estuary, significantly increasing the trophic state (Cabral et al., 2020; Silva et al., 2016). In this scenario, the algae proliferation promotes bad smell (Regulation & Maintenance) and particulate matter, fouls the bathing waters (Cultural) (Silva et al., 2016).

Overfishing, trawling and water pollution are responsible for the decreasing yield of fishing and fish quality in systems adjacent to BSCI (Martins et al, 2014, 2018; Prudencio et al., 2014), also affecting the preservation of key species for marine conservation (Macedo and Medeiros, 2018). The perception of traditional fishermen from the central region of BSCI is that the urbanization around the estuarine rivers affected the taste of the seafood (person. comm.). However, there is lack of data to support this and future studies are needed to quantify the dimension of the impact of eutrophication in the BSCI human welfare. Moreover, in order to understand the system's response to future pressures, reproducible fieldwork protocols (compatible with the reality of local institutions) and a database (allowing ecological modeling) are fundamental to provide management responses based on ecosystem services (Anzaldua et al., 2018; Newton and Weichselgartner, 2014).

4.3. Management responses

Eutrophication mitigation strategies on BSCI must include controlling nutrient and organic matter loads and ecosystem restoration in a sustainable way, as detail described in others systems (Biermann et al., 2016; Duarte and Krause-Jensen, 2018; Luk et al., 2019; Newton et al., 2003; Selman and Greenhalgh, 2009; Wu et al., 2017). The success of these strategies depend on social, economic, political and governmental characteristics, as well as the interference of environmental pressures that occur at larger spatial and temporal scales, such as climate change (Boesch, 2019; Duarte and Krause-Jensen, 2018; Gari et al., 2015). Environmental management as a response to eutrophication should be ecologically sustainable, socially desirable/tolerable, and economically viable; constructed by defensible ethic, cultural inclusion, and feasible technology; supported by legal, political and administrative permissible instruments; and guaranteed by popular participation and effective communication (Barnard and Elliott, 2015).

Based on these premises, the management of response actions in the BSCI should occur at the State level (Santa Catarina), since the system is influenced by the activities of nine municipalities, which are independent in the management of their space. The innermost municipalities, Águas Mornas, Sto Amaro da Imperatriz, São Pedro de Alcântara and Antônio Carlos, are economically supported by rural activities, which are important sources of N and P to the bays. Furthermore, they do not directly depend on the ecosystem services of the BSCI and are less motivated to restore the system, since they do not have a shoreline. According to the State legislation (State Law SC No. 5010/2006), these municipalities are not part of the coastal zone and are not considered for coastal management strategies, which is a barrier to defining responses that aim the integrity of marine ecosystem services (de Andrés et al., 2018). As opposed to the US and Europe, Brazil does not have specific policies to minimize the impact of eutrophication (Silva et al., 2018), as the Nitrate Directive, which aims to reduce water contamination by agricultural fertilizers (Bricker et al., 2008; Newton et al., 2003). Brazilian legislation sets the management of water resources by River Basin Committees (Brazilian Law No. 9.433/1997). The legal instrument for water quality (CONAMA No. 357/2005) is used to ensure the improvement of water quality. The Cubatão and Tijucas river basins, where three of the four innermost municipalities are located, have had basin committees for at least a decade, but actions of these management units are still ineffective, characterized by discontinuity, little social participation and attention to the improvement of environmental quality (Siebert, 2010).

The effectiveness of response actions to eutrophication requires social mobilization, cooperation between different institutions and governmental support (Newton and Elliott, 2016). Solutions must start from the understanding of the problem by society and the definition of future scenarios (Borja et al., 2016; Newton et al., 2016). The Management Strategies in the SC coastal zone have a legal instrument, the State Plan for Coastal Management of Santa Catarina (State Law SC No. 13553/2005), but there is no government commitment and no clear directives to implement it in practice (Guião and Scherer, 2018), as is the case of the Basin Committees. Management strategies must also involve broad social mobilization, dissemination of technical information, and effective participation of the various actors in the decisions.

Recent changes in the Brazilian Forest Law (Brazilian Law No. 12.727, 2012) began politically by the State of SC and promoted more land uses and less preservation of riparian forests and marine wetlands (Grasel et al., 2019; Pagliosa et al., 2012). The Florianópolis Municipal Master Plan foresees the use of wetlands for the expansion of the city in the next 20 years, which is also observed in other municipalities of SC and Brazil (de Andrés et al., 2018). The preservation and restoration of riparian forests and wetlands are responses used in many aquatic systems to minimize nutrient input and eutrophication (Wu et al., 2017). However, the current political trends point away from the social and environmental demands of BSCI basin, making it difficult to implement responses that minimize eutrophication in the short and long-term.

Despite of the difficulties reported, the economic losses associated with eutrophication (Proença et al., 2017) could stimulate social mobilization in BSCI and governmental management responses to control the state change in the ecosystem. The integrated management of the Cubatão, Biguaçú and Maruin rivers and the central region BSCI could be the initial step for an optimized strategy of regional management. Municipal Master Plans for urban occupation of coastal plains wetlands and bordering mangrove areas should be reconsidered, since these areas play an important role in mitigating eutrophication (Wu et al., 2017). The wetlands also provide space to be occupied by mangroves in a sea level rise scenario, protecting private and public goods and services (Schaeffer-Novelli et al., 2016).

In addition, to preserve target ecosystems and mitigate eutrophication (Onorevole et al., 2018; Rodriguez-Dominguez et al., 2020), bivalve aquaculture could promote its control as well. Bivalve production in BSCI, Pacific oyster *Crassostrea gigas* and the brown mussel *Perna perna*, might be cropping phytoplankton biomass due to their high filtration capacity (Silva et al., 2019) and promoting a top-down control of eutrophication (Cabral et al., 2020), as observed in others systems (Saurel et al., 2014). Macroalgae farms, of native *Ulva* sp. and exotic *Kappaphycus alvarezii*, have the potential to absorb N and produce biomass for food and pharma industries (Hayashi et al., 2011; Hayashi and Reis, 2012). However, non-native species may disturb the ecosystem, as loss of biodiversity (i.e. competition and pest introduction) and decreasing the mineralization potential in the sediment (Gallardi, 2014). Ecosystem-based management and nature-based solutions implemented in BSCI (Scherer and Asmus, 2016) may guarantee a sustainable aquaculture activity and boost the ecosystem service that has the potential to mitigate eutrophication (Gallardi, 2014; Silva et al., 2019; Suplicy et al., 2017).

The inherent uncertainties of this study, such as the low precision and accuracy of the methods and the lack of important ecological information (such as submerged shallow areas) should be considered in the BSCI. Some of these uncertainties are also cited in systems that have long-term monitoring and integrated scientific management to reduce the impact of eutrophication, such as the Baltic Sea (Boesch 2019; Malone and Newton, 2020). This reinforces the urgent need for environmental monitoring and assessment focusing on eutrophication in BSCI.

5. Conclusion

Human activities around the BSCI produce endogenous pressures from rural and urban sources, resulting in an input of 1998 t N and 155 t P per year and classifying the system as eury-active for N and meso-active for P, in relation to global trends. Thus the biogeochemical processes at the land-ocean interface of the BSCI ecosystem are fueled by both natural and anthropogenic inputs of nutrients, resulting in eutrophication. Rural activities accounted for twice the N load estimated for BSCI, when compared to urban sources. Despite the importance of rural activities for N inputs, the basins size (<100 km^2) and the high population density (>1100 inhab.km^{-2}), mostly in the shoreline of BSCI, indicate that urban sources generate strong pressures for the yield of N. In contrast, the load and the yield of P were related primarily with urban activities and the inputs of P into the watersheds.

Harmful blooms of *Dinophysis* spp and *Pseudo-nitzchia* sp. were associated with different scale of pressures, intensified by the southerly winds (winter) and high precipitation events (summer) , respectively. Eutrophication descriptors were correlated to ecosystem services loss in the BSCI, which depends on good water quality to provide aquaculture production, fisheries and develop tourism. The response and management strategies to prevent eutrophication in the BSCI should occur at the State level (Santa Catarina), since the governance of the system is the responsibility of nine municipalities that manage their space independently. However, there are major obstacles and barriers to develop efficient management strategies to control endogenous pressures leading to eutrophication. These include barriers at the national, state, and local levels such as: (i) Lack of clarity and directives from the national government with respect to environmental management, such as legislation that facilitates the human uses of riparian forest, marine wetlands and mangrove areas for agriculture and aquaculture; (ii) The coastal management plan is fragmented and does not include the participation of rural municipalities from the BSCI catchment area, which are potentially source of nutrients to the bay.; (iii) Poor implementation and commitment to established management plans; (iv) Poor social awareness and social participation in issues related to the improvement of environmental quality. Furthermore, according to the urban plans of the main cities, the catchment population will duplicate in the next decade. In this scenario, the reduction of pressures and the sustainability of marine resources associated with eutrophication in BSCI (SDG 14.1) are unlikely to be achieved by 2030.

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.

Acknowledgments

This work was supported by the Instituto Nacional de Ciência e Tecnologia de Transferência de Materiais Continente-Oceano (INCT-TMCOcean 573.601/2008-9). Alice Newton acknowledges IMBeR, Future Earth Coasts and Future Earth Ocean KAN. We also thank the anonymous reviewers and editor for comments and suggestions that improved the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2020.107109.

Author statement

Alessandra Larissa Fonseca: Conceptualization, Formal analysis, Writing - Review & Editing.

Alice Newton: Conceptualization, Writing - Review & Editing.

Alex Cabral: Data Curation, Writing - Review & Editing.

References

Alexander, T.J., Vonlanthen, P., Seehausen, O., 2017. Does eutrophication-driven evolution change aquatic ecosystems? Philos. Trans. R. Soc. B Biol. Sci. 372 https://doi.org/10.1098/rstb.2016.0041.

Alves, T.P., Schramm, M.A., Proença, L.A.O., Pinto, T.O., Mafra, L.L., 2018. Interannual variability in Dinophysis spp. abundance and toxin accumulation in farmed mussels (Perna perna) in a subtropical estuary. Environ. Monit. Assess. 190 https://doi.org/10.1007/s10661-018-6699-y.

Ana, 2015. Agência Nacional de Águas (ANA) - Conjuntura dos recursos hídricos no Brasil: regiões hidrográficas brasileiras. Available: http://www.snirh.gov.br/portal/snirh/centrais-de-conteudos/conjuntura-dos-recursos-hidricos.

Ana, 2017. Agência Nacional de Águas (ANA) - Atlas Esgotos - Despoluição de bacias hidrográficas. Available: http://atlasesgotos.ana.gov.br/.

Andrade, J., Scherer, M.E.G., 2014. Decálogo da gestão costeira para Santa Catarina: avaliando a estrutura estadual para o desenvolvimento do Programa Estadual de Gerenciamento Costeiro. Desenvolv. Meio Ambiente 29, 139–154. https://doi.org/10.5380/dma.v29i0.31405.

Anzaldua, G., Gerner, N.V., Lago, M., Abhold, K., Hinzmann, M., Beyer, S., Winking, C., Riegels, N., Krogsgaard Jensen, J., Termes, M., Amorós, J., Wencki, K., Strehl, C., Ugarelli, R., Hasenheit, M., Nafo, I., Hernandez, M., Vilanova, E., Damman, S., Brouwer, S., Rouillard, J., Schwesig, D., Birk, S., 2018. Getting into the water with the ecosystem services approach: the DESSIN ESS evaluation framework. Ecosyst. Serv. 30, 318–326. https://doi.org/10.1016/j.ecoser.2017.12.004.

Barcellos, D.R., Wolf, M.A.D., Sanches, S.R., de Quadro, M.F.L., 2020. Analysis of precipitation temporal variability in the Florianópolis/SC city. Cienc. Nat. 42, 1–18. https://doi.org/10.5902/2179460X55310.

Barletta, M., Lima, A.R.A., Costa, M.F., 2019. Distribution, sources and consequences of nutrients, persistent organic pollutants, metals and microplastics in South American estuaries. Sci. Total Environ. 651, 1199–1218. https://doi.org/10.1016/j.scitotenv.2018.09.276.

Barnard, S., Elliott, M., 2015. The 10-tenets of adaptive management and sustainability: an holistic framework for understanding and managing the socio-ecological system. Environ. Sci. Pol. 51, 181–191. https://doi.org/10.1016/j.envsci.2015.04.008.

Bernardino, A.F., Netto, S.A., Pagliosa, P.R., Barros, F., Christofoletti, R.A., Rosa Filho, J. S., Colling, A., Lana, P.C., 2015. Predicting ecological changes on benthic estuarine assemblages through decadal climate trends along Brazilian Marine Ecoregions. Estuar. Coast Shelf Sci. 166, 74–82. https://doi.org/10.1016/j.ecss.2015.05.021.

Biermann, F., Bai, X., Bondre, N., Broadgate, W., Arthur Chen, C.T., Dube, O.P., Erisman, J.W., Glaser, M., van der Hel, S., Lemos, M.C., Seitzinger, S., Seto, K.C., 2016. Down to Earth: contextualizing the anthropocene. Global Environ. Change 39, 341–350. https://doi.org/10.1016/j.gloenvcha.2015.11.004.

Boesch, D.F., 2019. Barriers and bridges in abating coastal eutrophication. Front. Mar. Sci. 6, 1–25. https://doi.org/10.3389/fmars.2019.00123.

Bonetti, C., Bonetti, J., Barcellos, R.L., 2007. Caracterização sedimentar e geoquímica de sistemas costeiros com ênfase na avaliação da influência de sítios de cultivo de moluscos, pp. 139–149.

Bordin, L.H., Machado, E. da C., Carvalho, M., Freire, A.S., Fonseca, A.L.D.O., 2019. Nutrient and carbon dynamics under the water mass seasonality on the continental shelf at the South Brazil Bight. J. Mar. Syst. 189, 22–35. https://doi.org/10.1016/j.jmarsys.2018.09.006.

Borja, A., Elliott, M., Andersen, J.H., Cardoso, A.C., Carstensen, J., Ferreira, J.G., Heiskanen, A.S., Marques, J.C., Neto, J.M., Teixeira, H., Uusitalo, L., Uyarra, M.C., Zampoukas, N., 2013. Good Environmental Status of marine ecosystems: what is it and how do we know when we have attained it? Mar. Pollut. Bull. 76, 16–27. https://doi.org/10.1016/j.marpolbul.2013.08.042.

Borja, Á., Elliott, M., Carstensen, J., Heiskanen, A.S., van de Bund, W., 2010. Marine management - towards an integrated implementation of the European marine strategy framework and the water framework directives. Mar. Pollut. Bull. 60, 2175–2186. https://doi.org/10.1016/j.marpolbul.2010.09.026.

Borja, A., Elliott, M., Snelgrove, P.V.R., Austen, M.C., Berg, T., Cochrane, S., Carstensen, J., Danovaro, R., Greenstreet, S., Heiskanen, A.S., Lynam, C.P., Mea, M., Newton, A., Patrício, J., Uusitalo, L., Uyarra, M.C., Wilson, C., 2016. Bridging the gap between policy and science in assessing the health status of marine ecosystems. Front. Mar. Sci. 3 https://doi.org/10.3389/fmars.2016.00175.

Braga, E.S., Chiozzini, V.C., Berbel, G.B.B., Maluf, J.C.C., Aguiar, V.M.C., Charo, M., Molina, D., Romero, S.I., Eichler, B.B., 2008. Nutrient distributions over the Southwestern South Atlantic continental shelf from Mar del Plata (Argentina) to Itajaí (Brazil): winter-summer aspects. Continent. Shelf Res. 28, 1649–1661. https://doi.org/10.1016/j.csr.2007.06.018.

Brauko, K, Cabral, A, Costa, N.V., Hayden, J, Dias, C.E., Leite, E.S., Westphal, R.D, Mueller, C.M., Hall-Spencer, J.M., Rodrigues, R.R., Rörig, L.R., Pagliosa, P.R., Fonseca, A.L., Fonseca, O.E., Horta, P.A., et al., 2020b. Marine Heatwaves, Sewage and Eutrophication Combine to Trigger Deoxygenation and Biodiversity Loss: A SW Atlantic Case Study. Front. Mar. Sci. https://doi.org/10.3389/fmars.2020.590258. In press.

Brauko, K.M., Cerveira, A.F., Freitas, R.F., Hadlich, H.L., Beras, T., Fonseca, A.L., Pagliosa, P.R., 2020. Comparing structural and functional approaches to distinguish benthic communities from eutrophic and non-eutrophic estuaries. Mar. Pollut. Bull. 157, 111290. https://doi.org/10.1016/j.marpolbul.2020.111290.

Brauko, K.M., Souza, F.M. de, Muniz, P., Camargo, M.G. de, Lana, P. da C., 2015. Spatial variability of three benthic indices for marine quality assessment in a subtropical estuary of Southern Brazil. Mar. Pollut. Bull. 91, 454–460. https://doi.org/10.1016/j.marpolbul.2014.10.025.

Brazilian Law No 9.433, 1997. of 8th January 1997. Available: http://www.planalto.gov.br/ccivil_03/leis/l9433.htm.

Brazilian Law No 12.727, 2012. of 17th October. Available: http://www.planalto.gov.br/ccivil_03/_ato2011-2014/2012/lei/l12727.htm.

Bricker, S.B., Longstaff, B., Dennison, W., Jones, A., Boicourt, K., Wicks, C., Woerner, J., 2008. Effects of nutrient enrichment in the nation's estuaries: a decade of change. Harmful Algae 8, 21–32. https://doi.org/10.1016/j.hal.2008.08.028.

Cabral, A., Bercovich, M.V., Fonseca, A., 2019. Implications of poor-regulated wastewater treatment systems in the water quality and nutrient fluxes of a subtropical coastal lagoon. Reg. Stud. Mar. Sci. 29, 100672. https://doi.org/10.1016/j.rsma.2019.100672.

Cabral, A., Bonetti, C.H.C., Garbossa, L.H.P., Pereira-Filho, J., Besen, K., Fonseca, A.L., 2020. Water masses seasonality and meteorological patterns drive the biogeochemical processes of a subtropical and urbanized watershed-bay-shelf continuum. Sci. Total Environ. 749, 141553. https://doi.org/10.1016/j.scitotenv.2020.141553.

Cappelini, C., Evangelista, M.C., Vieira, C., Carvalho, D., Leite, H., da Luz, V., 2011. Plano de desenvolvimento sustentável do turismo de Florianópolis (Florianópolis).

Carneiro, A.P., Soares, C.H.L., Manso, P.R.J., Pagliosa, P.R., 2020. Impact of marine heat waves and cold spell events on the bivalve Anomalocardia flexuosa: a seasonal comparison. Mar. Environ. Res. 156, 104898. https://doi.org/10.1016/j.marenvres.2020.104898.

Cloern, J.E., 2001. Our evolving conceptual model of the coastal eutrophication problem. Mar. Ecol. Prog. Ser. 210, 223–253.

Conama No 357, 2005. of 17th March 2005. Available: http://www2.mma.gov.br/port/conama/legiabre.cfm?codlegi=459.

Crouzet, P., Nixon, S., Laffon, L., Bøgestrand, J., Kristensen, P., Lallana, C., Izzo, G., Bokn, T., Bak, J., Lack, T.J., 1999. Nutrients in European Ecosystems, first ed. European Environment Agency, Copenhagen, Denmark.

de Andrés, M., Barragán, J.M., Scherer, M., 2018. Urban centres and coastal zone definition: which area should we manage? Land Use Pol. 71, 121–128. https://doi.org/10.1016/j.landusepol.2017.11.038.

De Lacerda, L.D., Vaisman, A.G., Maia, L.P., Ramos E Silva, C.A., Soares Cunha, E.M., 2006. Relative importance of nitrogen and phosphorus emissions from shrimp farming and other anthropogenic sources for six estuaries along the NE Brazilian coast. Aquaculture 253, 433–446. https://doi.org/10.1016/j.aquaculture.2005.09.005.

De Lima, A.D.S., Figueiroa, A.C., Gandra, T.B.R., Perez, B.H.M., Dos Santos, B.A.Q., Scherer, M.E.G., 2018. Ecosystem-based information as a support tool to the integrated coastal management of the Santa Catarina Island, Brazil. Desenvolv. Meio Ambiente 44, 20–35. https://doi.org/10.5380/dma.v44i0.54947.

de Melo, A.T., Soriano-Sierra, E.J., Paulino, L.A., 2011. Desenvolvimento espaço-temporal da cobertura vegetal de manguezal em área de aterro hidráulico: mapeamento e interpretação de aerofotografias digitais, e análise dos dados quantitativos. Biotemas 24, 59–73. https://doi.org/10.5007/2175-7925.2011v24n4p59.

de Paula Filho, F.J., Marins, R.V., de Lacerda, L.D., 2015. Natural and anthropogenic emissions of N and P to the Parnaíba river Delta in NE Brazil. Estuar. Coast Shelf Sci. 166, 34–44. https://doi.org/10.1016/j.ecss.2015.03.020.

de Souza, R.V., Campos, C.J.A., Garbossa, L.H.P., Seiffert, W.Q., 2018. Developing, cross-validating and applying regression models to predict the concentrations of faecal indicator organisms in coastal waters under different environmental scenarios. Sci. Total Environ. 630, 20–31. https://doi.org/10.1016/j.scitotenv.2018.02.139.

de Souza, R.V., Garbossa, L.H.P., Campos, C.J.A., Vianna, L.F.d.N., Vanz, A., Rupp, G.S., 2016. Metals and pesticides in commercial bivalve mollusc production areas in the North and South Bays, Santa Catarina (Brazil). Mar. Pollut. Bull. 105, 377–384. https://doi.org/10.1016/j.marpolbul.2016.02.024.

Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. Science 321, 926–929. https://doi.org/10.1126/science.1156401, 80.

Dolbeth, M., Stålnacke, P., Alves, F.L., Sousa, L.P., Gooch, G.D., Khokhlov, V., Tuchkovenko, Y., Lloret, J., Bielecka, M., Rózyński, G., Soares, J.A., Baggett, S., Margonski, P., Chubarenko, B.V., Lillebø, A.I., 2016. An integrated Pan-European perspective on coastal Lagoons management through a mosaic-DPSIR approach. Sci. Rep. 6, 1–12. https://doi.org/10.1038/srep19400.

Duarte, C.M., Krause-Jensen, D., 2018. Intervention options to accelerate ecosystem recovery from coastal eutrophication. Front. Mar. Sci. 5, 1–8. https://doi.org/10.3389/fmars.2018.00470.

Dupra, V.C., Smith, S.V., Crossland, J.I.M., Crossland, C.J., 2000. Estuarine Systems of the South American Region : Carbon , Nitrogen and Phosphorus Fluxes. Texel, The Netherlands. https://doi.org/10.13140/RG.2.1.2349.2322.

Eberhardt, D.S., Schiocchet, M.A., 2015. Recomendações para a produção de arroz irrigado em Santa Catarina (Sistema pré-germinado). EPAGRI, p. 92. Available: https://publicacoes.epagri.sc.gov.br/SP/article/download/433/327.

Elliott, M., Burdon, D., Atkins, J.P., Borja, A., Cormier, R., de Jonge, V.N., Turner, R.K., 2017. "And DPSIR begat DAPSI(W)R(M)!" - a unifying framework for marine environmental management. Mar. Pollut. Bull. 118, 27–40. https://doi.org/10.1016/j.marpolbul.2017.03.049.

Elliott, M., De Jonge, V.N., 2002. The management of nutrients and potential eutrophication in estuaries and other restricted water bodies. Hydrobiologia 475–476, 513–524. https://doi.org/10.1023/A:1020372316420.

Ferreira, J.G., Andersen, J.H., Borja, A., Bricker, S.B., Camp, J., Cardoso da Silva, M., Garcés, E., Heiskanen, A.S., Humborg, C., Ignatiades, L., Lancelot, C., Menesguen, A., Tett, P., Hoepffner, N., Claussen, U., 2011. Overview of eutrophication indicators to assess environmental status within the European Marine Strategy Framework Directive. Estuar. Coast Shelf Sci. 93, 117–131. https://doi.org/10.1016/j.ecss.2011.03.014.

FIESC, 2015. Santa Catarina Em Dados. Available: https://fiesc.com.br/sites/default/files/inline-files/sc_em_dados_site_2015.pdf.

Gallardi, D., 2014. Effects of bivalve aquaculture on the environment and their possible mitigation: a review. Fish. Aquacult. J. https://doi.org/10.4172/2150-3508.1000105, 05.

Garbossa, L.H.P., Souza, R.V., Campos, C.J.A., Vanz, A., Vianna, L.F.N., Rupp, G.S., 2017. Thermotolerant coliform loadings to coastal areas of Santa Catarina (Brazil) evidence the effect of growing urbanisation and insufficient provision of sewerage infrastructure. Environ. Monit. Assess. 189 https://doi.org/10.1007/s10661-016-5742-0.

Gari, S.R., Newton, A., Icely, J.D., 2015. A review of the application and evolution of the DPSIR framework with an emphasis on coastal social-ecological systems. Ocean Coast Manag. 103, 63–77. https://doi.org/10.1016/j.ocecoaman.2014.11.013.

Gouvêa, L.P., Schubert, N., Martins, C.D.L., Sissini, M., Ramlov, F., Rodrigues, E.R. de O., Bastos, E.O., Freire, V.C., Maraschin, M., Carlos Simonassi, J., Varela, D.A., Franco, D., Cassano, V., Fonseca, A.L., Barufi J, B., Horta, P.A., 2017. Interactive effects of marine heatwaves and eutrophication on the ecophysiology of a widespread and ecologically important macroalga. Limnol. Oceanogr. 62, 2056–2075. https://doi.org/10.1002/lno.10551.

Grasel, D., Fearnside, P.M., Rovai, A.S., Vitule, J.R.S., Rodrigues, R.R., Mormul, R.P., Sampaio, F.D.F., Jarenkow, J.A., 2019. Brazil's Native Vegetation Protection Law Jeopardizes Wetland Conservation: a Comment on Maltchik et al. Environ. Conserv. 46, 121–123. https://doi.org/10.1017/S0376892918000474.

Green, P.A., Vörösmarty, C.J., Meybeck, M., Galloway, J.N., Peterson, B.J., Boyer, E.W., 2004. Pre-industrial and contemporary fluxes of nitrogen through rivers: a global assessment based on typology. Biogeochemistry 68, 71–105. https://doi.org/10.1023/B:BIOG.0000025742.82155.92.

Griggs, D., 2013. Sustainable development goals for people and planet. Nature 495, 305–307.

Guião, V., Scherer, M.E.G., 2018. Analysis of the adequacy of the santa catarina coastal state plan of management to the relevant topics for integrated coastal management. Desenvolv. Meio Ambiente 44, 422–430. https://doi.org/10.5380/dma.v44i0.54967.

Haines-Young, R., Potschin-Young, M., Czúcz, B., 2018. Report on the Use of CICES to Identify and Characterise the Biophysical, Social and Monetary Dimensions of ES Assessments, p. 106. Deliv. D4.2, EU Horiz. 2020 Esmeralda Proj. Grant Agreem. No. 642007. 106.

Hattenrath-Lehmann, T.K., Marcoval, M.A., Mittlesdorf, H., Goleski, J.A., Wang, Z., Haynes, B., Morton, S.L., Gobler, C.J., 2015. Nitrogenous nutrients promote the growth and toxicity of dinophysis acuminata during estuarine bloom events. PloS One 10, 1–22. https://doi.org/10.1371/journal.pone.0124148.

Hayashi, L., Reis, R.P., 2012. Cultivation of the red algae Kappaphycus alvarezii in Brazil and its pharmacological potential. Brazil. J. Pharmacogn. 22, 748–752. https://doi.org/10.1590/S0102-695X2012005000055.

Hayashi, L., Santos, A.A., Faria, G.S.M., Nunes, B.G., Souza, M.S., Fonseca, A.L.D., Barreto, P.L.M., Oliveira, E.C., Bouzon, Z.L., 2011. Kappaphycus alvarezii (Rhodophyta, Areschougiaceae) cultivated in subtropical waters in Southern Brazil. J. Appl. Phycol. 23, 337–343. https://doi.org/10.1007/s10811-010-9543-5.

Hoinaski, L., Franco, D., Haas, R., Martins, R.F., Lisboa, H.D.M., 2014. Investigation of rainwater contamination sources in the southern part of Brazil. Environ. Technol. 35, 868–881. https://doi.org/10.1080/09593330.2013.854412.

IBGE, 2011. Atlas geográfico das zonas costeiras e oceânicas do Brasil. Rio de Janeiro. Available: https://biblioteca.ibge.gov.br/visualizacao/livros/liv55263.pdf.

IBGE, 2017. Censo Agropecuário: Resultados Definitivos 2017. Brasília. Available: https://biblioteca.ibge.gov.br/index.php/biblioteca-catalogo?view=detalhes&id=73096.

INMET, Instituto Nacional de Meteorologia. Available: https://mapas.inmet.gov.br/.

Kroeze, C., Bouwman, L., Seitzinger, S., 2012. Modeling global nutrient export from watersheds. Curr. Opin. Environ. Sustain. 4, 195–202. https://doi.org/10.1016/j.cosust.2012.01.009.

Lacerda, L.D., Molisani, M.M., Sena, D., Maia, L.P., 2008. Estimating the importance of natural and anthropogenic sources on N and P emission to estuaries along the Ceará State Coast NE Brazil. Environ. Monit. Assess. 141, 149–164. https://doi.org/10.1007/s10661-007-9884-y.

Lamego, F., da Silva, A.L.G., Simonassi, J.C., Nepomuceno, A., 2017. Reconstructing recent land–ocean changes in the Brazilian Southern Coast using sedimentary proxies and tracking airborne 210Pb. J. Radioanal. Nucl. Chem. 314, 2281–2299. https://doi.org/10.1007/s10967-017-5623-1.

Luk, S.Y., Hoagland, P., Rheuban, J.E., Costa, J.E., Doney, S.C., 2019. Modeling the effect of water quality on the recreational shellfishing cultural ecosystem service of Buzzards Bay, Massachusetts. Mar. Pollut. Bull. 140, 364–373. https://doi.org/10.1016/j.marpolbul.2018.12.047.

Macedo, H.S., Medeiros, R.P., 2018. Rethinking governance in a Brazilian multiple-use marine protected area. Mar. Pol. https://doi.org/10.1016/j.marpol.2018.08.019, 0–1.

Malone, T.C., Newton, A., 2020. The globalization of cultural eutrophication in the coastal ocean: causes and consequences. Front. Mar. Sci. 7, 1–30. https://doi.org/10.3389/fmars.2020.00670.

Martins, C.D.L., Arantes, N., Faveri, C., Batista, M.B., Oliveira, E.C., Pagliosa, P.R., Fonseca, A.L., Nunes, J.M.C., Chow, F., Pereira, S.B., Horta, P.A., 2012. The impact of coastal urbanization on the structure of phytobenthic communities in southern Brazil. Mar. Pollut. Bull. 64, 772–778. https://doi.org/10.1016/j.marpolbul.2012.01.031.

Martins, I.M., Medeiros, R.P., Di Domenico, M., Hanazaki, N., 2018. What Fishers' local ecological knowledge can reveal about the changes in exploited fish catches. Fish. Res. 198, 109–116. https://doi.org/10.1016/j.fishres.2017.10.008.

Martins, I.M., Medeiros, R.P., Hanazaki, N., 2014. From fish to ecosystems: the perceptions of fishermen neighboring a southern Brazilian marine protected area. Ocean Coast Manag. 91, 50–57. https://doi.org/10.1016/j.ocecoaman.2014.01.015.

Meybeck, M., Dürr, H.H., Vörösmarty, C.J., 2006. Global coastal segmentation and its river catchment contributors: a new look at land-ocean linkage. Global Biogeochem. Cycles 20, 1–15. https://doi.org/10.1029/2005GB002540.

Ministerio do Meio Ambiente (Mma), 2002. Biodiversidade Brasileira: avaliação e identificação de áreas e ações prioritárias para a conservação, utilização sustentável e repartição de benefícios da biodiversidade brasileira. Available: https://www.mma.gov.br/estruturas/chm/_arquivos/biodiversidade31.pdf.

Mizerkowski, B.D., Hesse, K.J., Ladwig, N., Da Costa MacHado, E., Rosa, R., Araujo, T., Koch, D., 2012. Sources, loads and dispersion of dissolved inorganic nutrients in paranaguá bay. Ocean Dynam. 62, 1409–1424. https://doi.org/10.1007/s10236-012-0569-x.

Molisani, M.M., De Assis Esteves, F., De Lacerda, L.D., De Rezende, C.E., 2013. Emissões naturais e antrópicas de nitrogênio, fósforo e metais para a bacia do rio Macaé (Macaé, RJ, Brasil) sob influência das atividades de exploração de petroleo e gás na Bacia de Campos. Quim. Nova 36, 27–33. https://doi.org/10.1590/S0100-40422013000100006.

Netto, S.A., Pagliosa, P.R., Colling, A., Fonseca, A.L., Brauko, K.M., 2018. Brazilian Estuaries. https://doi.org/10.1007/978-3-319-77779-5.

Newton, A., Elliott, M., 2016. A typology of stakeholders and guidelines for engagement in transdisciplinary, participatory processes. Front. Mar. Sci. 3, 1–13. https://doi.org/10.3389/fmars.2016.00230.

Newton, A., Harff, J., You, Z.J., Zhang, H., Wolanski, E., 2016. Sustainability of future coasts and estuaries: a synthesis. Estuar. Coast Shelf Sci. 183, 271–274. https://doi.org/10.1016/j.ecss.2016.11.017.

Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A.C., Colijn, F., Riva, S.D., Gertz, F., Hansen, J.W., Holmer, M., Ivanova, K., Leppäkoski, E., Canu, D.M., Mocenni, C., Mudge, S., Murray, N., Pejrup, M., Razinkovas, A., Reizopoulou, S., Pérez-Ruzafa, A., Schernewski, G., Schubert, H., Carr, L., Solidoro, C., Viaroli, P., Zaldívar, J.M., 2014. An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. Estuar. Coast Shelf Sci. 140, 95–122. https://doi.org/10.1016/j.ecss.2013.05.023.

Newton, A., Icely, J.D., Falcao, M., Nobre, A., Nunes, J.P., Ferreira, J.G., Vale, C., 2003. Evaluation of eutrophication in the Ria Formosa coastal lagoon, Portugal. Continent. Shelf Res. 23, 1945–1961. https://doi.org/10.1016/j.csr.2003.06.008.

Newton, A., Weichselgartner, J., 2014. Hotspots of coastal vulnerability: a DPSIR analysis to find societal pathways and responses. Estuar. Coast Shelf Sci. 140, 123–133. https://doi.org/10.1016/j.ecss.2013.10.010.

Noriega, C., Araujo, M., 2009. Nitrogen and phosphorus loading in coastal watersheds in northeastern Brazil. J. Coast Res. 871–875.

Norkko, J., Gammal, J., Hewitt, J.E., Josefson, A.B., Carstensen, J., Norkko, A., 2015. Seafloor ecosystem function relationships: in situ patterns of change across gradients of increasing hypoxic stress. Ecosystems 18, 1424–1439. https://doi.org/10.1007/s10021-015-9909-2.

Onorevole, K.M., Thompson, S.P., Piehler, M.F., 2018. Living shorelines enhance nitrogen removal capacity over time. Ecol. Eng. 120, 238–248. https://doi.org/10.1016/j.ecoleng.2018.05.017.

Pagliosa, P.R., Barbosa, F.A.R., 2006. Assessing the environment-benthic fauna coupling in protected and urban areas of southern Brazil. Biol. Conserv. 129, 408–417. https://doi.org/10.1016/j.biocon.2005.11.008.

Pagliosa, P.R., Fonseca, A., Bosquilha, G.E., Braga, E.S., Barbosa, F.A.R., 2005. Phosphorus dynamics in water and sediments in urbanized and non-urbanized rivers in Southern Brazil. Mar. Pollut. Bull. 50, 965–974. https://doi.org/10.1016/j.marpolbul.2005.04.005.

Pagliosa, P.R., Rovai, A.S., Fonseca, A.L., 2012. Carbon mismanagement in Brazil. Nat. Clim. Change 2, 764. https://doi.org/10.1038/nclimate1718.

Pereira Filho, J., Rörig, L.R., 2016. Balanço de massa para nutrientes inorgânicos (N e P) no Estuário do Rio Itajaí-Açu, SC. Rev. Bras. Recur. Hidricos 21, 168–187. https://doi.org/10.21168/rbrh.v21n1.p168-187.

Proença, L.A.O., Schramm, M.A., Alves, T.P., Piola, A.R., 2017. The extraordinary 2016 autumn DSP outbreak in Santa Catarina, Southern Brazil, explained by large-scale oceanographic processes. In: Proc. 17th Int. Conf. Harmful Algae. Int. Soc. Study Harmful Algae 2017, pp. 42–45.

Prudencio, J.M., Vieira, P.F., Fonseca, A.L. de O., 2014. Etnoconservação de recursos naturais na zona costeira catarinense: uma análise das transformações da paisagem na bacia do Rio da Madre, à luz do enfoque de ecodesenvolvimento. Desenvolv. Meio Ambiente 32. https://doi.org/10.5380/dma.v32i0.35553.

R Core Team, 2015. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org.

Rigotto, C., Victoria, M., Moresco, V., Kolesnikovas, C.K., Corrêa, A., Souza, D.S.M., Miagostovich, M.P., Simões, C.M.O., Barardi, C.R.M., 2010. Assessment of adenovirus, hepatitis A virus and rotavirus presence in environmental samples in Florianopolis, South Brazil. J. Appl. Microbiol. 109, 1979–1987. https://doi.org/10.1111/j.1365-2672.2010.04827.x.

Rodriguez-Dominguez, M.A., Konnerup, D., Brix, H., Arias, C.A., 2020. Constructed wetlands in Latin America and the Caribbean: a review of experiences during the last decade. Water (Switzerland) 12. https://doi.org/10.3390/w12061744.

Sabatini, M.E., Brandini, F.P., Calliari, D.L., 2018. Plankton Ecology of the Southwestern Atlantic. https://doi.org/10.1007/978-3-319-77869-3. Plankt. Ecol. Southwest. Atl.

Sáenz, L.A., Seibert, E.L., Zanette, J., Fiedler, H.D., Curtius, A.J., Ferreira, J.F., Alves de Almeida, E., Marques, M.R.F., Bainy, A.C.D., 2010. Biochemical biomarkers and metals in Perna perna mussels from mariculture zones of Santa Catarina, Brazil. Ecotoxicol. Environ. Saf. 73, 796–804. https://doi.org/10.1016/j.ecoenv.2010.02.015.

Saurel, C., Ferreira, J.G., Cheney, D., Suhrbier, A., Dewey, B., Davis, J., Cordell, J., 2014. Ecosystem goods and services from Manila clam culture in Puget Sound: a modelling analysis. Aquacult. Environ. Interact. 5, 255–270. https://doi.org/10.3354/aei00109.

Schaeffer-Novelli, Y., Soriano-Sierra, E.J., Do Vale, C.C., Bernini, E., Rovai, A.S., Pinheiro, M.A.A., Schmidt, A.J., De Almeida, R., Coelho Júnior, C., Menghini, R.P., Martinez, D., Abuchahla, G.M. de O., Cunha-Lignon, M., Charlier-Sarubo, S., Shirazawa-Freitas, J., Cintrón-Molero, G., 2016. Climate changes in mangrove forests and salt marshes. Braz. J. Oceanogr. 64, 37–52. https://doi.org/10.1590/S1679-875920160919064sp2.

Scherer, M.E.G., Asmus, M.L., 2016. Ecosystem-based knowledge and management as a tool for integrated coastal and ocean management: a Brazilian initiative. J. Coast Res. 75, 690–694. https://doi.org/10.2112/si75-138.1.

Schlemper, S.R. de M., 2002. As Cores das Marés: A construção cultural do conhecimento sobre as marés vermelhas. UFSC, Florianópolis.

Schwarz, G.E., Hoos, A.B., Alexander, R.B., Smith, R.A., 2006. The SPARROW surface water-quality model: theory, applications and user documentation. U.S. Geol. Surv. Tech. Methods 6-B3, 1–29.

Seitzinger, S.P., Bouwman, A.F., Kroeze, C., 2010a. Preface to special section on past and future trends in nutrient export from global watersheds and impacts on water quality and eutrophication. Global Biogeochem. Cycles 24, 1–2. https://doi.org/10.1029/2010GB003851.

Seitzinger, S.P., Kroeze, C., Bouwman, A.F., Caraco, N., Dentener, F., Styles, R.V., 2002. Global patterns of dissolved inorganic and particulate nitrogen inputs to coastal systems: recent conditions and future projections. Estuaries 25, 640–655. https://doi.org/10.1007/BF02804897.

Seitzinger, S.P., Mayorga, E., Bouwman, A.F., Kroeze, C., Beusen, A.H.W., Billen, G., Van Drecht, G., Dumont, E., Fekete, B.M., Garnier, J., Harrison, J.A., 2010b. Global river nutrient export: a scenario analysis of past and future trends. Global Biogeochem. Cycles 24. https://doi.org/10.1029/2009GB003587.

Selman, M., Greenhalgh, S., 2009. Eutrophication: Policies, Actions, and Strategies to Address Nutrient Pollution. WRI Policy note - Water Qual. Eutrophication hypoxia 16.

Silva, A.R. da, Fonseca, A.L.D.O., Monteiro, J.P., Santos, L.C.A. dos, 2018. A gestão e monitoramento das águas: uma abordagem das legislações em Portugal e no Brasil. Rev. Bras. Geogr. Física 11, 1512–1525.

Siebert, C., 2010. Institutional arrangements and trajectory of planning in Santa Catarina. Revista Paranaense de Desenvolvimento 119, 137–167.

Silva, A.R. da, Fonseca, A.L.D.O., 2016. Eutrofização dos recursos hídricos como ferramenta para a compreensão das doenças de vinculação hídrica. Geosul 31, 247. https://doi.org/10.5007/2177-5230.2016v31n62p247.

Silva, A.R. da, Fonseca, A.L.D.O., Rodrigues, C.J., Beltrame, Â. da V., 2016. Application of ecological indicators in coastal watershed under high pressure during summer period. Rbrh 21, 537–548. https://doi.org/10.1590/2318-0331.011615106.

Silva, E., Garbossa, L.H.P., Nuñer, A.P.O., Lapa, K.R., 2019. Hydrodynamic modelling of the dispersion and deposition of biodeposits from marine bivalve mollusc farming under neap and spring tides in Santa Catarina Island Bays. Aquaculture 501, 507–514. https://doi.org/10.1016/j.aquaculture.2018.11.001.

Sinha, E., Michalak, A.M., Balaji, V., 2017. Eutrophication will increase during the 21st century as a result of precipitation changes. Science 357, 1–5. https://doi.org/10.1126/science.aan2409, 80.

Smith, S., Swaney, D.P., Talaue-McManus, L., Bartley, J.D., Sandhei, P.T., McLaughlin, C. J., Dupra, V., Crossland, C.J., Buddemeier, R.W., Maxwell, B., Wulff, F., 2003. Humans, hydrology, and the distribution of inorganic nutrient loading to the ocean. Bioscience 53, 235. https://doi.org/10.1641/0006-3568(2003)053[0235:hhatdo]2.0.co;2.

Smith, S.K., Tayman, J., Swanson, D.A., 2013. A Practitioner's Guide to State and Local Population Projections. Springer Press, Dordrecht, Heidelberg, New York, London.

Smith, V.H., Schindler, D.W., 2009. Eutrophication science: where do we go from here? Trends Ecol. Evol. 24, 201–207. https://doi.org/10.1016/j.tree.2008.11.009.

Souza, E.M. de, 2011. Biomassa e estrutura da comunidade fitoplanctônica dos ecossistemas do Banco de Ambrolhos, adjacências e no Antlântico Sul (Brasil X África) 1–160.

State Law Sc No 5010, 2006. of 16th November 2005. Available: https://leisestaduais.com.br/sc/lei-ordinaria-n-13553-2005-santa-catarina-institui-o-plano-estadual-de-gerenciamento-costeiro.

State Law Sc No 13553, 2005. of 11th November 2005. Available: http://leis.alesc.sc.gov.br/html/2005/13550_2005_Lei.html.

Suplicy, F.M., 2018. Plano Estratégico para Desenvolvimento Sustentável da Maricultura Catarinense (2018- 2028). Epagri, Florianópolis, p. 93. Available: https://www.agricultura.sc.gov.br/index.php/arquivos/cederural/camara-setorial-da-maricultura/409-plano-estrategico-da-maricultura-catarinense-2018-2028/file.

Suplicy, F.M., Vianna, L.F.d.N., Rupp, G.S., Novaes, A.L.T., Garbossa, L.H.P., de Souza, R. V., Guzenski, J., da Costa, S.W., Silva, F.M., dos Santos, A.A., 2017. Planning and management for sustainable coastal aquaculture development in Santa Catarina State, south Brazil. Rev. Aquacult. 9, 107–124. https://doi.org/10.1111/raq.12107.

Tavares, J.F., Proença, L.A.O., Odebrecht, C., 2009. Assessing the harmful microalgae occurrence and temporal variation in a coastal aquaculture area, southern Brazil. Atlantica 31, 129–144. https://doi.org/10.5088/atl.

Teubner Junior, F.J., Lima, A.T.M., Barroso, G.F., 2018. Emission rates of nitrogen and phosphorus in a tropical coastal river basin: a strategic management approach. Environ. Monit. Assess. 190 https://doi.org/10.1007/s10661-018-7101-9.

Tibiriçá, C.E.J. de A., Fernandes, L.F., Mafra Junior, L.L., 2015. Seasonal and spatial patterns of toxigenic species of dinophysis and pseudo-nitzschia in a subtropical brazilian estuary. Braz. J. Oceanogr. 63, 17–32. https://doi.org/10.1590/S1679-87592015071906301.

Turra, A., Cróquer, A., Carranza, A., Mansilla, A., Areces, A.J., Werlinger, C., Martínez-Bayón, C., Nassar, C.A.G., Plastino, E., Schwindt, E., Scarabino, F., Chow, F., Figueroa, F.L., Berchez, F., Hall-Spencer, J.M., Soto, L.A., Buckeridge, M.S., Copertino, M.S., de Széchy, M.T.M., Ghilardi-Lopes, N.P., Horta, P., Coutinho, R., Fraschetti, S., Leão, Z.M. de A.N., 2013. Global environmental changes: setting priorities for Latin American coastal habitats. Global Change Biol. 19, 1965–1969. https://doi.org/10.1111/gcb.12186.

Van Der Struijk, L.F., Kroeze, C., 2010. Future trends in nutrient export to the coastal waters of South America: implications for occurrence of eutrophication. Global Biogeochem. Cycles 24, 1–14. https://doi.org/10.1029/2009GB003572.

Vianna, L.F. de N., Filho, J.B., 2018. Spatial analysis for site selection in marine aquaculture: an ecosystem approach applied to Baía Sul, Santa Catarina, Brazil. Aquaculture 489, 162–174. https://doi.org/10.1016/j.aquaculture.2017.12.039.

von Sperling, M., 2007. Wastewater Characteristics, Treatment and Disposal, Choice Reviews Online. https://doi.org/10.5860/choice.45-2633.

Weis, W.A., Soares, C.H.L., de Quadros, D.P.C., Scheneider, M., Pagliosa, P.R., 2017. Urbanization effects on different biological organization levels of an estuarine polychaete tolerant to pollution. Ecol. Indicat. 73, 698–707. https://doi.org/10.1016/j.ecolind.2016.10.029.

Wu, K., Dai, M., Li, X., Meng, F., Chen, J., Lin, J., 2017. Dynamics and production of dissolved organic carbon in a large continental shelf system under the influence of both river plume and coastal upwelling. Limnol. Oceanogr. 62, 973–988. https://doi.org/10.1002/lno.10479.