Geophysical Monograph Series

Geophysical Monograph 256

Coastal Ecosystems in Transition

A Comparative Analysis of the Northern Adriatic and Chesapeake Bay

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This Work is a co-publication of the American Geophysical Union and John Wiley and Sons, Inc.

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WILEY

This edition first published 2021

Published under the aegis of the AGU Publications Committee

Registered Office
John Wiley & Sons, Inc., 111 River Street, Hoboken, NJ 07030, USA

Editorial Office
111 River Street, Hoboken, NJ 07030, USA

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Library of Congress Cataloging-in-Publication Data

Names: Malone, Thomas C., editor. | Malej, Alenka, editor. | Faganeli, Jadran, editor.
Title: Coastal ecosystems in transition : a comparative analysis of the northern Adriatic and Chesapeake Bay / [edited by] Thomas C. Malone, Alenka Malej, Jadran Faganeli.
Description: First edition. | Hoboken, NJ : American Geophysical Union/Wiley, 2021. | Includes bibliographical references and index.
Identifiers: LCCN 2020022364 (print) | LCCN 2020022365 (ebook) | ISBN 9781119543589 (hardback) | ISBN 9781119543602 (adobe pdf) | ISBN 9781119543565 (epub)
Subjects: LCSH: Coastal ecosystem health–Chesapeake Bay (Md. and Va.) | Coastal ecosystem health–Adriatic Sea. | Marine ecosystem management.
Classification: LCC QH541.5.C65 C565 2020 (print) | LCC QH541.5.C65 (ebook) | DDC 333.95/60975518–dc23
LC record available at https://lccn.loc.gov/2020022364
LC ebook record available at https://lccn.loc.gov/2020022365

Cover Design: Wiley
Cover Image: © Martina Orlando Bonaca

Set in 10/12pt Times New Roman by SPi Global, Pondicherry, India

10 9 8 7 6 5 4 3 2 1

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PREFACE

A series of workshops was hosted in the 1990s by the Marine Biology Station Piran of the National Institute of Biology (Slovenia), the Centre for Marine Research of the Ruđer Bošković Institute Rovinj (Croatia), and the Horn Point Laboratory of the University of Maryland Center for Environmental Science (USA). Their purpose was to advance our understanding of how coastal ecosystems are responding to cultural eutrophication, coastal development, and fishing pressure through a comparative analysis of the Northern Adriatic Sea and Chesapeake Bay, two river-dominated systems with urbanized watersheds that support extensive industrial agriculture.

These workshops led to the 1999 publication of *Ecosystems at the Land–Sea Margin: Watershed to the Coastal Sea* as part of the AGU Estuarine and Coastal Sciences Series. The comparative analysis was undertaken in order to improve our understanding of how coastal ecosystems are responding to the pressures of human expansion. The focus was on impacts of local anthropogenic pressures that are occurring globally (coastal development, habitat loss, nutrient pollution, and fisheries) and was based on research conducted during the 1980s and 1990s.

Revisiting these two ecosystems two decades later provides an opportunity to assess changes in anthropogenic pressures (including climate-driven changes) that have occurred in the past two decades and to inform ecosystem-based approaches to managing multiple anthropogenic pressures on coastal marine ecosystem services. In addition, we hope that this publication will foster international collaboration and information exchange on the ecology and value of coastal ecosystems in the Anthropocene.

The chapters that follow include updates on current anthropogenic pressures with an emphasis on the effects of nutrient enrichment and climate change on the extent and condition of critical coastal habitats, patterns of stratification and circulation, food-web dynamics from phytoplankton to fish, nutrient cycling, water quality, and harmful algal events. A common theme running throughout is the causes and consequences of interannual variability and secular trends in annual cycles and means.

Publication of this book commemorates the 50th anniversary of Slovenia's Marine Biology Station Piran, the only institution for marine research and monitoring of seawater quality in Slovenia. We gratefully acknowledge financial support from the following: Long Term Ecological Research Network in Italy and Slovenia (LTER-Italy, LTER-Slovenia), Slovenian Research Agency, Croatian Ministry of the Science, Environmental Agency of Slovenia, Croatian Meteorological and Hydrological Service, the European Environmental Agency, the District Po River Basin Authority, the Regional Environmental Protection Agencies of Emilia Romagna, European Commission, US Environmental Protection Agency, US Geological Survey, US National Oceanic and Atmospheric Administration, and US National Science Foundation.

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2

Recent Status and Long-Term Trends in Freshwater Discharge and Nutrient Inputs

Qian Zhang[1], Stefano Cozzi[2], Cindy Palinkas[3], and Michele Giani[4]

ABSTRACT

Anthropogenic inputs of nutrients via river runoff are the primary drivers of ecosystem degradation in Chesapeake Bay (CB) and the northern Adriatic Sea (NAS). The annual cycle of river flow is typically unimodal in CB (seasonal peak during spring) and bimodal in the NAS (peaks during April–June and October–December). Dissolved inorganic nitrogen accounts for most of the total nitrogen (TN) in both systems. During 1985–2015, annual loads of TN to CB tended to decrease while total phosphorus (TP) loads tended to increase. In contrast, annual loads of TN to the NAS tended to increase while TP loads tended to decrease. However, these annual input trends were significant only for dissolved inorganic P in the NAS, whereas in the case of N they were masked by interannual changes of the runoff. Climate-driven changes in the water cycle may bring new challenges of controlling nutrient loading in CB, where annual rainfall is expected to increase. In contrast, annual rainfall is projected to decrease in the NAS region, which would aid efforts to control nutrients. An additional challenge unique to CB is the filling up of Conowingo Reservoir on the Susquehanna River, which resulted in increased P and sediment loads due to reduced trapping efficiency.

2.1. INTRODUCTION

Increasing anthropogenic inputs of nitrogen (N), phosphorus (P), and sediments to the coastal ocean via river discharge over the past 100 years are primary drivers of ecosystem degradation in many estuarine and coastal systems worldwide, including Chesapeake Bay (CB) and the northern Adriatic Sea (NAS) (Degobbis, 1989; Giani et al., 2012; Hagy et al., 2004; Kemp et al., 2005; Murphy et al., 2011; Salvetti et al., 2006; Testa et al., 2014; Zhang et al., 2018). The effects of these inputs include the annual recurrence of seasonal hypoxia, declines in water transparency, habitat loss, and loss of biodiversity (Boesch et al., 2001; Breitburg et al., 2018; Cloern, 2001; Degobbis, 1989; Diaz & Rosenberg, 2008; Giani et al., 2012; Kemp et al., 2005; Testa et al., 2019). Consequently, reducing land-based inputs of N, P, and sediments have long been a management priority for both CB and the NAS.

In CB, severe bottom-water hypoxia and loss of submerged aquatic vegetation (SAV) were first evident in the 1950s and 1960s, respectively (Kemp et al., 2005). In subsequent decades, restoration of SAV was a largely uncoordinated voluntary effort. In 1983, the US Environmental Protection Agency (USEPA) signed the first Chesapeake Bay Agreement with four jurisdictions in the bay's watershed, and the Chesapeake Bay Program

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Coastal Ecosystems in Transition: A Comparative Analysis of the Northern Adriatic and Chesapeake Bay, Geophysical Monograph 256, First Edition.
Edited by Thomas C. Malone, Alenka Malej, and Jadran Faganeli.

DOI: 10.1002/9781119543626.ch2

was formed to coordinate and facilitate multijurisdictional efforts to restore CB by reducing nutrient and sediment inputs. Subsequent agreements set goals of reducing nutrient inputs by 40% by 2000 and to improve CB water quality sufficiently to remove it from the "dirty waters list" by 2010 (Boesch et al., 2001). Years later, it was realized that this deadline would not be met. Consequently, the USEPA established the Total Maximum Daily Load for CB (US Environmental Protection Agency, 2010), which mandates state-wide efforts to establish watershed implementation plans to reduce nutrient and sediment runoff (Linker, Batuik, et al., 2013; Shenk & Linker, 2013). In 2014, the Chesapeake Bay Watershed Agreement established goals and outcomes for clean water, sustainable fisheries, vital habitats, toxic contaminants, healthy watersheds, stewardship, land conservation, public access, environmental literacy, and climate resiliency (Chesapeake Bay Program, 2014).

Since the 1970s, seasonal hypoxic and anoxic events in the NAS have been observed along the western coast and in the northernmost Gulf of Trieste, with episodic events occurring offshore (Alvisi & Cozzi, 2016; Djakovac et al., 2012; Stachowitsch, 2014). The quality of marine waters was also degraded by toxic dinoflagellate blooms and massive accumulations of mucilaginous aggregates (Djakovac et al., 2012; Giani et al., 2012). The economic impacts of these events (primarily on tourism) resulted in Italian regulations in 1986 to reduce polyphosphates in detergents and in the establishment of the Po Basin Authority in 1989 to manage nutrients inputs to the Po River, the largest tributary of the NAS (Seagle et al., 1999). In 2000, the Water Framework Directive 2000/60/EC (WFD) of the European Union (EU) established a framework for member states to achieve good ecological and chemical status objectives for inland surface waters, estuaries, and coastal waters within 1 nautical mile from shore through watershed management by 2015 (Teodosiu et al., 2003). In 2013, the Management Plan of the Po River (PdGPo 2010) was approved, which opened a new phase for water management and for the reduction of nutrient loads, through the realization of spill basins for agriculture and manure wastes and the implementation of the wastewater collection and depuration systems (Bortone, 2014). However, 15 years after the directive was agreed to, achieving its objectives remains a challenge, with 47% of EU surface waters not reaching good status in 2015 (Voulvoulis et al., 2017). To achieve the objectives of the WFD, a more integrated understanding of the relationships between land-use practices in coastal watersheds and the status of surface waters is needed.

The main objective of this chapter is to review and compare the current status, seasonality, and long-term trends of freshwater and nutrient inputs to CB and the NAS. We begin with an overview of the two watersheds followed by a comparison of freshwater inputs in terms of their seasonality and long-term trends. We then compare nutrient and sediment loads from the watersheds, elucidate the controls of nutrient and sediment export, highlight some of the major challenges to achieving reductions in land-based inputs, and conclude with recommendations for the management and restoration of CB and the NAS.

2.2. OVERVIEW OF THE WATERSHED AND FRESHWATER INPUTS

Chesapeake Bay is a large estuary in the Mid-Atlantic region of the United States. Among its many tributaries, nine account for over 90% of river flow into CB (Chanat et al., 2016; Moyer et al., 2012). The watershed of the Susquehanna River, the largest river discharging directly into the mainstem bay, comprises about 43% of total CB watershed and is dominated by forested areas (~65%). The NAS is a shallow, semienclosed arm of the NE Mediterranean Sea. The Po River, the largest river discharging into the NAS, has a watershed that comprises 67% of the total NAS watershed and hosts large urban and industrial settlements, as well as extended areas of intensive cropping and livestock activities (Seagle et al., 1999).

River flows into CB, and associated inputs of nutrients and sediments, are monitored by the US Geological Survey (US Geological Survey, 2018). For the NAS, flow rates of Italian rivers are monitored the Hydrographic and Mareographic National Service of Italy (1917–1990s) and the Regional Environmental Protection Agencies (1990s to today). Nutrient data were obtained from the scientific literature, monitoring programs, and past projects (Cozzi & Giani, 2011; Cozzi et al., 2019). Data for the Istrian Rivers were provided by the Environmental Agency of the Republic of Slovenia, the Croatian Meteorological and Hydrological Service, and the European Environmental Agency.

The volume transports (Q) of major rivers flowing into CB and the NAS are similar and exhibited strong interannual variability during 1985–2015 (Figure 2.1). On average, the Susquehanna accounts for 62% of riverine inputs of freshwater while the Po accounts for 69%. The highest transports in CB watershed occurred during the years 1996, 2004, and 2011 in association with major hurricanes and tropical storms (Figure 2.1a). River flows to the NAS were characterized by a maximum during 2014 and lows during the drought period of 2003–2007 (Figure 2.1b).

The volume transports of major rivers flowing into CB and the NAS show strong seasonal variability. Seasonal peaks tended to occur during January–March and April–June in CB rivers (Figure 2.1c). In comparison, seasonal

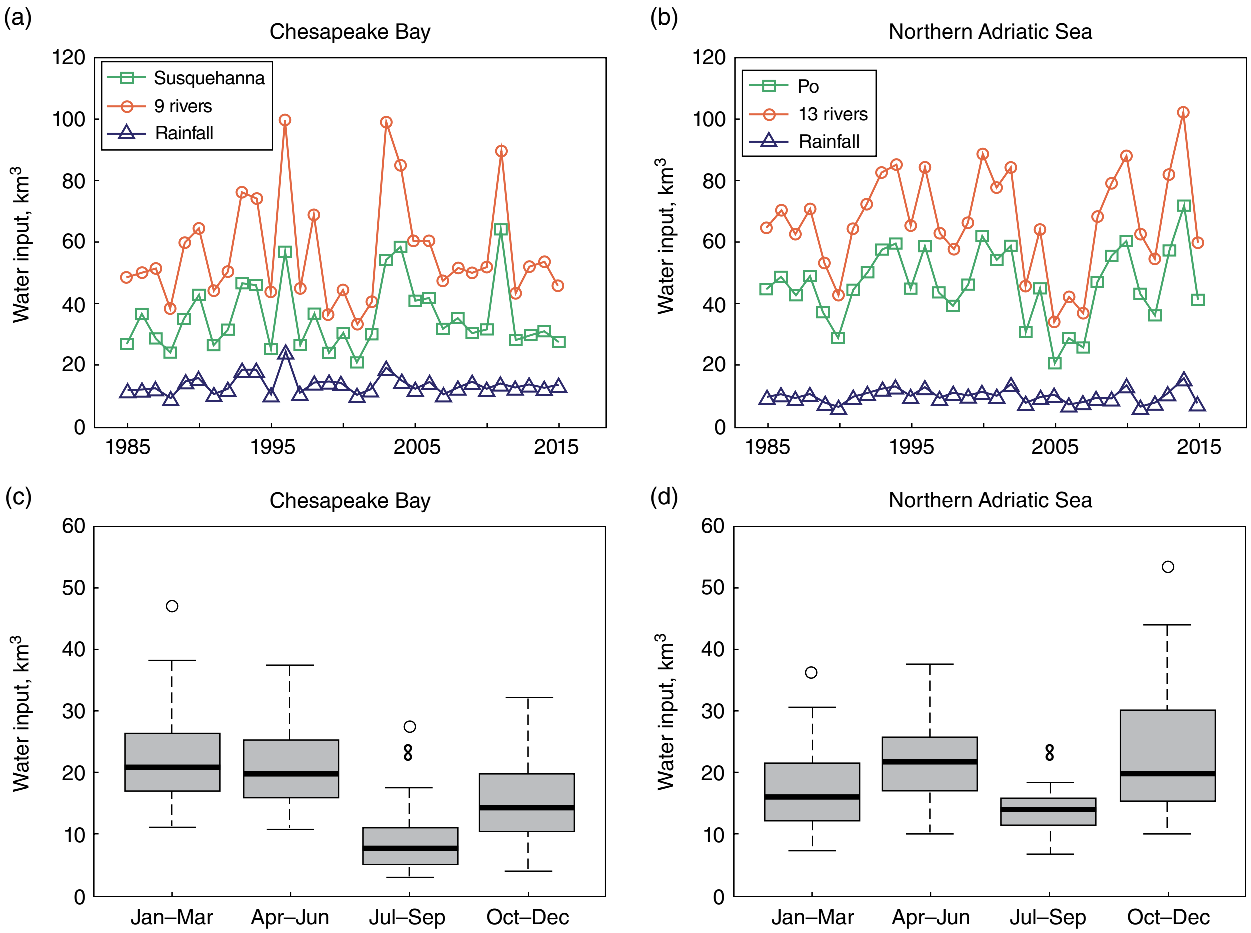

Figure 2.1 Time series of annual freshwater input to (a) Chesapeake Bay and (b) northern Adriatic Sea and boxplots of seasonal freshwater input to (c) Chesapeake Bay and (d) northern Adriatic Sea in the period of 1985–2015, including input from the largest river (Susquehanna and Po, respectively), input from all major tributaries, and direct wet precipitation.

peaks tended to occur during April–June and October–December in the NAS (Figure 2.1d). Direct precipitation to the sea surface was estimated to account for 14–29% of seasonal freshwater input to CB and for 10–23% of the input to the NAS. For both systems, direct precipitation is more important in July–September relative to other months that have higher Q.

Freshwater inputs to CB and the NAS were further compared on a centennial scale using data from the Susquehanna and Po Rivers. During 1900–2015, the Susquehanna annual Q has an estimated Mann–Kendall (MK) trend slope of -0.0063 km^3 year^{-1} (p = 0.79). On a seasonal basis, Susquehanna Q had negative slopes in all four seasons, i.e., -0.010, -0.038, -0.095, and -0.025 km^3 year^{-1} in January–March, April–June, July–September, and October–December, respectively, with the latter three trends being statistically significant. During 1917–2015, the Po annual Q has an estimated trend slope of 0.016 km^3 year^{-1} (p = 0.86). On a seasonal basis, Po Q was estimated to have positive slopes in January–March (0.0050 km^3 year^{-1}) and October–December (0.0006 km^3 year^{-1}) and negative slopes in April–June (-0.012 km^3 year^{-1}) and July–September (-0.015 km^3 year^{-1}), with the July–September trend being statistically significant.

2.3. NUTRIENT INPUTS

2.3.1. Recent Status: 2004–2012

Median annual inputs of nutrients and sediment in 2004–2012 to CB and the NAS are summarized in Table 2.1. NO_x (nitrate + nitrite) accounts for most of the TN load, whereas dissolved inorganic P (DIP) is a minor fraction of the TP load. Among the nine major tributaries, the Susquehanna contributes about 65% of TN and 46% of TP (Zhang et al., 2015). Relative to riverine inputs, inputs from direct precipitation to CB are small (7% of TN and 17% of TP).

Table 2.1 Median annual inputs of freshwater (Q, km^3) and nutrients (10^6 kg $year^{-1}$) to Chesapeake Bay (CB) and the northern Adriatic Sea (NAS) due to river runoff and precipitation (PPT) for the years 2004–2012

	CB			NAS				
Parameter	Rivers	PPT	Total	Western rivers	Northern rivers	Eastern rivers	PPT	Total
Q	54.7	13.8	68.5	52.6	5.9	3.2	9.4	71.1
TN	84.9	5.7	90.6	145.0	16.0	6.3	13.6	180.9
NO_x	56.7	2.5	59.2	101.0	12.0	4.3	5.0	122.3
TP	5.2	0.9	6.1	8.1	0.4	0.1	1.1	9.7
DIP	0.7	0.2	0.9	2.7	0.1	0.04	0.9	3.7

Note: TN, total nitrogen; NO_x = nitrate + nitrite; TP, total phosphorus; DIP, dissolved inorganic phosphorus.

Compared with CB, riverine inputs of nutrients to the NAS are much higher (Table 2.1), with riverine inputs of anthropogenic nutrients to the western NAS (dominated by the Po River) accounting for 80%, 83%, 84%, and 73% of annual inputs of TN, NO_x, TP, and DIP, respectively. This reflects the higher population density in river watersheds of the western shore. Notably, nutrient inputs to the NAS are much lower than those to CB when normalized to the volume of the receiving water bodies (~1812 × 10^3 kg N km^{-3} $year^{-1}$ for CB vs. ~285 × 10^3 kg N km^{-3} $year^{-1}$ for the NAS). In addition, the TN/TP molar ratio for CB is much lower than for the NAS (33:1 vs. 41:1) while the NO_x/DIP molar ratio for CB is much higher than for the NAS (141:1 vs. 72:1).

2.3.2. Seasonality (2004–2012)

For both CB and the NAS, nutrient and sediment loads show strong seasonal variability (Figure 2.2). For CB, the annual input cycles of freshwater, TN, TP, and suspended sediment (SS) are typically unimodal with maxima during March–April and minima during July–August. This pattern has been reported for TN in all the major tributaries to CB (Zhang et al., 2015). For the NAS, the annual input cycles of freshwater, TN, and TP tend to be more bimodal with high inputs during April–June and October–December and low inputs during January–March and July–September.

2.3.3. Long-Term Trends (1985–2015)

Interannual variations in Susquehanna annual loads of TN and NO_x have negative MK slopes while TP and SS have positive slopes (Figure 2.3). Although these slopes are not statistically significant, their directions are consistent with results of flow-normalized loads (Zhang et al., 2015) that account for interannual variability in river flow (Hirsch et al., 2010). Declines in TN and NO_x were partially due to upgraded wastewater treatment (Boynton et al., 2008) and decreases in atmospheric deposition due to the Clean Air Act (Eshleman et al., 2013; Linker, Dennis, et al., 2013). By contrast, TP and SS loads have increased since the late 1990s, likely due to declining trapping efficiency of the Conowingo Dam in the lower Susquehanna (Hirsch, 2012; Langland, 2015; Zhang, Ball, et al., 2016; Zhang et al., 2013).

Interannual variations in the Po River annual load of TN had a positive MK slope while loads of NO_x, TP, DIP, and SS had negative slopes (Figure 2.3). The only statistically significant trend for Po is with DIP, which shows a long-term reduction after the peak in the 1980s. This pattern can be attributed to a reduction of P content in fertilizers and detergents, as well as improved management of wastewaters (Cozzi & Giani, 2011; Viaroli et al., 2018). By contrast, N loads have been driven by interannual oscillations from persistent anthropogenic N emission in the watershed and by interannual changes in river flow, particularly during the extreme drought of 2003–2007 (Cozzi et al., 2019). Current transport of SS by the Po is high compared to river flow, due to large SS contributions by the Apennine tributaries and the absence of dams in the lower river (Tesi et al., 2013). Transport of SS to the NAS is critical for the maintenance of delta and alongshore habitats, as well as for sedimentation processes in the western Adriatic Sea (Frignani et al., 2005). Despite SS transport decreases during the previous century, the present estimates suggest that SS loads have not changed significantly since the 1980s.

2.4. CONTROLS OF NUTRIENT EXPORT

2.4.1. Nutrient Sources

Watershed export of nutrients is complex due to heterogeneities in their sources, fates, and transports. In terms of sources, agriculture nonpoint sources, atmospheric deposition, urban (storm water) sources, as well as point sources (wastewater treatment plants) account for most inputs to CB and the NAS watersheds (Ator et al., 2011; Palmeri et al., 2005; Salvetti et al., 2006; Viaroli et al., 2018; Volf et al., 2013). Globally, there is a significant linear correlation between net anthropogenic N supplies to coastal watersheds and total riverine

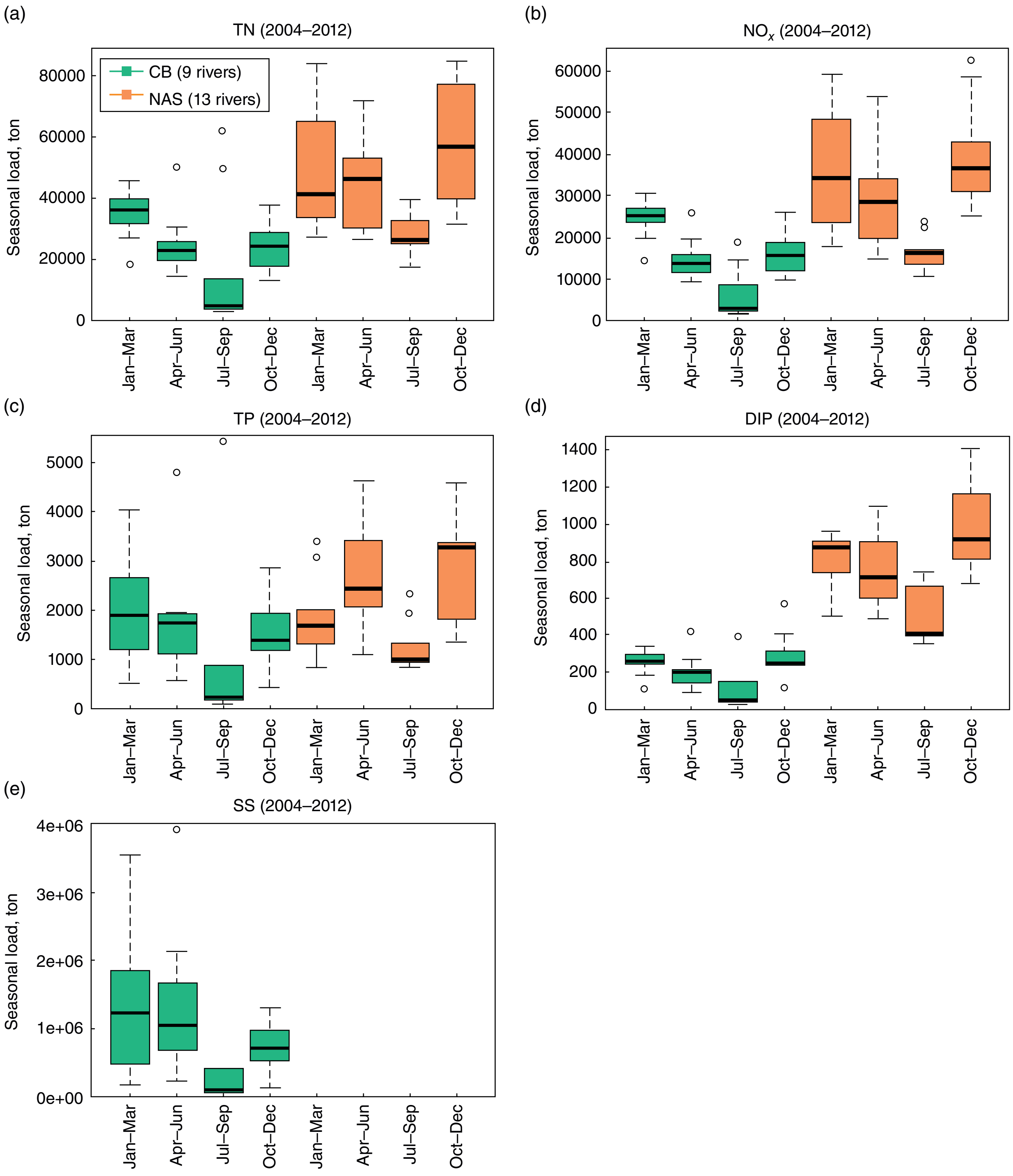

Figure 2.2 Boxplots showing seasonal loads of (a) total nitrogen (TN), (b) nitrate + nitrite (NO_x), (c) total phosphorus (TP), (d) dissolved inorganic phosphorus (DIP), and (e) suspended sediment (SS) to Chesapeake Bay (four boxes on the left) and the northern Adriatic Sea (four boxes on the right) from tributaries with available data (9 and 13 tributaries, respectively) in 2004–2012.

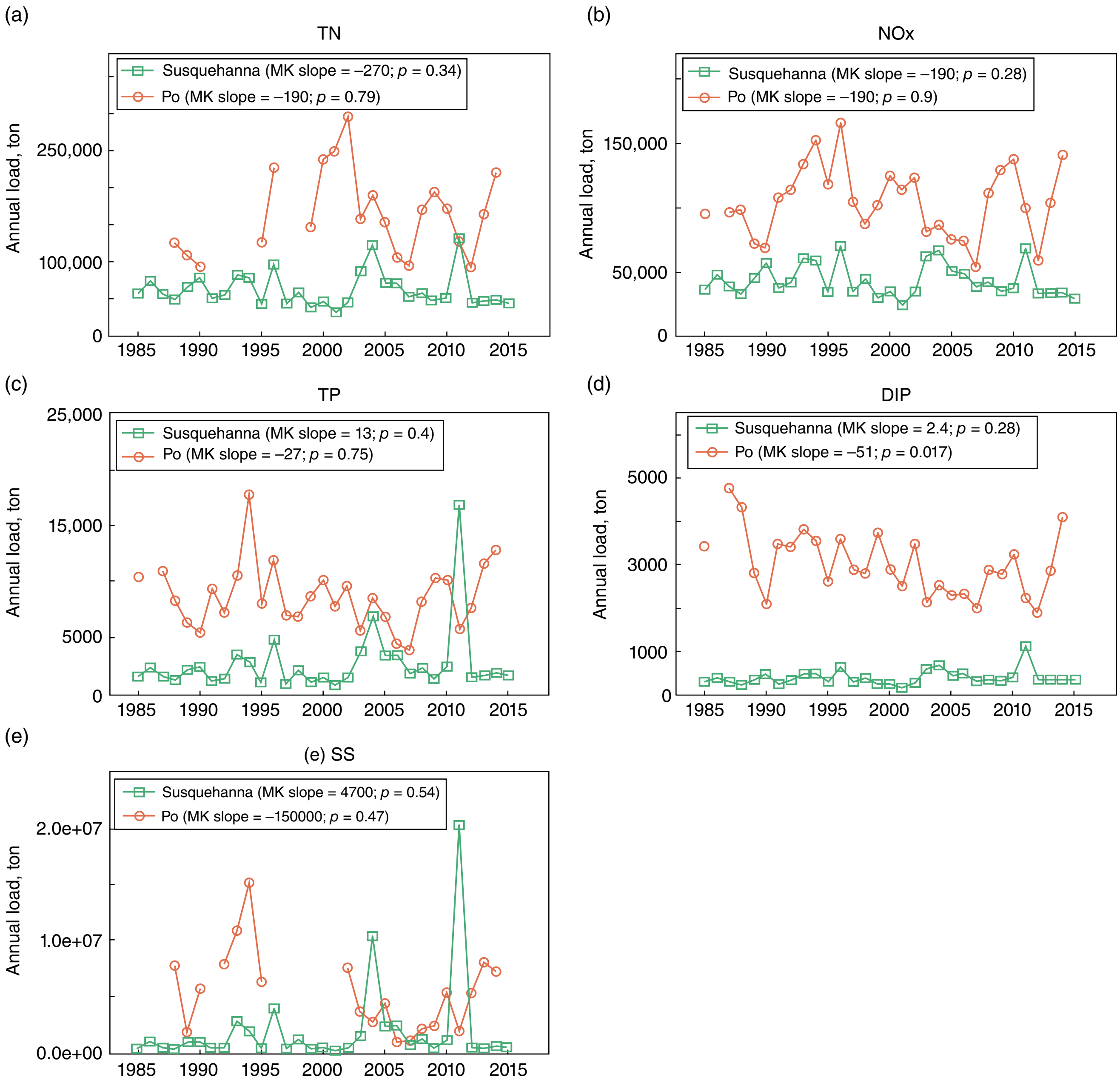

Figure 2.3 Time series of annual loads of (a) total nitrogen (TN), (b) nitrate + nitrite (NO_x), (c) total phosphorus (TP), (d) dissolved inorganic phosphorus (DIP), and (e) suspended sediment (SS) to Chesapeake Bay and the northern Adriatic Sea from their largest tributaries (i.e., Susquehanna and Po, respectively) in 1985–2015. Mann-Kendall (MK) trend slope and significance (*p*) values are shown in legend.

nitrogen export to the coastal ocean (Boyer & Howarth, 2008). However, it should be noted that the major sources and their relative contributions can vary significantly both spatially (as a function of watershed characteristics such as land use, climate, and geology) and temporally (as a function of watershed management, urbanization) (Ator et al., 2011; Carpenter et al., 1998).

For the CB watershed, riverine export of TN was dominated by agriculture nonpoint sources (fertilizer and manure 54%), followed by contributions from atmospheric deposition (17%), point sources (16%), and urban sources (12%). Riverine export of TP was dominated by agriculture nonpoint sources (43%), followed by point sources (32%) and urban sources (11%) (Ator et al., 2011). Typically, nutrients accumulate in watersheds during dry periods of low flows and are transported to receiving waters via groundwater discharge and surface water runoff during wet periods and storm events (Shields et al., 2008; Tesi et al., 2013). Groundwater can represent a major fraction of riverine load (especially N). Bachman et al. (1998) estimated that base flow (a proxy of groundwater input) accounted for 17–80% (median 48%) of the TN load at 36 CB monitoring sites. In addition, riverine export can be strongly modulated by reservoirs. For example, Conowingo Reservoir and two others in the lower Susquehanna River historically trapped about 2%, 45%, and 70% of annual N, P, and SS load, respectively (Langland & Hainly, 1997).

Riverine inputs of TN to the NAS were also dominated by agriculture nonpoint sources (40%) followed by point and urban sources (27%), and groundwater (29%). The TP load mainly originated from point sources and urban sources (43%) and agriculture nonpoint sources (36%) (Volf et al., 2013). For the Po watershed, TN load comes from point sources (40%), nonpoint sources (20%), and groundwater and springs (40%), whereas TP loads come from point sources (80%) and nonpoint sources (20%) (Salvetti et al., 2006).

2.4.2. Controlling Factors

In the CB watershed, several controlling factors have been identified for nutrient export in the Susquehanna River (Zhang, Ball, et al., 2016). First, river flow dominates the interannual variability of constituent export. Second, land-use patterns strongly affect the relative contribution of the subwatersheds. Specifically, long-term median yields of N, P, and SS all correlate positively with the fraction of the watershed that is not forested. Third, riverine loads from different Susquehanna subwatersheds have generally declined due to reductions in source input over the same period. Finally, dams and reservoirs can strongly modulate the storage and release of particulate constituents and the extent of modulation has varied considerably over time as reservoirs fill. Two decades ago, the Conowingo Reservoir and two others in the lower Susquehanna River trapped about 2%, 45%, and 70% of annual N, P, and SS load, respectively (Langland & Hainly, 1997). Currently, the reservoir system is no longer an effective trap of these constituents (Hirsch, 2012; Langland, 2015; Zhang et al., 2013; Zhang, Hirsch, et al., 2016).

In the NAS watershed, the annual export of nutrient and sediment is directly linked to river flow and can be estimated using Po River loads as a reference (Cozzi & Giani, 2011). However, the short-term dynamics of riverine loads are complicated during freshets by the effects of flow on the erosion, groundwater inputs, dilution, and biological processes in the river environment (Marchina et al., 2015; Tesi et al., 2013). The N and P loads in the marine environment have long-term trends consistent with watershed inputs from industrial activities, agriculture, livestock, urban settlements, and atmospheric deposition; dynamics of their delivery are also modulated by the variable retention of these elements in soils and aquifers (Cozzi et al., 2019; Palmeri et al., 2005; Salvetti et al., 2006; Viaroli et al., 2018). In general, retention mechanisms differ considerably between N and P—sedimentation may be the major retention mechanism for particulate P during overland flow conditions (Cirmo & McDonnell, 1997; Hoffmann et al., 2009), whereas sorption/desorption reactions and denitrification are more important for P and N, respectively, during subsurface flow conditions (House, 2003; Withers & Jarvie, 2008).

2.4.3. Watershed Management

For CB, coordinated efforts have been implemented to reduce pollutant inputs since the first Chesapeake Bay Agreement was signed in 1983. Here we provide a brief overview of historical changes in management and practices associated with point, agricultural, and stormwater sources in the Susquehanna River (Zhang et al., 2013). For point sources, the more important historical controls were the P ban in detergents (Litke, 1999) and the adoption of increasingly effective nutrient removal technologies at wastewater treatment plants (Chesapeake Executive Council, 1988). For agriculture nonpoint sources, many strategies have focused on controlling fertilizer and manure applications, including regulations on storage and usage of animal manure and regulations on concentrated animal operations and feeding operations (New York State Department of Environmental Conservation, 2007; Pennsylvania Department of Environmental Protection, 2004; US Department of Agriculture and US Environmental Protection Agency, 1999). For stormwater sources, the USEPA initiated the National Pollutant Discharge Elimination System Phase I regulations in 1990 for Municipalities with Separate Storm Sewer Systems (MS4s) serving populations of 100,000 or more (US Environmental Protection Agency, 2000) and expanded the program in 1999 to include smaller MS4s (US Environmental Protection Agency, 2000). States also took actions to promulgate regulations on stormwater discharges in the late 1990s to early 2000s (New York State Department of Environmental Conservation, 2007; Pennsylvania Department of Environmental Protection, 2004).

For the NAS, efforts were made by Italian regulators to progressively reduce P content in detergents to 1% (6% for automatic laundry detergents) in 1989 (Marchetti et al., 1989; Rinaldi, 2014). In that year, industries also substituted sodium-triphosphate-based detergents with zeolite A (Glennie et al., 2002). The control of N loads has been improved recently with the adoption of the European Nitrates Directive (91/676/EEC). However, the reduction of N loads was less effective compared to that of P loads, because N is largely contributed by nonpoint sources (Viaroli et al., 2010, 2018). The creation of designated Authorities for specific hydrographic basins and of Regional Environmental Protection Agencies in 1994 further supported a coordinated monitoring of continental waters in Italy. The main measures adopted to reduce eutrophication from river loads were (a) establishment of spill basins and manure wastes for crop and livestock farming, (b) implementation of good farming practices in vulnerable areas, and (c) implementation of the Urban Waste Water Treatment Directive (91/271/EEC) (Bortone, 2014).

The N and P inputs in the NAS are not limited to riverine inputs. Direct inputs to the NAS occur via groundwater discharge and treatment plants, but are poorly quantified. Groundwater aquifers, particularly in the NE Adriatic region, are very sensitive to external pollution due to their large draining capacity of surface waters and low self-cleaning potential (The EU.WATER Project, 2010). For treatment plants, several underwater pipelines have been built since the 1980s, especially in the NE Adriatic region. Despite a gradual upgrade of treatment plants from secondary to tertiary treatment, wastewater loads continue to be an important source of nutrients in the NAS (Cozzi et al., 2014; Scroccaro et al., 2010; Sekulić et al., 2004; Volf et al., 2018).

2.5. MAJOR CHALLENGES

2.5.1. Legacy Sources

Many restoration efforts around the world have not yet achieved significant progress in reducing riverine loads of nutrient and sediment due to challenges such as legacy inputs, which accumulate and are stored in groundwater aquifers and sediments. Such effects have been documented for watersheds in North America (e.g., Chesapeake Bay, Mississippi River, and Lake Erie) and Europe (Basu et al., 2010; Jarvie et al., 2013; Sharpley et al., 2013; Van Meter et al., 2017; Van Meter et al., 2017, 2018; Vero et al., 2017). For CB, there is strong evidence for the importance of legacy sources. For example, riverine loads in the Susquehanna remained relatively constant in the past 30 years despite strong reductions in anthropogenic inputs to watersheds (Zhang, Ball, et al., 2016). Such patterns may reflect the effects of legacy sources (Basu et al., 2010; Thompson et al., 2011). Van Meter et al. (2017) reported that N dynamics in the Susquehanna are dominated by groundwater legacies, with 18% of the current annual N input to the river being at least 10 years old. Apparent ages of groundwater in the CB watershed can reach 20 years or more (Focazio et al., 1997) and base flow accounts for a major fraction of riverine N load at many CB sites (Bachman et al., 1998). For this region, the legacy stores are comprised primarily of groundwater for N (Bachman et al., 1998; Sanford & Pope, 2013), surface soils and river sediments for P (Ator et al., 2011; Sharpley et al., 2013), and stream corridors and reservoir beds for sediment (Gellis et al., 2008; Pizzuto et al., 2014; Walter & Merritts, 2008). These results suggest the importance of considering lag time between implementation of management actions and achievement of water-quality improvement. For the NAS, budget estimates indicate the accumulation in river watersheds of inorganic and organic N and P from anthropogenic sources that still negatively affect the quality of freshwater systems (Giani et al., 2012; Viaroli et al., 2018; Volf et al., 2018) and river-dominated coastal areas (Alvisi & Cozzi, 2016).

2.5.2. Climate Change

Climate change is another major challenge to ecosystem restoration (Charlton et al., 2018; Forber et al., 2018; Meier et al., 2018; Rankinen et al., 2016; Sinha et al., 2017). In general, climate change is expected to result in increased air and water temperature and an acceleration of the water cycle (Bloschl et al., 2017; Milly et al., 2005; Najjar et al., 2010; Rice & Jastram, 2014; Rice et al., 2017), which can alter the volume transport of freshwater and inputs of nutrients and sediments. For example, Sinha et al. (2017) estimated that climate-change-induced precipitation changes alone will substantially increase (19 ± 14%) riverine inputs of TN within the continental United States by the end of the century. In addition, the effects of climate change can differ among seasons. For CB, projected acceleration of the water cycle is expected to increase river runoff and associated inputs of nutrients and sediments during winter–spring and to decrease runoff during summer–fall (Wagena et al., 2018). Thus, management strategies for CB need to account for the impact of projected climate change on water quality. In this context, modeling and assessment is underway in the Chesapeake Bay Program partnership to evaluate the effects of climate change on nutrient export, efficacy of best management practices, and water quality in the estuary.

In contrast, climate-driven changes in the water cycle in the NAS watershed may tend toward persistent periods of low runoff alternating with episodic events of high discharge. Climate change appears to be increasing the frequency of heavy precipitation events (Alcamo et al., 2007). This has yet to induce long-term changes in the annual discharge of the Po River, which has greatly oscillated over the past three decades without showing clear trends (Cozzi & Giani, 2011). However, flow dynamics of Po River are characterized by a shift towards early spring peaks of runoff (Zampieri et al., 2015) and a decline in summer flows (Cozzi et al., 2019). At the same time, the other NAS rivers have shown a strong reduction in flow (Cozzi et al., 2012). It is important to note that reductions in river flow can result from both a greater anthropogenic use of continental waters as well as from climate-driven changes. For example, annual runoff to Adriatic rivers of Slovenia were reduced (6%) in 1971–2000 due to increased evapotranspiration of the soils (11%), even in the presence of relatively constant precipitation (Frantar, 2007).

2.5.3. Reservoir Filling

A major challenge that is unique to CB is the filling of the Conowingo Reservoir of the Susquehanna River which has neared its sediment storage capacity after 90 years of operation. As sediment accumulates in this reservoir, the cross-sectional area available for flow, and the vertical depth from water surface to sediment bed, decreases, thereby increasing the average horizontal flow velocity. Consequently, sediment trapping by the reservoir decreases and sediment load to CB increases. Numerous studies have demonstrated the declining trapping performance of this reservoir in recent decades (Hirsch, 2012; Langland, 2015; Zhang et al., 2013; Zhang, Hirsch, et al., 2016). Moreover, Zhang, Hirsch, et al. (2016) reported that such decline in reservoir trapping has occurred under a wide range of flow conditions. These changes, if not addressed, can hinder the attainment of the Chesapeake Bay Total Maximum Daily Load goals because the reservoir was expected to continue trapping sediments and nutrients at historical rates for another 20–30 years when those goals were established in 2010. Thus, the Chesapeake Bay Program partnership has worked to incorporate recent scientific understanding in upgrading its watershed model to better capture the temporal changes in reservoir function (Linker, Batuik, et al., 2013; Shenk & Linker, 2013), which will be used to adjust the goals of nutrient and sediment reductions by each jurisdiction.

2.6. IMPLICATIONS AND RECOMMENDATIONS

Anthropogenic riverine inputs of N, P, and sediment have led to undesirable consequences in the coastal marine environment, including eutrophication and associated oxygen depletion, declines in water transparency, loss of submerged aquatic vegetation, and shifts in the composition of plankton communities (Boesch et al., 2001; Breitburg et al., 2018; Cloern, 2001; Degobbis, 1989; Diaz & Rosenberg, 2008; Giani et al., 2012; Kemp et al., 2005). Therefore, reduction of watershed inputs has been a management priority for many coastal marine systems, including CB and the NAS. A review of parallel time-series data on hypoxia and watershed loading rates in coastal ecosystems shows that oxygen conditions tend to improve rapidly and linearly when the primary driver targeted for control is nutrients from wastewater treatment plants (Kemp et al., 2009). In larger more open systems, where nonpoint nutrient loads are more important in fueling eutrophication, responses to remediation tend to be nonlinear with hysteresis and time-lags. Nonetheless, there have been some signs of ecosystem recovery. For CB, water quality improved with time during 1985–2016, which is statistically linked to the reduction of riverine inputs of TN (Zhang et al., 2018). For the NAS, the reduction of riverine loads of P has been an effective method to alleviate eutrophication, even with high inputs of N and silicates (Djakovac et al., 2012; Giani et al., 2012). However, ecosystem conditions in this posteutrophic phase are still not comparable to those in pristine environments due to the occurrence of hypoxia and degraded benthic habitats in shallow coastal zones (Alvisi & Cozzi, 2016; Stachowitsch, 2014). Thus, continued reduction of watershed loads is indispensable for both CB and the NAS.

After decades of management efforts, the goals of CB and the NAS restoration have not yet been fulfilled (Volf et al., 2018; Zhang et al., 2018). Moving forward, we provide the following recommendations:

- continue monitoring river flows and water quality in the major tributaries to CB and the NAS;
- improve statistical approaches for quantifying riverine constituent loads and trends, including associated uncertainties;
- increase understanding of watershed factors that influence riverborne loads and trends (e.g., land use, hydrology, source controls) and their relative importance;
- develop consensus and solutions among stakeholders to address the major challenges that hinder the achievement of restoration goals in a timely fashion (e.g., legacy sources, climate change, and reservoir filling);
- increase understanding of the effects of land-based inputs on downstream water quality and ecological responses (e.g., dissolved oxygen, water clarity, chlorophyll-*a*);
- enhance public awareness of the impacts of anthropogenic nutrient loading, management goals and actions, progress toward achieving these goals, and major challenges.

In a world with seemingly ubiquitous nutrient enrichment and water-quality degradation, past and future advancement in our scientific understanding on these two coastal ecosystems can be valuable resources that may guide and facilitate the protection and restoration of estuarine and coastal ecosystems in other geographical locations.

ACKNOWLEDGMENTS

The authors thank many institutions for making the river monitoring data available, including the US Geological Survey, the Chesapeake Bay Program, the Environmental Agency of the Republic of Slovenia, the Croatian Meteorological and Hydrological Service, the European Environmental Agency, the Autorità di Bacino Distrettuale del fiume Po, the Regional Environmental Protection Agencies of Emilia Romagna, Veneto and Friuli Venezia Giulia for the NAS rivers. The editors and anonymous reviewers are greatly appreciated for their comments on an earlier draft of this chapter. This is contribution CN5869 of the University of Maryland Center for Environmental Science.

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