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Review

An 'extreme' future for estuaries? Effects of extreme climatic events on estuarine water quality and ecology

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ABSTRACT

Recent climate observations suggest that extreme climatic events (ECE; droughts, floods, tropical cyclones, heat waves) have increased in frequency and/or intensity in certain world regions, consistent with climate model projections that account for man's influence on the global climate system. A synthesis of existing literature is presented and shows that ECE affect estuarine water quality by altering: (1) the delivery and processing of nutrients and organic matter, (2) physical–chemical properties of estuaries, and (3) ecosystem structure and function. From the standpoint of estuarine scientists and resource managers, a major scientific challenge will be to project the estuarine response to ECE that will co-occur with other important environmental changes (i.e., natural climate variability, global warming, sea level rise, eutrophication), as this will affect the provisioning of important ecosystem services provided by estuaries.

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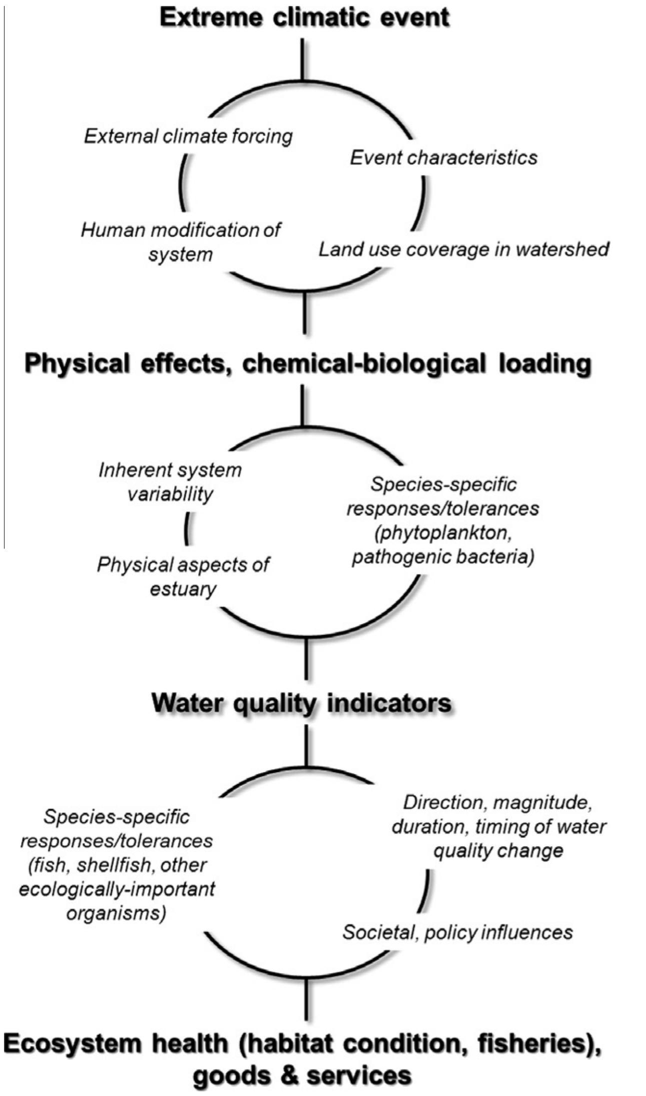

Fig. 1. Schematic representation of the effects of extreme climatic events and potential modulating factors (in italics).

1. Background

Estuaries represent one of the world's most vital aquatic resources, providing food resources and habitat for ecologically and economically important fish and shellfish species, recreational opportunities, scientific and educational experiences, and other important ecosystem services (Costanza et al., 1997; Hobbie, 2000; Pendleton, 2008; Yoskowitz et al., 2010). But estuaries worldwide are being exposed to an increasingly complex suite of environmental perturbations originating in their watersheds (e.g., anthropogenic nutrient loading, land use change, hydrologic modification) and externally from climatic forcings (e.g., tropical cyclones, droughts, global warming, sea level rise). The cumulative effect(s) of these perturbations often includes declining water quality (eutrophication) and deleterious changes in ecosystem structure and trophic dynamics in many estuaries (Cloern, 2001; Paerl et al., 2006; Rabalais et al., 2009). This, in turn, may alter or compromise the quantity and quality of goods and services provided by estuaries.

Climate variability, and specifically extreme climatic events (ECE), strongly influences the delivery of freshwater and associated nutrients and organic matter to the coastal zone (Scavia et al., 2002; Paerl et al., 2006). Despite evidence highlighting large-scale ecological effects of ECE on the coastal zone, our knowledge is limited in terms of mechanistic linkages between particular events and subsequent estuarine water quality and ecosystem responses. This, in turn, hinders the ability of scientists and resource managers to project the trajectory and magnitude of future water quality changes in estuaries and thus effectively manage these vital aquatic resources. Here we summarize the effects of ECE on estuarine water quality and important ecosystem services. We follow a conceptual model that examines effects of ECE on external chemical–biological loading to estuaries as well as effects on estuarine physical dynamics. We then document subsequent impacts on water quality indicators and highlight the cumulative effects of these changes on estuarine ecosystem health, goods and services (especially fisheries), all while attempting to identify important modulating factors (Fig. 1). The emphasis will be on tropical cyclones, floods, drought, and heat waves. We focus on the following questions:

1. How do ECE affect estuarine water quality, and does this depend on the characteristics of a particular event (e.g. timing, duration, trajectory in the case of tropical cyclones)?
2. How long do the water quality changes caused by ECE persist, and does this depend on the tidal regime of affected estuaries?
3. Is there a relationship between land-use coverage in estuarine watersheds and ECE-induced water quality changes?

These questions must be answered to effectively project and plan for the effects of future changes in frequency or intensity of ECE and the eventual impacts on the provisioning of ecosystem services.

2. Effects of extreme climatic events on estuarine water quality

2.1. How do ECE affect estuarine water quality, and does this depend on the characteristics of a particular event?

2.1.1. Tropical cyclones and floods

Tropical cyclones (e.g., Paerl et al., 1998, 2001; Williams et al., 2008) and floods (Wong et al., 2010) can mobilize significant quantities of organic matter and suspended solids which are subsequently deposited in estuaries. The combination of high organic matter loads and stratification caused by the freshwater input can interact to promote hypoxia/anoxia in subpycnocline water and occasionally in the whole water column (Paerl et al., 1998; Mallin et al., 2002; Burkholder et al., 2004; Tomasko et al., 2006). In some cases, freshwater itself may contain relatively low (i.e., hypoxic) oxygen levels that exacerbate these conditions (e.g., Mallin et al., 2002; Tomasko et al., 2006). For example, Hurricanes Fran (Cat. 3) and Bertha (Cat. 2) caused abrupt, adverse changes in the water quality of freshwater and estuarine ecosystems along the US Atlantic coast in 1996. Runoff generated by these storms delivered considerable amounts of terrestrial organic matter, nitrogen (N)-rich organic matter from wastewater treatment facilities, and inorganic nutrients into North Carolina's Cape Fear River (Mallin et al., 1999). This led to hypoxia/anoxia that persisted for nearly 1 month (Mallin et al., 1999, 2002). Large fish kills were reported in the river, and further downstream in the Cape Fear estuary, hypoxic conditions lasted several weeks and had deleterious effects on the benthic invertebrate community (Mallin et al., 1999, 2002). In the nearby Neuse River Estuary, runoff from Fran delivered a large volume of low oxygen, organic matter-rich freshwater to the system, also resulting in development of hypoxic/anoxic conditions and fish mortalities (Paerl et al., 1998).

Tropical cyclones (Bales, 2003; Peierls et al., 2003; Burkholder et al., 2004) and floods (e.g., Magnien et al., 1992; Hickel et al.,

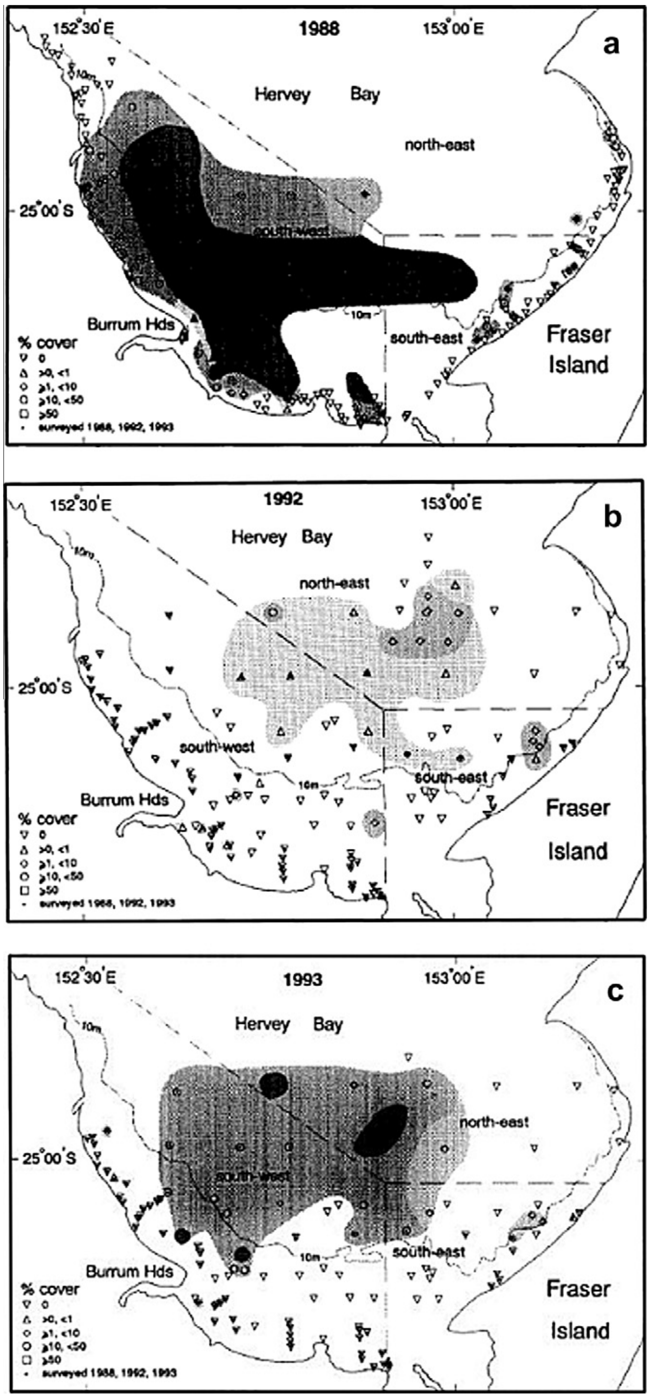

Fig. 2. Abundance of seagrass (% cover) in Hervey Bay, Australia, prior to (a), ~10 months after (b), and ~22 months after massive flooding and passage of a tropical cyclone (c). Stippling represents seagrass cover generalized from sampling sites. These events led to severe light limitation and significant sediment resuspension that prolonged recovery of seagrass beds. Reprinted from: Preen, A.R., Lee Long, W.J., Coles, R.G., 1995. Flood and cyclone related loss, and partial recovery, of more than 1000 km^2 of seagrass in Hervey Bay, Queensland, Australia. Aquatic Botany 52, 3–17, with permission from Elsevier.

1993; Correll et al., 1999; Eyre and Ferguson, 2006; Cook et al., 2010; Voynova and Sharp, 2012) can also lead to significant nutrient loading in estuaries, where phytoplankton growth is often nutrient limited (Cloern, 2001). Despite this, the phytoplankton response to these nutrient-loading events is not straightforward. For instance, high turbidity in flood waters and rapid flushing, which tend to be most pronounced in the days and weeks following these events, can create conditions that are unfavorable for phytoplankton growth. Several studies have noted reductions in phytoplankton productivity in estuaries immediately following tropical cyclones (e.g., Paerl et al., 1998; Mallin et al., 2002) and floods (Eyre and Ferguson, 2006; Murrell et al., 2007) due to reduced light availability or excessive flushing. Certain taxa of phytoplankton do appear to be favored under post-cyclone or flood conditions (i.e., relatively low light, high nutrient and organic matter concentrations), including harmful species belonging to the Raphidophyceae and Dinophyceae (e.g., Kempton et al., 2008; Hall et al., 2008). Because of the mixotrophic capabilities of many of these organisms, they are capable of supplementing energetic demands under low light conditions through utilization of organic matter and can thus outcompete other phytoplankton such as diatoms and flagellates (Burkholder et al., 2008; Jeong, 2011). When light limitation is not a factor, tropical cyclones may rapidly (<1 week after passage) stimulate phytoplankton productivity and bloom development as a result of wind-driven resuspension of nutrients (e.g., Miller et al., 2006; Wetz and Paerl, 2008). Phytoplankton blooms have also been noted to occur after a lag phase of weeks to months post-cyclone or flood (Paerl et al., 2001; Peierls et al., 2003; Eyre and Ferguson, 2006; Hagy et al., 2006; Murrell et al., 2007; Cook et al., 2010; Voynova and Sharp, 2012), which allows ample time for the light environment to become favorable again and flushing to lessen.

Seagrasses can also be affected by changes in the light environment that often accompanies tropical cyclones and floods. Light stress caused by these events can lead to massive seagrass die-off if low light conditions persist for many weeks to months (e.g., Preen et al., 1995; Longstaff and Dennison, 1999; Campbell and Mckenzie, 2004; Carlson et al., 2010; Fig. 2). In contrast, shorter-lived events tend to have minimal impact on seagrasses (e.g., Longstaff and Dennison, 1999; Anton et al., 2009). Furthermore, seagrasses appear to be fairly resilient to the extreme physical stresses caused by tropical cyclones (Steward et al., 2006; Anton et al., 2009; Carlson et al., 2010), and several authors have noted that these events may actually lead to seagrass range expansion by dispersal of seeds (Kendall et al., 2004; Bell et al., 2008).

There is no question that tropical cyclones and floods can affect estuarine water quality, but interestingly the degree to which this occurs may be dependent on the characteristics of a particular event (e.g., Mallin and Corbett, 2006; Wetz and Paerl, 2008; Briceno and Boyer, 2010). For instance, it has been proposed that tropical cyclones passing through an estuary's watershed may mobilize more nutrients and/or organic matter than those passing over or skirting an estuary, thereby leading to more severe effects on estuarine water quality (Mallin et al., 2002; Mallin and Corbett, 2006). The effects of floods and tropical cyclones may also depend on post-event environmental conditions. For instance, Hickel et al. (1993) observed large phytoplankton blooms in the German Bight following floods in 1981, but not following floods in 1987–1988. They attributed this differential response to differences in wind conditions following the floods. Calm winds followed the 1981 floods, which interacted with freshwater input to allow for development of stratification and favorable light conditions for phytoplankton. In contrast, strong winds followed the 1987–1988 floods, leading to significant vertical mixing and poor light conditions. Mallin et al. (2002) suggested that tropical cyclones which strike temperate estuaries in late fall may have a less severe effect on water quality than do those that occur during warmer summer months. Cooler temperatures found in late fall should not only promote higher oxygen solubility, but also lead to reduced bacterial growth and degradation of freshwater-derived organic matter (Mallin et al., 2002). Furthermore, light levels are lower in late fall, which may prevent large-scale phytoplankton blooms from forming. Unfortunately, little information is available on the influence of tropical cyclone or flood characteristics on estuarine water

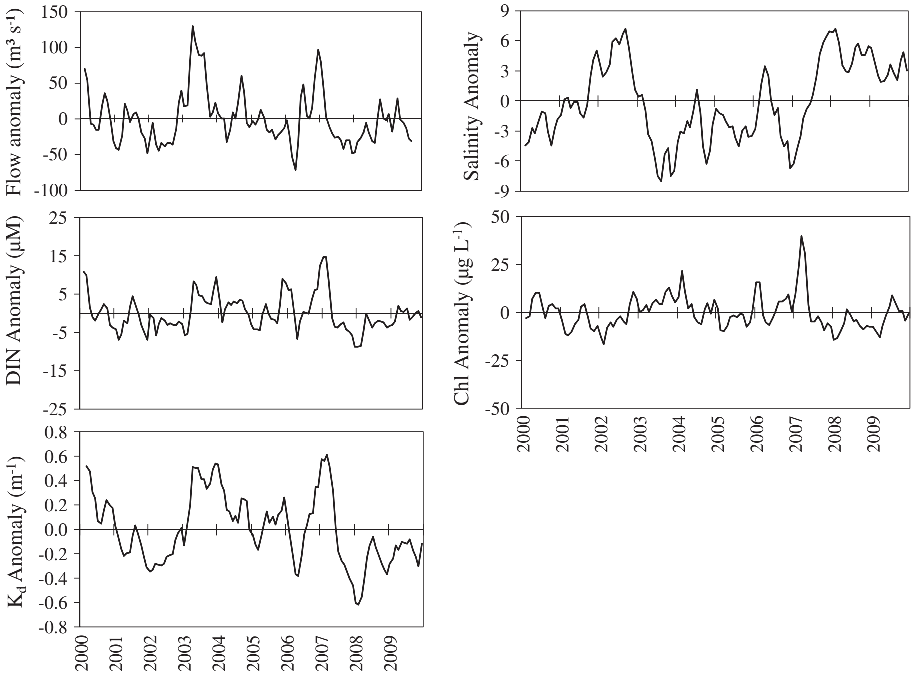

Fig. 3. Mean monthly anomalies of river flow, salinity, dissolved inorganic nitrogen, chlorophyll a, and light attenuation (K_d) in North Carolina's Neuse River Estuary. "Anomaly" refers to the deviation from a long-term (2000–2009) monthly average. Positive K_d indicates higher than average light attenuation. Included in this time series are two drought periods (2001–2002, 2007–2008) and two periods of above average precipitation and river flow (2003, late 2006–early 2007). For additional details on study design, see: Wetz, M.S., Hutchinson, E., Lunetta, R., Paerl, H.W., Taylor, J.C., 2011. Severe droughts reduce planktonic production in estuaries with cascading effects on higher trophic levels. Limnol. Oceanogr. 56, 627–638.

quality. Thus many of the aforementioned relationships need further verification through sustained observations that allow for comparison of the effects of different events on the water quality of a given system.

In addition to the nutrient and organic matter loading that may accompany tropical cyclones and floods, there is growing concern about other contaminants such as microbial pathogens. A number of studies have noted significant loading and/or enhancement of *in situ* growth of fecal coliforms (Weiskel et al., 1996; Lipp et al., 2001; Chigbu et al., 2004; Coulliette and Noble, 2008), enteroviruses (Lipp et al., 2001), and *Vibrio* spp. (Wetz et al., 2008; Lara et al., 2009) following tropical cyclone passage or floods. As will be discussed later, loading of these pathogens via stormwater appears to be exacerbated in highly urbanized settings.

2.1.2. Drought

Limited studies on extreme low freshwater inflow events suggest that these events may affect estuarine water quality in a manner dissimilar to high freshwater inflow events, namely by improving it. Less freshwater inflow should theoretically equate to reduced allochthonous organic matter and nutrient inputs, and also reduced phytoplankton bloom activity.

Several studies have in fact shown a general decrease in nutrient loading, phytoplankton productivity and/or biomass during extended periods of below average freshwater inflow (e.g., Cloern et al., 1983; Nixon, 2003; Boynton et al., 2008; Lin et al., 2008; Abreu et al., 2010; Phlips et al., 2010; Wetz et al., 2011; Fig. 3). In a situation perhaps analogous to a prolonged drought, freshwater discharge to the Nile Delta decreased by >90% and seasonal phytoplankton blooms ceased to develop after construction of the Aswan Dam on the Nile River in the 1960s (Nixon, 2003). In recent years, the productivity of the Nile Delta has recovered somewhat, not as a result of restored freshwater inflow but instead from significant increases in anthropogenic nutrient loadings (Nixon, 2003). This is an interesting scenario that suggests enhanced anthropogenic nutrient loading may sustain ecosystem productivity during extreme low freshwater inflow events. However, it remains to be determined whether the ecosystem behaves the same in these conditions as opposed to under more normal freshwater inflow regimes, especially considering that point-source anthropogenic nutrients are qualitatively/functionally different than watershed-derived nutrients and may lead to phytoplankton blooms that are harmful to the ecosystem (e.g., Anderson et al., 2002; Nixon, 2003).

There are examples of estuarine systems that do not follow the general trend of reduced phytoplankton biomass or productivity during low freshwater inflow conditions. One complicating factor is that freshwater inflow often has a significant modulating effect on residence time as well. In the eutrophic Hudson River Estuary for example, persistent high nutrient loads only lead to significant phytoplankton productivity during extended periods of low flow. This reduces advection of phytoplankton via increased residence time and also increases stratification that improves the light environment for phytoplankton (Howarth et al., 2000). Temporarily open/closed coastal (estuarine) lakes of South Africa, Australia and New Zealand are affected not only by freshwater inflow but also by flushing associated with intermittent opening of their mouth, allowing for exchange with coastal seawater. As such, complex water quality and phytoplankton responses are often observed that depend on both freshwater inflow and exchange with coastal waters, with closing of the mouths often leading to increased residence time, accumulation of nutrients and phytoplankton biomass (e.g., Taljaard et al., 2009; Perissinotto et al., 2010; Schallenberg et al., 2010).

Water quality data from Wetz et al. (2011) highlights the effects of variable freshwater inflow on water quality in a spatially-explicit manner, indicating that while the scenario of drought-induced reductions in nutrient inputs and phytoplankton activity generally holds true, the relationship is somewhat more complicated. For instance, during two droughts that affected North

Carolina's Neuse River Estuary during the 2000s, phytoplankton productivity and biomass were below average and bottom water dissolved oxygen was above average over a substantial portion of the estuary during drought, but phytoplankton productivity was actually enhanced and bottom water dissolved oxygen was below average in the upper estuary due to upstream migration of the estuarine chlorophyll maxima (Wetz et al., 2011). Thus from an estuarine habitat management perspective, water quality may improve in some areas of an estuary while deteriorating in others as a result of drought.

Several authors have also noted above average estuarine dissolved oxygen levels during drought (e.g., Livingston et al., 1997; Attrill and Power, 2000a; Boynton et al., 2008; Lin et al., 2008; Wetz et al., 2011), attributing this to reduced stratification and reduced deposition of phytodetritus. Another possibility may be increased benthic primary production. Increased light penetration is often an artifact of reduced freshwater inflow and/or reduced water column phytoplankton biomass (Livingston et al., 1997; Wetz et al., 2011; Fig. 3) and may favor the growth of, and oxygen production by, benthic microalgae or other benthic primary producers (Rask et al., 1999; Fear et al., 2004; Stutes et al., 2006; Krause-Jensen et al., 2007; Murrell et al., 2009). There is almost no information available on the response of benthic primary producers to sharp reductions in freshwater inflow, though the aforementioned studies point to a necessity for more research focusing on coupled benthic and pelagic primary producer responses to changes in freshwater inflow.

Despite generally positive effects on estuarine water quality, low freshwater inflow events may still have deleterious effects on living resources. Most apparent are effects of these events on the physical–chemical conditions in estuaries (e.g., salinity, marsh chemistry, etc.) and subsequent implications for habitat suitability. For example, several recent studies have noted dieback of salt marsh grasses during droughts (Mckee et al., 2004; Alber et al., 2008). Although the cause of these diebacks is unknown, at least one study suggested that there are direct and negative effects of drought on marsh grass physiology (Brown et al., 2006). Similarly, a recent study documented major declines in seagrass biomass during drought (Cardoso et al., 2008).

Drought also appears to create conditions favorable for certain bivalve pathogens, namely *Haplosporidium* and *Perkinsus*, as has been noted by the inter- and intra-estuary range expansion of these organisms during droughts (Burreson and Calvo, 1996; Burreson and Ford, 2004; Bushek et al., 2012). Drought also favors many predators of bivalves, hence during drought severe mortality of commercially and ecologically-important shellfish is often observed due to both predation and disease (e.g., Buzan et al., 2009; Petes et al., 2012). In regards to human pathogens, recent studies conducted in several North Carolina estuaries suggests that the abundance of the pathogen *Vibrio vulnificus* may be greatly reduced during drought relative to above average freshwater inflow conditions (Wetz and Noble, submitted for publication; Froelich et al., 2012).

2.1.3. Interactions between droughts and tropical cyclones/floods

Few studies have examined the interactive effects of ECE, though there are intriguing examples which suggest that the sequence of drought–cyclone/flood events may significantly affect estuarine water quality. For example, Kaushal et al. (2008) observed high nitrogen export during wet conditions that followed a drought in a Chesapeake Bay watershed, implying that nitrogen is stored in soils and other reservoirs within the watershed during drought. Likewise, Sigleo and Frick (2007) observed very high nitrate export from a coastal Oregon watershed during the first storm that followed a prolonged drought. Thus, the sequence of events may affect the relative magnitude of nutrient loading accompanying ECE (see also Baldwin et al., 2005). A biological example pointing to the importance of drought–flood sequences comes from the lower Chesapeake Bay and tributaries. In 2007, a massive bloom of the icthyotoxic dinoflagellate, *Cochlodinium polykrikoides*, developed coincident with a period of heavy rains that followed a drought and low nutrient inputs (Mulholland et al., 2009). The authors suggest this sequence allowed *C. polykrikoides* to gain relative importance over other phytoplankton because of its ability to utilize organic nutrients during the drought; hence, when even more favorable conditions developed (i.e., high inorganic and organic nutrient loads accompanying the heavy rains), it was primed for bloom formation. These limited examples highlight the importance of not only quantifying the effects of specific events, but also the necessity for examining differences in estuarine water quality response that may be attributed to the *sequence* of events. This will require sustained observations and highlights the importance of coastal time-series studies that focus on key organisms and ecological processes.

2.1.4. Heat waves

A number of important water quality-related processes are positively correlated with temperature in estuaries, including (but not limited to) bacterial respiration and oxygen demand in subpycnocline waters (Stanley and Nixon, 1992; Cowan and Boynton, 1996; Borsuk et al., 2001) and microbial pathogen growth (Motes et al., 1998; Pfeffer et al., 2003; Hsieh et al., 2008). Yet few studies have specifically focused on the effects of heat waves. The few studies that are available all suggest a generally negative effect of heat waves on estuarine water quality. For instance, several potentially toxic cyanobacteria species thrive at high temperatures (reviewed by Paerl et al. (2011)). One example comes from northern Europe, where during a recent heat wave massive blooms of *Microcystis*, a toxin producing cyanobacteria, were observed in estuarine waters (Johnk et al., 2008). Concerns are also mounting over the synergistic effects of high temperatures and increasing carbon dioxide levels in the atmosphere, which may further act to intensify cyanobacterial blooms (Paerl et al., 2011). Stimulatory effects of high temperatures have also been noted on pathogenic microbies in coastal waters. For example, in Israeli coastal waters, a major outbreak of *V. vulnificus* occurred coincident with a record heat wave (Paz et al., 2007). These outbreaks of harmful microbes point to the risks associated with heat waves and emphasize the need for further study of time-series data that emphasizes species-specific responses to elucidate the ecological effects of heat waves. These examples also highlight the importance of developing emerging technologies for rapid quantification of harmful species and/or their toxins.

2.1.5. Summary

- Tropical cyclones and floods often have a net negative effect on estuarine water quality by causing significant organic matter, nutrient and pathogen loading, as well as deoxygenation of estuarine waters. One positive aspect of tropical cyclone passage is the wind-driven disruption of subpycnocline hypoxia that is facilitated by stratification (Rabalais et al., 2009), though this immediate benefit may be negated by the aforementioned longer-term negative effects of tropical cyclones.
- Severity of water quality changes caused by tropical cyclone passage may depend on location of landfall and cyclone trajectory.
- Effects of tropical cyclones and floods may depend on their timing as well as post-event environmental conditions.
- Droughts influence water quality by altering freshwater inflow, watershed nutrient/organic matter loading, residence time and ultimately phytoplankton activity. Positive and negative effects of drought on water quality have been documented.

- Droughts can reduce habitat suitability for important estuarine organisms by induction of physiological stress or promotion of pathogen outbreaks.
- The effects of drought and tropical cyclones/floods are modulated by the sequence in which these events occur.
- High temperatures, such as during heat waves, stimulate a number of important ecological processes, many of which have negative consequences for estuarine water quality.

2.2. How long do the effects of ECE persist, and how does this depend on the tidal regime of affected estuaries?

Tropical cyclones and floods tend to elicit fairly short-lived water quality changes in estuaries (Valiela et al., 1998; Eyre and Ferguson, 2006; Hagy et al., 2006; Tomasko et al., 2006; Dix et al., 2008). For instance, Tomasko et al. (2006) and Dix et al. (2008) examined the effects of several major tropical cyclones that passed through Florida in 2004. In central Florida, Tomasko et al. (2006) found that cyclone-derived changes in water quality lasted <2 weeks. As a result of the same cyclones passing over a north Florida estuary, Dix et al. (2008) found that cyclone-induced organic matter and nutrient loading lasted for <1 month. Following Hurricane Ivan's passage over Florida's Pensacola Bay, Hagy et al. (2006) observed development of a modest phytoplankton bloom, which again lasted <1 month. Likewise, cyclone-induced blooms noted in Wetz and Paerl (2008) also tended to last <1 month. However, back-to-back cyclones may elicit a much more severe water quality response. For instance, three cyclones struck eastern North Carolina in late summer–fall 1999, leading to severe flooding that lasted several months and that stimulated large phytoplankton blooms in a coastal embayment, Pamlico Sound, that lasted ~6 months (Paerl et al., 2001; Peierls et al., 2003). The longest lived effects of tropical cyclones and floods may be on seagrass communities. In those instances where severe loss of seagrass beds was noted to occur following tropical cyclone passage or flooding, it took several years for the seagrass to recover (Preen et al., 1995; Campbell and Mckenzie, 2004).

The overall susceptibility of an estuarine ecosystem to tropical cyclone and flood effects may depend on an estuary's tidal regime. In some cases, affected estuarine waters can be rapidly diluted via exchange with relatively oligotrophic coastal waters following tropical cyclone passage (Tomasko et al., 2006; Edmiston et al., 2008). Similarly, Caffrey et al. (2007) observed stimulation of phytoplankton production but not biomass in response to freshwater nutrient pulses in Elkhorn Slough estuary, California. Those authors argued that the lack of phytoplankton biomass accumulation was due to the rapid tidal flushing of this estuary, which is consistent with the analysis of Monbet (1992). Cross-ecosystem comparative studies are merited to better understand the relationship between tidal flushing and cyclone or flood effects. This mechanistic understanding will be necessary to assess the vulnerability to these effects for the broad range of estuaries as classified according to degree of tidal flushing.

When one compares the duration (~3–6 months) of water quality perturbations caused by even the most severe back-to-back tropical cyclones with those of recent droughts (>6 months), it appears that drought elicits a much more prolonged ecosystem response. Wetz et al. (2011) found that the ecological effects of a drought that began in mid-2007 and officially ended mid-2008 actually lasted well into 2009. Paleoclimate records suggest that naturally occurring droughts lasting 10 years or more have occurred quite regularly in the eastern United States during the past 1600 years (Stahle et al., 1988; Cronin et al., 2000), accompanied by water quality changes in affected estuaries (Cronin and Vann, 2003).

2.2.1. Summary

- Water quality changes induced by tropical cyclones and floods are of much shorter duration (days to few months) than those from drought (months to years).
- Differences may exist in the water quality response within macrotidal versus microtidal estuaries to cyclone or flood events.

2.3. Is there a relationship between land-use coverage in estuarine watersheds and ECE-induced water quality changes?

Conversion of natural forested lands to urban area or agricultural land can alter water quality of adjacent rivers and estuaries (Hopkinson and Vallino, 1995; Carpenter et al., 1998). For example, Peierls et al. (1991) demonstrated that nitrate export in rivers increased precipitously as human population in watersheds increased. More recently, efforts have been put forth to better understand the effects of different types of land use on material export. A number of studies have now observed relatively higher organic carbon and/or organic nitrogen loads associated with agricultural or urbanized watersheds compared to forested watersheds (Howarth et al., 1991; Jordan et al., 2003; Rothenberger et al., 2009; but see Mallin et al., 2009). Similarly, inorganic nutrient (N and P) export tends to be elevated in agricultural or urbanized watersheds compared to forested watersheds (Vernberg et al., 1992; Bowen and Valiela, 2001; Jordan et al., 2003; Handler et al., 2006; Kaushal et al., 2008; Mallin et al., 2009; Rothenberger et al., 2009). In addition to affecting biogeochemical dynamics, land use coverage and change can significantly affect microbial pathogen loading. There is ample evidence pointing to agricultural or urbanized lands as being a relatively greater source of fecal coliform loading to adjacent waterbodies than forested lands (Vernberg et al., 1992; Mallin et al., 2000, 2001, 2009; Holland et al., 2004; Handler et al., 2006; Campos and Cachola, 2007; DiDonato et al., 2009). Impervious surface associated with urbanization is a particularly important driver of water quality. A study in eastern North Carolina showed that streams and estuaries adjacent to highly urbanized areas experienced proportionally greater stormwater runoff, nutrient and fecal coliform loads and biological oxygen demand than those with forested watersheds (e.g., Mallin et al., 2009). However, in contrast to those findings from microtidal estuaries in North Carolina, van Dolah et al. (2008) did not observe strong relationships between watershed urbanization and water quality in South Carolina tidal creeks, perhaps suggesting that tidal flushing may limit prolonged deleterious effects of stormwater pulses.

Through the synergism between land use coverage, stormwater pulses, and tidal flushing, future increases in ECE may cause as yet unpredictable effects on estuarine ecosystems. The contrasting results of van Dolah et al. (2008) and Mallin et al. (2009) for example, suggest that more studies are needed on the relationship between land use coverage, estuarine tidal flushing and water quality, perhaps through comparative studies of estuaries that vary in terms of their degree of tidal flushing. To date, there have been few studies that compare water quality responses to ECE across multiple estuaries that vary in land use coverage or tidal flushing.

2.3.1. Summary

- Estuaries with agricultural or urbanized watersheds experience relatively more severe water quality degradation from tropical cyclone passage or flooding than estuaries with forested watersheds.
- The degree of tidal flushing appears to affect the magnitude of water quality changes induced by tropical cyclone passage or flood events. Limited evidence suggests that microtidal estuar-

ies with urbanized watersheds may experience relatively more severe water quality degradation from tropical cyclone passage or flooding than macrotidal estuaries with urbanized watersheds.

3. An extreme future for estuaries?

Climate projections suggest the possibility of more frequent and/or intense high precipitation events, drought and heat waves in the near future as a result of anthropogenic greenhouse gas emissions (Meehl et al., 2007). For example, model projections from the most recent Intergovernmental Panel on Climate Change (IPCC) report suggest that air temperatures in coastal regions worldwide are expected to rise from 2 to 5 °C during the 21st century (Christensen et al., 2007). As such, the models project that prolonged heat waves will become more common in the future (Meehl et al., 2007). Concurrently, drought may become more frequent due to higher evapotranspiration rates and shifting precipitation patterns. Regions of particular concern for intensification of drought include both northern and southern Africa, southern Europe and the greater Mediterranean region, west-central Asia (Middle East), the western Gulf of Mexico, greater Caribbean region, and southern Australia (Christensen et al., 2007). Climate models also project a trend towards more intense precipitation events, at least a seasonal basis and possibly throughout the year, in regions such as west central and east central Africa, northern Europe, Asia (east, northeast, southern), the US Atlantic coast, and possibly eastern South America (Christensen et al., 2007). Finally, there are indications that tropical cyclones may become more intense but less frequent over the coming century (e.g., Elsner et al., 2008; Knutson et al., 2010). Consequently, it is conceivable that ECE will be an ever more important driver of estuarine ecosystem function and water quality in the near future. It is important to note that evidence is emerging that many of the projected changes to heat wave frequency, drought and flood cycles may already be underway (e.g., Min et al., 2011; Pall et al., 2011; Trenberth and Fasullo, 2012; Trenberth, 2012), adding a degree of urgency to efforts to understand their effects on ecosystems, including estuaries.

Anthropogenic climate change will be superimposed on substantial increases in human populations living in coastal watersheds. According to the 2010 revision of the United Nations population projections, the world population will surpass 9 billion people by the year 2050 and 10 billion people by 2100 (United Nations, 2011). At present it is estimated that 40% of the world's population lives within 100 km of the coast (CIESIN, 2012). Assuming no changes in these percentages and population growth estimates, 4 billion people could be living along the world's coasts by 2100 versus 2.8 billion today. Anthropogenic climate change and projected increases in human populations in the coastal zone will have far reaching implications for water cycles and estuarine ecosystem dynamics, as climate drivers such as ECE interact with anthropogenic change (land use, nutrient loading) to strongly affect estuarine water quality and overall estuarine ecosystem health (Scavia et al., 2002; Flemer and Champ, 2006; Paerl et al., 2006). These changes may be tempered or amplified by natural climate cycles, such as those associated with ENSO, the North Atlantic Oscillation, Pacific Decadal Oscillation and other modes of climate variability (e.g., Cloern et al., 2007; Kimmel et al., 2009; Barbosa et al., 2010; Pollack et al., 2011; Bushek et al., 2012), pointing to a need for considering these factors in totality, as opposed to in isolation (Fig. 1; Table 1). An important long-term goal for estuarine resource managers is to project estuarine ecosystem responses to various environmental change scenarios, including more frequent or intense ECE, ever increasing human freshwater demands, land use changes (urbanization) and nutrient enrichment of coastal watersheds (e.g., Najjar et al., 2010). The ability to accomplish this important task necessitates improvement of regional climate projections (Najjar et al., 2010), as global climate models are of limited use at regional or single-estuary scale due to uncertainties involved in downscaling precipitation patterns (Schiermeier, 2010; Maraun et al., 2010) and in tropical cyclone projections (Bender et al., 2010).

As highlighted in our review, there are fundamental gaps in our scientific understanding of mechanistic linkages between climate variability (incl. ECE), other environmental changes (i.e., land use), estuarine water quality and ecological dynamics that hinder efforts to project future changes to estuarine ecosystem dynamics. One reason may be a lack of dedicated funding due to the potential damaging effects on research infrastructure and significant risks involved (in the case of tropical cyclones and floods) (e.g., Lindemayer et al., 2010). Lack of coastal time-series studies encompassing a sufficient suite of systems, environmental parameters, and sampling locations within particular systems and their watersheds creates an additional challenge to quantifying environmental change in the coastal zone, and more specifically for understanding effects of ECE. For example, the most extensive US water quality monitoring program, the National Estuarine Research Reserve (NERR) program, includes study sites from nearly every coastal state in the US. At present, most NERR sites deploy instrumentation to continuously monitor basic water quality parameters (i.e., temperature, salinity, pH, dissolved oxygen), but could address a broader suite of ecological questions if given the ability to incorporate modern, but practical, technologies for continuous monitoring of other parameters (i.e., chlorophyll *a* and/or biomarker pigments, Richardson et al., 2010; nutrients and organic matter, Conmy et al., 2004; Jannasch et al., 2008; Plant et al., 2009). Additionally, many NERR locations are constrained by logistics and funding to only deploying monitoring equipment and/or collecting discrete samples at a few fixed sites. But if these efforts could be expanded to include a further subset of sampling sites along the estuarine salinity gradient, this could prove to be a powerful tool for characterizing the whole estuary response to climatic and/or anthropogenic changes. Similar constraints have been documented in European estuaries as well (de Jonge et al., 2006). At a larger scale, creation of integrated regional and continental scale sampling networks that include not only estuaries but their watersheds would represent a major step forward towards recognizing and understanding the critical role that watersheds will play in modulating the future direction of environmental changes in estuaries (e.g., de Jonge et al., 2006; Hopkinson et al., 2008).

Despite challenges associated with projecting the response of estuaries to ECE, here we offer two potential scenarios based upon general historic or contemporaneous patterns observed in estuaries, as previously highlighted in our review. The first emphasizes the potential effects of more intense tropical cyclone strikes or flood events, while the second emphasizes the potential effects of prolonged low freshwater inflow conditions.

3.1. Effects of more intense tropical cyclones and floods

Model projections from several recent studies converge on a solution of more intense cyclones and high precipitation events as a result of global climate change (Meehl et al., 2007; Elsner et al., 2008; Bender et al., 2010). Thus it is worthwhile to offer a broad scenario for the effects of stronger cyclones or precipitation (flood) events. It is conceivable that infrastructural damage (i.e., to municipal wastewater facilities, animal waste lagoons) will become more common as a result of winds and flooding at inland locations and as a result of winds, flooding and enhanced storm surge (resulting from sea-level rise) near the coast. This problem is likely to be most pronounced in developing nations, which are

Table 1
Summary of the short- and long-term effects of extreme climatic events on select water quality indicators. ↓ indicates negative effect, ↔ indicates neutral effect, ↑ indicates positive effect, sp. indicates potential for species-specific responses, and ? indicates not enough data available to pass judgment.

Water quality indicator	Short-term (days–weeks) effects	Long-term (months–years) effects
Phytoplankton and harmful algal bloom growth (via interaction between nutrient loading, light attenuation, stratification and residence time)	Cyclone and flood: ↓	↑↔↓
	Drought:	↓
	Heat wave: ↑sp.	↑?sp.
Bottom water oxygenation (via interaction between organic matter loading, stratification, temperature and residence time)	Cyclone and flood: ↑↔↓	↔
	Drought:	↑↔
	Heat wave: ↓?	↓?
Microbial pathogen growth (via interactions between pathogen loading, salinity and temperature)	Cyclone and flood: ↑	?
	Drought:	↑↓sp.
	Heat wave: ↑?	↑?
Seagrass or marsh grass health (via interactions between physical damage, physiological stress and light attenuation)	Cyclone and flood: ↓↔	↓↔
	Drought:	↓
	Heat wave: ?	?

expected to see a major increase in wastewater facilities and associated nutrient loadings over the coming decades due to population growth (van Drecht et al., 2009), but which may lack resources to build weather-resistant facilities. It is also conceivable that on land, soil mobilization and flushing may be enhanced, while scouring of sediments may be enhanced in riverine or estuarine waterbodies (DeLaune and White, 2012; Yoon and Raymond, 2012). The net effect of infrastructural damages as well as increased soil and sediment scouring will be enhanced point- and non-point source nutrient and organic matter loadings to estuaries. Possible regulating factors include the frequency at which these extreme precipitation events occur and/or prior climatic conditions, as well as the rate and direction of land use changes in affected coastal watersheds. Stronger cyclones or flood events will likely also lead to large-scale, prolonged stratification in estuaries as a result of freshwater input. In the short-term (days–weeks), flushing and low light levels may prevent phytoplankton growth, though allochthonous organic matter loading will likely be significant. Strong stratification and elevated allochthonous organic matter inputs, coupled with projected global temperature increases by the end of the 21st century, will create conditions favorable for bottom water hypoxia formation that is rapid in development, broadly distributed and persistent in time. Sediment resuspension and runoff-driven loading of pathogenic microbes, coupled with warmer temperatures, create conditions favoring widespread outbreaks of these organisms (Lipp et al., 2001; Coulliette and Noble, 2008; Fries et al., 2008; Wetz et al., 2008).

In the longer-term (weeks–months), enhanced nutrient inputs and relaxation of severe flushing conditions may act to stimulate widespread phytoplankton blooms, possibly including harmful algal bloom species. Upon reaching senescence, these blooms may further stimulate biological oxygen demand and hypoxia formation or, in the case of harmful algal blooms, may release deleterious compounds into the water column. From an ecosystem goods and services standpoint, a reasonable focus is on the habitat and fisheries implications of more severe tropical cyclones and floods. Hypoxic conditions alone can severely reduce habitat for a number of commercially-important fish and shellfish species and lead to significant localized, short-term mortality (e.g., Paerl et al., 2001). It is quite possible that high, sustained loadings of sediment, allochthonous (riverine) and autochthonous (phytoplankton) organic matter will lead to a long-term (months) decline in light penetration. This, in conjunction with other environmental stressors (i.e., sea level rise and associated erosional forces, unfavorable water quality conditions, etc.; Duarte, 2002), may have severe negative effects on seagrasses and associated biological communities in affected estuarine systems (Preen et al., 1995). In conjunction with increased hypoxia potential, loss of critical seagrass habitat would represent an additional stressor upon coastal fish communities (Waycott et al., 2009). Coincident with any negative effects on coastal fisheries that may develop, side effects such as loss of tourism or recreational fishing activity in response to the harmful water quality changes induced by stronger tropical cyclones or flood events should be anticipated. The potential impact on a broader suite of ecosystem services must also be considered, such as reduction in coastal protection, erosion control, water purification, recreational opportunities, carbon sequestration, and water purification (Barbier et al., 2011). These benefits can be significant, as pollution and poor water quality can reduce the value of trips to beaches and for recreational fishing (Freeman, 1995; Kaoru, 1995; Massey et al., 2006), and can go as far as impacting residential land prices (Leggett and Bockstael, 2000).

3.2. Effects of intensified and/or more frequent low freshwater inflow events

Drought coupled with burgeoning human population growth in coastal watersheds puts severe strain on freshwater supplies and greatly reduces freshwater inflows to estuaries, especially when coincident with seasonal peaks in human freshwater demand (e.g., Livingston et al., 1997; Meyer et al., 1999). Extreme low freshwater inflow events are becoming a global concern, with human activity (water diversion and usage) and climate change (drought) interacting to modify the global water cycle and river flows, thereby affecting delivery of freshwater to coastal regions (Meyer et al., 1999; Vorosmarty et al., 2000; Milliman et al., 2008; Palmer et al., 2008). Future projections of more intense drought due to climate change coupled with increasing human freshwater demand in many world regions magnify these concerns (Gibson et al., 2005; Flemer and Champ, 2006; Meehl et al., 2007; McDonald et al., 2011).

Freshwater contains nutrients and organic matter that, upon delivery to the coastal zone, fuels the rich productivity of coastal ecosystems and shapes critical fish habitat through effects on salinity gradients and stratification (e.g., Alber, 2002; Kimmerer, 2002; Nixon, 2003). Thus, extreme low freshwater inflow events have the potential to significantly alter estuarine ecosystem structure, function, and overall water quality (Livingston et al., 1997;

Attrill and Power, 2000b; Pollack et al., 2011; Wetz et al., 2011). In Texas, a natural climatic gradient exists where freshwater inflow decreases >50-fold moving south from the Louisiana to Mexico border (Montagna et al., 2007). Nitrogen loads decrease and salinity increases concurrent with the decreasing freshwater inflow, and likewise shellfish harvests decrease by 20-fold, though the mechanistic linkages between freshwater inflow and changes in shellfish harvest are not well understood. In North Carolina's Neuse River Estuary, nitrogen loads, phytoplankton productivity and zooplankton biomass were all sharply lower during drought conditions as opposed to non-drought conditions, adult fish mortality was higher by >2-order of magnitude, and commercial fish landings were lower during drought (Wetz et al., 2011; Wetz, unpubl. data). Similarly, during a prolonged drought in Apalachicola Bay, Florida, Livingston et al. (1997) observed significant reductions in the biomass of several higher trophic level functional guilds and speculated that this was due, in part, to reduced primary and secondary productivity and consequently food for the higher trophic levels. Livingston et al. (2000) noted a strong negative effect of drought on oysters, presumably due to higher salinities, increased disease and predation. After construction of the Aswan Dam on Egypt's Nile River, freshwater discharge to the Nile delta decreased by >90%, seasonal phytoplankton blooms no longer developed and fish and shrimp landings decreased by 50–100% relative to predam conditions (Nixon, 2003). In the US southwest, reductions of freshwater inflow to the Colorado River Delta have been linked to significant (~94%) reductions in bivalve densities compared to historical periods of natural river flow (Kowalewski et al., 2000). In Texas, a 10-year drought that began in the late 1940s also led to significant declines in estuarine oyster, shrimp and blue crab harvests (Copeland, 1966).

These findings highlight both the habitat and trophic importance of adequate freshwater inflow conditions. Ultimately, effects of reduced freshwater inflow on ecosystem goods and services may range from the obvious (i.e., altered nutrient cycling; lost recreational and commercial fishing and shellfishing opportunities) to more subtle (i.e., changes in coastal erosion and protection, Koch et al., 2009). Nonetheless, there is a clear inherent value in adequate freshwater inflow to estuaries, yet this continues to be overlooked in the growing policy debate over human versus ecosystem necessity for this vital resource (see e.g., Gillson, 2011).

4. Conclusions

ECE are not only major drivers of estuarine water quality and ecological dynamics, but also affect a number of important ecosystem goods and services derived from estuaries, such as fisheries production, recreational opportunities, erosion control, and carbon sequestration. Even considering uncertainties in projections of future ECE, contemporaneously they represent a naturally occurring aspect of the global climate system. Furthermore, the likelihood of reaching a global agreement in the near-term to curb CO_2 emissions and prevent projected changes in ECE frequency or intensity remains uncertain at best, and evidence has arisen suggesting that these changes may already be occurring. Thus resource managers may face a growing challenge of balancing the provisioning of important estuarine ecosystem goods and services with changes in ecosystem structure and function as a result of ECE and other co-occurring environmental changes. Proactive efforts for increasing the resiliency of estuaries to harmful effects of ECE will need to be conducted in the context of broader ecosystem-based management efforts which integrate, among other factors, the various stressors facing particular systems (i.e., climate change, eutrophication, freshwater withdrawals, sea level rise, etc.), man's needs from a goods and services standpoint that are derived from or affect functioning of a particular estuarine system, and societal actions that may arise in response to managed and unmanaged changes to estuarine ecosystems and their services (see e.g., Atkins et al., 2011; Fig. 1). Implicit in this is the necessity for understanding mechanistic linkages between ECE and estuarine water quality and ecosystem dynamics.

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