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Environmental links to interannual variability in shellfish toxicity in Cobscook Bay and eastern Maine, a strongly tidally mixed coastal region

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ABSTRACT

The Gulf of Maine experiences annual closures of shellfish harvesting due to the accumulation of toxins produced by dinoflagellates of the genus *Alexandrium*. Factors controlling the timing, location, and magnitude of these events in eastern Maine remain poorly understood. Previous work identified possible linkages between interannual variability of oceanographic variables and shellfish toxicity along the western Maine coastline but no such linkages were evident along the eastern Maine coast in the vicinity of Cobscook Bay, where strong tidal mixing tends to reduce seasonal variability in oceanographic properties. Using 21 years (1985–2005) of shellfish toxicity data, interannual variability in two metrics of annual toxicity, maximum magnitude and total annual toxicity, from stations in the Cobscook Bay region are examined for relationships to a suite of available environmental variables. Consistent with earlier work, no (or only weak) correlations were found between toxicity and oceanographic variables, even those very proximate to the stations such as local sea surface temperature. Similarly no correlations were evident between toxicity and air temperature, precipitation or relative humidity. The data suggest possible connections to local river discharge, but plausible mechanisms are not obvious. Correlations between toxicity and two variables indicative of local meteorological conditions, dew point and atmospheric pressure, both suggest a link between increased toxicity in these eastern Maine stations and weather conditions characterized by clearer skies/drier air (or less stormy/humid conditions). As no correlation of opposite sign was evident between toxicity and local precipitation, one plausible link is through light availability and its positive impact on phytoplankton production in this persistently foggy section of coast. These preliminary findings point to both the value of maintaining long-term shellfish toxicity sampling and a need for inclusion of weather variability in future modeling studies aimed at development of a more mechanistic understanding of factors controlling interannual differences in eastern Gulf of Maine shellfish toxicity.

1. Introduction

Harmful algal blooms (HABs) caused by dinoflagellates of the genus *Alexandrium* spp. necessitate annual shellfish bed closures along coastlines of the Gulf of Maine. *Alexandrium* spp. produce a number of toxins that are often referred to as saxitoxin equivalents (Anderson et al., 1994). Among these toxins are neurotoxins that if ingested in a large enough quantity can cause paralytic shellfish poisoning (PSP) in vertebrate consumers, including humans. Closures of shellfish beds due to PSP toxicity in Maine are patchy in both space and time throughout the spring, summer, and fall (Anderson, 1997). To protect human health, the Maine Department of Marine Resources (DMR) monitors shellfish each season throughout the Maine coast for toxicity levels (Bean et al., 2005). Samples are collected roughly weekly at approximately 100 stations along the coast, with expanded sampling efforts used to isolate events in time and space. Samples are processed using Association of Official Analytical Chemists procedures (AOAC) for mouse bioassay that determine a toxicity score (Hallegraeff et al., 1995). Scores approaching 80 μg STX equivalents 100 g^{-1} tissue, the quarantine toxicity level set by the World Health Organization, result in shellfish bed harvesting closures. Economic losses due to closures can be substantial, totaling ~$5 M in lost revenue for Maine businesses and $18 M in Massachusetts in 2005 alone (Jin et al., 2008).

Links between the space and time variability of *Alexandrium* cell distributions in the Gulf of Maine, environmental variability and coastal shellfish toxicity levels are the subject of ongoing research (Anderson, 1997; Anderson et al., 2005). Townsend et al. (2001, 2005) show that offshore cell distributions in the open Gulf of Maine have links to increased nutrient concentrations and light availability imposed by stratification, both modulated by overall

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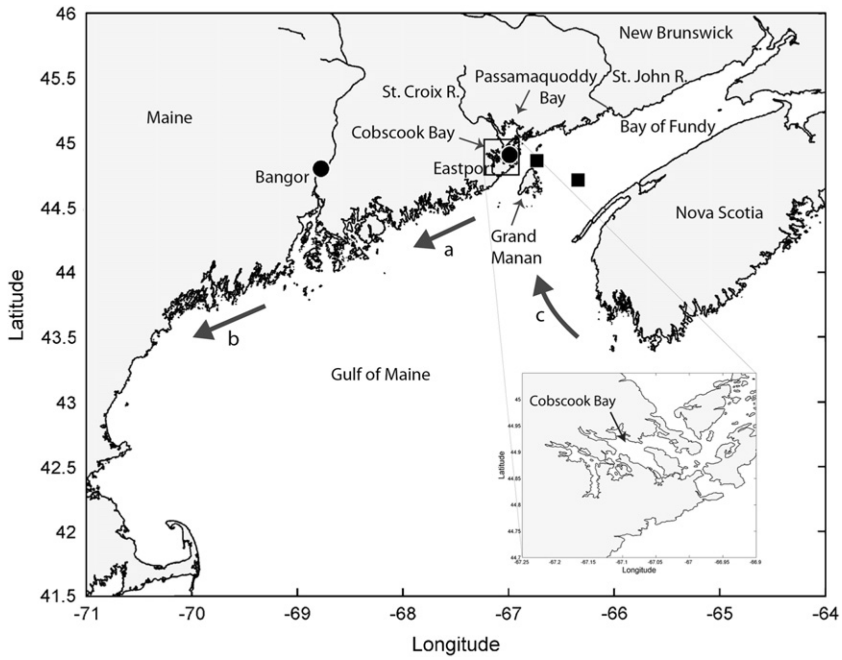

Fig. 1. The Gulf of Maine showing the location of Cobscook Bay, the St. Croix and St. John Rivers, other relevant geographic sites, the locations where surface temperature records (black squares) and meteorological records (black circles) are measured, and the direction of residual surface circulation in the coastal Gulf of Maine (a, Eastern Maine Coastal Current, b, Western Maine Coastal Current, and c, flow on the southern Scotian shelf). Inset shows detail of Cobscook Bay region. We subjectively identify regions in the vicinity of a, as eastern Maine and those in the vicinity of b, as western Maine.

circulation patterns. Overall cell distributions near shore appear to be strongly controlled by the Maine coastal currents (Keafer et al., 2005). Concentrations develop earliest in the season in warmer western portions of the Gulf and later in the east, with the possibility that surface distributions are initially fed by a series of cyst beds along the coast (McGillicuddy et al., 2005a) and then modulated by the hydrographic conditions of that year (McGillicuddy et al., 2011). Establishing links between interannual environmental variability and *Alexandrium* cell numbers/distribution is difficult due to the complex physical–biological interactions regulating these blooms and the relatively few large-scale surveys of cell counts. Furthermore, McGillicuddy et al. (2005b) show that overall offshore populations were relatively stable from year to year over the six years they examined (between 1993 and 2002), suggesting that interannual variability in coastal shellfish toxicity might largely be regulated by transport processes and/or local processes close to shore. Here we make use of over two decades of shellfish toxicity records at multiple coastal sites to investigate possible links between toxicity and environmental variability focusing on a region within which no previous links have been found.

Previous work shows strong ecological heterogeneity along the coast of Maine (Fig. 1) among a wide assortment of benthic marine distributions and processes (e.g. Hale, 2010) as well as HAB impacts (Hurst and Yentsch, 1981; Bean et al., 2005; Thomas et al., 2010), most likely linked to strong gradients in oceanographic conditions along the coast (e.g. Brooks, 1985; Pettigrew et al., 2005; Townsend et al., 2005). Superimposed on the residual cyclonic circulation of the gulf that creates southwestward flow along the Maine coast is tidal forcing that increases from the southwest to the northeast. Southwestern portions of the coast (Fig. 1) are therefore downstream of our study area, experience reduced tidal mixing and strong seasonal surface temperature cycles, with warm, stratified, nutrient deplete conditions and relatively weak flow of the Western Maine Coastal Current prevailing at the surface throughout the summer. Eastern portions of the coast are upstream, experience some of the strongest tidal mixing in the world, are subjected to relatively strong advective alongshore flow of the cold, well-mixed Eastern Maine Coastal Current (Fig. 1), and have a reduced surface seasonal temperature cycle, remaining relatively cold, nutrient replete, and well mixed throughout the year (Townsend et al., 2006).

Cobscook Bay is a highly denticulated inlet lying near the Canadian border at the eastern end of the Maine coast near the mouth of the Bay of Fundy (Fig. 1). With a mean tidal range of 5.7 m, tidal forcing is extremely strong in this region. Cobscook Bay has an average flushing time of about two days and all water entering the bay flows through a complex island archipelago and a narrow entrance channel and is therefore strongly mixed (Brooks et al., 1999). Water entering the bay is potentially influenced by a number of sources. Freshwater influences include drainage into Cobscook Bay itself which is relatively minor, discharge from the St. Croix River at the head of Passamaquoddy Bay (Fig. 1) and influences from the St. John River, located upstream of the region, the largest direct riverine freshwater input into the Gulf of Maine (Fig. 1). Oceanic water influencing Cobscook Bay and the nearby coast arrives from the mouth of the Bay of Fundy. These waters originate from the well-mixed southern Scotian Shelf (Fig. 1) and waters within the Bay of Fundy, and are influenced by Gulf of Maine water at the upstream portion of the Eastern Maine Coastal Current.

Toxicity levels in shellfish along the Maine coast are strongly seasonal, peaking in summer (Shumway et al., 1988) and not present (or minimal) in winter. Early work suggested the influence of oceanographic variables controlling coastal toxicity variability in western Gulf of Maine regions, including the hypothesis that coastal freshwater advection modulated by wind forcing might be related to toxicity along the western coast of Maine (Franks and Anderson, 1992a,b). Luerssen et al. (2005) suggested that alongshore flow structure evident in surface temperature patterns was linked to western coastal toxicity. A survey of 21 years of interannual variability of shellfish toxicity along the whole Maine coast (Thomas et al., 2010) showed strong and coherent geographic pattern among stations with similar interannual variability. Stations

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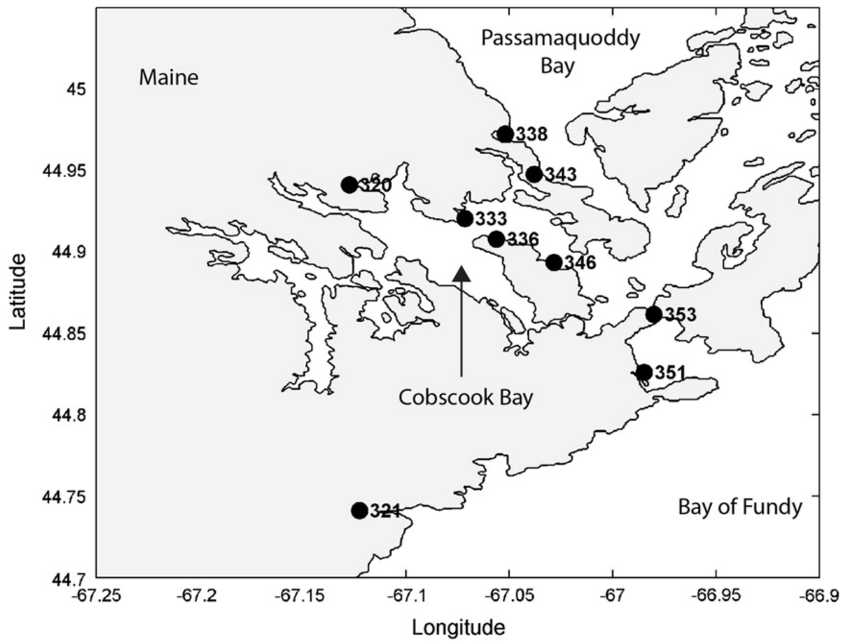

Fig. 2. The Cobscook Bay and eastern Maine study area showing toxicity sampling station locations, their reference numbers and local geographic names.

close in space often behaved similarly, with a clear separation between station groups in the colder eastern coast of Maine and those in the warmer west. Along the western, downstream portion of the Maine coast, Thomas et al. (2010) found links between toxicity and both sea surface temperature patterns and surface Ekman transport driven by wind stress. However, this work could not identify any oceanographic factors linked to coastal toxicity variability in regions along the colder, upstream eastern Maine coast, including stations in Cobscook Bay. Here we expand on the work reported in Thomas et al. (2010) in three ways. We focus specifically on toxicity variability at stations in the far eastern regions of the Gulf of Maine, especially Cobscook Bay (Fig. 2). We analyze interannual toxicity variability at individual stations rather than means among statistically clustered groups of stations. Lastly, we compare station variability to an expanded list of oceanographic and meteorological factors more local to these eastern Maine shellfish toxicity sampling sites.

2. Data and methods

Twenty-one years of Maine coastal shellfish toxicity data from 1985 to 2005 were obtained from the Maine Department of Marine Resources. The original data comprised over 64,000 records, measured in over 10 different species, with approximately 350 stations sampled at least once between 1985 and 2005. Of these stations, however, about 100 were sampled ~weekly in most years providing a relatively clear picture of seasonal patterns within each year. Nine of these stations are in the far eastern Maine–Cobscook Bay area (Fig. 2) with the others spread along the rest of the Maine coast. Toxicity is monitored between March 1 and November 1 of each year. To avoid biases based on species-specific rates of filtering, toxin uptake and depuration, we use only toxicity records from the genus *Mytilus*. These, however, account for approximately 58 percent of the records.

Following protocols outlined in Thomas et al. (2010) the toxicity time series at each station, in each year, is reduced to two metrics of interannual variability, maximum magnitude of toxicity and total annual toxicity. The toxicity time series at each station and year is first averaged, where necessary, to a single measurement for each week. Maximum magnitude of toxicity is calculated as the mean of the three highest weekly toxicity measurements at a station for a given year. We use three values to reduce biases created by single measurements. This metric was calculated only for stations and years with at least 10 toxicity measurements. Total annual toxicity is calculated as the sum of all toxicity values in a year, performed for stations and years with at least five toxicity measurements. Use of fewer valid measurements in each station/year allows more stations and years to enter the analysis. Although those stations/years with fewer samples have potentially low-biased metric values of total annual toxicity, the nature of Maine DMR sampling and coastal toxicity mitigates this bias. Specifically, stations in any year that persistently record zero (or very low) toxicity are sampled less frequently and represent locations and years of low toxicity anyway. Together, these metrics provide potentially related, but differing expressions of how toxic an individual station was in each year over our study period.

Daily river discharge data from the St. John (site 1014000) and St. Croix (site 1021000) Rivers (Fig. 1) are available from the United States Geological Survey over the study period. From these we calculated mean monthly discharge in each calendar month in each study year for each river.

Oceanographic data inside Cobscook Bay over the study period are not available. Sea surface temperatures in the vicinity of Cobscook Bay were subsampled from archived satellite data. Multiple daily satellite-measured sea surface temperature (SST) records of the Gulf of Maine region were received and processed by the Satellite Oceanography Laboratory at the University of Maine and produced into monthly mean images. Monthly data spanning the period 1985–2005 were subsampled as means over 25 km^2 at two stations, one located near the northern end of Grand Manan Island and the other over central western Bay of Fundy (Fig. 1). The spatial resolution of the satellite SST data is not sufficient to retrieve measurements inside Cobscook Bay, but strong tidal exchange and vertical mixing likely minimize differences, especially at the Grand Manan site.

Air temperature and precipitation records are from the Eastport International Airport (Fig. 1). Precipitation data are received as monthly (water equivalent) totals over the study period. Air temperatures are monthly means. Historical records over the study period of additional meteorological metrics are not readily available from Eastport. Relative humidity, atmospheric pressure, and dew point are available from the Bangor International Airport (Fig. 1) as daily averages over the study period, from which we calculated mean monthly values. Although further inland (~40 km)

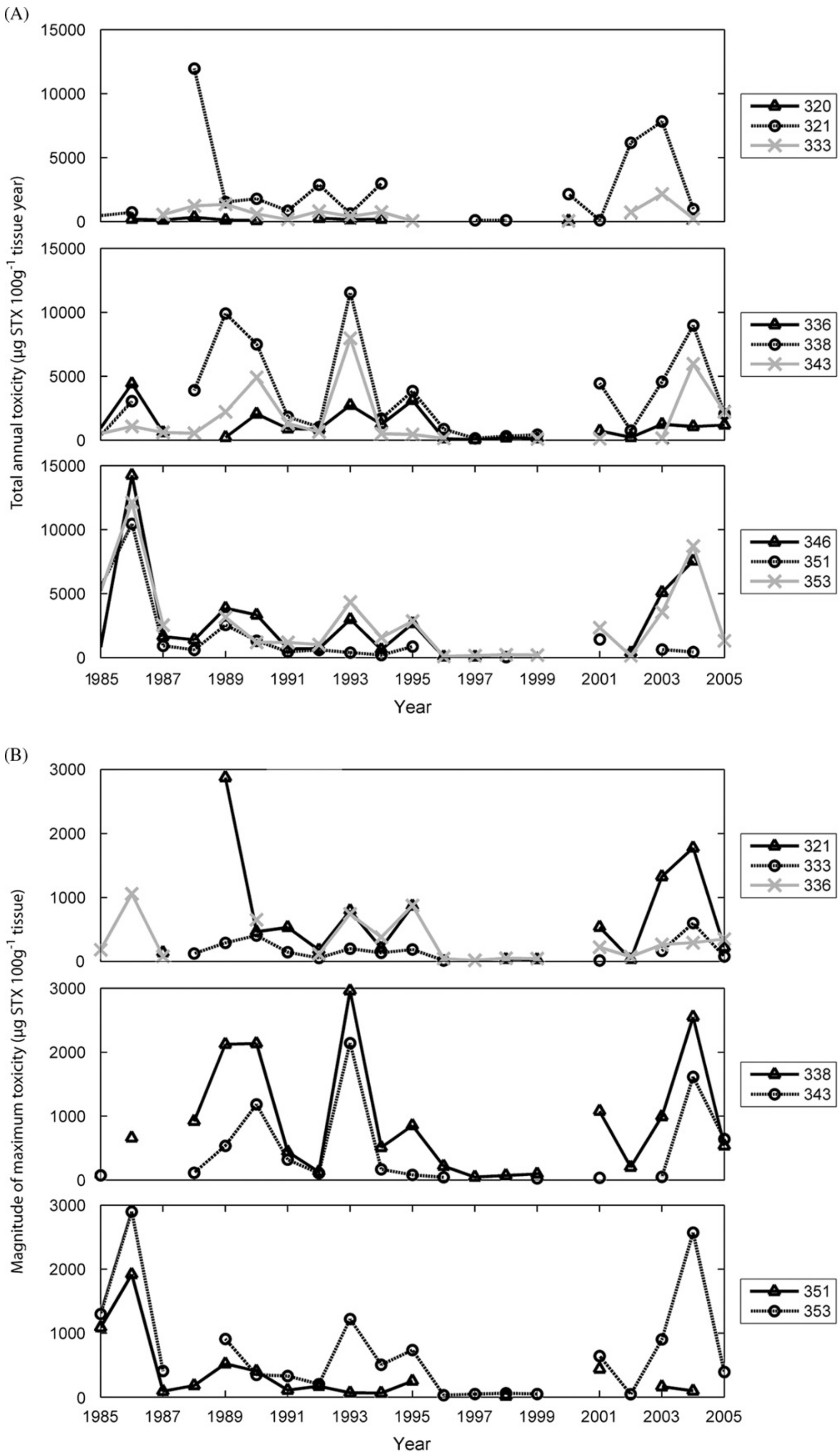

Fig. 3. Interannual variability of shellfish toxicity over the study period at each station for each of the two metrics, (a) total annual toxicity and (b) magnitude of maximum toxicity. Missing values are those years when insufficient samples were taken to form the metric.

Table 1
Correlations between environmental metrics and total annual toxicity.

Environmental metric	Station	Month					
		March	April	May	June	July	August
Sea surface temperature Grand Manan	320	−0.11	0.00	−0.21	−0.18	−0.04	0.36
	321	−0.24	−0.13	0.14	0.13	−0.06	0.11
	333	−0.29	−0.11	0.14	0.00	0.04	0.30
	336	−0.10	0.06	0.05	−0.18	0.11	0.38
	338	−0.42	−0.20	−0.05	−0.16	−0.22	0.21
	343	−0.26	−0.08	−0.14	−0.28	−0.25	0.10
	346	−0.07	0.10	0.27	−0.14	−0.05	0.01
	351	0.23	0.20	0.03	−0.18	0.00	−0.13
	353	−0.01	0.15	0.19	−0.22	−0.01	−0.13
Sea surface temperature Bay of Fundy	320	**0.49**	0.30	−0.25	−0.19	−0.02	0.26
	321	−0.05	−0.16	−0.12	−0.07	−0.19	−0.01
	333	0.01	−0.06	−0.14	−0.12	−0.05	0.33
	336	**0.38**	0.35	0.13	−0.16	0.23	0.36
	338	0.20	−0.10	−0.22	−0.31	−0.16	0.19
	343	0.29	0.09	−0.22	−0.36	−0.08	0.24
	346	−0.03	0.20	0.22	−0.14	−0.13	−0.05
	351	−0.04	0.20	0.23	−0.02	−0.04	−0.26
	353	0.08	0.31	0.19	−0.15	−0.03	−0.19
St. Croix river discharge	320	**−0.42**	0.04	0.22	0.13	−0.18	−0.19
	321	**−0.43**	−0.15	0.36	0.26	−0.11	0.09
	333	−0.28	−0.11	0.18	0.20	−0.21	0.09
	336	−0.30	−0.03	−0.13	0.05	−0.25	0.07
	338	**−0.47**	−0.04	0.23	0.22	−0.20	0.03
	343	−0.33	−0.15	0.13	0.17	−0.17	0.03
	346	−0.33	−0.16	−0.10	−0.01	−0.22	0.21
	351	−0.28	−0.36	−0.16	−0.12	−0.18	0.14
	353	**−0.47**	−0.25	−0.20	−0.16	−0.29	0.03
St. John river discharge	320	−0.19	−0.10	−0.09	0.27	**−0.41**	−0.29
	321	−0.36	−0.35	−0.04	0.11	−0.27	0.31
	333	−0.15	−0.27	−0.10	0.12	−0.24	**0.47**
	336	−0.09	−0.06	−0.17	0.28	−0.07	**0.40**
	338	−0.33	−0.29	−0.17	0.28	−0.30	0.32
	343	−0.20	−0.00	−0.20	0.29	−0.22	0.30
	346	−0.24	−0.15	−0.28	0.04	0.04	**0.76**
	351	−0.21	−0.17	−0.17	−0.01	0.09	**0.38**
	353	**−0.38**	−0.20	−0.29	0.11	0.07	**0.68**
Air temperature	320	−0.19	−0.19	0.16	0.16	−0.04	−0.08
	321	**−0.39**	**−0.47**	0.10	−0.05	0.01	−0.03
	333	−0.29	**−0.40**	−0.06	−0.26	0.23	0.22
	336	−0.12	0.07	−0.31	−0.09	−0.10	−0.23
	338	**−0.41**	−0.35	−0.00	−0.17	−0.08	0.10
	343	−0.34	−0.10	−0.11	−0.34	0.01	0.12
	346	−0.20	0.04	−0.21	**−0.40**	−0.21	**−0.42**
	351	−0.04	0.32	−0.06	−0.22	−0.29	**−0.62**
	353	−0.26	0.09	−0.15	**−0.41**	−0.21	**−0.47**
Relative humidity	320	0.07	0.29	−0.17	−0.28	−0.24	−0.24
	321	0.14	0.08	0.23	−0.24	−0.28	0.07
	333	−0.11	0.04	−0.01	**−0.38**	−0.14	0.14
	336	0.20	0.04	−0.10	−0.34	−0.14	**−0.41**
	338	−0.04	0.21	0.08	−0.33	−0.20	0.04
	343	−0.19	0.10	−0.17	−0.31	−0.21	−0.04
	346	0.22	0.21	0.11	−0.34	0.01	0.11
	351	−0.08	0.14	0.10	0.01	0.00	0.08
	353	0.06	0.03	0.05	−0.27	−0.00	0.00
Precipitation	320	−0.34	−0.21	**0.39**	0.32	0.10	−0.25
	321	−0.22	−0.27	**0.43**	0.21	0.05	0.10
	333	**−0.41**	−0.16	**0.41**	−0.00	0.32	0.30
	336	−0.04	−0.04	−0.06	0.01	**0.51**	0.11
	338	**−0.46**	−0.21	0.35	0.20	0.22	−0.00
	343	**−0.41**	−0.07	0.20	0.02	0.16	0.07
	346	−0.05	−0.02	0.07	−0.19	**0.44**	0.35
	351	−0.13	−0.10	−0.03	−0.25	0.28	0.26
	353	−0.13	−0.11	−0.09	−0.11	0.16	0.35
Dew point	320	−0.09	0.17	0.16	−0.06	0.03	−0.28
	321	−0.16	−0.22	0.32	−0.28	−0.36	0.02
	333	−0.15	−0.07	0.03	**−0.53**	−0.08	0.26
	336	0.08	0.21	−0.08	−0.22	0.03	−0.22
	338	−0.29	0.04	0.22	**−0.38**	−0.16	0.23
	343	−0.23	0.16	−0.08	**−0.45**	−0.16	0.16

Table 1 (*continued*)

Environmental metric	Station	Month					
		March	April	May	June	July	August
	346	*0.11*	*0.31*	*0.10*	**−0.54**	*−0.16*	*0.00*
	351	*−0.04*	*0.27*	*0.16*	*−0.16*	*−0.12*	*−0.12*
	353	*0.02*	*0.17*	*0.09*	**−0.47**	*−0.18*	*−0.12*
Atmospheric pressure	**320**	**0.47**	*0.15*	*−0.03*	*0.08*	**0.50**	*−0.14*
	321	**0.48**	*0.17*	*0.11*	*0.21*	*0.31*	*−0.30*
	333	**0.54**	*0.08*	*0.03*	*0.07*	***0.41***	*−0.11*
	336	*0.22*	*0.31*	*−0.02*	*−0.09*	*0.18*	*0.23*
	338	**0.54**	*0.19*	*0.04*	*0.09*	*0.31*	*−0.06*
	343	***0.43***	*0.20*	*0.01*	*0.07*	*0.25*	*0.16*
	346	***0.40***	*0.15*	*0.11*	*−0.14*	*0.24*	*0.00*
	351	*0.12*	*0.11*	*−0.03*	*−0.24*	*0.18*	*0.19*
	353	*0.28*	*0.21*	*0.07*	*−0.12*	*0.22*	*0.08*

Significance levels: **95**%, ***90***%, and *not significant*.

than Eastport and approximately 140 km away, over the monthly averaged time scales we deal with here, the Bangor records are likely at least indicative of coastal interannual variability in these additional meteorological conditions.

Spearman rank correlations are used to quantify relationships between concurrent interannual variability of the two toxicity metrics and the suite of monthly environmental metrics over the 21-year study period. Because we are unsure of lags in possible impact or affect, we examine correlations between interannual time series of each spring and summer month and the annual toxicity records. We use nonparametric statistics to avoid assumptions of the underlying distribution of the toxicity data, as these did not always pass tests for a normal distribution (Thomas et al., 2010). Correlations do not imply causation, nor do they necessarily indicate scientific significance. We use the correlations as exploratory tools for a region within which no previous relationships between toxicity and the environment were found. Stronger correlations suggest relationships that warrant further investigation.

Our use of multiple correlations to explore possible similarity between toxicity and environmental variables increases the probability of making a Type 1 statistical error. While adjusting for this using the Bonferroni correction is often attempted, reducing Type 1 errors, this approach significantly increases the probability of making Type 2 statistical errors (Nakagawa, 2004; Narum, 2006; Perneger, 1998). Other approaches are possible (Narum, 2006) and here we use a bootstrap technique modeled after Barton et al. (2003) that makes use of the availability of multiple geographic sampling sites, expected patterns, and that has been applied previously to toxicity data (Thomas et al., 2010). With many correlation calculations, chance will result in some values exceeding the statistical significance level (Type 1 errors). We can expect such occurrences to appear in our correlation matrix randomly if truly due to chance. However, if many significant correlations are apparent, and if these are grouped in a systematic way, we can estimate the probability that the observed number and grouping would occur by chance. When an apparently large number of significant correlations occur between station toxicities and an environmental variable, we make the assumption that any relationship between toxicity in one year and the same environmental variable value from other years should be strictly random. The environmental time series in question is randomized and its correlation to each station toxicity time series is calculated, for 1000 iterations. From these results, a ranked look-up-table of correlations values is created and the number of instances a significant relationship occurs is noted to estimate a 95% confidence level. The number of observed significant correlations in the actual data is compared to the number of instances of random correlations to estimate the probability that the observed number could occur by chance. We then address only those toxicity—environmental metric comparisons that produce a larger number of significant correlations than the bootstrap 95% significance value. While strictly a statistical view with no mechanistic insight, this approach provides some guidance whether any observed systematic patterns of significant correlations in a large table might be due to chance.

3. Results and discussion

Shellfish toxicity values in each year from the two metrics at each station are shown in Fig. 3. Total annual toxicity (Fig. 3a), varies between 0 and $>14{,}000\ \mu g$ STX $100\ g^{-1}$ tissue over the study period across the 9 stations, with Station 320 showing the lowest overall values. In general, total annual toxicity appears relatively high through the late 1980s and early 1990s, low or zero in the late 1990s, and then increases again at the end of the study period, a pattern consistent with that shown by Thomas et al. (2010). The magnitude of maximum toxicity (Fig. 3b) at the 7 stations varies from 0 to $\sim 3000\ \mu g$ STX $100\ g^{-1}$ tissue, lowest overall at Stations 336 and 351. Superimposed on the differences between stations is a similar overall interannual pattern as that of total toxicity, with reduced values in the late 1990s.

Over the months examined, neither of the SST time series have a strong relationship with either metric of interannual toxicity (Tables 1 and 2). Similarly, air temperature, relative humidity and precipitation showed no, or only isolated, relationships with the two toxicity metrics (Tables 1 and 2) over the interannual time series. Within these environmental variables, a few stations show significant correlations in some months, but do not pass our bootstrap test of pattern significance. Here we seek evidence of systematic correlations across many stations. Overall, these results are consistent with those of Thomas et al. (2010) who found relationships between oceanographic signals and coastal toxicity at stations along the western Maine coast, but not between the mean toxicity variability of statistically grouped eastern Maine stations and central Gulf of Maine SST and patterns derived from SST. Here we show similar results but tested against sea surface temperature locations more proximate to the eastern stations and tested against toxicity records from each individual station. One possible mechanism explaining this lack of relationships at eastern Maine stations is the strong and annually recurrent tidal mixing that would reduce the expression of any interannual SST variability in the vicinity of Cobscook Bay.

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Table 2
Correlations between environmental metrics and magnitude of maximum annual toxicity.

Environmental metric	Station	Month					
		March	April	May	June	July	August
Sea surface temperature Grand Manan	**321**	−0.24	−0.14	0.11	0.12	−0.05	0.16
	333	−0.28	−0.09	0.12	0.02	0.06	0.32
	336	−0.15	0.07	0.05	−0.17	0.11	0.45
	338	−0.41	−0.15	−0.04	−0.17	−0.21	0.21
	343	−0.27	−0.10	−0.12	−0.25	−0.22	0.09
	351	0.17	0.15	0.01	−0.15	0.02	−0.07
	353	−0.03	0.13	0.20	−0.21	−0.00	−0.12
Sea surface temperature Bay of Fundy	**321**	−0.00	−0.15	−0.16	−0.09	−0.17	0.04
	333	−0.01	−0.04	−0.14	−0.10	−0.03	0.36
	336	***0.41***	0.36	0.10	−0.19	0.25	***0.42***
	338	0.19	−0.07	−0.23	−0.33	−0.15	0.22
	343	0.30	0.05	−0.23	−0.35	−0.05	0.25
	351	−0.07	0.14	−0.21	0.00	−0.02	−0.22
	353	0.08	0.27	−0.16	−0.16	−0.02	−0.16
St. Croix river discharge	**321**	−***0.43***	−0.14	***0.37***	0.26	−0.11	0.08
	333	−0.25	−0.09	0.18	0.22	−0.21	0.10
	336	−0.25	−0.00	−0.07	0.11	−0.25	−0.00
	338	−**0.43**	−0.00	0.20	0.20	−0.21	0.01
	343	−0.30	−0.18	0.16	0.18	−0.15	0.00
	351	−0.29	−***0.41***	−0.16	−0.10	−0.20	0.14
	353	−**0.47**	−0.24	−0.15	−0.14	−0.29	0.02
St. John river discharge	**321**	−0.33	−0.35	−0.03	0.12	−0.31	0.25
	333	−0.10	−0.26	−0.08	0.15	−0.20	**0.48**
	336	−0.02	−0.11	−0.14	0.36	−0.06	0.35
	338	−0.29	−0.25	−0.17	0.29	−0.28	0.34
	343	−0.20	−0.00	−0.15	0.31	−0.22	0.26
	351	−0.20	−0.23	−0.15	−0.00	0.07	***0.38***
	353	−***0.40***	−0.23	−0.26	0.15	0.06	**0.69**
Air temperature	**321**	−***0.37***	−**0.47**	0.12	−0.01	0.04	0.01
	333	−0.28	−0.37	−0.10	−0.27	0.27	0.24
	336	−0.13	0.04	−***0.41***	−0.11	−0.04	−0.12
	338	−***0.38***	−0.32	−0.04	−0.21	−0.02	0.17
	343	−0.34	−0.14	−0.10	−0.31	0.05	0.20
	351	−0.05	0.27	−0.05	−0.18	−0.28	−**0.56**
	353	−0.28	0.05	−0.15	−***0.42***	−0.16	−***0.40***
Relative humidity	**321**	0.13	0.07	0.20	−0.25	−0.31	0.04
	333	−0.13	0.03	−0.04	−0.36	−0.12	0.14
	336	0.21	0.05	0.00	−0.26	−0.04	−***0.41***
	338	−0.05	0.19	0.04	−0.33	−0.16	0.05
	343	−0.20	0.06	−0.15	−0.30	−0.22	−0.04
	351	−0.12	0.13	0.12	0.02	0.00	0.09
	353	0.04	0.01	0.07	−0.30	−0.02	0.03
Precipitation	**321**	−0.25	−0.28	**0.44**	0.22	0.07	0.08
	333	−***0.41***	−0.15	***0.42***	−0.02	0.34	0.29
	336	−0.06	−0.01	0.01	0.07	**0.49**	−0.04
	338	−**0.44**	−0.18	0.34	0.16	0.22	0.01
	343	−***0.39***	−0.05	0.19	0.05	0.10	0.04
	351	−0.19	−0.15	−0.01	−0.22	0.32	0.26
	353	−0.16	−0.10	−0.05	−0.10	0.13	0.36
Dew point	**321**	−0.16	−0.23	0.32	−0.26	−0.34	0.02
	333	−0.15	−0.05	−0.03	−**0.51**	−0.05	0.26
	336	0.07	0.20	−0.08	−0.13	0.17	−0.19
	338	−0.26	0.07	0.16	−***0.40***	−0.11	0.27
	343	−0.25	0.10	−0.07	−***0.42***	−0.14	0.19
	351	−0.09	0.23	0.18	−0.13	−0.11	−0.06
	353	−0.01	0.12	0.09	−**0.50**	−0.17	−0.07
Atmospheric pressure	**321**	**0.49**	0.17	0.08	0.23	0.32	−0.29
	333	**0.52**	0.08	0.02	0.07	***0.42***	−0.07
	336	0.27	0.23	−0.11	−0.03	0.19	0.23
	338	**0.54**	0.19	0.01	0.09	0.31	−0.04
	343	***0.38***	0.17	−0.02	0.10	0.21	0.14
	351	0.13	0.12	−0.03	−0.25	0.19	0.20
	353	0.30	0.19	0.03	−0.08	0.23	0.06

Significance levels: **95**%, ***90***%, and *not significant*.

Station toxicity correlations with interannual variability in river discharge during one of the months from both the St. John River and the St. Croix River (Tables 1 and 2) rise above the bootstrap significance level. For March St. Croix River discharge (Fig. 4), four (three) stations have a negative relationship with total annual (maximum magnitude of) toxicity. Discharge in

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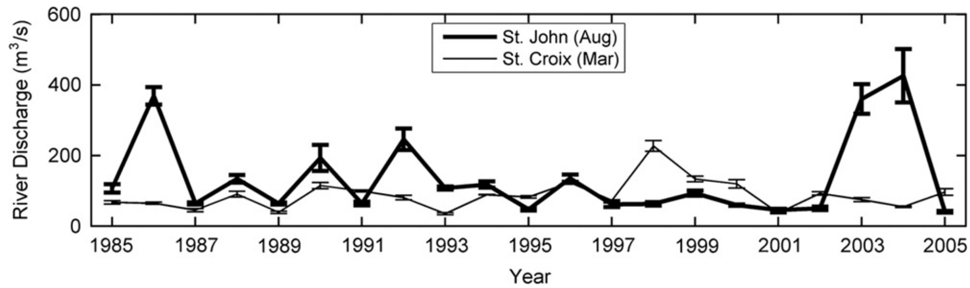

Fig. 4. Interannual variability of monthly mean river discharge for the St. Croix (March) and St. John (August) rivers. Error bars are standard errors of the daily means.

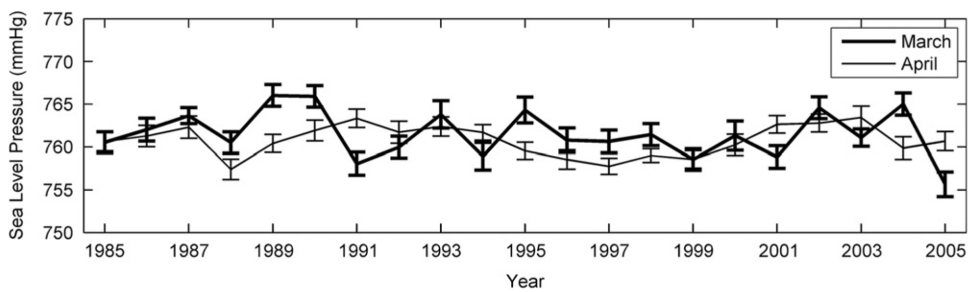

Fig. 5. Interannual variability in monthly mean early spring atmospheric pressure for March, for which there is a statistical correlation to toxicity, and April, an example of a month in which no correlation was evident. Error bars are standard errors of the daily means.

March, however, averages a third less than that which occurs during the annual peak of April, the month of strongest interannual variability. We note that correlations of both toxicity metrics with April (or May, June, July) are very weak (and do not pass our bootstrap significant test). If St. Croix River discharge plays a role in interannual variability of toxicity in the region, we might expect a link to April discharge. So while the data suggest a possible connection to seasonal toxicity levels, a mechanism is not readily apparent. Interannual variability in August St. John River discharge (Fig. 4) is positively related to five (three) stations for total annual (maximum magnitude of) toxicity (Tables 1 and 2). However, August discharge is at its seasonal minimum, only ~15% of the spring discharge peak, making relatively small (in the seasonal context) interannual changes in discharge volume appear important. No (or only weak) correlation is seen for July (or other months). We also note that most toxicity events in the region occur in mid-summer, prior to possible impacts of changes in August discharge. Stations that do show correlation to these rivers are a mix of those most proximate and most distant from possible river influence. We therefore regard the correlations of Cobscook Bay region toxicity with local river discharge as inconclusive, but leave open the possibility of a relationship that warrants further investigation. These results expand on those of Thomas et al. (2010) who used Penobscot River discharge, whose mouth is downstream from this region, as indicative of overall Gulf of Maine river input. They found no relationship between this river input and interannual variability in regionally averaged toxicity metrics.

Monthly mean atmospheric pressure in early spring (March, Fig. 5) has systematic positive correlations with both total annual toxicity (6 stations) and the magnitude of maximum toxicity (4 stations) in our study area (Tables 1 and 2, Fig. 6), a pattern well above the significance threshold of the bootstrap test. The March atmospheric pressure time series shows maximum values in 1989 and 1990, minimums in 1991, 1994 and 2005; a pattern different from that of April (Fig. 5) that does not show correlations. Within the seasonal cycle, March is earlier than the normal June onset of toxicity (Thomas et al., 2010) and so a direct mechanistic link is tenuous. However, atmospheric pressure is indicative of overall weather conditions, and monthly means characterize interannual differences in early spring conditions. Lower atmospheric pressure generally indicates increased storm activity and overcast weather conditions while higher atmospheric pressure indicates fewer storms and clearer skies. The sign of the correlation suggests that years of higher spring atmospheric pressure (or fewer stormy days) are associated with years of increased toxicity.

Mean early summer (June) dew point over the study period (Fig. 7) has a negative correlation with toxicity in the vicinity of Cobscook Bay (Tables 1 and 2, Fig. 8). Dew point is a measure of the moisture content of air and a more consistent metric of overall atmospheric moisture content than temperature-dependent relative humidity. Lower dew point indicates a drier air mass with poorer cloud and/or fog forming potential. The negative correlation indicates that on interannual scales, increased early summer monthly mean dew point (greater air moisture) is associated with decreased toxicity.

We examine whether the relationships shown above are unique to the Cobscook Bay and eastern Maine study region or applicable along the whole coast of Maine into regions where Thomas et al. (2010) show toxicity variability has links to onshore transport by wind forcing and its associated sea surface temperature patterns. Relationships between the two meteorological factors and toxicity metrics at all stations along the Maine coast (Figs. 9 and 10) show that correlations are weak and/or non-significant along the rest of the coast, strongest along the eastern coast, and concentrated in our study area. The pattern of these results is further evidence, but not proof, that the strong correlations in the Cobscook Bay region are not a statistical artifact. These results are also indicative of toxicity signals in different regions responding to differing environmental factors, with a similar disconnect between interannual toxicity patterns at stations in far eastern regions and those along the western Maine coast as that discussed in Thomas et al. (2010). One possibility is that this isolation of eastern regions, independence of toxicity variability and the resulting differences in environmental linkages result from the strongly differing hydrographic conditions due to

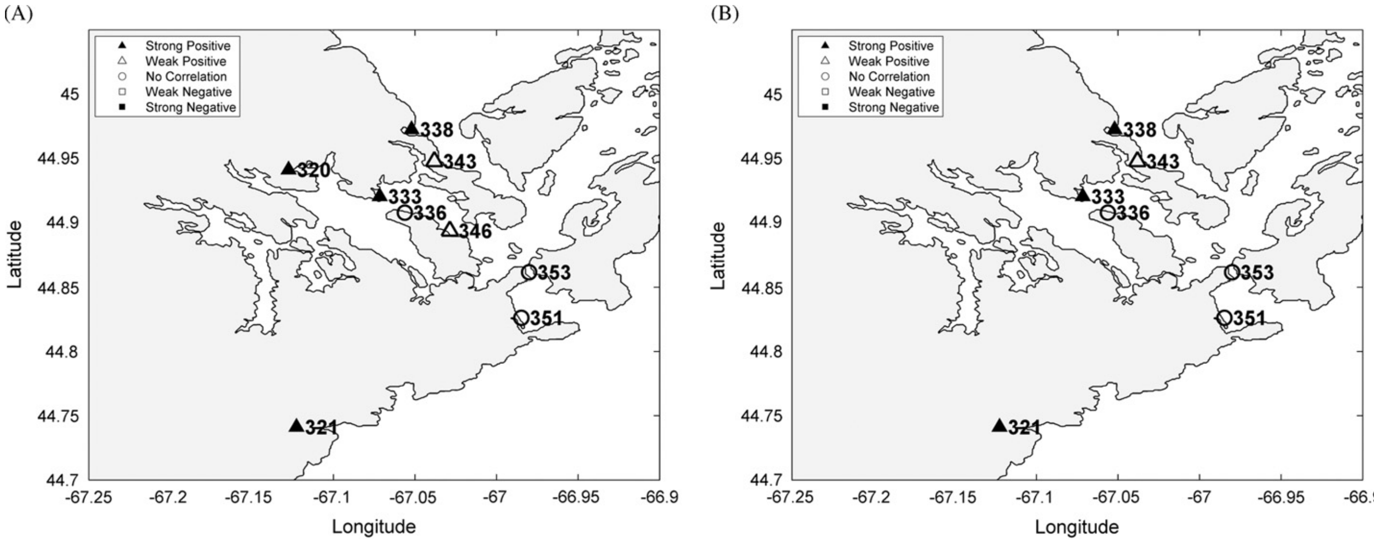

Fig. 6. Cobscook Bay area showing station-specific correlations between early spring (March) atmospheric pressure and (a) total annual toxicity and (b) magnitude of maximum toxicity. Strong correlations are significant at the 95% level, weak correlations are significant at the 90% level.

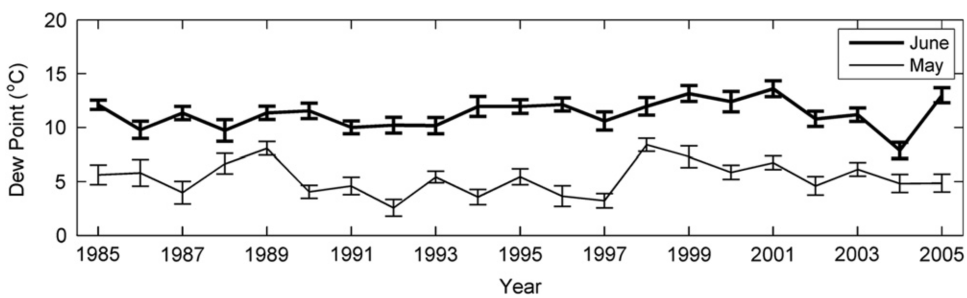

Fig. 7. Interannual variability in monthly mean summer dew point for June, for which there is a statistical correlation to toxicity, and May, an example of a month in which no correlation was evident. Error bars are standard errors of the daily means.

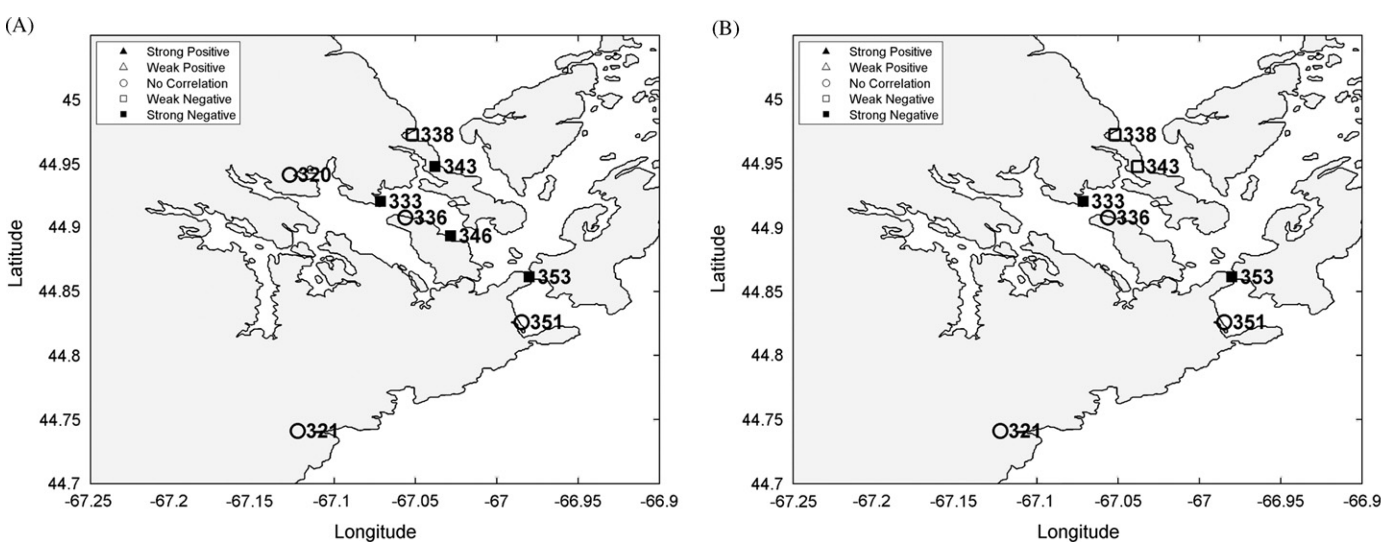

Fig. 8. Cobscook Bay area showing station-specific correlations between mean June dew point and (a) total annual toxicity and (b) magnitude of maximum toxicity. Strong and weak correlations are as per Fig. 6.

tidal mixing. These appear to override any down-stream coastal toxicity correlation that might have been expected due to along-shore transport by Maine coastal currents.

Both meteorological metrics with correlations in our study area suggest increased atmospheric water content and increased cloudy/stormy weather are associated with a decrease in toxicity. Links between interannual variability in meteorological conditions and ocean biological parameters in the study region are becoming increasingly well documented. Numerous studies now link wind forcing to coastal toxicity in the western Gulf of Maine (e.g. Franks and Anderson, 1992a,b; Keafer et al., 2005; Thomas et al., 2010) and monthly mean 700 mb pressure, a metric of local weather patterns, emerges as the strongest contributor to the leading mode of atmospheric variability influencing lobster larval settlement in the

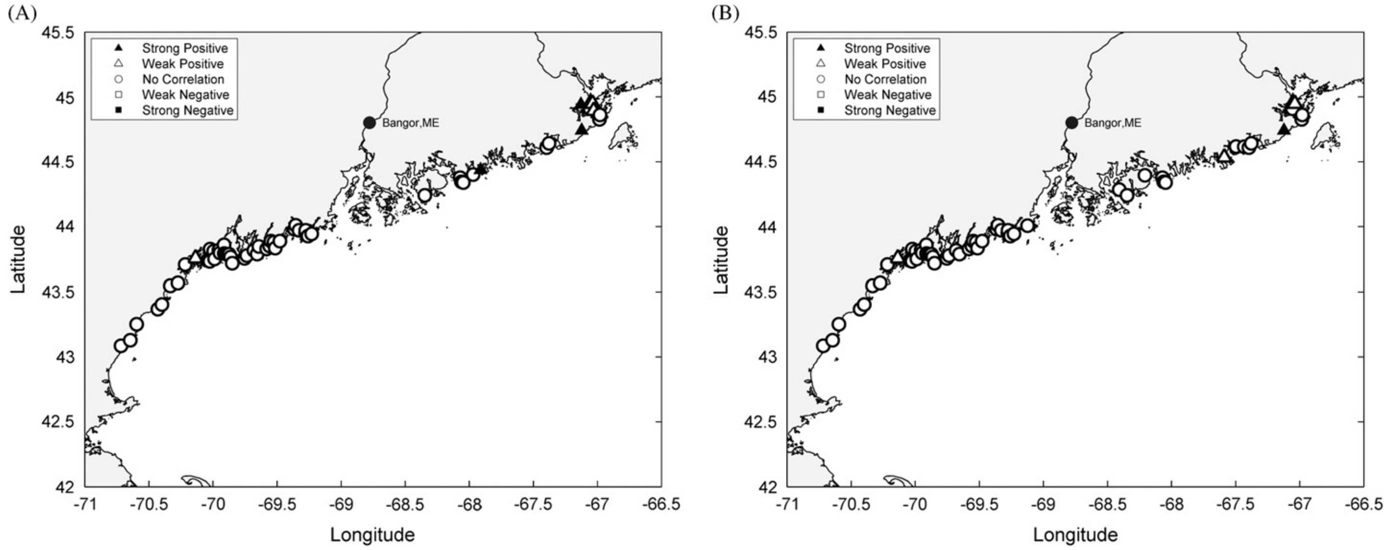

Fig. 9. Correlations of early spring (March) mean atmospheric pressure and (a) total annual toxicity and (b) magnitude of maximum toxicity at stations along the whole Maine coast. Strong and weak correlations are as per Fig. 6.

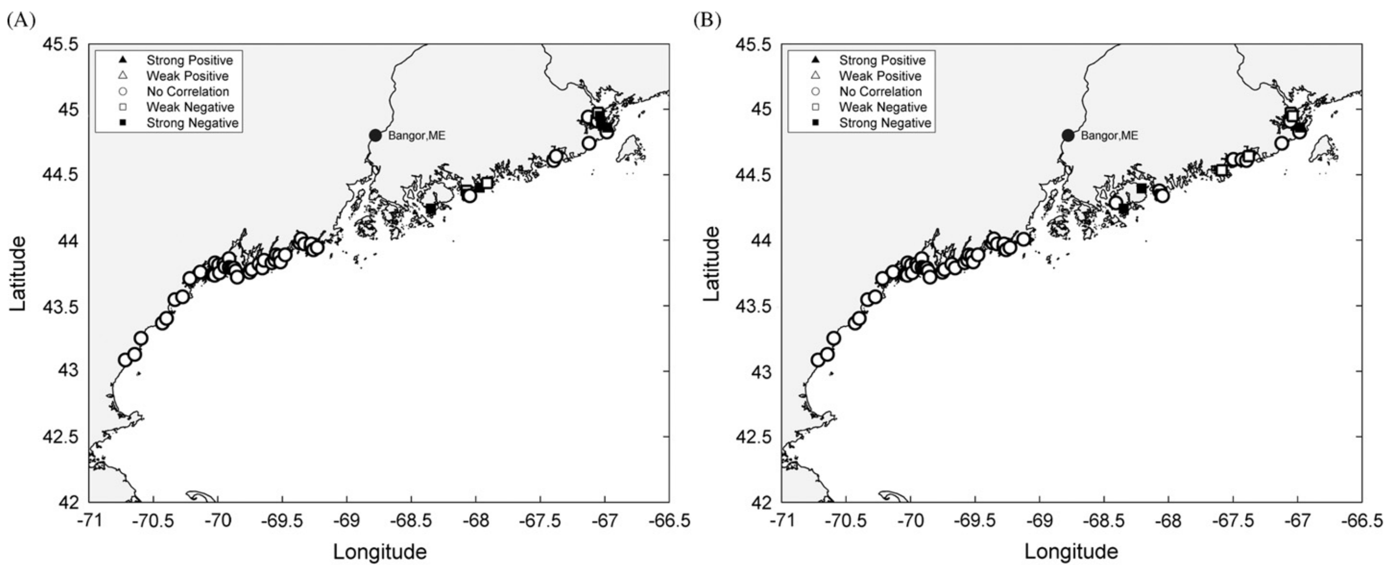

Fig. 10. Correlations of mean June dew point and (a) total annual toxicity and (b) magnitude of maximum toxicity at stations along the whole Maine coast. Strong and weak correlations are as per Fig. 6.

Gulf of Maine (Pershing et al., 2012). The data presented here are not capable of inferring causality and the results are strictly correlational, however, plausible mechanisms can be suggested. Linkages might be between the meteorological conditions and (a) the shellfish themselves (e.g. feeding and or depuration behavior), (b) the toxicity of cells within relatively constant cell populations (McGillicuddy et al., 2005b), or (c) with actual cell numbers of Alexandrium. Given the strong tidal forcing of the region and complex coastline morphology, we suggest it is less likely that the meteorological conditions change coastal oceanographic properties (e.g. currents, vertical structure, horizontal transport) significantly. Years of reduced clouds and/or fog would increase light availability, possibly increasing coastal ocean phytoplankton production in well-mixed and more nutrient replete regions such as the eastern Maine coast. Increased phytoplankton productivity could include *Alexandrium*, subjecting shellfish to increased HAB concentrations. Another possible mechanism suggested by the observed relationships is that increased sunlight may affect the toxicity of individual cells. Previous work has shown a link between the toxicity of *Alexandrium* cells and irradiance that may contribute to interannual variability (Etheridge and Roesler, 2005). Toxin concentrations within *Alexandrium* cells also vary among species and with physiological conditions (Anderson et al., 1994). Environmental conditions may allow for variability of dominant strains and/or species within *Alexandrium* populations affecting the toxicity values.

4. Conclusions

Previous work examining links between interannual variability in Maine coastal shellfish toxicity and environmental variability found no relationships between regionally averaged Cobscook Bay area measurements and Gulf of Maine oceanographic metrics. Our results confirm these findings, even after treating individual site locations within the Cobscook Bay area and examining very local oceanographic measurements. However, two metrics indicative of meteorological conditions are correlated with the 21 years of shellfish toxicity. Correlations between toxicity and atmospheric pressure and dew point both suggest that clearer and drier weather is associated with higher toxicity levels in the Cobscook Bay area. Such correlations were not evident at stations further west along the Maine coast where previous work has shown toxicity interannual variability has closer links to Gulf of Maine oceanographic variables. We suggest that the strong tidal mixing in the eastern Maine–Cobscook Bay region reduces interannual variability of most oceanographic signals. Our results found no parallel correlations between shellfish toxicity and local precipitation or air temperatures, suggesting that possible linkages to meteorological conditions may be through light availability. This region is characterized by recurrent spring/summer fog, a phenomenon potentially reduced in years of higher atmospheric pressure and lower air moisture content. Identifying mechanisms of interaction are beyond the scope of the present work, but in this nutrient replete region, increased light availability is consistent with increased phytoplankton productivity. The data also indicate that local river runoff may play a role in controlling toxicity variability, but these results are not as strong as those for the meteorological metrics, suggesting further investigation is warranted. Results reported here support the continued monitoring and archiving of shellfish toxicity scores from a diversity of regions along the Maine coast. It is evident that strong geographic differences in both oceanographic interannual variability and the interaction of *Alexandrium*-induced shellfish toxicity with environmental conditions make the application of models developed in only one region unlikely to be broadly applicable to such a heterogeneous coastline. More importantly, factors beyond ocean conditions, such as local weather, likely need to be taken into account in future studies focused on assisting management forecasts.

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