

Relationships of environmental factors to toxic dinoflagellate blooms in the Bay of Fundy

Alan W. White

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Shellfish toxicity data for the Bay of Fundy extend back to 1944, representing the longest time series of this kind in the world. The data reflect the abundance of the toxic dinoflagellate *Gonyaulax excavata* in the water and thus the pattern of its annual blooms over the years. Relationships between shellfish toxicity and environmental data over the 40-year period are being examined. Results at this stage show some significant correlations between environmental factors during pre-bloom months and summer toxicity. Correlation with the 18.6-year cycle of lunar modulation of the tide is of particular interest. Pre-bloom environmental variables account for 28-97% of the variation in summer toxicity, but only a few of the correlations are statistically significant ($P < 0.05$).

La collecte des données sur la toxicité des mollusques dans la baie de Fundy se fait depuis 1944; il n'y a pas, dans le monde, de manifestation pour laquelle des données ont été recueillies pendant si longtemps. Les données reflètent l'abondance du dinoflagellé toxique *Gonyaulax excavata* dans l'eau et, en conséquence, la distribution des floraisons annuelles au cours des années. Les rapports entre la toxicité des mollusques et les données de l'environnement recueillies pendant 40 ans sont étudiés; il s'en dégage certaines corrélations importantes entre les facteurs de l'environnement au cours des mois précédant la floraison et la toxicité pendant l'été. A ce propos, il convient de signaler l'influence du cycle de 18.6 ans de modulation des marées par la lune. Les variables de l'environnement avant la floraison influent, dans une proportion de 28 à 97 pour cent, sur la variation de la toxicité pendant l'été, mais seules quelques-unes des corrélations sont significatives sur le plan statistique ($P < 0.05$).

Alan W. White: Fisheries and Environmental Sciences, Fisheries Research Branch, Department of Fisheries and Oceans, Biological Station, St. Andrews, New Brunswick, EOG 2X0 Canada. Present address: Sea Grant Program, Woods Hole Oceanographic Institution, Woods Hole, Massachusetts 02543, USA.

Introduction

For many years toxic *Gonyaulax* blooms and shellfish toxicity have been annual events in the Bay of Fundy (Needler, 1949; Prakash *et al.*, 1971; White, 1982). In recent years the blooms have intensified and caused fish kills as well as shellfish toxicity (White, 1977, 1980, 1982, 1984). There is good agreement between the timing and intensity of summer shellfish toxicity in the Bay of Fundy and the number of *Gonyaulax* cells in the plankton community. Thus shellfish toxicity data closely reflect the status of *Gonyaulax* blooms. Shellfish serve as indicators in time and space of *Gonyaulax* populations.

Since 1944, shellfish toxicity has been monitored continually at several stations in the southwestern Bay of

Fundy, and at others since 1959. Shellfish toxicity and environmental data have been consolidated and statistical methods are now being used to examine the relationships between the toxic blooms and environmental factors. Initial results are reported here.

Materials and methods

A shellfish toxicity surveillance programme was established in the Bay of Fundy in 1943, and toxicity has been determined by mouse bioassay (AOAC, 1980) with only minor modification over the years. Data are available in reports issued annually by the Fish Inspection Laboratory, Department of Fisheries and Oceans, Black's Harbour, New Brunswick. Until 1959, toxicity was ex-

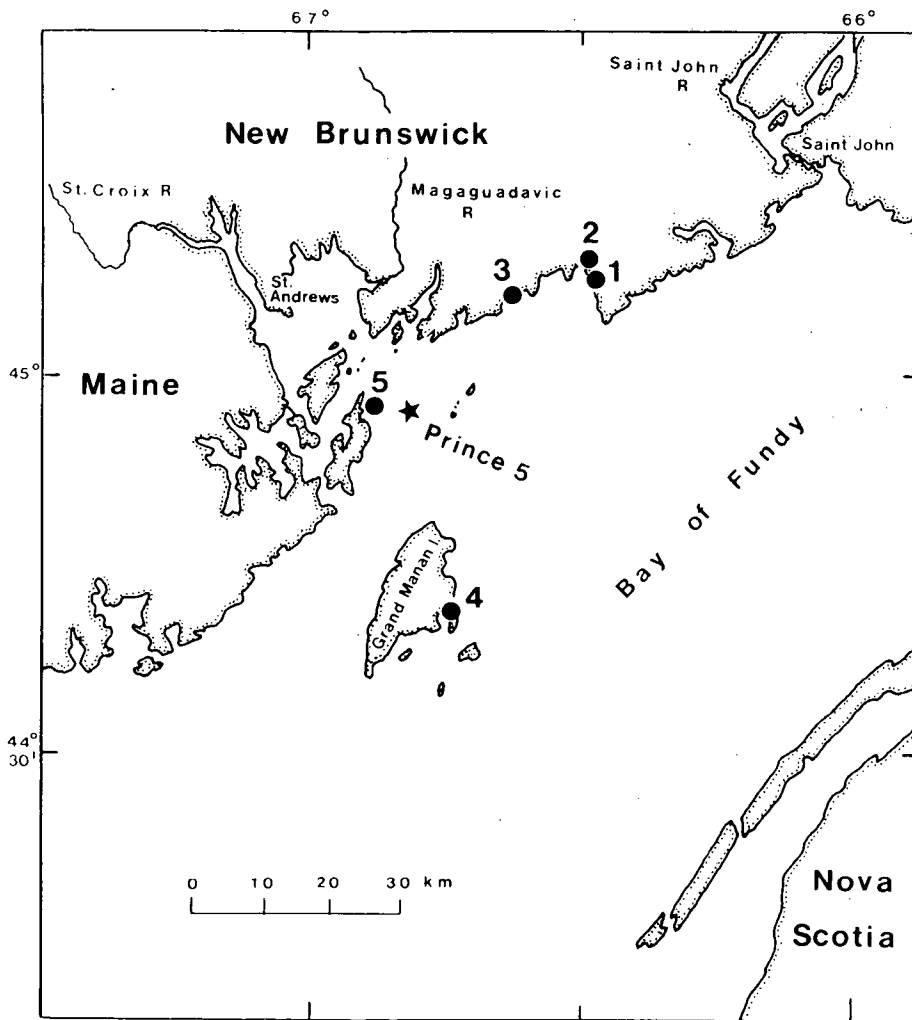


Figure 1. Locations of the five shellfish sampling stations: 1) Lepreau Basin, 2) Lepreau Harbour, 3) Crow Harbour, 4) Ross Island Thoroughfare, 5) Head Harbour. Temperature and salinity data are from station Prince 5. River discharge data are from the three rivers shown.

pressed as "mouse units". For uniformity with later values, the early data were converted to "micrograms of saxitoxin equivalent" using the factor 0.16.

Long-term toxicity records are available for soft-shell clams (*Mya arenaria* L.) from five stations, for blue mussels (*Mytilus edulis* L.) from two stations, and for horse mussels (*Modiolus modiolus* L.) from one station. Station locations are shown in Figure 1. Mean monthly toxicity values (January 1944–October 1983) were calculated for each set of toxicity data and were entered in a data file along with environmental variables. Environmental data included surface water temperature and salinity, sunshine duration, wind speed, river discharge, and the nodal modulation factor. Unfortunately, long-term nutrient data are not available for the Bay of Fundy.

The variables discussed in this report are defined below (see Fig. 1 for locations):

LBM, Lepreau Basin mussel toxicity; LBC, Lepreau Basin clam toxicity; LHC, Lepreau Harbour clam toxicity; CHC, Crow Harbour clam toxicity; RIHM, Ross Island Thoroughfare horse mussel toxicity; RIC, Ross Island Thoroughfare clam toxicity; HHM, Head Harbour mussel toxicity; T, surface water temperature at station Prince 5 (mid-month); S, surface salinity at station Prince 5 (mid-month); SUN, total monthly hours of sunshine at Saint John Airport; SPEED, mean monthly wind speed at Saint John Airport; MAG, mean monthly discharge from Magaguadavic River; CROIX, mean monthly discharge from St. Croix River; RSJOHN, mean monthly dis-

Table 1. Simple correlation coefficients between toxicities; n = 121-339.

	LBM	LBC	LHC	CHC	RIHM	RIC	HHM
LBM	-	0.877	0.768	0.889	0.551	0.640	-
LBC	-	-	0.750	0.806	0.433	0.531	0.394
LHC	-	-	-	0.736	0.512	0.640	0.367
CHC	-	-	-	-	0.667	0.801	-
RIHM	-	-	-	-	-	0.764	-
RIC	-	-	-	-	-	-	-
HHM	-	-	-	-	-	-	-

charge from Saint John River; NODMOD, inverse of the nodal modulation factor.

Table 2. Values of R² for multiple correlations among toxicities.

Variables = LBM LBC LHC CHC RIHM RIC (n = 123; 1959-1983)

LBM	0.879
LBC	0.811
LHC	0.680
CHC	0.909
RIHM	0.665
RIC	0.805

Variables = LBC LHC HHM (n = 115; 1944-1958)

LBC	0.858
LHC	0.854
HHM	0.156

Results and discussion

There is overall similarity in the toxicity patterns over the years among the toxicity data sets, as indicated by high correlation coefficients between them (Table 1) and by high coefficients of determination for multiple correlations (Table 2). The similarity can be seen in Figures 2 and 3 which depict Lepreau Basin mussel and clam toxicities (graphs for the other toxicity data are available in White, 1982). It is apparent that shellfish toxicity in the southwestern Bay of Fundy behaves

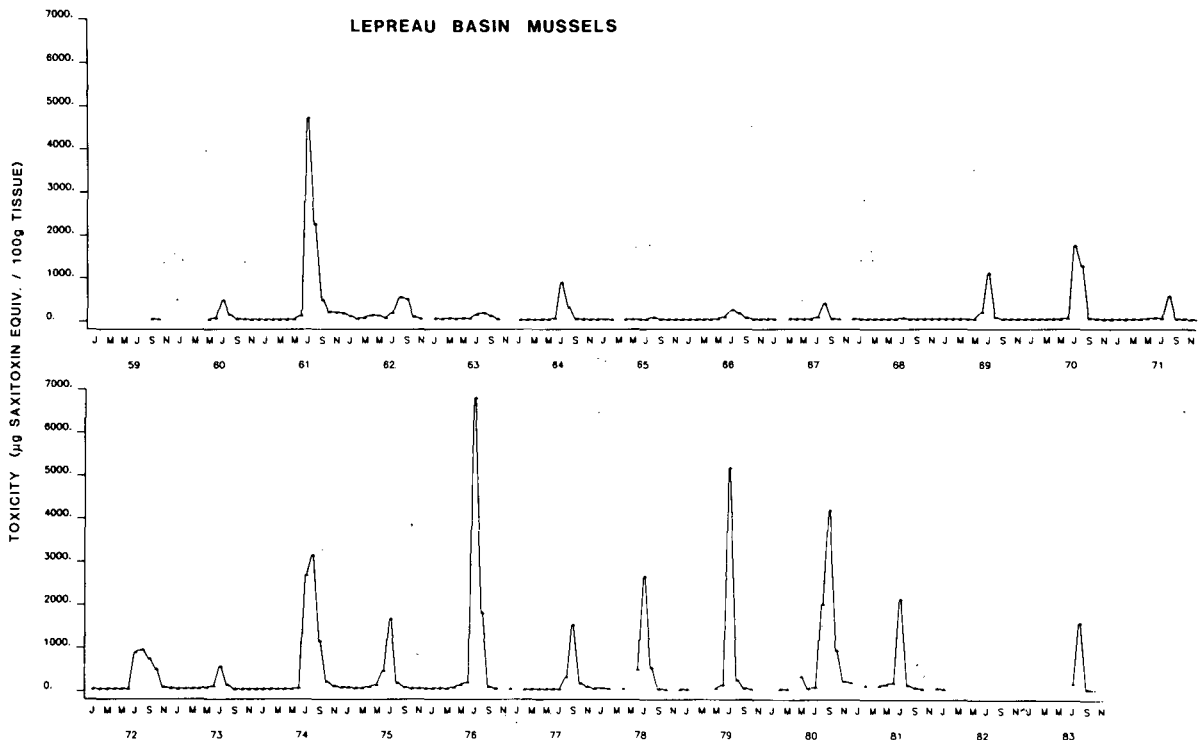


Figure 2. Mean monthly toxicity of blue mussels (*Mytilus edulis*) at Lepreau Basin, 1959 to 1983.

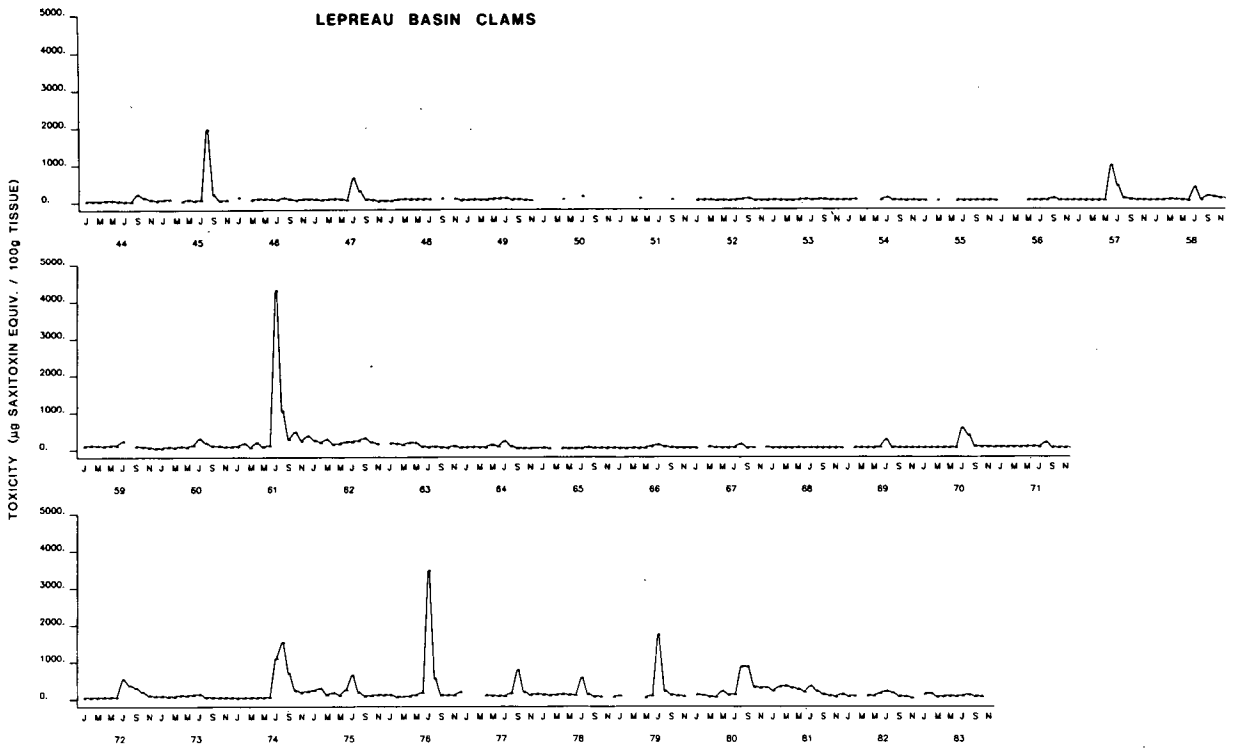


Figure 3. Mean monthly toxicity of soft-shell clams (*Mya arenaria*) at Lepreau Basin, 1944 to 1983.

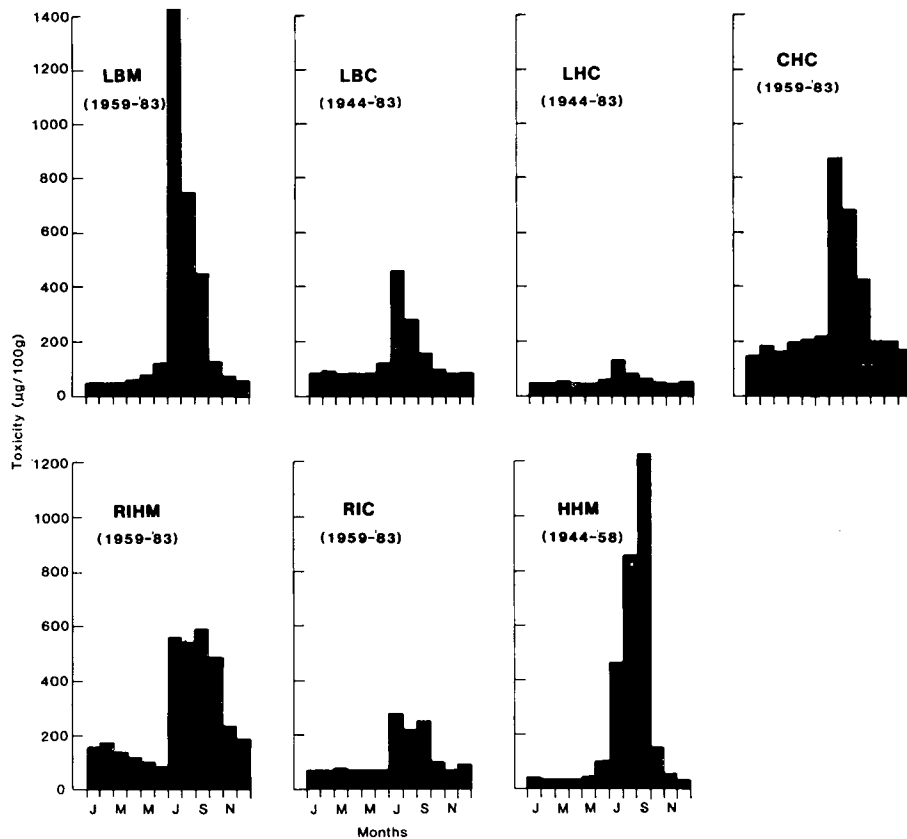


Figure 4. Mean toxicity over the years for Lepreau Basin mussels (LBM), Lepreau Basin clams (LBC), Lepreau Harbour clams (LHC), Crow Harbour clams (CHC), Ross Island Thoroughfare horse mussels (RIHM), Ross Island Thoroughfare clams (RIC), and Head Harbour mussels (HHM).

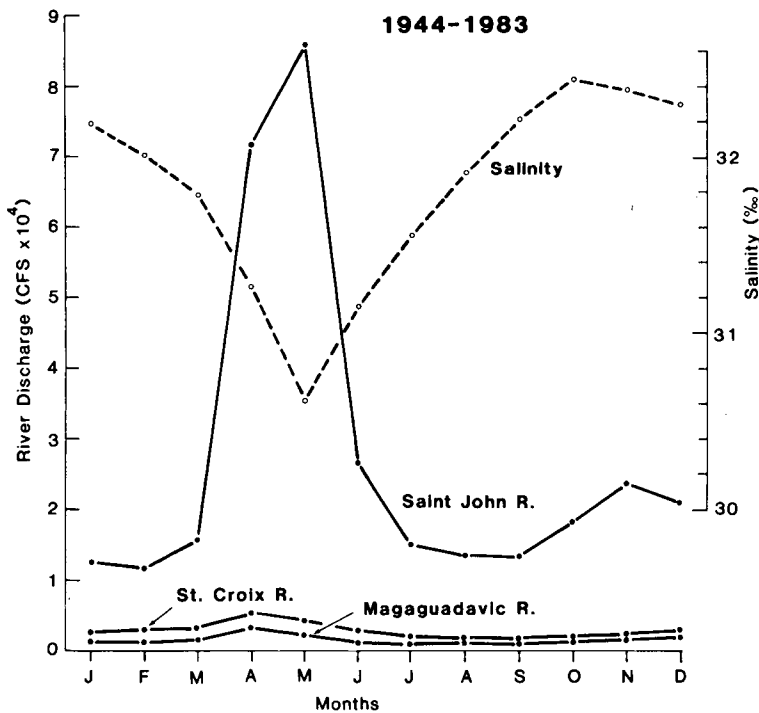
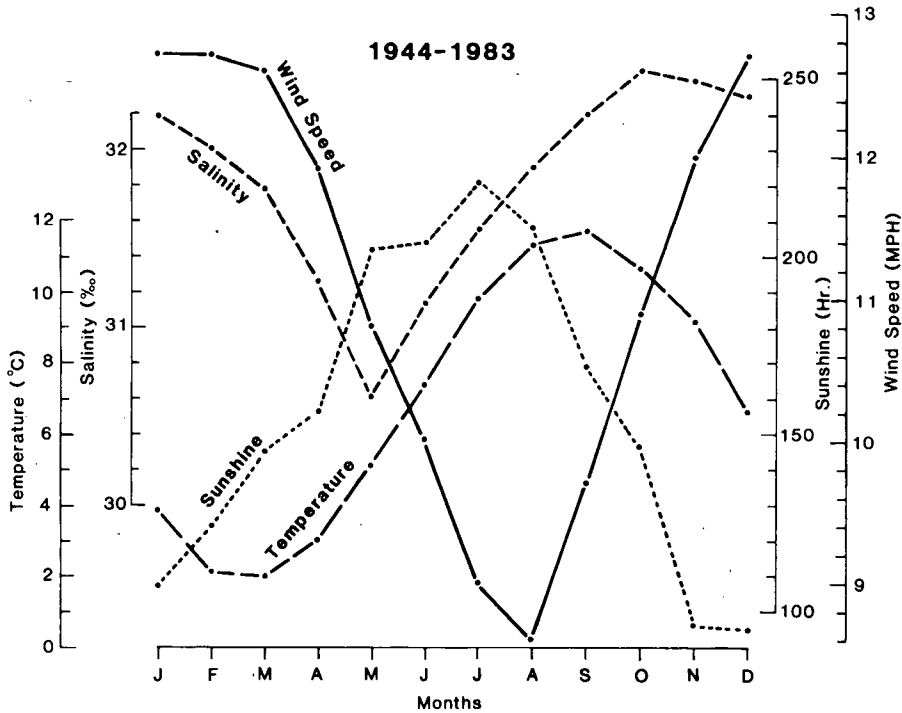


Figure 5. Mean environmental data from 1944 to 1983. Surface water temperature and salinity are for station Prince 5. Sunshine and wind speed are for Saint John Airport.

largely as a unit, with the exception of Head Harbour mussel toxicity.

Head Harbour mussel toxicity data (HHM), available only between 1944 and 1958, do not correlate well with

data from the other stations (Tables 1 and 2). The independability of Head Harbour as an indicator station for the southwestern Bay of Fundy was recognized in the late 1950s, so monitoring at the station was termi-

nated, and Lepreau Basin and Crow Harbour were selected as indicator stations (Prakash *et al.*, 1971). Dissimilarity between toxicity data at Head Harbour and at the other locations may well be related to the fact that Head Harbour is located within a zone of pronounced tidal mixing (Fig. 1). More than 20 years ago, mussel toxicity data at this location were investigated in relation to some environmental factors, although with inconclusive results (Prakash and Medcof, 1962).

Gonyaulax blooms and high shellfish toxicity levels are summer events in the Bay of Fundy, occurring between June and October (Fig. 4). Generally, the blooms last for three or four weeks and then dissipate rapidly. The increase in shellfish toxicity follows the rise in *Gonyaulax* numbers rather closely. The decrease in toxicity lags behind the disappearance of *Gonyaulax* by several weeks because of the time required for self-cleansing (depuration) of the shellfish.

Mean annual patterns of environmental factors over the 40-year period (1944–1983) are shown in Figure 5. Based on what is known about dinoflagellate biology, it could be hypothesized that any or all of these factors are important in determining the timing and intensity of summer *Gonyaulax* blooms and shellfish toxicity and deserve consideration in this initial, exploratory analysis.

The observation that periods of greatest toxicity seem to occur around 1945, 1961, and the late 1970s (see Figs. 2, 3, and White, 1982) suggests that *Gonyaulax* blooms and shellfish toxicity may be linked to a cyclical phenomenon. There is an 18.6-year tidal cycle caused by the change in the inclination of the moon to the equator (Loder and Garrett, 1978). Although the variation in

the tide-producing force, and thus current speed, is only $\pm 3.7\%$ (Garrett, 1977), tidal mixing parameters which delimit the location of frontal zones are related to the cube of the current speed. Since dinoflagellate blooms in many parts of the world occur in areas of high tidal energy dissipation (and the Bay of Fundy is certainly one), there would seem to be some plausibility in the idea that blooms and toxicity may be related to this cycle. Figure 6 shows the cycle, expressed as the inverse of the nodal modulation factor (NODMOD). In this form, maximum tide-producing force occurs when the sine wave is at a minimum.

As a first step, simple correlations were computed between toxicity and environmental data for the months of June through October when the blooms and toxicity occur. Statistically significant correlations ($P < 0.05$) were few, occurring most often as negative correlations with wind speed and NODMOD (Table 3). In several instances (some not listed in the table), there were positive correlations between wind speed and Head Harbour mussel toxicity, providing some evidence of the dissimilar response of HHM to the other toxicities.

The intensity of the summer blooms may be approximated by the sum of June-through-October toxicities. Further, in attempting to develop predictive capability we must examine environmental events that precede summer toxicity. Accordingly, simple correlations were computed between environmental data for the pre-bloom months of January to June and the sum of the subsequent monthly toxicities from June through October (designated as JOLBM, JOLBC, etc.). A number of statistically significant correlations ($P < 0.05$) occur between January-and-February salinity, sunshine, wind

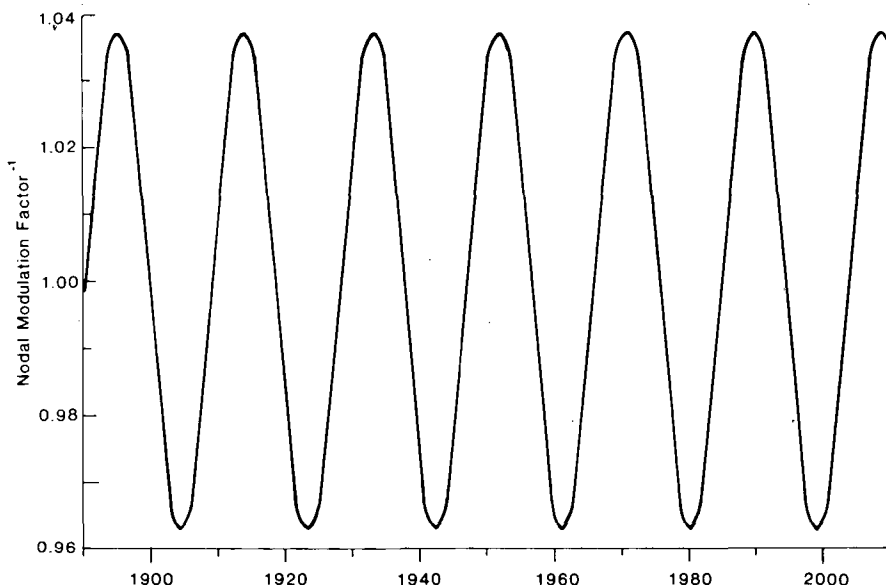


Figure 6. Inverse of the nodal modulation factor representing the influence on the tides of the 18.6-year cycle of the moon's inclination to the equator. Maximum tidal strength and mixing occur at the minimum points of the sine wave.

Table 3. Simple correlation coefficients ($P < 0.05$) between environmental variables and toxicity; $n = 13-37$.

	T	S	SUN	SPEED	MAG	CROIX	RSJOHN	NODMOD
Jun								
LBM	-	-	-	-	-	-	-	-
LBC	-	-	-	-	-	-	-	-
LHC	-	-	-	-0.326	-	-	-	-
CHC	0.483	-	-	-0.626	-	-	-	-
RIHM	-	-	-	-	-	-	-	-0.381
RIC	-	-	-	-0.691	-	-	-	-0.487
HHM	-	-	-	0.632	-	-	-	-0.516
Jul								
LBM	-	-	-	-0.352	-	-	-	-0.364
LBC	-	-	-	-	-	-	-	-0.333
LHC	-	-	-	-	-	-	-	-0.435
CHC	-	-	-	-0.374	-	-	-	-0.416
RIHM	-	-	-	-0.465	-	-	-	-0.429
RIC	-	-	-	-0.489	-	-	-	-0.452
HHM	-	-	-0.468	-	-	-	-	-0.472
Aug								
LBM	-	-	0.407	-	-	-	-	-
LBC	-	-	0.304	-	-	-	-	-
LHC	-	-	-	-0.452	-	-	-	-0.385
CHC	-	0.358	0.420	-0.372	-	-	-	-
RIHM	0.411	-	-	-0.670	-	-	-	-0.474
RIC	-	-	-	-0.501	-	-	-	-
HHM	-	-	-	-	-	-	-	-
Sep								
LBM	-	-	-	-	-	-	-	-
LBC	-	-	-0.300	-	-	-	-	-0.279
LHC	-	-	-	-	-	-	-	-
CHC	-	-	-	-	-	-	-	-
RIHM	-	0.433	-	-	-	-	-	-
RIC	-	-	-	-	-	-	-	-
HHM	-	-	-	-	-	-	-	-
Oct								
LBM	-	-	-	-0.395	-	-	-	-
LBC	-	-0.372	-	-	0.354	-	-	-0.440
LHC	-	-	-	-	0.323	-	-	-
CHC	-	-	0.360	-0.436	-	-	-	-
RIHM	-	-	-	-	0.576	0.480	0.447	-
RIC	-	-	-	-0.372	0.384	-	-	-
HHM	-	-	-	-	-	-	-	-0.572
Jun-Oct ($n = 113-183$)								
LBM	-	-	0.269	-0.337	-	-	-	-0.162
LBC	-	-	0.122	-0.160	-	-	-	-0.210
LHC	-	-	0.133	-0.233	-	-	-	-0.268
CHC	-	-	0.307	-0.393	-	-	-	-0.159
RIHM	0.252	0.165	-	-0.282	-	-	-	-0.264
RIC	0.153	-	-	-0.338	-	-	-	-0.236
HHM	-	-	-	-	-	-	-	-0.206

speed, nodal modulation factor, and subsequent summer toxicities (Table 4). Other significant correlations occur inconsistently throughout the pre-bloom months. As before, there are negative correlations with wind speed and NODMOD, with the exception of HHM which is positively correlated with wind speed on several occasions.

Using the same data (January-to-June environmental data and June-through-October toxicity sums), the concerted effects of all eight environmental factors on summer toxicity were examined. Squares of the multiple correlation coefficients (R^2) are listed in Table 5. They indicate that the January-to-June environmental data account for 28-97% of the variation in summer toxic-

Table 4. Simple correlation coefficients ($P < 0.05$) between lagged environmental variables and the sum of June-through-October toxicity; $n = 11-36$.

	T	S	SUN	SPEED	MAG	CROIX	RSJOHN	NODMOD
Jan								
JOLBM	0.437	0.599	-	-	-	-	-	-0.407
JOLBC	-	-	0.344	-	-	-	-	-0.423
JOLHC	-	0.422	0.358	-	-	-	-	-0.576
JOCHC	-	0.551	-	-	-	-	-	-
JORIHM	-	0.605	0.535	-0.578	-	-	-	-0.501
JORIC	0.494	0.602	0.514	-0.427	-	-	-	-0.471
JOHHM	-	-	-	-	-	-	-	-
Feb								
JOLBM	-	0.515	0.461	-	-	-	-	-0.471
JOLBC	-	0.502	-	-	-	-	-	-0.471
JOLHC	-	0.618	0.364	-	-	-	-	-0.471
JOCHC	0.438	0.523	-	-0.586	-	-	-	-0.471
JORIHM	-	0.530	-	-0.806	-	-	-	-0.471
JORIC	0.391	0.623	-	-0.747	-	-	-	-0.471
JOHHM	-	-	0.567	0.544	-	-	-	-0.471
Mar								
JOLBM	0.381	-	-	-	-	-	-	-0.471
JOLBC	-	-	-	-	-	-	-	-0.471
JOLHC	-	-	-	-	-	-	0.412	-0.471
JOCHC	-	-	-	-	-	-	-	-0.471
JORIHM	-	-	-	-0.518	-	-	-	-0.471
JORIC	-	0.442	-	-0.390	-	-	-	-0.471
JOHHM	-	0.392	-	0.705	-	-	-0.490	-0.471
Apr								
JOLBM	-	0.408	-0.414	-0.579	-	-	-	-0.471
JOLBC	-	-	-	-	-	-	-	-0.471
JOLHC	-0.390	-	-	-	-	-	-	-0.471
JOCHC	-	0.606	-	-	-	-	-	-0.471
JORIHM	-	-	-	-0.469	-	-	-	-0.471
JORIC	-	-	-	-0.409	-	-0.477	-0.562	-0.471
JOHHM	-0.517	-	0.480	-	-	-	-	-0.471
May								
JOLBM	-	-	-	-	-	-	-	-0.471
JOLBC	-	-	-	-	0.475	0.449	0.372	-0.471
JOLHC	-	-	-	-	-	-	-	-0.471
JOCHC	-	-	-	-	-	-	-	-0.471
JORIHM	0.467	-	-	-	-	-	-	-0.471
JORIC	-	-	-	-	-	-	-	-0.471
JOHHM	-	-	0.672	-	-	-	-	-0.471
Jun								
JOLBM	-	-	-	-0.560	-	-	-	-0.471
JOLBC	-	-	-	-	-	0.362	0.322	-0.471
JOLHC	-	-	-	-	-	0.418	-	-0.471
JOCHC	-	-	-	-0.463	-	-	-	-0.471
JORIHM	0.593	-	-	-0.560	0.440	-	-	-0.471
JORIC	0.588	-	-	-0.486	0.386	-	-	-0.471
JOHHM	-	-	-	-	-	-	-	-0.471

ities. Only a few of the correlations are statistically significant ($P < 0.05$).

These exploratory statistical analyses reveal that much of the variation in summer toxicities can be accounted for by the multiple interactions of the eight environmental variables during pre-bloom months. Fur-

ther, considering the environmental variables singly, salinity, wind speed, and the nodal modulation factor have the most consistent, statistically significant relationships with summer toxicity. It is enigmatic how salinity and wind speed during January and February could influence summer toxicity. Here it must be

Table 5. Values of R^2 for multiple correlation of all environmental variables with the sum of June-through-October toxicity.

	Jan	Feb	Mar	Apr	May	Jun
JOLBM	0.649	0.608	*0.669	*0.725	0.473	0.462
JOLBC	0.328	0.457	0.352	0.373	*0.526	0.314
JOLCH	0.494	*0.584	*0.709	*0.589	*0.588	*0.577
JOCHC	0.637	0.688	0.618	0.667	0.281	0.382
JORIHM	*0.896	*0.902	0.716	*0.831	0.660	0.632
JORIC	*0.805	*0.831	0.655	0.611	0.622	0.615
JOHHM	0.775	0.969	*0.909	0.538	0.847	0.905

* $P < 0.05$.

stressed that one is not to ascribe causality to these statistical relationships. But the statistical treatment may perhaps steer us to some underlying causal relationships.

In this connection, the rather consistent significant correlations of toxicity with the 18.6-year cycle of lunar motion is intriguing. It is conceivable that there are oceanographical or physiological mechanisms in operation here. Increased tidal energy dissipation (mixing) may influence the distribution of *Gonyaulax* aggregations (in frontal zones?) and their transport to shellfish areas or may influence the growth of *Gonyaulax* populations. The trend of this relationship would be considerably more convincing if it continues to be observed over another cycle (Fig. 6) – which, unfortunately, will not occur for another 12 years!

It does not appear that it will be a simple task to develop predictive capability for *Gonyaulax* blooms and shellfish toxicity. The data were not collected with this purpose in mind and may be inappropriate for the task because of location and time scale. Other correlative attempts at red-tide prediction have been made, also with obscure results (Baldrige, 1975; Ouchi, 1982; Rounsefell and Dragovich, 1966). It may be optimistic to think that predictive capability can be developed through such an empirical approach. But the alternative is to wait until we have gained a much fuller understanding of the mechanisms underlying the dynamics of bloom initiation and development. At least the empirical process helps generate ideas, define questions, and focus hypotheses along the way.

Acknowledgements

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References

- Association of Official Analytical Chemists. 1980. Paralytic shellfish poison biological method (32). In Official methods of analysis, pp. 298–299. Ed. by W. Horwitz. AOAC, Washington, D.C. 1018 pp.
- Baldrige, H. D. 1975. Temperature patterns in the long-range prediction of red tide in Florida waters. In Proceedings of the first international conference on toxic dinoflagellate blooms, pp. 69–79. Ed. by V.R. LoCicero. Mass. Sci. Technol. Found., Wakefield, Mass., USA. 541 pp.
- Garrett, C. J. R. 1977. Tidal influences on the physical oceanography of the Bay of Fundy and the Gulf of Maine. In Fundy tidal power and the environment, pp. 101–115. Ed. by G.R. Daborn. The Acadia University Institute, Wolfville, Nova Scotia. 304 pp.
- Loder, J. W., and Garrett, C. 1978. The 18.6-year cycle of sea surface temperature in shallow seas due to variations in tidal mixing. J. Geophysical Res., 83: 1967–1970.
- Needler, A. B. 1949. Paralytic shellfish poisoning and *Gonyaulax tamarensis*. J. Fish. Res. Bd Can., 7: 490–504.
- Ouchi, A. 1982. Prediction of red tide occurrence by means of multiple linear regression model. Bull. Jap. Soc. Sci. Fish., 48: 1245–1250.
- Prakash, A., and Medcof, J. C. 1962. Hydrographic and meteorological factors affecting shellfish toxicity at Head Harbour, New Brunswick. J. Fish. Res. Bd Can., 19: 101–112.
- Prakash, A., Medcof, J. C., and Tennant, A. D. 1971. Paralytic shellfish poisoning in eastern Canada. Bull. Fish. Res. Bd Can., 168: 68 pp.
- Rounsefell, G. A., and Dragovich, A. 1966. Correlation between oceanographic factors and abundance of the Florida red tide (*Gymnodinium breve* Davis), 1954–61. Bull. Mar. Sci., 16: 404–422.
- White, A. W. 1977. Dinoflagellate toxins as probable cause of an Atlantic herring (*Clupea harengus harengus*) kill, and pteropods as apparent vector. J. Fish. Res. Bd Can., 34: 2421–2424.
- White, A. W. 1980. Recurrence of kills of Atlantic herring (*Clupea harengus harengus*) caused by dinoflagellate toxins transferred through herbivorous zooplankton. Can. J. Fish. Aquat. Sci., 37: 2262–2265.
- White, A. W. 1982. Toxic dinoflagellate blooms in the Bay of Fundy since 1944. ICES CM 1982/L:13. 13 pp.
- White, A. W. 1984. Paralytic shellfish toxins and finfish. In Seafood toxins, pp. 171–180. Ed. by E.P. Ragelis. ACS Symposium Series 262. Amer. Chem. Soc., Washington, D.C. 460 pp.