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HISTORICAL EUTROPHICATION IN A RIVER–ESTUARY COMPLEX IN MID-COAST MAINE

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Abstract. European settlement of New England brought about a novel disturbance regime that impacted rivers and estuaries through overfishing, deforestation, dams, and water pollution. The negative consequences of these activities intensified with industrialization in the 19th and 20th centuries, often resulting in ecosystem degradation. Since environmental legislation was implemented in the 1970s, improvement in water quality has been tangible and widespread; however, ecological recovery can require substantial amounts of time and may never be complete. To document the natural baseline conditions and investigate the recovery of a severely degraded river–estuary complex in mid-coast Maine, we examined diatoms, pollen, total organic carbon, total nitrogen, stable isotopes, total phosphorus, biogenic silica, and trace metals in intertidal sediments and established a chronology with ¹⁴C, ²¹⁰Pb, and indicator pollen horizons. Both climate variability and human effects were evident in the sedimentary record of Merrymeeting Bay, the freshwater tidal portion of the Kennebec estuary. Natural climate variability was apparent in an episode of high sedimentation and altered diatom abundance during the 12th and 13th centuries and in changing pollen abundances between the 16th and 19th centuries, indicative of regional cooling. During the 18th century, colonial land clearance began an era of high sedimentation and eutrophication that strongly intensified with industrialization during the late 19th and 20th centuries. Improvements in water quality over the past 30 years in response to environmental regulation had little effect on ecosystem recovery as represented by the sedimentary record. Diatom composition and productivity and high fluxes of organic C, total P, and biogenic Si in recent sediments indicate that rates of nutrient loading remain high. These environmental proxies imply that aquatic productivity in Merrymeeting Bay was originally nutrient limited and water clarity high, relative to today. Further recovery may require more stringent regulation of nutrient inputs from industrial and municipal point sources. This historical study can contribute to public debate about the environmental management of this unusual river–estuary complex by describing its long-term natural baseline, thereby illustrating the upper limit of its potential for recovery.

Key words: coastal ecosystem; diatom record; ecosystem degradation; ecosystem recovery; eutrophication; historical ecology; Maine, USA; pollution; shifting baseline.

INTRODUCTION

European settlement of the Atlantic coast of North America brought about a novel disturbance regime that was widespread and prolonged (Foster et al. 1998). Overharvest of wild populations, deforestation, agriculture, and raw industrial and municipal pollution altered terrestrial and aquatic ecosystems (Foster et al. 2003, Lotze and Milewski 2004). Because rivers provided a means of transportation, abundant natural resources, and waterpower, key locations at waterfalls and heads-of-tide developed into early industrial centers (Wells 1869). By the mid-20th century, the cumulative effects of a century and a half of commercial, municipal, and industrial growth resulted in severe degradation of

several major “industrial” rivers in New England (Walker 1931, Steinberg 1991). Although the egregious pollution of the 19th and 20th centuries has abated, rivers and estuaries continue to receive excess nutrients from paper mills, wastewater treatment facilities, runoff, and atmospheric deposition (Roman et al. 2000). Numerous dams still alter the flow regime of rivers and prevent anadromous fish from reaching their spawning habitat (Poff et al. 1997), and trophic web structure has been altered by species loss and vastly reduced population sizes (Jackson et al. 2001, Lotze and Milewski 2004).

In coastal Maine, the Kennebec and Androscoggin Rivers played important roles during European settlement and later industrialization of the mid-coast region (Fig. 1). In 1753, the first dam spanning the Androscoggin at Brunswick-Topsham was constructed to power sawmills (Wheeler and Wheeler 1878). Tanneries,

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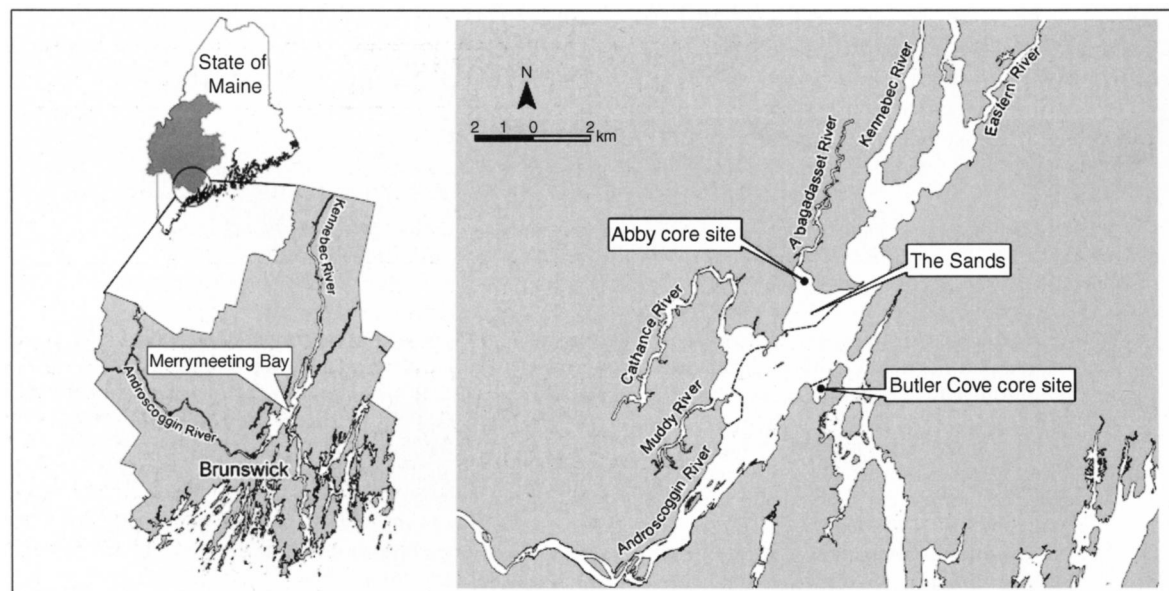


FIG. 1. Map of Merrymeeting Bay region showing core sites and location on the coast of Maine, USA. Bordering towns of Brunswick, Topsham, and Bath are shown. Gray shaded area in the inset map shows the combined watershed of the Kennebec and Androscoggin Rivers.

textile mills, and foundries soon followed. By 1815, migrating salmon could no longer pass the dam at Brunswick to upstream spawning habitat (Foster and Atkins 1869). In 1837, a dam completed across the Kennebec at Augusta to power several sawmills, a grist mill, and a machine shop (North 1870) caused the immediate collapse of the commercial salmon fishery throughout the watershed (Foster and Atkins 1869). Severe water pollution began in the latter 19th century, and by the mid-20th century, contributed to the final collapse of the remaining commercial fisheries. Pulp and paper mills discharged large quantities of organic matter, which resulted in anoxia, H_2S production, and frequent fish kills. Industry harnessed the waterpower of the rivers, and the rivers conveniently disposed of industry's waste (Steinberg 1991).

Although flagrant organic pollution no longer causes widespread oxygen depletion in the Kennebec and Androscoggin Rivers (Lichter et al. 2006), portions of the Androscoggin River periodically experience algal blooms and reduced oxygen concentrations (Maine Department of Environmental Protection 2000). Of the limiting nutrients entering the Androscoggin River, ~77% of P originates from treatment of paper mill effluents, 13% derives from municipal wastewater treatment facilities, and 10% from runoff. Although less impacted, the Kennebec River also experiences occasional algal blooms (Maine Department of Environmental Protection 2000).

There is widespread public interest in restoring Maine's rivers and estuaries. However, a significant problem faced in garnering public support for more stringent environmental regulation is the "shifting

baseline" syndrome described by Pauly (1995). In the case of Maine's rivers and estuaries, public and institutional memory does not extend back in time far enough to understand and value the true potential of a degraded ecosystem, only far enough to recognize improvement over the appalling conditions of the middle 20th century. As such, a shifting baseline lowers expectations for recovery. Historical ecology can inform public debate about environmental policy by documenting the long-term natural baseline and in so doing, illustrate the upper endpoint of the potential for ecosystem recovery. In the case of coastal eutrophication, understanding the potential for clean water, restored aquatic habitat, and increased secondary production can contribute to public debate about the benefits and costs of environmental regulation by illustrating the potential opportunity cost of failing to reduce nutrient inputs with the best available technologies. This may be especially relevant for Maine, where tourism and recreation are vital economic activities that would benefit from restoration of coastal ecosystems.

To document natural baseline conditions and ecosystem response to the Clean Water Act of 1972 for a river-estuary complex in coastal Maine, we reconstructed an environmental history of Merrymeeting Bay based on diatom, pollen, stratigraphic, and chemical records preserved in intertidal sediments. Merrymeeting Bay is the freshwater tidal portion of the Kennebec estuary and receives water from the Kennebec and Androscoggin Rivers as well as four minor rivers (Fig. 1). Our objectives were to (1) determine baseline conditions associated with natural climatic variability, (2) investigate the effects of European settlement and industrial-

ization within the watershed, and (3) examine ecosystem response to environmental regulation in the late 20th century. This historical study increases our understanding of the causes and consequences of coastal eutrophication in Maine, and contributes to ongoing discussion of the benefits and costs of environmental regulation.

STUDY REGION

Merrymeeting Bay is formed by the confluence of the Kennebec, Androscoggin, Cathance, Abbagadasset, Muddy, and Eastern rivers, 30 km inland from the Atlantic coast of Maine (Fig. 1). Water draining one-fourth of the area of Maine and part of New Hampshire (24 755 km²) exits Merrymeeting Bay into the lower Kennebec estuary through a narrow (215-m wide) channel called "the Chops." The substantial river flow through this narrow channel and a series of underwater sills constrain the upstream mixing of seawater into Merrymeeting Bay (Wong and Townsend 1999, Kistner and Pettigrew 2001). Although elevated salinities occur seasonally during periods of low river flow, salinity values generally range below the 5 g/L threshold that distinguishes freshwater from brackish water ecosystems (Odum et al. 1984). The Kennebec estuary is a drowned river valley typical of New England estuaries. Tides range up to 3.5 m at the mouth of the Kennebec but attenuate to ~1.8 m at the Chops. Wild rice (*Zizanea aquatica*) and other freshwater emergents dominate expansive tidal marshes in Merrymeeting Bay and provide forage for migratory waterfowl. Historically, Merrymeeting Bay also provided important spawning and nursery habitat for several species of anadromous fish. Until the early 19th century, the Kennebec was famous for its spawning runs of Atlantic salmon (*Salmo salar*) and hosted prolific spawning runs of alewife (*Alosa pseudoharengus*), American shad (*Alosa sapidissima*), blue-back herring (*Alosa aestivalis*), and striped bass (*Morone saxatilis*; Foster and Atkins 1869).

Although Europeans first settled in the region during the early 17th century, the harsh physical environment and recurrent wars with the native Abenaki limited immigration and population growth (Ghere 1995, Bourque 2001). The nearby towns of Brunswick, Topsham, Woolwich, Cork (later Dresden), Richmond, and Bowdoinham were being settled by 1717 (McKeen 1853, Wheeler and Wheeler 1878, Allen 1931); and by 1760, the area surrounding Merrymeeting Bay was a mixture of farms, houses, and forests (Allen 1931). By 1800, Brunswick and Topsham were industrial centers, as was Augusta on the upstream Kennebec (Wheeler and Wheeler 1878, North 1870). Sawmills loaded the rivers with sawdust, textile mills discharged fibers and dyes, and foundries dispersed trace metals. The most destructive sources of pollution were the pulp and paper mills. The sulfite pulp process in use between 1880 and the 1960s discharged tons of lignosulphonic acid (Sutermeister 1941) weekly into both the Kennebec and Androscoggin rivers (Water Improvement Commission

1960, 1966, Lawrence 1967). The rivers became anoxic as bacteria metabolized the organic matter stimulating anaerobic bacteria to reduce the sulfite wastes to H₂S. As early as 1893, the Androscoggin had a reputation for being polluted (Owen 1936). By the 1940s, the river achieved infamy as H₂S emanating from its waters blackened the paint on nearby buildings and the distinctive rotten egg odor could be smelled miles away (Chase 1949). As a remedy for the odor, 6694 tons of CaNO₃ were dumped into the river upstream of the Lewiston-Auburn between 1948 and 1960 (Lawrence 1967). Denitrifying bacteria made use of the NO₃ as an electron acceptor, outcompeting the sulfate-reducing bacteria and effectively halting H₂S production. Although this quick fix made the river less obnoxious, it probably magnified downstream eutrophication. Widespread fish kills occurred in 1947, 1957, 1963, 1965, and 1973 (Water Improvement Commission 1960, 1966, Maine Department of Environmental Protection 1971). With the installation and operation of primary treatment facilities at several towns and at the pulp and paper mills, seasonal anoxia subsided and water quality improved (Lichter et al. 2006). Today, the Kennebec is rated a class B river (i.e., dissolved oxygen ≥ 7 mg/L and *Escherichia coli* mean counts $< 64/100$ mL), whereas the Androscoggin is rated a class C river (i.e., dissolved oxygen ≥ 5 mg/L and *Escherichia coli* mean counts $< 142/100$ mL).

METHODS

Sedimentary record and analyses

To reconstruct an environmental history, we collected two cores of intertidal sediments at each of two sites in Merrymeeting Bay. The first coring site was near the mouth of the Abagadasset River in an area known locally as "The Sands" (Fig. 1). The Sands is a deltaic structure formed by upriver sediment transport driven by strong flood tides. It was noted as an area of shallow water in the 17th century prior to widespread land clearance (Wheeler and Wheeler 1878). Examination of a series of aerial photographs dating back to 1953 indicates that neither the Abagadasset River channel nor any connecting tidal channels have migrated laterally across the core location over this time period. The second coring site, Butler Cove, is a small embayment lacking stream or tidal channels (Fig. 1). Both coring sites are dominated by wild rice (*Zizanea aquatica*) and consist of sandy sediments with organic matter percentages ranging between 5 and 10 percent. We vibrated a 7.6 cm diameter aluminum tube into the intertidal sediments to a depth of 1 m, plugged the top, and retrieved each core with a hand jack. Visible stratigraphy was recorded and cores were sectioned into 1-cm slices and stored at 4°C.

A chronology was established for each site using ¹⁴C, ²¹⁰Pb, and ¹³⁷Cs dating, as well as *Rumex* and *Ambrosia* pollen horizons indicative of land settlement. Beta Analytic, Miami, Florida, USA, analyzed plant tissue

and small pieces of wood for ^{14}C age. GEOTOP laboratory, Montreal, Canada, analyzed prepared sediments for ^{210}Pb and ^{137}Cs activity. The ^{210}Pb concentration was inferred by measurement of the activity of the daughter product ^{210}Po with alpha spectrometry assuming secular equilibrium between the two isotopes (Flynn 1968). ^{210}Pb ages were estimated assuming a constant rate of supply of ^{210}Pb to the sediments (Binford 1990). The ^{137}Cs peak identified sediments deposited in 1963–1964, the period of maximum activity of atmospheric nuclear weapon testing (Milan et al. 1995).

Diatom analysis was carried out in 4-cm steps on one of the Abagadasset River cores (i.e., Abby-1). To prepare the samples, 1 cm³ of sediment was treated overnight with 25 mL of a 50:50 solution of H_2SO_4 and HNO_3 and then digested for three hours at 90°C. Samples were washed in several steps with deionized water and allowed to settle for 24 hours between washings. This procedure was repeated until the sample was neutralized. To remove sand, samples were mixed and allowed to settle for 10 seconds; the decanted solution was used for diatom counting. A known concentration of microspheres was added to each sample solution, and 750 μL was dried on a cover glass and mounted in Naphrax (a synthetic resin dissolved in toluene) on microscope slides. A minimum of 500 diatom valves was counted with 1000 \times magnification using an Olympus BX51 microscope (Center Valley, Pennsylvania, USA) with phase contrast. Diatom taxa were identified to the species level using freshwater and coastal references (Krammer and Lange-Bertalot 1986, 1988, 1991a, b, Witkowski et al. 2000). These references also provided information about salinity and habitat tolerances of individual diatom species. Diatom assemblages were divided into four zones based on an optimal partitioning algorithm implemented with the software ZONE (Lotter and Juggins 1991).

Patterns in diatom assemblages, as well as their relation to other proxies measured in the sediments, were explored using several ordination techniques. Because detrended correspondence analysis (DCA) determined that the variation in diatom data was <2 SD, we analyzed diatom assemblages with linear methods (Birks 1995), such as DCA for identification of changes over time and redundancy analysis (RDA) for identification of correlations with other sedimentary proxies. Principal component analysis (PCA) was used for summarizing patterns in pollen, as the variation in these data was >2 SD (Birks 1995). Ordinations were carried out with CANOCO software version 4.5 (ter Braak and Šmilauer 2002).

Pollen analysis was completed on 16 samples from the upper 80 cm of one core from each of the two sites. Because of the high sand content in the estuary sediments, we used a pollen floatation technique developed for sandy sediments (Vandergoes 2000). After deflocculation, samples were sieved through 125- μm

Nitex mesh, and then suspended in viscous sodium polytungstate solution with a specific gravity of 2.32. Pollen and organic particles float in the polytungstate and are decanted for treatment with glacial acetic acid to dehydrate cells and remove cellulose (Faegri et al. 1989). Fine-grained silts and clays were removed by sieving the sample through 6- μm Nitex sieving mesh (Cwynar et al. 1979). The remaining sediment was permanently suspended in silicon oil. At 40 \times magnification, a minimum of 300 arboreal and herbaceous pollen cells were counted and identified. Pollen data were plotted as relative percentage using Tilia and TGView 2.0.2 software (E. Grimm, Illinois State Museum, Research and Collections Center, Springfield, Illinois, USA).

Elemental carbon and nitrogen stable-isotope analyses were conducted on sediments from both core locations. Sediments were dried for 48 h in a convection oven and subsamples ground to a fine powder with a ball mill. C, N, ^{13}C , and ^{15}N concentrations were measured by the University of California, Davis stable isotope laboratory using continuous flow isotope-ratio mass spectrometry (IRMS, 20-20 mass spectrometer, PDZEuropa, Northwich, UK) after combusting samples at 1000°C in an on-line elemental analyzer (ANCA-GSL, PDZEuropa). Total phosphorus (TP) and biogenic silica (BSi) were analyzed for both sediment cores. The laboratory methods used for determining TP and BSi in sediments are described in the standard operating procedures of the NOAA/Great Lakes Environmental Research laboratory (Johengen 1996).

RESULTS

Chronology and sedimentation rate

The Abagadasset River core (i.e., Abby-1) that was analyzed for a sedimentary record exhibited excess ^{210}Pb activity from the sediment surface to a depth of 34 cm (Fig. 2). A strong linear relationship between log-transformed excess ^{210}Pb activity and cumulative sediment dry mass between 0 and 34 cm depth ($R^2 = 0.97$, $P < 0.001$) indicates a lack of physical and biological disturbance (e.g., Zimmerman and Canuel 2000). The resulting ^{210}Pb chronology is consistent with the timing of the onset of rapid deposition of numerous trace metals and fly-ash particles in the late 19th century (P. Lea, *unpublished data*), which reflect increasing industrialization and technological change along the tributary rivers. The ^{137}Cs profile of the Abby-1 core, however, is inconsistent with the fallout history of ^{137}Cs , suggesting post-depositional mobility and delayed inputs of ^{137}Cs (Davis et al. 1984). ^{137}Cs first appears at 24 cm depth, which corresponds to a ^{210}Pb date of 1912, much earlier than the beginning of atmospheric fallout of ^{137}Cs derived from thermonuclear testing in 1952. Also, excess ^{137}Cs activity does not reflect episodes of peak atmospheric fallout in 1959 and 1963, but instead is correlated with sediment organic matter content ($r = 0.72$, $P < 0.001$). These inconsistencies in the ^{137}Cs record are explained by (1) mobility of ^{137}Cs in

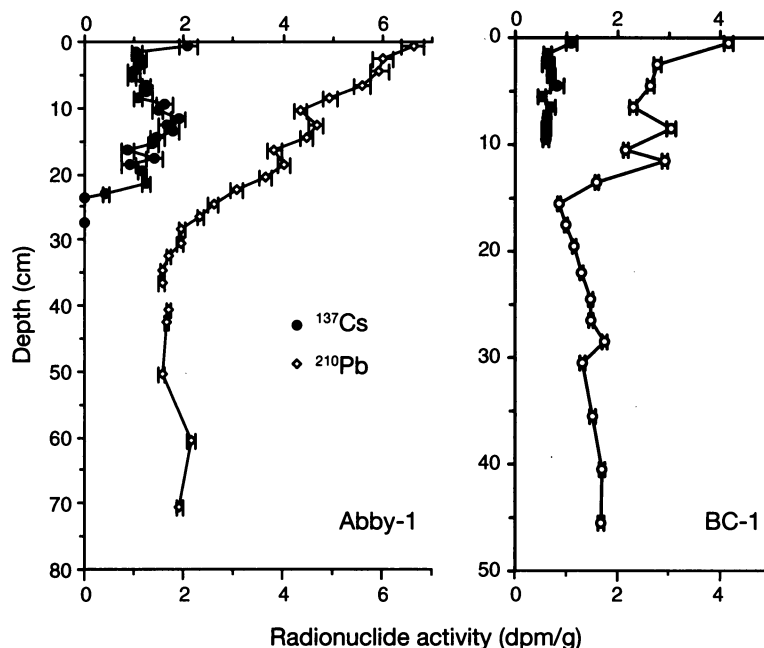


FIG. 2. Total activity (disintegrations per minute per gram) of ^{210}Pb and ^{137}Cs in the sediments of the Abby-1 and BC-1 cores. Open symbols represent ^{210}Pb measurements, and closed symbols represent ^{137}Cs measurements. [Conversion to SI units: 1 Ci = 2.22×10^{12} dpm; 1 Ci = 3.7×10^{10} Bq; 1 Bq = 1 dps.]

sediments with low clay content, (2) delayed inputs of ^{137}Cs via transport of sediments from upriver, and (3) ^{137}Cs adsorption to organic matter (Davis et al. 1984, Crusius and Anderson 1995, Milan et al. 1995). Generally, Cs is immobilized in sediments by adsorption to clay minerals at the time of deposition (Davis et al. 1984). However, in sediments of low clay content, which is indicated by low aluminum content (<1%) in the Abby-1 sediments, ^{137}Cs can diffuse freely both upward and downward within the sediment column. Delayed deposition of ^{137}Cs occurs when sediments deposited upriver are eroded, transported downriver, and redeposited in Merrymeeting Bay at a later date than the original atmospheric fallout. Such a time delay would smear the ^{137}Cs signal upward in the sediment column. ^{210}Pb is generally less mobile in sediments, and because it derives from continuous atmospheric deposition rather than episodic fallout events, it is less affected by delayed transport of sediments from upriver. The upward mobility of ^{137}Cs and/or delayed deposition of sediments washing in from upriver are illustrated by the strong ^{137}Cs peak at the sediment surface in both of our cores, despite a decline in atmospheric fallout following the end of thermonuclear testing in 1982 (Crusius and Anderson 1995, Milan et al. 1995). These problems with the ^{137}Cs record in the Abby-1 core invalidate its use as a complementary dating technique to ^{210}Pb for the recent sediments.

Excess ^{210}Pb activity was more variable at the Butler Cove site (i.e., BC-1 core) than at the Abagadasset site and indicates sediment mixing (Fig. 2). A linear

regression of log-transformed excess ^{210}Pb activity on cumulative sediment dry mass between 0 and 14 cm depth indicates a poor linear relationship ($R^2 = 0.46$, $P = 0.067$). This clear evidence of sediment mixing in the BC-1 core precludes its use as an environmental record. However, comparison with the Abby-1 ^{210}Pb profile supports our conclusion that the Abby-1 core represents a continuous environmental record without episodes of sediment mixing (Fig. 2). The ^{137}Cs profile of the BC-1 core exhibited similar post-depositional mobility, delayed inputs, and correlation with sediment organic matter as the Abby-1 ^{137}Cs profile. We therefore based our environmental history entirely on the Abby-1 core. Its location at the head of the Sands area, which accumulates sediments delivered by both of the major rivers, provided a high sedimentation rate and more consistent depositional environment than the Butler Cove site. The overall average sedimentation rate was 0.046 cm/yr for the Abby-1 core and 0.029 cm/yr for the BC-1 core.

Pollen indicators of local land clearance, *Rumex* and *Ambrosia*, first occur at 44 cm depth in the Abby-1 core. We assigned a date of AD 1717 to this depth because several towns in the area were settled at this time (Wheeler and Wheeler 1878, Baxter 1916, Allen 1931). A distinct sediment color change occurred below the settlement horizon at 48 cm depth. Sediments below 48 cm depth were dark organic, while those above 48 cm depth were light gray-brown. A gradual transition back to darker organic sediments occurred at 20 cm depth. We dated plant tissue from sediments at 48 cm depth to

TABLE 1. Radiocarbon analyses.

Core	Depth (cm)	Lab. no.	Material dated	¹⁴ C age yr BP	Confidence interval
Abby-1	48	Beta-205684	plant tissue	370 ± 40	AD 1440–1640
Abby-1	52	Beta-202185	plant tissue	740 ± 40	AD 1230–1300
Abby-1	64	Beta-202186	plant tissue	810 ± 60	AD 1050–1100 and AD 1140–1290
Abby-1	82	Beta-199575	wood	1800 ± 40	AD 120–340
BC-1	62	Beta-199576	wood	2140 ± 40	360–280 BC and 240–60 BC

Notes: The confidence interval is given as the 2-sigma range of the calibrated calendar dates. Analyses were conducted by Beta Analytic, Miami, Florida, USA.

estimate the timing of the sediment color change, sediments at 52 cm and 64 cm depths to bracket a period of change in the diatom record, and basal sediments at 82 cm depth. The timing of the sediment color change was estimated to be AD 1540 (95% CI = AD 1440–1640), and the period of change in diatom record between 52 cm and 64 cm depths was estimated to be AD 1170 (AD 1050–1240) and AD 1265 (AD 1230–1300; Table 1). A basal radiocarbon date at 82 cm indicated that the portion of the sedimentary record that is the focus of this study spanned the past 1800 ± 40 years. The overall sedimentation rates were low throughout much of the core except for the period between AD 1170 and 1265 and during the 18th, 19th, and 20th centuries (Fig. 3). The sand content ranged from ~10% throughout most of the core to 30% between 30 cm and 40 cm depth (data not shown).

Diatom record

Fossil diatom analysis distinguished four zones (I–IV) that were characterized by differences in species composition and abundance (Fig. 4). Sediments dated between AD 230 and 1170 (Zone I) were dominated by *Fragilaria subsalina* (Grunow) Lange-Bertalot and *F.*

geocollegarum Witkowski et Lange-Bertalot with lower abundances of *Achnanthes minutissima* Kützing and *A. delicatula* aggr. (Kützing) Grunow. Between AD 1170 and 1265 (Zone II), the relative abundance of planktonic species, such as *Cyclotella meneghiniana* Kützing and *Actinocyclus normanii* (Gregory) Hustedt, increased temporarily. After AD 1265 (Zone III), diatom assemblages returned to the species composition of Zone I, except that *F. geocollegarum* was replaced by *F. construens* var. *venter* (Ehrenberg) Grunow, *F. pulchella* (Ralfs) Lange-Bertalot, and *Achnanthes* spp. Following local land clearance in the 18th century, diatom abundance increased above the long-term baseline, especially benthic species. Soon after AD 1900, planktonic diatoms such as *C. meneghiniana*, *Cyclotella* sp. 1, and *A. normanii* increased abruptly along with some benthic taxa to dominate the present assemblage (Zone IV). Also, several species tolerant of pollution and eutrophication first appeared or increased in abundance in Zone IV, including *Navicula gregaria* Donkin, *Nitzschia palea* (Kützing) W. Smith, *Nitzschia tubicola* Grunow, and *Stephanodiscus* spp., whereas *F. pulchella*, *A. delicatula*, and *F. subsalina* disappear from the

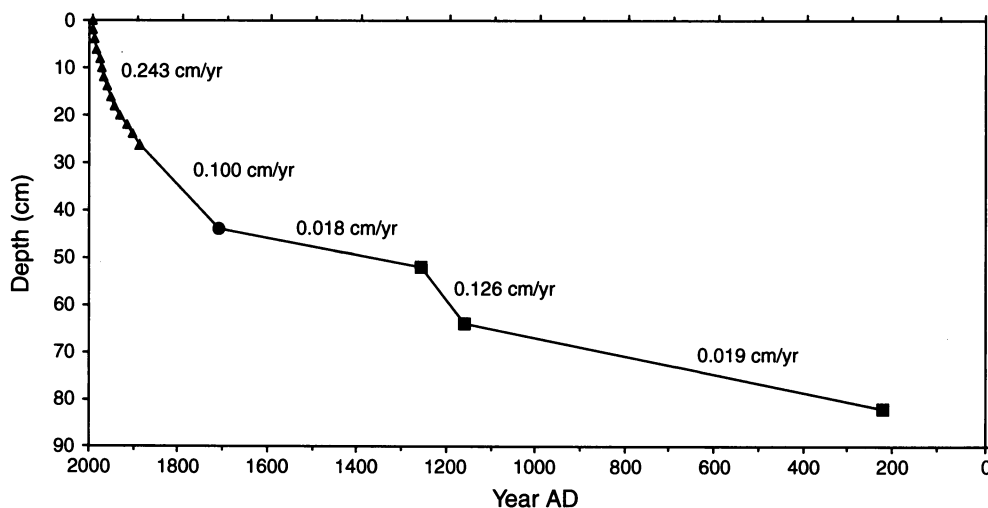


FIG. 3. Age-depth model for the Abby-1 core showing accelerated rates of sedimentation between AD 1170 and 1265, and in the 18th, 19th, and 20th centuries. Triangles represent ²¹⁰Pb dates, the circle indicates the pollen-based settlement horizon, and squares represent radiocarbon dates. Average sedimentation rates for sediments between dated horizons are given. Sediment ages between the settlement date and the oldest ²¹⁰Pb date were estimated by interpolation using a second-order polynomial.

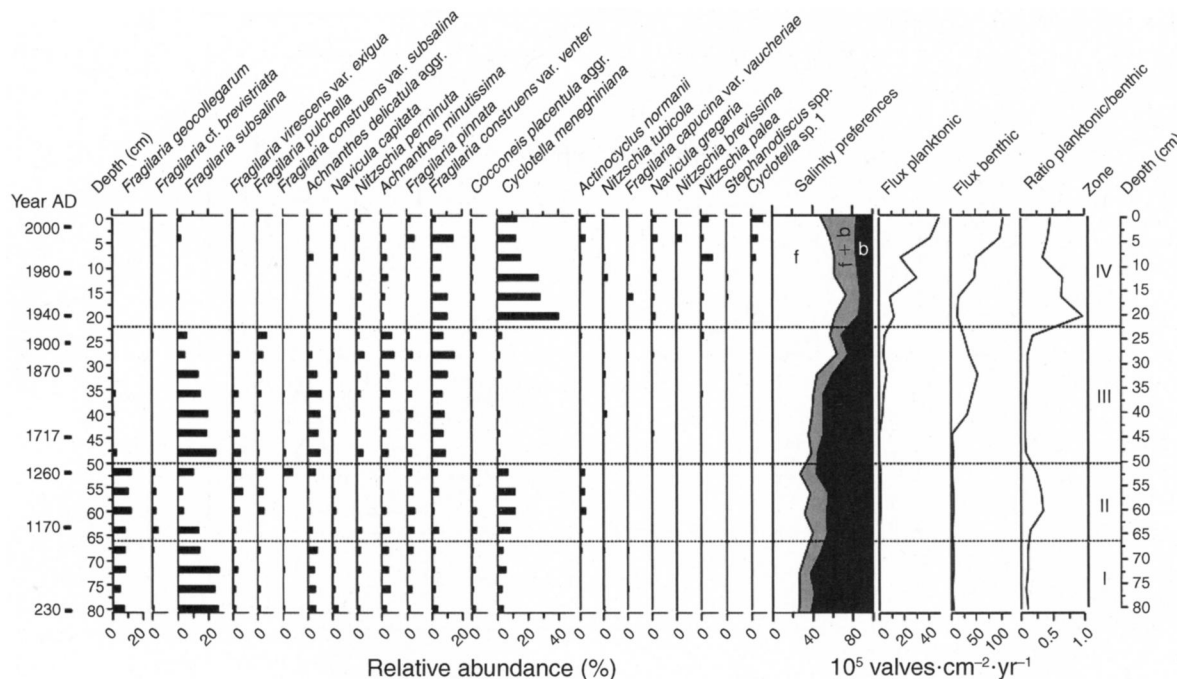


FIG. 4. Diatom stratigraphy of the Abby-1 core with salinity preferences and fluxes of planktonic and benthic species. Relative abundances of diatom species are ordered from left to right corresponding to their chronological appearance in the core from the past to the present. Major diatom zones are indicated by Roman numerals (right). Salinity preferences are shown as the proportion of freshwater (f), freshwater and brackish (f + b), and brackish water (b) species, represented by white, gray, and black areas, respectively. The numbers of planktonic and benthic diatoms are given as fluxes in units of $10^5 \text{ valves} \cdot \text{cm}^{-2} \cdot \text{yr}^{-1}$.

diatom assemblage. *F. subsalina* reappears in low abundances in the most recent samples.

The percentage of the diatom taxa characterized as freshwater species increased from 20% to 30% in Zone I to 62% in Zone II, followed by a decrease to 30% at the start of Zone III. Later in Zone III, freshwater taxa rose to a peak at 60% at depths of 24 cm and 28 cm, followed by declining relative proportions ranging between 30% and 45% in Zone IV. These changes are inversely proportional to the portion of brackish species for most of the core, except in the upper 15 cm, where species common to both freshwater and brackish water replaced some of the freshwater species.

The ordination results of the diatom data support the separation of four zones established by optimal partitioning (Fig. 5a). Two major excursions from the baseline conditions occurred from AD 1170 to 1265 (Zone II) and from AD 1920 to the present. Analysis of diatom–environment relationships with redundancy analysis (RDA) revealed that the major source of variation (eigenvalue 0.46) in diatom assemblage was accounted for by indicators of productivity such as total phosphorus, biogenic silica, percent carbon, and diatom flux, which loaded positively, and by diatom diversity, which loaded negatively (Fig. 5b). Variables loading on the second axis include sand content, presumed to increase with increasing land clearance, as well as with variation in flow. Pollen assemblage accounted for a

significant, but lower, proportion of variation, indicated by the eigenvalue of the second axis (0.21).

Pollen record

The earliest change in pollen abundance was exhibited by *Picea* sp. (spruce), which began to increase above its baseline percentages between AD 1265 and 1540 (Fig. 6). In the 17th century, pollen of *Pinus strobus* (white pine), *Tsuga canadensis* (eastern hemlock), and *Fagus grandifolia* (American beech) declined in relative abundance. Coincident with these decreases in tree pollen was a decline in Poaceae pollen in the 32–38 μm size class, which we assume to be *Zizanea* (wild rice) because of cell size (Lee et al. 2004), its continuous presence throughout the core, and the vigorous population of *Zizanea* in Merrymeeting Bay today. Pollen records representing terrestrial landscapes in Maine show little grass pollen prior to land clearance (see Russell et al. 1993). Along with *Rumex* and *Ambrosia*, alder (*Alnus* sp.) and hazelnut (*Corylus* sp.) increase at the time of local land clearance. Pollen of the aquatic macrophyte *Isoetes* (quillwort) also appears in the record at AD 1717, rising to a peak at AD 1880, followed by a sudden disappearance. However, this increase was not observed in the sediment core from Butler Cove, suggesting that the *Isoetes* rise in Abby-1 was a local phenomenon. Spruce and hemlock pollen declined after AD 1900, and *Zizanea* pollen increased to pre-AD 1540 levels.

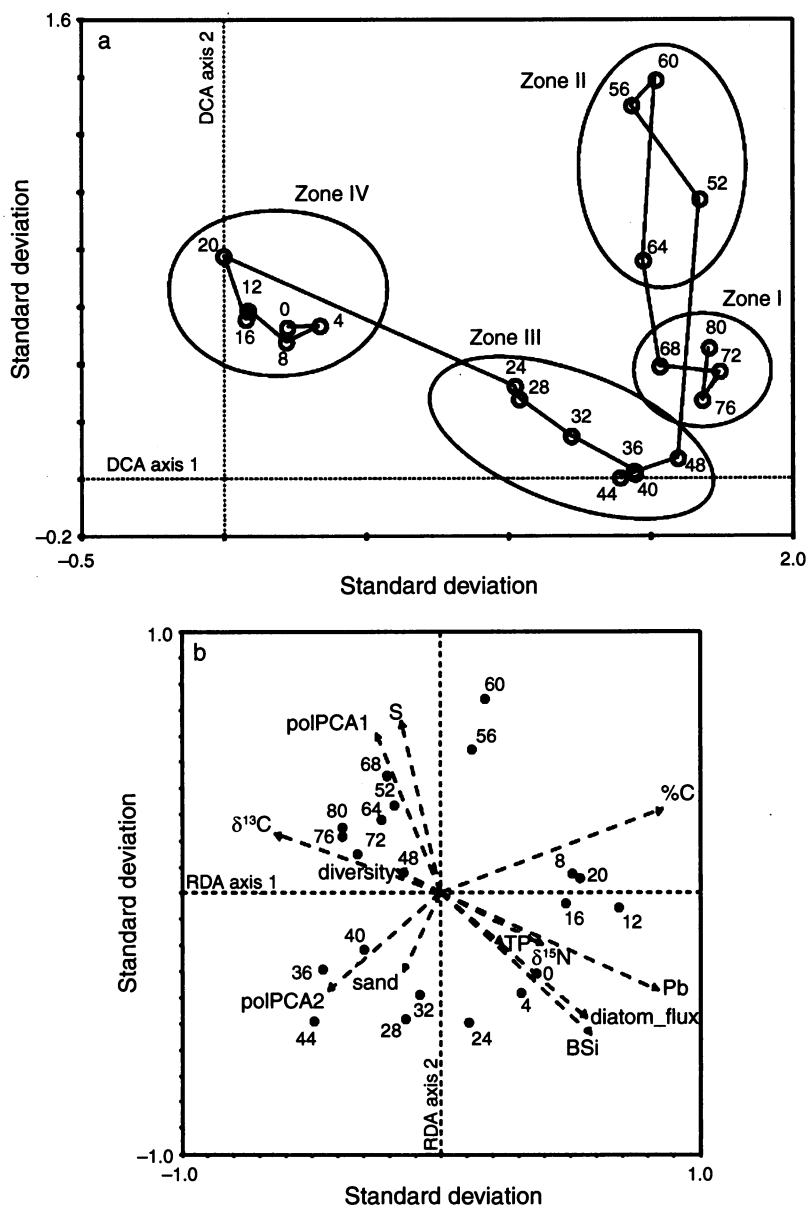


FIG. 5. Results of ordination analyses to discriminate patterns in fossil diatom assemblages and to evaluate relationships between variation in diatom assemblages and environmental variables: (a) detrended correspondence analysis plot showing four diatom zones consistent with results of optimal partitioning (Fig. 4); (b) redundancy analysis (RDA) plot showing that indicators of productivity (i.e., total phosphorus [TP], biogenic silica [BSi], organic C, and diatom flux) accounted for much of the variation in diatom assemblage and that indicators of land clearance accounted for a lesser proportion of the variation in diatom assemblage. polPCA1 and polPCA2 summarize the main patterns in fossil pollen data. They represent the sample scores of the samples on axes 1 and 2, respectively, on a PCA run with the fossil pollen data only.

Sediment chemistry

Changes in C and N content and in ^{13}C and ^{15}N isotopic composition observed between AD 1170 and 1265, and again during the 19th and 20th centuries (Fig. 7), corresponded to diatom zone changes (Fig. 4). Between AD 1170 and 1265, C and N percentage, organic C flux, total P flux, biogenic Si flux, and diatom abundance indicate greater primary productivity than during the time periods immediately before and after.

After AD 1540, sediment C and N percentages declined coinciding with a change to lighter colored sediments that lasted into the 20th century. Around AD 1880, organic C, total P, biogenic Si, and diatom abundance rose rapidly to their highest values in the 1800-year record. Stable isotopes also changed directionally with decreasing $\delta^{13}\text{C}$, indicating proportionally more of the lighter isotope. Conversely, increasing $\delta^{15}\text{N}$ indicated proportionally more of the heavy isotope.

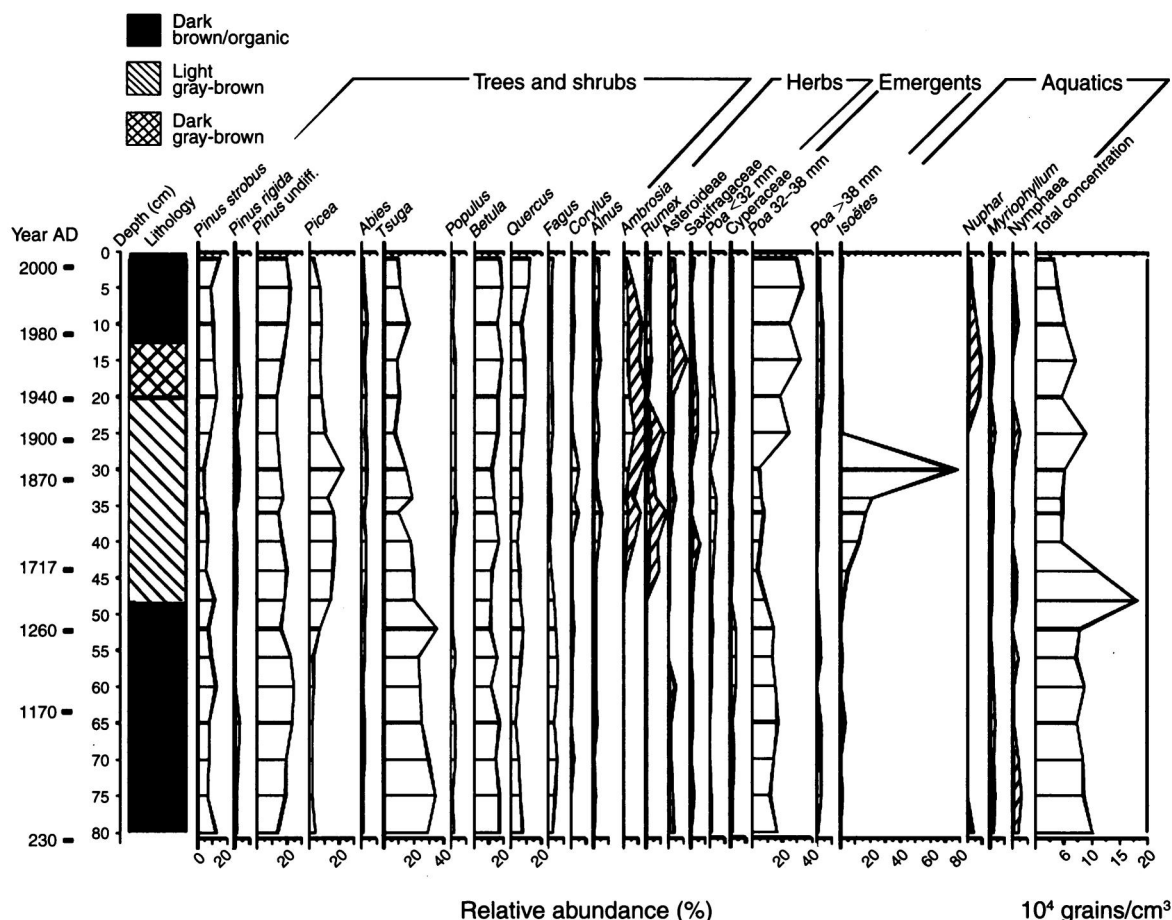


FIG. 6. Relative pollen abundances for the Abby-1 core. The total upland pollen count comprises tree, shrub, and herbaceous species. Abundances of aquatic and emergent species are calculated relative to total upland pollen. Hatched areas are exaggerated by 10 times.

DISCUSSION

Natural climatic variability, as well as human land use, influenced the sedimentary record of Merrymeeting Bay. Climate variability is evident in the episode of accelerated sedimentation and change in diatom abundances in the 12th and 13th centuries and in a longer period of climatic cooling coincident with the Little Ice Age (i.e., AD 1500–1900). Human land-use effects began with European settlement in the 17th or early 18th century and intensified with industrialization and municipal growth in the late 19th and 20th centuries. The diatom record showed little evidence of ecosystem recovery following implementation of environmental legislation in the 1960s and 1970s. These results are consistent with regional vegetation histories indicating broad-scale climate trends in New England (e.g., Russell et al. 1993, Fuller et al. 1998, Pederson et al. 2005), historical studies documenting eutrophication in estuaries throughout the 19th and 20th centuries (Cooper and Brush 1993, Cooper 1995, Cooper et al. 2004), and with biogeochemical studies quantifying rising nutrient con-

centrations in rivers and estuaries throughout the latter 20th century (Jaworski et al. 1997, Cloern 2001).

Natural climatic variability is evident in changes in sedimentation rate, diatom abundance and composition, the ratio of planktonic to benthic diatoms, and the pollen record. Between AD 1170 and 1265, sedimentation increased above background levels, and the abundances of several diatom taxa varied from previous centuries. An increase in the ratio of planktonic to benthic diatoms indicates increased turbidity associated with greater sediment influx and algal productivity (Cooper and Brush 1993). Pederson et al. (2005) interpret a similar episode of increased sedimentation in a Hudson River salt marsh, as indicative of regional drought during the Medieval Warm Period. Drought may increase tree mortality and fire frequency, leading to increased erosion and sedimentation. Between AD 1265 and 1540, spruce pollen increased from low relative percentages to amounts exceeding 10% of total tree pollen. Similar increases in the relative percentage of spruce pollen have been related to lower temperatures and higher precipitation in northeastern North America

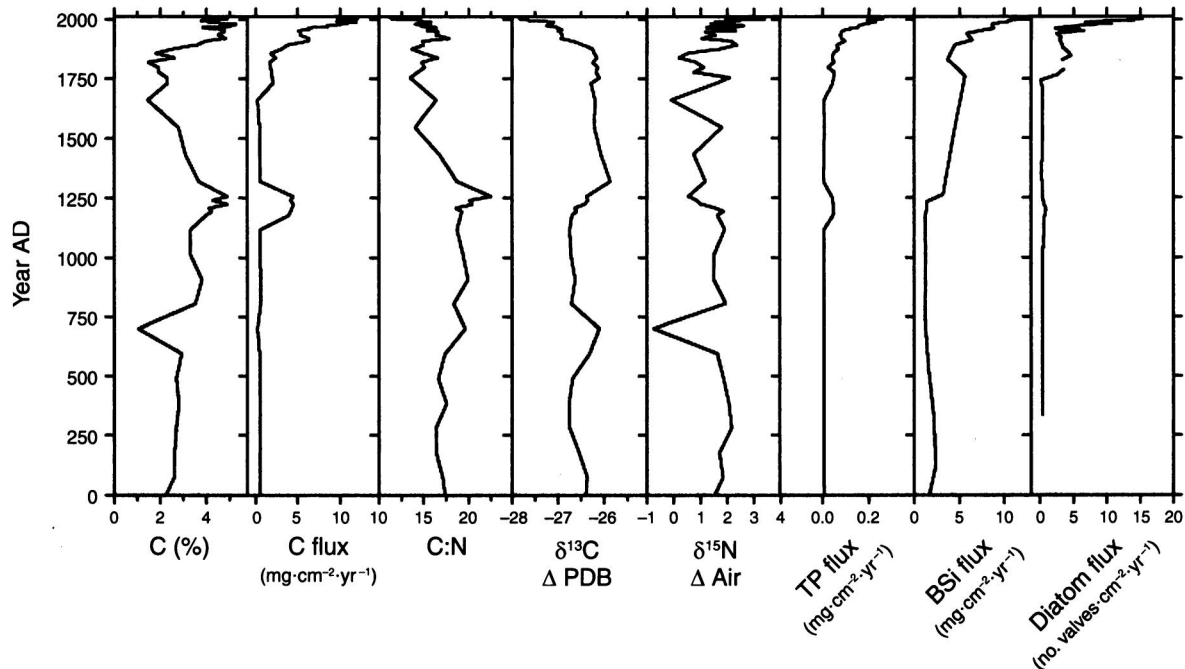


FIG. 7. Sediment chemistry for the Abby-1 core. From left to right, percentage carbon (%C), C flux, carbon to nitrogen ratio (C:N), ^{13}C and ^{15}N stable isotopes, total phosphorus (TP), biogenic silica (BSi), and diatom fluxes are illustrated. $\Delta = [R_{\text{sam}}/R_{\text{std}} - 1] \times 1000$, with R representing the $^{13}\text{C}/^{12}\text{C}$ and $^{15}\text{N}/^{14}\text{N}$ ratios in the sample and the PDB (Pee Dee Belemnite) and Air (air stable isotope ratio) standard, respectively. The numbers of planktonic and benthic diatoms are given as fluxes in units of $10000 \text{ valves} \cdot \text{cm}^{-2} \cdot \text{yr}^{-1}$.

(Russell et al. 1993, Schauffler and Jacobsen 2002, Lindbladh et al. 2003). Also at this time, *Zizanea* pollen declined in relative abundance suggesting a shorter ice-free growing season that reduced the establishment and growth of this annual grass on the intertidal mudflats.

The section of lighter colored sediments between 20 cm and 48 cm depths is related to greater inputs of mineral sediments relative to organic matter than in sediments above and below. The 95% confidence interval of the radiocarbon estimate of the timing of this change ranged from AD 1440 to 1640, suggesting that the initial increase in sedimentation was either associated with land-use changes very early in the Colonial Era or with natural climate variability that destroyed vegetation and promoted erosion. English families began settling along the tidal Androscoggin and Kennebec rivers in the 1650s (Churchill 2001); however, there were few settlers before 1640. Alternatively, a cooler climate associated with the Little Ice Age may have increased the duration of winter ice cover, thereby decreasing vegetative growth and in situ organic matter production on the intertidal flats, resulting in the observed change in sediment color.

Colonial impacts on the landscape are unambiguous at 44 cm depth, where pollen indicators of land clearance first appear, sedimentation accelerates, and diatom productivity increases. Several studies have implicated land clearance and agriculture as the underlying causes of increased sedimentation in estuaries (e.g., Brush 1989, Neubauer et al. 2002, Cooper et al. 2004), although

accelerated sea-level rise along the Maine coast beginning around AD 1800 may also have contributed to increased sedimentation rates (Gehrels et al. 2002). Local historians reported that land-use changes along the lower Androscoggin and other rivers resulted in the filling in of Merrymeeting Bay with sand during the 18th century (McKeen 1853). Whereas the largest ships of the 17th century could navigate through Merrymeeting Bay and up the lower Androscoggin River to the falls at Brunswick, ships had to unload a part of their cargo at Merrymeeting Bay by the mid-19th century before proceeding up the Androscoggin because of the shallowness of its channel (Wheeler and Wheeler 1878). By AD 1800, benthic diatom productivity increased markedly along with a relative increase in the proportion of freshwater diatom taxa. Increased nutrient inputs and run-off from cleared land probably triggered these early changes in diatom composition (e.g., Cooper 1995).

A larger and more abrupt change in diatom assemblage and sediment chemistry occurred at the end of the 19th century corresponding with population growth and accelerating industrial activity within the watershed (Babcock et al. 1995). Benthic diatoms indicative of polluted waters (e.g., *Navicula gregaria* and *Nitzschia palea*) became abundant, as did planktonic species often dominant in eutrophic waters (e.g., *Cyclotella meneghiniana* and *Actinocyclus normanii*). The productivity of benthic taxa declined temporarily between AD 1900 and 1940, probably because of increased turbidity associated

with pulp-mill effluents and frequent blooms of blue-green algae (Walker 1931, Maine Department of Environmental Protection 1979). Planktonic diatoms increased in both relative and absolute abundance throughout the 20th century, suggesting that nutrient loading has continued without reduction.

C flux to sediments supports the evidence of eutrophication in the diatom record. High organic content beginning in the mid-19th century and lasting throughout the 20th century indicates high primary productivity and probably increasing inputs of organic matter from industry and growing municipalities (Lichter et al. 2006). Although decomposition is likely to reduce the C content of the very recent sediments, the late 20th century produced the highest quantities of organic C throughout the sedimentary record. Since the mid-19th century, $\delta^{15}\text{N}$ has undergone an oscillating increase, while $\delta^{13}\text{C}$ has declined by $\sim 1.5\%$. Increasing $\delta^{15}\text{N}$ values are consistent with high productivity (Bratton et al. 2003) as well as wastewater input (Hodell and Schelske 1998). Declines in $\delta^{13}\text{C}$ in recent sediments of Chesapeake Bay have been interpreted as increases in terrestrial relative to marine sources of organic C associated with wetter climates (Bratton et al. 2003). However, the decline in C:N ratio of organic matter in the Merrymeeting Bay sediments indicates that algal productivity increased relative to terrestrial C inputs during the last century, which is supported by increases in diatom flux, total P, and biogenic Si. It is possible that the decline in sediment $\delta^{13}\text{C}$ resulted from a decrease in $\delta^{13}\text{C}$ of atmospheric CO_2 . The timing of the shift in the upper sediments is consistent with a decline in atmospheric $\delta^{13}\text{C}$ attributed to fossil fuel burning (Francey et al. 1999).

The evidence of eutrophication in Merrymeeting Bay conforms with historical studies of Chesapeake Bay (Cooper and Brush 1993, Zimmerman and Canuel 2002), New Bedford Harbor (Chmura et al. 2004), and of the Pamlico and Neuse estuaries (Cooper et al. 2004). Like these studies, indicators of eutrophication showed little evidence of ecosystem recovery following the Clean Water Act. Benthic diatom productivity increased after a period of low abundance between AD 1900 and 1970, suggesting that primary wastewater treatment improved water clarity. However, high sedimentation rate, high diatom productivity, high planktonic to benthic diatom ratio, and high levels of organic C, total P, and biogenic Si indicate that the ecosystem is more eutrophic than its long-term baseline conditions. The primary sources of N in coastal Maine are atmospheric deposition and municipal wastewater facilities (Roman et al. 2000), whereas the primary sources of P are emissions from paper mills (McCubbin Consultants 2003). It is also possible that P is recycled from sediments (Coelho et al. 2004). Fertilization experiments indicate that both N and P limit phytoplankton productivity within Merrymeeting Bay (Lichter, unpublished data), and P emissions from paper mills have been implicated as causes of upriver algal blooms (McCubbin Consultants 2003).

Implications for management and restoration

The accelerated sedimentation and cultural eutrophication that characterized the last two centuries of Merrymeeting Bay history had important consequences for the ecosystem. In addition to algal blooms, oxygen depletion, fish kill, and loss of biodiversity, increased turbidity is likely to have reduced the abundance of submerged aquatic vegetation (SAV) and consequently, altered the benthic food web. Aerial photographs show that SAV in Merrymeeting Bay currently inhabits $<5\%$ of shallows <3 m deep at low tide. Reductions in SAV have been attributed to increased turbidity associated with eutrophication and high sedimentation in the Hudson River estuary (Nieder et al. 2004), Chesapeake Bay (Orth and Moore 1984, Stevenson et al. 1993, Orth et al. 2002), the Potomac River (Carter and Rybicki 1986), and Waquoit Bay (Short and Burdick 1996). SAV provides vital habitat and food for invertebrates and fish (Menzie 1980, Catling et al. 1994, Fluharty 2000), food for waterfowl (Noordhuis et al. 2002), and beneficial ecosystem effects such as nutrient uptake, oxygen generation, and sediment stabilization (Barko et al. 1991, Carter and Rybicki 1991). As such, the loss of SAV may hinder efforts to restore populations of native anadromous fish such as American shad and striped bass because of the lack of nursery habitat and forage for juvenile fish.

Although the extent to which turbidity associated with ongoing nutrient loading interferes with the recovery of SAV in Merrymeeting Bay is unclear, the diatom record indicates that current nutrient inputs plus potential recycling of P from sediments are sufficient to maintain the high diatom productivity and altered community composition that characterized the era of raw pollution during the mid-20th century. This is somewhat enigmatic because region-wide analyses of nutrient-loading rates in New England estuaries indicate that the Kennebec estuary is one of the least impacted by nonpoint nutrient sources because the watersheds of the Androscoggin and Kennebec rivers are largely forested with little agriculture and few urban areas (Boyer et al. 2002). It is likely that prehistoric nutrient inputs and productivity were extremely low in the undisturbed watershed (Nixon 1997). Consequently, even the relatively low levels of anthropogenic nutrient inputs into the Androscoggin and Kennebec rivers may be sufficient to maintain the altered community composition and elevated productivity observed in the diatom record of Merrymeeting Bay.

Much research would be needed to evaluate the effects of ongoing nutrient loading on the water clarity of Merrymeeting Bay. There are particular areas within Merrymeeting Bay and its small tributaries that are characterized by excessive turbidity. In addition to high algal productivity, turbidity may result from the activity of introduced benthivorous fish such as carp (*Cyprinus carpio*; Zambrano et al. 2001) as well as tidal resuspension of sediments. However, should it be determined

that algal productivity strongly influences water turbidity, nutrient inputs to Merrymeeting Bay and its tributaries would have to be reduced. While primary and secondary wastewater treatments effectively eliminate organic matter, tertiary treatment is required to remove significant percentages of the nutrients in municipal wastewaters. Much of the P emitted by paper mills on the Androscoggin could also be eliminated as technology is available that would reduce P discharges by as much as 65% (McCubbin Consultants 2003). While it is true that environmental regulation can be costly, the opportunity cost of continuing to pollute rivers and estuaries should be considered in making management decisions. Because the paper industry constitutes an important source of economic activity in Maine and provides relatively high-paying jobs that are essential in rural areas, it wields considerable influence in the politics of environmental management. However, tourism, recreation, and shellfish fisheries are also vital components of the Maine economy, and all would benefit from sustained recovery of coastal ecosystems.

A shifting baseline may contribute to the political difficulty of legislating more stringent regulation of nutrient pollution. Public and institutional memory does not extend back in time far enough to recognize the potential for ecosystem recovery, only far enough to recognize the substantial improvement since the time when Merrymeeting Bay and its tributary rivers were described as open sewers. Public debate continues to focus on whether or not industry and municipalities meet the minimum standards, rather than what this unusual and important river–estuary complex could be and how to best manage it to achieve long-term restoration goals. Environmental history can inform debate about ecosystem management in light of economic tradeoffs by documenting the long-term baseline of the ecosystem, and thereby illustrate the upper limit of the potential for ecosystem recovery.

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LITERATURE CITED

- Allen, C. E. 1931. History of Dresden, Maine. Kennebec Journal Print Shop, Augusta, Maine, USA.
- Babcock, R. H., Y. Frenette, C. A. Scontras, and E. Eagan. 1995. Work and workers in the Industrial Age, 1865–1930. Pages 448–479 in R. W. Judd, E. A. Churchill, and J. W. Eastman, editors. Maine: The Pine Tree State from prehistory to the present. University of Maine Press, Orono, Maine, USA.
- Barko, J. W., D. Gunnison, and S. R. Carpenter. 1991. Sediment interactions with submersed macrophyte growth and community dynamics. *Aquatic Botany* 41:41–65.
- Baxter, J. P. 1916. The Baxter manuscripts. Documentary history of the State of Maine. Volume XXII. Portland, Maine Historical Society, Portland, Maine, USA.
- Binford, M. W. 1990. Calculation and uncertainty analysis of ^{210}Pb dates for PIRLA project lake sediment cores. *Journal of Paleolimnology* 3:253–267.
- Birks, H. J. B. 1995. Quantitative paleoenvironmental reconstructions. Pages 161–254 in D. Maddy and J. S. Brew, editors. Statistical modeling of Quaternary Science data. Quaternary Research Association, Cambridge, UK.
- Bourque, B. J. 2001. Twelve thousand years: American Indians in Maine. University of Nebraska Press, Lincoln, Nebraska, USA.
- Boyer, E. W., C. L. Goodale, N. A. Jaworski, and R. W. Howarth. 2002. Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U.S.A. *Biogeochemistry* 57:137–169.
- Bratton, J. F., S. M. Colman, and R. R. Seal. 2003. Eutrophication and carbon sources in Chesapeake Bay over the last 2700 yr: human impacts in context. *Geochimica Et Cosmochimica Acta* 67:3385–3402.
- Brush, G. S. 1989. Rates and patterns of estuarine sediment accumulation. *Limnology and Oceanography* 34:1235–1246.
- Carter, V., and N. B. Rybicki. 1990. Light attenuation and submersed macrophyte distribution in the tidal Potomac River. *Estuaries* 13:441–452.
- Catling, P. M., K. W. Spicer, M. Biernacki, and J. L. Doust. 1994. The biology of Canadian weeds. 103. *Vallisneria spiralis* Michx. *Canadian Journal of Plant Science* 74: 883–897.
- Chase, E. S. 1949. Pollution of the Androscoggin River by industrial wastes and control measures thereof. *Boston Society of Civil Engineering* 36:357–381.
- Chmura, G. L., A. Santos, V. Pospelova, Z. Spasojevic, R. Lam, and J. S. Latimer. 2004. Response of three paleo-primary production proxy measures to development of an urban estuary. *Science of the Total Environment* 320:225–243.
- Churchill, E. A. 2001. English beachheads in Seventeenth-Century Maine. Pages 51–75 in R. W. Judd, E. A. Churchill, and J. W. Eastman, editors. Maine: The Pine Tree State from prehistory to the present. University of Maine Press, Orono, Maine, USA.
- Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210:223–253.
- Coelho, J. P., M. R. Flindt, H. S. Jensen, A. I. Lillebo, and M. A. Pardal. 2004. Phosphorus speciation and availability in intertidal sediments of a temperate estuary: relation to eutrophication and annual P-fluxes. *Estuarine, Coastal and Shelf Science* 61:583–590.
- Cooper, S. R. 1995. Chesapeake Bay watershed historical land-use–impact on water quality and diatom communities. *Ecological Applications* 5:703–723.
- Cooper, S. R., and G. S. Brush. 1993. A 2500-year history of anoxia and eutrophication in Chesapeake Bay. *Estuaries* 16: 617–626.
- Cooper, S. R., S. K. McGlothlin, M. Madritch, and D. L. Jones. 2004. Paleoecological evidence of human impacts on the Neuse and Pamlico Estuaries of North Carolina, USA. *Estuaries* 27:617–633.
- Crusius, J., and R. F. Anderson. 1995. Evaluating the mobility of Cs-137, Pu-239+240, and Pb-210 from their distributions in laminated lake sediments. *Journal of Paleolimnology* 13: 119–141.
- Cwynar, L. C., E. Burden, and J. H. McAndrews. 1979. An inexpensive sieving method for concentrating pollen and

- spores from fine-grained sediments. *Canadian Journal of Earth Sciences* 16:1115–1120.
- Davis, R. B., C. T. Hess, S. A. Norton, D. W. Hanson, K. D. Hoagland, and D. S. Anderson. 1984. ^{137}Cs and ^{210}Pb dating of sediments from soft-water lakes in New England (USA) and Scandinavia, a failure of Cs dating. *Chemical Geology* 44:151–185.
- Faegri, K., P. E. Kaland, and K. Krzywinski. 1989. Textbook of pollen analysis. Fourth edition. J. Wiley and Sons, New York, New York, USA.
- Fluharty, D. 2000. Habitat protection, ecological issues, and implementation of the Sustainable Fisheries Act. *Ecological Applications* 10:325–337.
- Flynn, W. W. 1968. Determination of low levels of polonium-210 in environmental materials. *Analytical Chimica Acta* 43: 221–227.
- Foster, D. R., G. Motzkin, and B. Slater. 1998. Land-use history as long-term broad-scale disturbance: regional forest dynamics in central New England. *Ecosystems* 1:96–119.
- Foster, D. R., F. Swanson, J. Aber, I. Burke, N. Brokaw, D. Tilman, and A. Knapp. 2003. The importance of land-use legacies to ecology and conservation. *Bioscience* 53:77–88.
- Foster, N. W., and C. G. Atkins. 1869. Reports of the Commissioners of Fisheries of the State of Maine for the years 1867 and 1868. Owen and Nash, Augusta, Maine, USA.
- Francey, R. J., C. E. Allison, D. M. Etheridge, C. M. Trudinger, I. G. Enting, M. Leuenberger, R. L. Langenfelds, E. Michel, and L. P. Steele. 1999. A 1000-year high precision record of $\delta^{13}\text{C}$ in atmospheric CO_2 . *Tellus* 51B: 170–193.
- Fuller, J. L., D. R. Foster, J. S. McLachlan, and N. Drake. 1998. Impact of human activity on regional forest composition and dynamics in central New England. *Ecosystems* 1:76–95.
- Gehrels, W. R., D. F. Belknap, S. Black, and R. M. Newnham. 2002. Rapid sea-level rise in the Gulf of Maine, USA, since AD 1800. *The Holocene* 12:383–389.
- Ghere, D. L. 1995. Diplomacy and war on the Maine frontier, 1678–1759. Pages 120–142 in R. W. Judd, E. A. Churchill, and J. W. Eastman, editors. *Maine: The Pine Tree State from prehistory to the present*. University of Maine Press, Orono, Maine, USA.
- Hodell, D. A., and C. L. Schelske. 1998. Production, sedimentation, and isotopic composition of organic matter in Lake Ontario. *Limnology and Oceanography* 43:200–214.
- Jackson, J. B. C., et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293:629–638.
- Jaworski, N. A., R. W. Howarth, and L. J. Hetling. 1997. Atmospheric deposition of nitrogen oxides onto the landscape contributes to coastal eutrophication in the Northeast United States. *Environmental Science and Technology* 31: 1995–2004.
- Johengen, T. 1996. Standard operating procedures for determining total phosphorus, available phosphorus, and biogenic silica concentrations of Lake Michigan sediments and sediment trap material. NOAA/Great Lakes Environmental Research Lab, Ann Arbor, Michigan, USA.
- Kistner, D. A., and N. R. Pettigrew. 2001. A variable turbidity maximum in the Kennebec estuary, Maine. *Estuaries* 24:680–687.
- Krammer, K., and H. Lange-Bertalot. 1986. *Bacillariophyceae*, 1. Teil: Naviculaceae. Gustav Fischer Verlag, Stuttgart, Germany.
- Krammer, K., and H. Lange-Bertalot. 1988. *Bacillariophyceae*, 2. Teil: Bacillariaceae, Epithemiaceae, Surirellaceae. Gustav Fischer Verlag, Stuttgart, Germany.
- Krammer, K., and H. Lange-Bertalot. 1991a. *Bacillariophyceae*, 3. Teil: Centrales, Fragilariaceae, Eunotiaceae. Gustav Fischer Verlag, Stuttgart, Germany.
- Krammer, K., and H. Lange-Bertalot. 1991b. *Bacillariophyceae*, 4. Teil: Achnantheaceae. Gustav Fischer Verlag, Stuttgart, Germany.
- Lawrance, W. A. 1967. A twenty-five year review of Androscoggin River pollution control activities. A report to the Androscoggin River Technical Committee. Sanitary Water Board, State of Maine, Lewiston, Maine, USA.
- Lee, G.-A., A. M. Davis, D. G. Smith, and J. H. McAndrews. 2004. Identifying fossil wild rice (*Zizanea*) pollen from Cootes Paradise, Ontario: a new approach using scanning electron microscopy. *Journal of Archeological Science* 31: 411–421.
- Lichter, J., H. Caron, T. S. Pasakarnis, S. L. Rodgers, T. S. Squiers, Jr., and C. S. Todd. 2006. The ecological collapse and partial recovery of a freshwater tidal ecosystem. *Northeastern Naturalist* 13:153–178.
- Lindbladh, M., G. L. Jacobson, Jr., and M. Schaffner. 2003. The postglacial history of three *Picea* species in New England, USA. *Quaternary Research* 59:61–69.
- Lotter, A., and S. Juggins. 1991. POLPROF, TRAN and ZONE: Programs for plotting, editing and zoning pollen and diatom data. *INQUA Commission for the Study of the Holocene Newsletter* 6:17–24.
- Lotze, H. K., and I. Milewski. 2004. Two centuries of multiple human impacts and successive changes in a North Atlantic food web. *Ecological Applications* 14:1428–1447.
- Maine Department of Environmental Protection. 1971. Kennebec Basin interim water quality plan. Environmental Improvement Commission. Augusta, Maine, USA.
- Maine Department of Environmental Protection. 1979. The Kennebec River, 1978 water quality, a historic perspective. Maine Department of Environmental Protection, Augusta, Maine, USA.
- Maine Department of Environmental Protection. 2000. Kennebec River modeling report. Bureau of Land and Water Quality, Division of Environmental Assessment, Augusta, Maine, USA.
- McCubbin Consultants. 2003. Current technology for control of phosphorus and BOD discharges in effluents from three Kraft pulp mills on the Androscoggin River. Report prepared for the State of Maine Department of Environmental Protection. Augusta, Maine, USA.
- McKeen, J. 1853. Some accounts of the early settlements at Sagadahock and on the Androscoggin river. Article V. Collection of the Maine Historical Society V. III. Portland, Maine, USA.
- Menzie, C. A. 1980. The chironomid (Insecta: Diptera) and other fauna of a *Myriophyllum spicatum* L. plant bed in the lower Hudson River. *Estuaries* 3:38–54.
- Milan, C. S., E. M. Swenson, R. E. Turner, and J. M. Lee. 1995. Assessment of the Cs-137 method for estimating sediment accumulation rates—Louisiana Salt Marshes. *Journal of Coastal Research* 11:296–307.
- Neubauer, S. C., I. C. Anderson, J. A. Constantine, and S. A. Kuehl. 2002. Sediment deposition and accretion in a mid-Atlantic (USA) freshwater marsh. *Estuarine, Coastal and Shelf Science* 54:713–727.
- Nieder, W. C., E. Barnaba, S. E. G. Findlay, S. Hoskins, N. Holochuck, and E. A. Blair. 2004. Distribution and abundance of submerged aquatic vegetation and *Trapa natans* in the Hudson River estuary. *Journal of Coastal Research Special Issue* 45:150–161.
- Nixon, S. W. 1997. Prehistoric nutrient inputs and productivity in Narragansett Bay. *Estuaries* 20:253–261.
- Noordhuis, R., D. T. van der Molen, and M. S. van der Berg. 2002. Response of herbivorous water-birds to the return of Chara in Lake Veluwemeer, The Netherlands. *Aquatic Botany* 72:349–367.
- North, J. W. 1870. The history of Augusta. Clapp and North, Augusta, Maine, USA.

- Odum, W. E., T. J. Smith, J. K. Hoover, and C. C. McIvor. 1984. The ecology of tidal freshwater marshes of the United States east coast: a community profile. FWS/OBS-83117. U.S. Fish and Wildlife Service, Washington, D.C., USA.
- Orth, R. J., R. A. Batiuk, P. W. Bergstrom, and K. A. Moore. 2002. A perspective on two decades of policies and regulations influencing the protection and restoration of submerged aquatic vegetation in Chesapeake Bay, USA. *Bulletin of Marine Science* 71:1391–1403.
- Orth, R. J., and K. A. Moore. 1984. Chesapeake Bay: an unprecedented decline in submerged aquatic vegetation. *Science* 222:51–53.
- Owen, H. W., Jr. 1936. The Edward Clarence Plummer history of Bath, Maine. The Times Company, Bath, Maine, USA.
- Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. *Trends in Ecology and Evolution* 10:430.
- Pederson, D. C., D. M. Peteet, D. Kurdyla, and T. Guilderson. 2005. Medieval warming, Little Ice Age, and European impact on the environment during the last millennium in the lower Hudson Valley, New York, USA. *Quaternary Research* 63:238–249.
- Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegard, B. D. Richter, R. E. Sparks, and J. S. Stromberg. 1997. The natural flow regime. *Bioscience* 47: 769–784.
- Roman, C. T., N. Jaworski, F. T. Short, S. Findlay, and R. S. Warren. 2000. Estuaries of the northeastern United States: habitat and land use signatures. *Estuaries* 23:743–764.
- Russell, E. W. B., R. B. Davis, R. S. Anderson, T. E. Rhodes, and D. S. Anderson. 1993. Recent centuries of vegetational change in the glaciated north-eastern United States. *Journal of Ecology* 81:647–664.
- Schaffler, M., and G. L. Jacobson. 2002. Persistence of coastal spruce refugia during the Holocene in northern New England, USA, detected by stand-scale pollen stratigraphies. *Journal of Ecology* 90:235–250.
- Short, F. T., and D. M. Burdick. 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. *Estuaries* 19:730–739.
- Steinberg, T. 1991. *Nature incorporated: industrialization and the waters of New England*. Cambridge University Press, Cambridge, UK.
- Stevenson, J. C., L. W. Staver, and K. W. Staver. 1993. Water quality associated with survival of submersed aquatic vegetation along an estuarine gradient. *Estuaries* 16:346–361.
- Sutermeyer, E. 1941. *Chemistry of pulp and paper making*. John Wiley and Sons, New York, New York, USA.
- ter Braak, C. J. F., and P. Smilauer. 2002. *CANOCO reference manual and CanoDraw for Windows user's guide: software for canonical community ordination (v. 4.5)*. Microcomputer Power, Ithaca, New York, USA.
- Vandergoes, M. J. 2000. High resolution record of late Quaternary vegetation and climate change, south Westland, New Zealand. Dissertation. University of Otago, Dunedin, New Zealand.
- Walker, C. L. 1931. Survey and report of river and stream conditions in the State of Maine 1930. Biennial Report to the Governor. State of Maine, Augusta, Maine, USA.
- Water Improvement Commission. 1960. The Kennebec River water classification report. Augusta, Maine, USA.
- Water Improvement Commission. 1966. Addendum to 1960 Kennebec water classification report. Augusta, Maine, USA.
- Wells, W. 1869. *Water-power of Maine*. Hydrographic survey of Maine. Sprague, Owen, and Nash, Augusta, Maine, USA.
- Wheeler, G. A., and H. W. Wheeler. 1878. *History of Brunswick, Topsham, and Harpswell, Maine*. Alfred Mudge and Son, Boston, Massachusetts, USA.
- Witkowski, A., H. Lange-Bertalot, and D. Metzeltin. 2000. *Diatom flora of marine coasts 1*. Koeltz Scientific Books, Königstein, Germany.
- Wong, M. W., and D. W. Townsend. 1999. Phytoplankton and hydrography of the Kennebec estuary, Maine, USA. *Marine Ecology Progress Series* 178:133–144.
- Zambrano, L., M. Scheffer, and M. Martinez-Ramos. 2001. Catastrophic response of lakes to benthivorous fish introduction. *Oikos* 94:344–350.
- Zimmerman, A. R., and E. A. Canuel. 2000. A geochemical record of eutrophication and anoxia in Chesapeake Bay sediments: anthropogenic influence on organic matter composition. *Marine Chemistry* 69:117–137.
- Zimmerman, A. R., and E. A. Canuel. 2002. Sediment geochemical records of eutrophication in the mesohaline Chesapeake Bay. *Limnology and Oceanography* 47:1084–1093.