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The impact of the zebra mussel (*Dreissena polymorpha*) on the limnology, geochemistry and sedimentology of Seneca Lake, New York

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Abstract

Seneca Lake is very similar to and provides a useful analog for Cayuga Lake. We present ongoing limnological and preliminary hydrogeochemical data to assess the impact of zebra mussels (*Dreissena polymorpha*) on the limnology, hydrochemistry and sedimentology of Seneca Lake. The data includes lake water analysis for dissolved reactive phosphate, dissolved reactive silica, nitrate, chlorophyll-a, plankton enumerations, and Secchi disc depths; stream analysis for calcium and nutrient fluxes; and sediment analysis for calcium fluxes to the lake floor.

Previous presentations based on pre-1998 limnological data discussed the dramatic increase in water clarity (Secchi disc depths increased from a few meters to almost 10 m), and decrease in plankton biomass (chlorophyll-a concentrations dropped from a few $\mu\text{g L}^{-1}$ to under $1 \mu\text{g L}^{-1}$) since the introduction of the exotic bivalve in 1992. However, these trends reversed direction during 1998 and the reversal continued through 1999 (e.g., 1998 Secchi disc depths average 6 m). We attribute the reversal to an increase in nutrient recycling of dead zebra mussels and their accumulated feces since 1997 although other factors may be important. This annual-scale variability highlights the importance of continued, long-term monitoring to assess the changing environmental status of Seneca and Cayuga Lakes.

Hydrochemical data focused on quantifying the inputs to, outputs from and concentration of calcium in the lake. We assumed that groundwater flow was negligible, and our selected streams and sediment samples are representative of the watershed. Our preliminary budget indicates that zebra mussels remove approximately 200 metric tons of calcium year⁻¹, equivalent to 5% of the annual calcium flux to the lake floor. This is consistent with an observed decrease in bulk carbonate content within the uppermost (<1 cm) sediments.

Introduction

The Finger Lakes of central New York State occupy 11 north-south trending elongated basins. Seneca and Cayuga Lakes comprise an important end-member of the Finger Lakes because they are very similar physiographically, limnologically, hydrochemically and sedimentologically. Combined, these two lakes contain approximately 80% of the water and 60 % of the surface area and drainage area among the 11 Finger Lakes. They are also the deepest Finger Lakes, with maximum water depths of 186 and 132 meters, respectively. The trophic status of both is borderline oligotrophic/mesotrophic with Cayuga slightly more productive along its southern shore near Ithaca, NY (e.g., Schaffner and Oglesby, 1978). Both contain higher concentrations of chloride and sodium than the other Finger Lakes due to their great depth and groundwater communication with evaporite bedrock beneath the lake floor (Wing et al., 1995). Finally, both basins are filled with over 200 meters of primarily glacial sediments with a thin (few meters) overlying veneer of postglacial muds (Mullins et al., 1996; Halfman and Herrick, 1998). These and other similarities indicate that Seneca Lake is an ideal analogue and benchmark for comparison to limnological, hydrochemical, sedimentological and other processes in Cayuga Lake.

These lakes are also an outstanding resource for the local area. For example, Seneca Lake provides Class AA drinking water for over 70,000 residents. It also has a tremendous impact on the local economy through a \$100 million tourist industry and the lake-influenced tax base of over \$1 billion. Thus, steps should be taken to preserve and protect the lake. The watershed is also an ideal natural laboratory because 29 subwatersheds, some combining a number of very small tributaries, drain into the lake. Each subwatershed drains different soil types, bedrock, landuse practices, and surface areas. The variability provides a mechanism to assess the source of various chemical constituents, including pollutants (e.g., Halfman et al., in press).

Zebra mussels (*Dreissena polymorpha*) were first detected in Seneca Lake (and presumably Cayuga Lake as well) during the late summer of 1992. Soon afterwards, zebra mussels colonized most of the suitable substrates throughout the lake. A field program was initiated just prior to the arrival of zebra mussels to assess its impact on the limnology and hydrochemistry of Seneca Lake at Hobart and William Smith Colleges through class projects and independent study endeavors (e.g., Acquisto, 1995). Since then, the number of analyzed parameters and frequency

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of cruises have increased to the present day. This paper assesses the impact of zebra mussels on the lake based on the available limnological data from this decade. Preliminary calcium mass-balance calculations are also presented to assess the impact of zebra mussels on the hydrochemistry and sedimentology of the lake. Our ultimate goal is to provide a benchmark for comparison with Cayuga Lake, and the pending environmental focus on its watershed.

Methods

The current water quality and limnological monitoring program at Hobart and William Smith Colleges (HWS) utilizes undergraduate students to collect and analyze water samples from primarily four offshore sites (Sites 1 – 4) within the northern portion of the lake (Fig. 1). These four sites maximize the diversity in the lake while minimizing the length of the field day. In 1998, monthly data were also collected from an additional five sites along the length of the lake (Sites 5 – 9, Fig. 1). The sample frequency increased over the past decade from occasional cruises during the year to our present-day weekly cruises during the non-winter months. The fieldwork uses the Colleges' 65-ft research vessel, the HWS Explorer.

Six creeks were also sampled since the early spring of 1995 for water discharge, calcium hardness, nutrient concentrations (phosphate, nitrate and silica) and other water quality parameters in conjunction with classroom and independent study projects. Calcium concentrations were analyzed on site, and stream water was brought back to the lab for filtration and analysis of the dissolved nutrient concentrations. The selected creeks represent an array of subwatershed surface areas from 40 to 100 km², landuse practices from primarily agricultural to primarily forested landscapes, and different soil / bedrock lithologies, which allow comparisons to investigate the impact of these characteristics on the flux of calcium to the lake.

At each lake site, a Secchi disc depth was measured on the shady side of the vessel and surface water was collected and analyzed for nutrients (dissolved phosphates, dissolved silica and nitrates), chlorophyll-a, calcium hardness, and plankton enumerations. We also routinely analyzed lake water for pH, chloride, alkalinity, total suspended solids, and dissolved oxygen concentrations. Bottom water was collected from approximately 2 meters above the lake floor in a 10-Liter Nansen bottle and analyzed for the same parameters (Sites 1, 3, 6 – 9). In addition, a water column profile of conductivity, temperature, pH (SBE 18), dissolved oxygen (SBE 23Y), and water transparency (Sea Tech 25-cm Transmissometer) was collected at each station using a Sea Bird Electronics CTD (Model SBE-19). In all, over 700 lake water samples and over 250 stream samples were analyzed during the past decade.

The water was analyzed following standard limnological procedures (e.g., Parsons et al., 1984, Wetzel and Likens, 1991). Soluble reactive phosphate was analyzed in the lab at 885 nm after reacting field filtered (HA Millipore 0.45 micron filter) water samples with a molybdate and trivalent antimony mixture. Samples are measured relative to potassium dihydrogen phosphate standard using a 10-cm cell, and reported in $\mu\text{g L}^{-1}$ - P. Soluble reactive silica was analyzed in the lab at 810 nm after reacting field filtered (HA Millipore 0.45 micron filter) water samples with a molybdate mixture. Samples were measured relative to a silicofluoride standard using a 1-cm cell, and reported in $\mu\text{g L}^{-1}$ Si. Nitrates were measured on unfiltered water on site with a Hach Low Range Nitrate Kit (Model NI-14) and reported in $\mu\text{g L}^{-1}$ N. Chlorophyll-a samples were determined by filtering approximately 1 L of sample water through a HA Millipore (0.45 μm pore diameter) filter in the field. The filtrate was used for the nutrient analyses. Each filter was subsequently dissolved in acetone and analyzed in the lab following the trichromatic method using a 1-cm cell (reported in $\mu\text{g L}^{-1}$). Calcium hardness was measured with a LaMotte Student Titration Kit (Model 4824-DR), and reported in ppm (mg L^{-1}) Ca. Plankton was collected with an 80- μm plankton net towed horizontally along the surface of the lake. At least 100 individuals were identified to genus, and reported as the relative percentage of the identified genera. Colonies were counted as a single individual. Some of these parameters were not measured on each cruise, especially earlier in the decade.

Precision was estimated seasonally by analyzing five samples from the same site. One standard deviation of the results for phosphate, silica, nitrate, chlorophyll-a, calcium hardness, Secchi disc depths, and plankton enumerations never exceeded 0.01 $\mu\text{g L}^{-1}$, 5.0 $\mu\text{g L}^{-1}$, 6.0 $\mu\text{g L}^{-1}$, 0.5 $\mu\text{g L}^{-1}$, 2.0 $\mu\text{g L}^{-1}$, 0.5 m, and 1.0%, respectively.

Over 50 grab samples were collected over the past decade from primarily the northern half of the lake. If present, approximately 10 to 20 grams of postglacial mud was subsampled from the surface 3 to 4 cm and analyzed for the calcium in the authigenic carbonate by loss-on-ignition (Dean, 1974). The results were reported as grams of calcium per gram of dry mud. All zebra mussel carapaces recovered by a dredge were removed, saved and subsequently analyzed in the lab for the equivalent mass of calcium in the shells by loss-on-ignition and reported as grams of calcium per dredge sample. The precision of the technique, based on triplicate analyses, was 0.1%. A

water depth of 50 meters typically divides Seneca Lake into deeper areas where postglacial muds accumulate and shallower areas where zebra mussels colonize.

Zebra mussel impact on Seneca Lake limnology

Significant changes are observed in the limnological data over the past decade (Table 1, Figs. 2a – e) beyond the expected seasonal variability. Two important multiyear trends exist. The first trend starts in 1992 and continues through the end of 1997. The second trend starts in 1998 and continues through the end 1999. Annual average Secchi disc depths increased from 4 m in the early 1990's to over 8 m by the end of 1997, then decreased to 6 to 7 m through 1999. Chlorophyll-a concentrations decreased from an annual average of $4.5 \mu\text{g L}^{-1}$ in the early 1990's to $0.6 \mu\text{g L}^{-1}$ by 1997, and then returned to 1.5 and $1.2 \mu\text{g L}^{-1}$ in 1998 and 1999. Yearly average dissolved silica concentrations increased from near $100 \mu\text{g L}^{-1}$ to $1400 \mu\text{g L}^{-1}$ in 1997, to subsequently return to below $220 \mu\text{g L}^{-1}$ by the end of 1999. Nitrate and phosphate data reveal small changes from 1992 through 1997 although the nitrate data are sparse. Both nutrient concentrations significantly increase (more than double) after 1997 through the end of 1999. The annual average concentrations changed from $140 \mu\text{g L}^{-1}$ before 1998 to 210 or $280 \mu\text{g L}^{-1}$ afterwards for nitrate, and from below $0.5 \mu\text{g L}^{-1}$ before 1998 to 2.4 and $3.3 \mu\text{g L}^{-1}$ afterwards for phosphate.

The 1992 through 1997 increase in water clarity (Secchi disc data) and decrease in phytoplankton biomass (chlorophyll-a data) is attributed to increased grazing by the growing population of the filter-feeding exotic bivalve. In Seneca Lake, Secchi disc data are primarily a function of plankton biomass (Fig. 3) but are also influenced by occasional zooplankton blooms and suspended sediments a day or so after significant precipitation events (Halfman et al., in press). Counts of zebra mussels recovered by sediment grab samples in water depths shallower than 50 m in the lake reveal increasing densities of zebra mussels from 1996 to 1998, consistent with our interpretation (Halfman, unpublished data). Similar water quality changes are observed elsewhere in North America that are attributed to the increased grazing pressures by growing zebra mussel populations (e.g., Reeders et al., 1989, Herbert et al., 1991, MacIsaac et al., 1992, Nalepa and Fahnenstiel, 1995, Fahnenstiel et al., 1995, Caraco et al., 1997).

Trends in the nutrient concentrations from 1992 through 1997 do not co-vary in the lake. The dissolved silica concentrations significantly increase over this time period. The annual mean nitrate concentrations decrease and phosphate concentrations increase but these trends are not very significant due to missing nitrate data and scattered phosphate data. We interpret the increase in dissolved silica to reflect the filter feeding of diatoms by zebra mussels. Diatoms dominate the spring (*Asterionella*) and fall (*Fragilaria*) phytoplankton in Seneca Lake. The reduction in diatom biomass decreases the algal removal of dissolved silica and thus allows dissolved silica concentrations to build up in the lake over time. Perhaps large increases were not observed in the phosphate and nitrate concentrations because the uptake of phosphates and nitrates was transferred from the pelagic phytoplankton to littoral macrophytes. This hypothesis is consistent with the observed increase in dissolved silica because macrophytes do not assimilate silica to the extent required by diatoms. Unfortunately, macrophyte densities were not systematically measured over the past 10 years to confirm this hypothesis.

Since 1997, Secchi disc depths and dissolved silica concentrations have decreased, whereas chlorophyll-a, phosphate, nitrate concentrations have dramatically increased from 1997 to 1998 and remain high through 1999. These water quality trends are counter to the increased ecosystem stress applied by grazing zebra mussels described above. A number of hypotheses may account for the change.

Hypothesis One: The lake is becoming more eutrophic with time due to an increase in the nutrient loading from outside the lake. Minimal evidence supports this hypothesis, however. Landuse practices have not changes considerably in the watershed over the past decade. In fact, Secchi disc depths have increased from the 1960s and 1970 to 1991, and the increase is attributed to better waste-water treatment and elimination of phosphates and detergents. A few farms have recently converted from growing crops to hog farming, a change that could increase the runoff of nutrients. However, recent water quality monitoring of a number of streams in the watershed indicate no difference in water quality between streams that drain hog farms and streams that drain other agricultural activities in the watershed (Spitzer, 1999). Spitzer (1999) concluded that the density of hog farms is too small to influence this watershed. A significant change beyond seasonal variability was not observed in stream discharge or stream nutrient concentrations over the past as well (Halfman, unpublished data).

Hypothesis Two: The number of zebra mussels has dramatically reduced from 1997 through 1999, and thus reduced their grazing pressures through this transition. Data from class projects reveal that the average number of zebra mussels collected in sediment dredges has increased 10 times from 1996 to 1998. Unfortunately, the number of zebra mussels recovered by a dredge is also influenced by variable sediment recovery in nearshore, coarse-grained sediments by the dredge, and recovery of suitable substrates for zebra mussel colonization. But this increase

and the calcium budget presented below suggest that zebra mussels are a significant and growing force in the lake through 1997. It changes after 1998 as sediments recovered in 1999 reveal, for the first time, dead zebra mussels. Lakeshore property owners have also complained about, for the first time, large numbers of dead zebra mussel shells littering the shoreline. Perhaps zebra mussels began to die in significant numbers during 1998 due to a lack of food or old age.

Hypothesis Three: The natural decomposition of dead zebra mussels and their accumulated feces released previously sequestered nutrients back into the water column during 1998 and 1999. Zebra mussel life spans and the quantity of nutrients sequestered by zebra mussels are critical to this hypothesis. The life span of zebra mussels is typically 2 to 9 years, with shorter life spans in North American lakes and longer life spans in European lakes (Mackie, 1993). If zebra mussels took a few years to get established in Seneca Lake, then the first significant accumulation of dead zebra mussels would not occur until 2 to 3 years after their establishment or 1997 or 1998. This timing fits the hypothesis. Evidence from the literature indicates that zebra mussels sequester half or more of the available total phosphorus during their establishment in an area (e.g., Kuenzler, 1961, Stanczykowski and Lewandowski, 1993, Johengen et al., 1995, Anderson et al., 1998). The sequestered phosphate is isolated from the lake in the soft tissue and feces of zebra mussels.

A final factor is also critical for this hypothesis. Summer temperatures, and lake water temperatures, were unusually warm in 1998 and again but to a slightly lesser extent in 1999. The depth of the thermocline was significantly deeper in 1998 than previous years as well. Both factors contribute to the highest calculated August heat budget in 1998 than any August in the previous 5 years by 10% or more (Ahrnsbrak, personal communication, 1999). Warmer temperatures promote faster and more complete bacterial recycling of dead organic matter. The sequestered nutrients in the dead zebra mussels are also released directly to the epilimnion. We suspect that a combination of starving zebra mussels, recycling of the sequestered nutrients, and climate were all significant in the observed limnological changes from 1997 to 1998 and through 1999.

The plankton community experienced some changes from 1997 to 1998. Previously copepods were the dominant zooplankton. They are now replaced with cladocerans and rotifers. Perhaps zebra mussels preferentially consumed the copepods because they are larger than the other zooplankton. Larger plankton were selectively removed by zebra mussels in growth chamber experiments (Mellina et al., 1995). Two phytoplankton species, *Ecballocystis* and *Anabaena*, bloomed in the spring of 1998 and fall of 1999, respectively, for the first time this decade. Any link between these changes and zebra mussels is unclear at present and may, instead or in combination, reflect the larger concentrations of dissolved phosphates in the lake during 1998 or the warmer temperatures. Otherwise, *Asterionella*, *Ceratium*, and *Fragillari* continue to dominate the phytoplankton populations in the lake during the spring, summer and fall, respectively.

In summary, the limnological character of Seneca Lake has changed considerably after the invasion of zebra mussels in 1992. It indicates that any one sample does not characterize the long-term, year-to-year limnology of a lake. This is important for the pending environmental focus on Cayuga Lake because it probably experienced similar swings in the water clarity, nutrient concentrations and plankton dynamics. Clearly, the lake should be closely and routinely monitored to assess the environmental status of the watershed.

Zebra mussel impact on Seneca Lake calcium budgets and sedimentology

Stream discharge and hydrochemical data were collected for various classroom and research activities starting in 1995. These data are suitable for a back-of-the-envelope estimate of the impact of zebra mussels on the calcium budget and, thus the sedimentology of the lake. In this section of the paper, the focus is on the flux of calcium to the lake floor through the sedimentation of the authigenic calcite, and the potential redirection of this calcium flux to zebra mussel carapaces (Fig. 4).

The flux of calcium to the lake floor by the sedimentation of authigenic calcite (a water-column precipitate) was estimated from mass accumulation rates of the authigenic calcite within the recent (upper 3 to 4 cm or previous 15 to 20 years) postglacial sediments and the area of the lake floor that accumulates postglacial sediments.

Flux $Ca_{Authigenic} = C_{CaMuds} S (1 - \phi) \rho A_{Muds}$, where:

Flux $Ca_{Authigenic}$ is the flux of calcium to the lake floor within authigenic calcite,

C_{CaMuds} is the calcium concentration in the authigenic calcite fraction of the recent muds
 S is the lake wide average linear sedimentation rate of the recent muds,

ϕ is the lake wide average sediment porosity of the recent muds,
 ρ is the average sediment density, and

A_{Muds} is the lake floor area accumulating postglacial muds.

The mass of calcium in the authigenic carbonate in the recent postglacial muds averages 0.03 ± 0.01 g Ca g⁻¹ dry mud. The average sedimentation rate for the profundal sediments is estimated at 0.20 cm year⁻¹ based on Cs-137 and Pb-210 sedimentation rates of a box core (Callinan, personal communication, 1998). This rate, as expected, is higher than published Holocene average postglacial sedimentation rates (e.g., Mullins et al., 1996, Anderson et al., 1997, Halfman and Herrick, 1998). The average porosity is 0.82 and was determined by weight loss after oven drying the sediment samples. We assumed an average sediment density of 2.65 g cm⁻³. The estimated annual flux of calcium to the lake floor by authigenic calcite is approximately 4,000 metric tonnes year⁻¹.

A pre-zebra mussel calcium mass balance provides a test for this estimated flux (Fig. 4). This mass balance requires two additional fluxes beyond the flux to the authigenic carbonate, the influx of calcium by streams and the efflux of calcium through the outlet. We assumed that calcium influx or efflux by groundwater and/or sediment pore waters was negligible. We also assumed an annual stream discharge to the lake and annual outlet discharge from the lake of 863×10^6 m³ year⁻¹ and 760×10^6 m³ year⁻¹, respectively (Wing et al., 1995).

The calcium input by streams was extrapolated from the available stream discharge and calcium concentration data collected from 6 streams since 1995. Each stream reveals unique discharge and calcium concentrations but some generalizations are possible to calculate annual average values for the watershed. Average discharge in these 6 streams is proportional to basin area but not calcium content (Fig. 5). Thus, the annual stream discharge for the entire basin is assumed proportional to the watershed area.

The selected creeks also span the range of factors that may influence calcium concentrations in surface runoff. These factors include landuse, bedrock lithology, and soil type in the watershed. The calcium concentration in these 6 streams is roughly proportional to the percentage of agricultural land in the subwatershed (Fig. 6), although Reeder Creek is a notable exception. If landuse controls calcium input and Reeder Creek is ignored, then a linear interpolation of the basin-wide agricultural landuse of 53% yields an average calcium concentration of 48 ppm in surface runoff to the lake. Alternatively, bedrock and soil composition may control the calcium concentration in streams, because Reeder, Wilson and Kashong Creeks are underlain by limestones and calcium rich soils. These streams reveal consistently higher calcium concentrations than the other streams that lack carbonates in the watershed. Assuming the concentration of calcium in subwatersheds underlain by carbonates is the average of that measured at Reeder, Wilson and Kashong Creeks, and the calcium concentration in the southern creeks is the average of that measured at Keuka, Plum Point, and Big Stream, the annual average calcium concentration for the entire watershed is 40 ppm. These two predictions are close, but the Reeder Creek fit to the lithology and the calcium budget presented below suggest that the bedrock / soil type scenario is the better option.

The annual influx of calcium to the lake is then 34,400 metric tonnes of calcium (a concentration of 48 ppm yields 41,400 mt year⁻¹). The annual calcium efflux through the outlet is the average lake concentration (41 ppm) times the annual discharge of water through the outlet (760×10^6 m³ year⁻¹) or 31,000 metric tonnes of calcium per year. The input of 34,400 metric tonnes year⁻¹ is very close to the sum of the two outputs, 31,000 plus 4,000 metric tonnes year⁻¹. However, more data are needed to confirm this estimate.

The flux of calcium into zebra mussels was estimated from the average mass of calcium in the zebra mussels collected by a sediment dredge, the surface area of the lake where zebra mussels accumulate, the surface area of the dredge, and the average life span for a zebra mussel.

Flux $Ca_{ZM} = (C_{CaZM} A_{ZM}) / (A_D Age_{ZM})$, where:

Flux Ca_{ZM} is the flux of calcium to the lake floor incorporated into zebra mussels,

C_{CaZM} is the average mass of calcium in the zebra mussels collected by a dredge,

A_{ZM} is the area of the lake floor colonized by zebra mussels,

Age_{ZM} is the average lifespan of a zebra mussel.

The average mass of calcium in the zebra mussel dredge samples is 1.2 ± 0.5 g Ca dredge⁻¹. The large variability in the mass reflects the presence or absence of suitable substrates for colonization by zebra mussels in each dredge. Large variability in zebra mussel densities is also seen elsewhere in North America and Europe (Ramcharan et al., 1992, Mellina and Rasmussen, 1994). The lake floor area is 50 km², the dredge surface area is 550 cm², and the

average zebra mussel life span is assumed 3 years. The estimated annual flux is 200 metric tonnes year⁻¹ (or 300 mt year⁻¹ if the average life span is 2 years).

The zebra mussel calcium flux is 5% of the total calcium flux to the lake floor. Since the calcium concentration in the streams and lake have not changed over the past 6 years, we conclude that zebra mussels have redirected the flux of calcium to the lake floor. The redirection is confirmed by analyses of recent muds in a box core, the carbonate content decreases from under 3% detected at the sediment surface to over 4% 1 cm below the sediment surface (Mullins, personal communication, 1999). More research is required to confirm these estimates. For example, we extrapolated one sedimentation rate across the entire basin, the stream influx must resolve whether landuse or bedrock / soil type dictates calcium concentrations, the errors in the zebra mussel calcium mass must be reduced, and the assumption of minimal groundwater or pore water influence must also be investigated.

Conclusions

Limnological data from Seneca Lake reveal significant annual-scale swings in Secchi disc depths, chlorophyll-a concentrations, and nutrient concentrations over the past decade beyond the expected seasonal variability. The trends are divided into two time periods. The limnological changes from 1992 through 1997 reveal the classic increase in water clarity and decrease in phytoplankton concentrations that has been observed in others lakes across North America under the influence of grazing zebra mussels. Since then, a reversal in these trends was observed through 1999, shown by an increase in phosphate and chlorophyll-a concentrations. Perhaps nutrients previously sequestered by zebra mussels were subsequently recycled in 1998 and 1999 by bacteria. The nutrient recycling was probably stimulated by unusually warm climates in 1998 and to a lesser extent in 1999, and promoted the observed increase in plankton biomass and nutrient concentrations. Finally, preliminary calcium mass-balance calculations suggest that zebra mussels have redirected a portion of the calcium flux to the lake floor from the deepwater authigenic carbonate phase to shallow-water zebra mussel carapaces. Similar changes are probably occurring in Cayuga Lake as well.

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Table 1. Annual average limnological data from Seneca Lake.

Year	1991	1992	1993	1994	1995	1996	1997	1998	1999
Secchi Disc Depth (m)	4.4	3.7	4.4	5.7	7.8	6.9	8.3	6.2	7.2
Alkalinity (mg L ⁻¹ CaCO ₃)						114	115	105	103
Calcium Hardness (mg L ⁻¹ Ca)				42	39	38	43	40	41
Phosphate (µg L ⁻¹ P)	0.4	0.3	0.5	1.2	1.5		0.5	2.4	3.3
Nitrate (mg L ⁻¹ N)		0.43					0.14	0.21	0.28
Silica (µg L ⁻¹ Si)		100		125	471	966	1426	267	216
Chlorophyll-a (µg L ⁻¹)	3.7	4.5	4.6	2.2	2.0	2.8	0.6	1.5	1.2

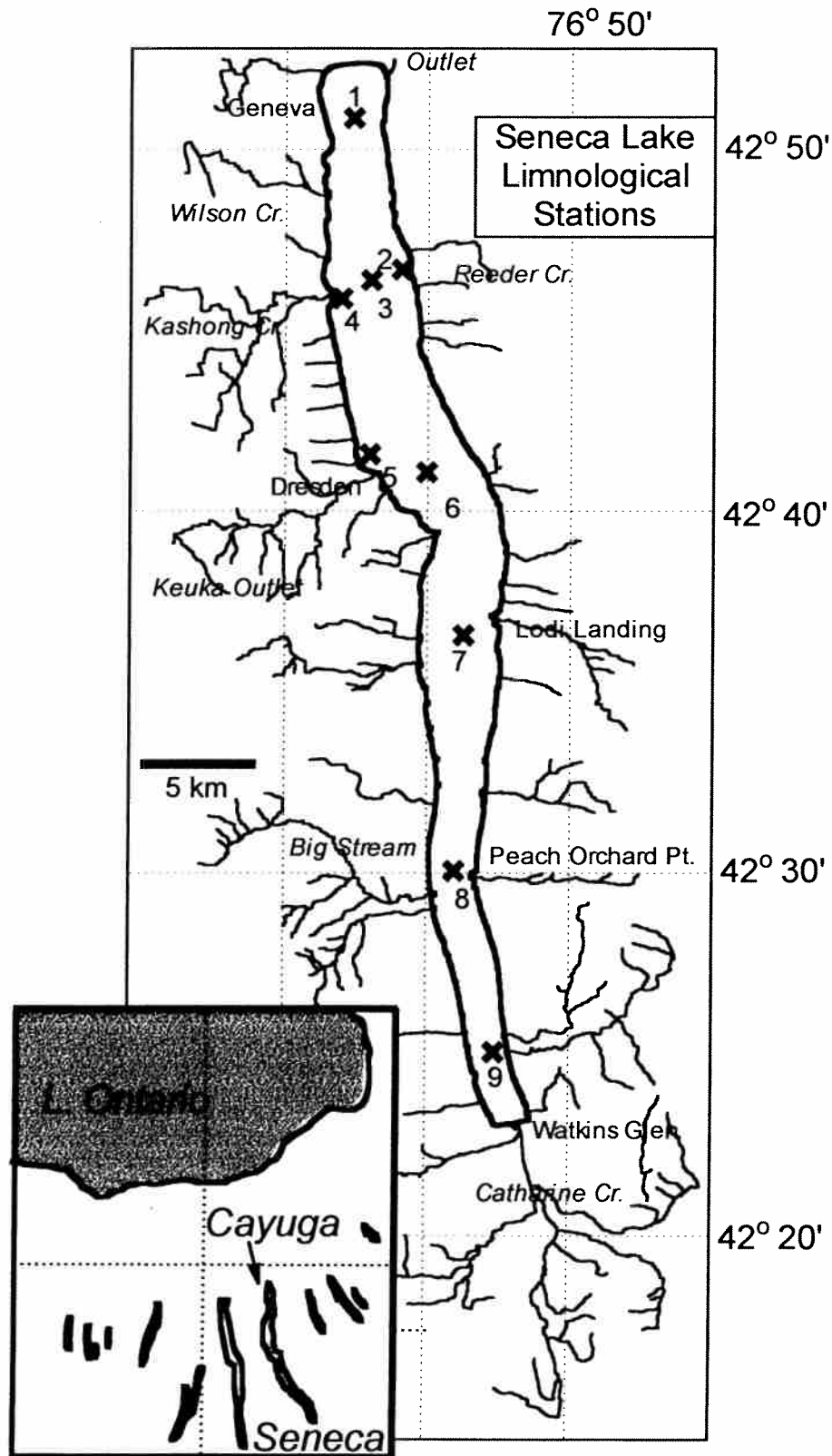


Figure 1. Map of Seneca Lake showing the location of the limnological stations and selected streams in the watershed mentioned in this study. The insert shows the location of Seneca Lake to Cayuga Lake and the other Finger Lakes in central New York.

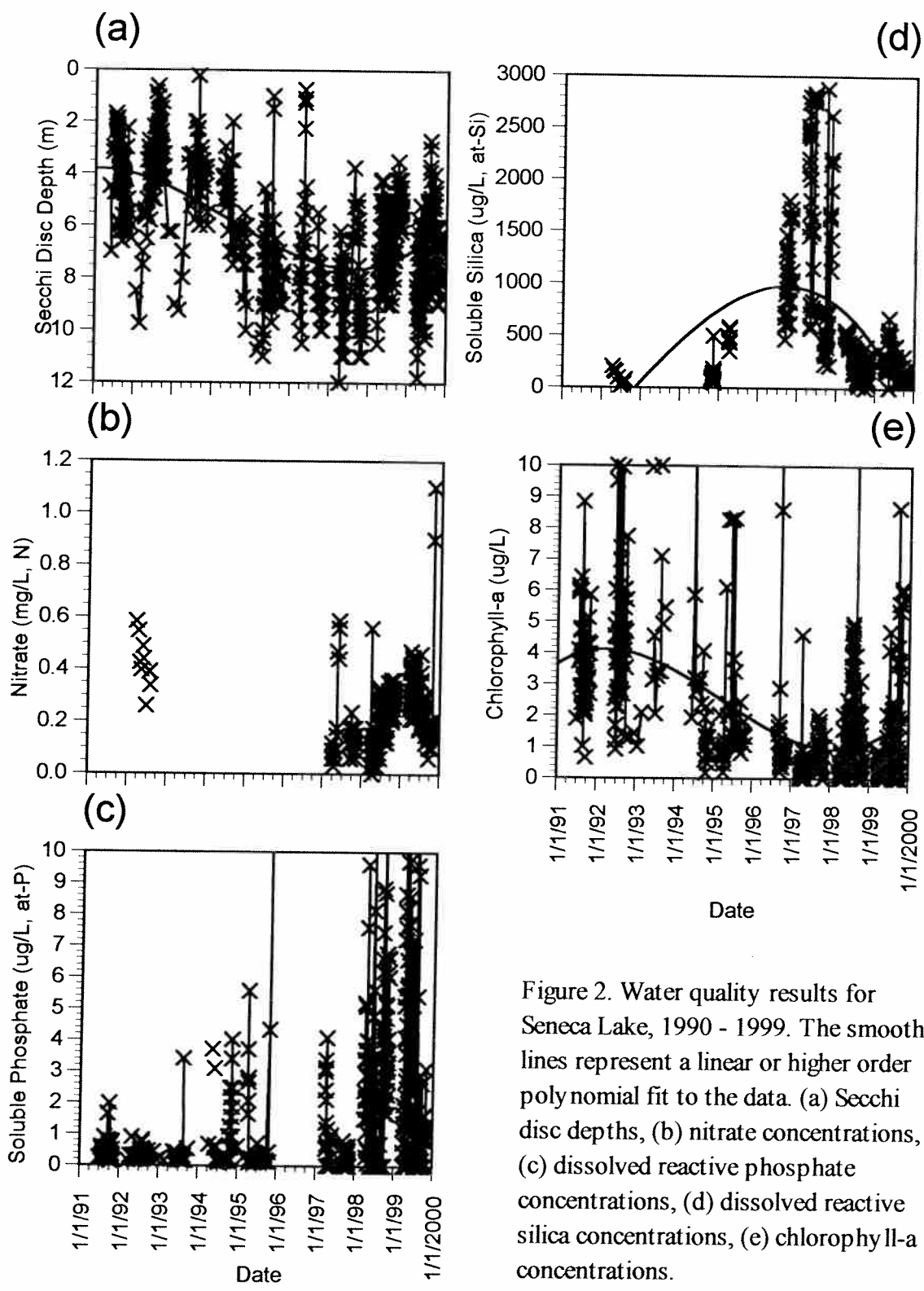


Figure 2. Water quality results for Seneca Lake, 1990 - 1999. The smooth lines represent a linear or higher order polynomial fit to the data. (a) Secchi disc depths, (b) nitrate concentrations, (c) dissolved reactive phosphate concentrations, (d) dissolved reactive silica concentrations, (e) chlorophyll-a concentrations.

Chlorophyll-a vs Secchi Disc Depths

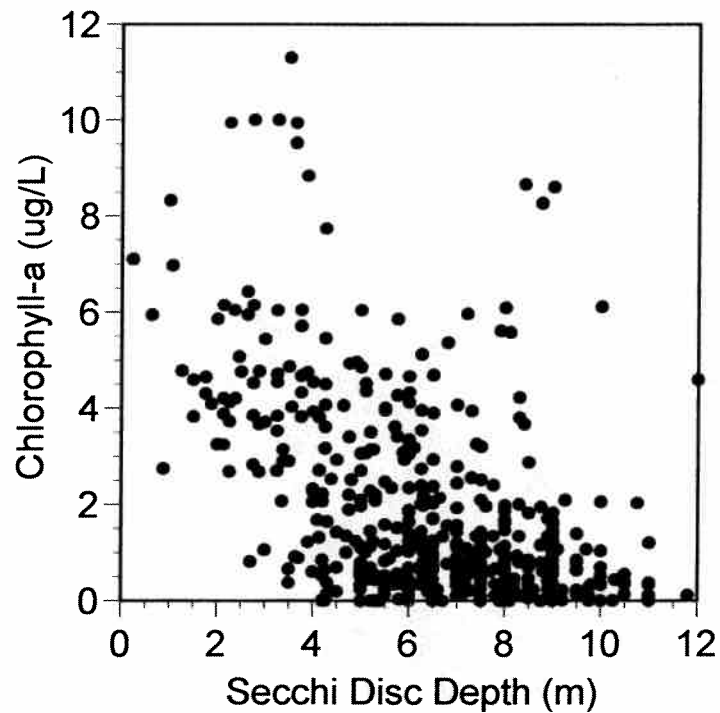


Figure 3. Secchi disc depths and chlorophyll-a concentrations. The plot indicates that water transparency is inversely proportional to plankton biomass in Seneca Lake.

Seneca Lake Mass Balance Calcium Model

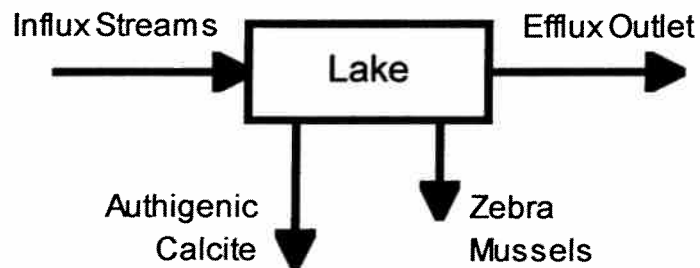
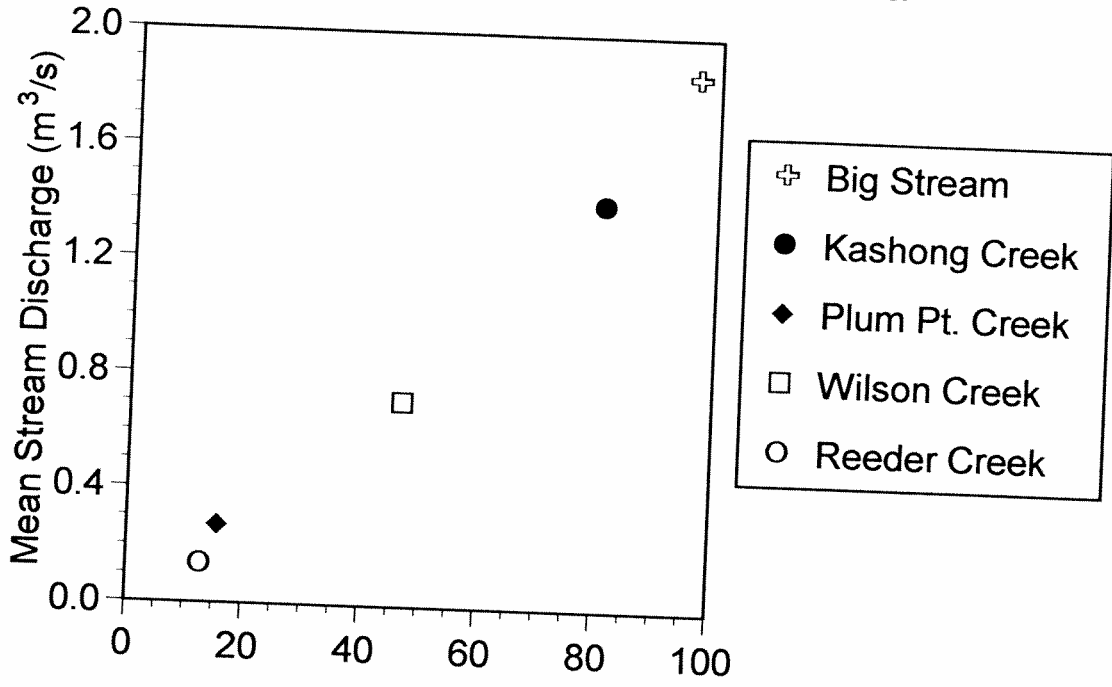


Figure 4. A first-order calcium mass-balance model for Seneca Lake showing the impact on the budget by zebra mussels.

Mean Discharge vs Subwatershed Area



Calcium Concentration vs Subwatershed Area

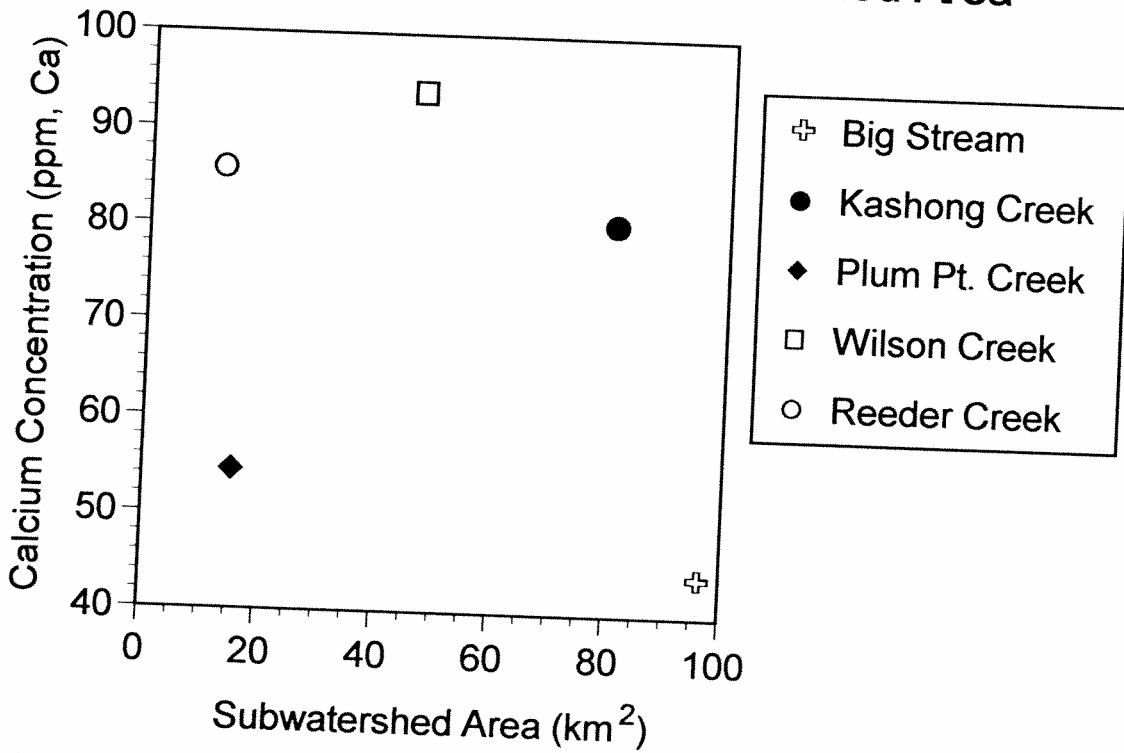


Figure 5. Mean annual discharge and mean calcium concentrations vs. subwatershed areas for 6 selected streams. The data suggest that stream discharge is roughly proportional to subwatershed area, whereas calcium concentration must reflect additional factors besides subwatershed area.

Landuse vs Calcium Concentration

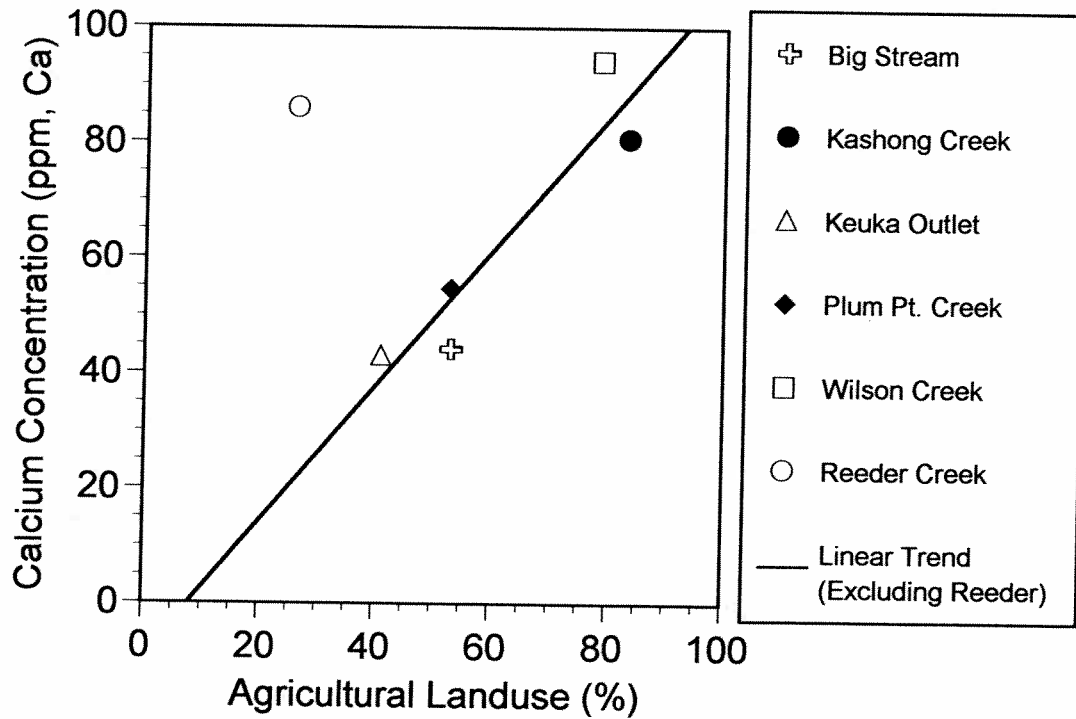


Figure 6. Mean annual calcium concentration and percentage of agricultural land for 6 selected subwatersheds. Substrate lithology is perhaps a better predictor of calcium concentrations (see text).