Aquatic Survey of
NASA Goddard Space Flight Center

Main and East Campuses

2002

By

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December 5, 2003

INTRODUCTION

This study is part of an ongoing partnership between the National Aeronautics and Space Administration (NASA) and the Environmental Science and Policy Department (ESP) at George Mason University (GMU). The goals of this partnership are to assist NASA in facilities management at its mid-Atlantic bases and to support NASA's commitment to Chesapeake Bay restoration.

The aquatic survey reported in this document addresses the former goal, seeking to provide valuable baseline information on the aquatic resources of the NASA Goddard Space Flight Center facility which will allow more informed management of this facility. The goal of the 2002 studies was to inventory and describe the water quality and biological communities at the facility. The current report addresses water quality during 2002 and plankton communities in three representative ponds on the Goddard campus. Future reports will address macroinvertebrate studies of the streams at Goddard which began in 2002 and continued through 2003 as well as detailed studies of stratification-water quality interactions in Pond P which were conducted during 2003.

NASA Goddard Space Flight Center is located in Prince George's County, Maryland approximately 11 miles (18 km) from the U.S. Capitol building in Washington, DC (Figure 1). The site is bordered to the south and east by residential development, to the west by residential development and the Baltimore-Washington Parkway, and to the north by the Beltsville Agricultural Research Center of the U.S. Department of Agriculture. Goddard is located in the Coastal Plain physiographic province of Maryland. The facility is located on a drainage divide with surface runoff from the north and central portions of the site flowing to Beaverdam Creek, a tributary of the Anacostia and Potomac Rivers. The southern portion of the property drains to the Patuxent River.

The author wishes to acknowledge the assistance of numerous individuals who contributed to the study. Thanks to Odean Serano and Ann Clarke of NASA Headquarters who were instrumental in developing the NASA-GMU partnership and obtaining the contracts necessary to fund this research. Darlene Squibb of NASA Goddard provided background information on the Goddard site and aided in access to Goddard for sampling personnel. Andrea Barberry conducted laboratory water quality analyses and assisted in the field work. Theresa Connor counted the zooplankton samples and Saiful Islam enumerated phytoplankton. Leslie Orzetti assisted with field work. This report is dedicated to the memory of Fred, my faithful companion for over 13 years.

METHODS

I. Streams

Seven stream sites were monitored on a monthly basis for water quality. All major stream reaches on the base were sampled and the individual sample sites were selected for representativeness and accessibility (Figure 2, Table 1). All stream sites were located on small first order tributaries. Watershed areas ranged from 8.5 to 80.4 hectares (21-201 acres). During the initial sampling trip longitude and latitude were determined at each site using Trimble GPS with real-time differential correction. At each site on each sampling date, temperature, dissolved oxygen, specific conductance, and pH were determined using a Hydrolab Datasonde and Surveyor 4a. A 1 liter water sample was collected at each site. From this an aliquot (125 mL) was filtered through a GF/F filter and the filtrate retained for analysis of ammonia-nitrogen, nitrate-nitrogen, and soluble reactive phosphorus. The filtered samples and the remaining whole water samples were kept on ice for return to the lab. Samples were refrigerated in the lab pending nitrogen, phosphorus, suspended solids, and alkalinity determination. Lab analysis techniques are described below under Ponds.

II. Ponds

Three of the four larger ponds at the NASA Goddard facility were surveyed during 2002. The ponds ranged in surface area from 0.18 – 2.37 hectares (0.45 – 5.9 acres) and in maximum depth from 0.5-2.5 m. The largest of these was on the Main Campus and was named Pond P for the purposes of this study (Figure 2, Table 2). Two smaller ponds were surveyed on the East Campus. One of these appeared to be an old farm pond and was located about 200 m NW of Building 25. This was called Pond Q. A more recently constructed stormwater management pond located between Buildings 32 and 33 was named Pond R. Trimble GPS with real time differential correction was used to ascertain the exact coordinates of each sampling site.

At each pond a canoe was used to sample at a fixed station located at the deepest point or if the pond was fairly uniform in depth, at a central point. At Pond P a deep spot between the island and the dam was chosen. At Pond Q a central point about 10 m from the dam was utilized. At Pond R a point near the center was selected.

At each sampling location, a Hydrolab Datasonde and Surveyor 4a display were used to assay temperature, specific conductance, dissolved oxygen, and pH near the surface (either 0.1 or 0.3 m) and at half meter intervals extending to just above the bottom. The lowest measurements were generally 2.4 and 1.4 m at Ponds P and R, respectively. At Pond Q, only one measurement was taken since it was so shallow.

Light attenuation was measured at each sampling location by two methods. A secchi disk was employed to determine water transparency. The disk was lowered until it just disappeared and raised until it just appeared. The mean of those two depths was taken as the secchi depth. Light attenuation coefficient was determined by taking ambient light readings using a LI-Cor PAR (photosynthetically active radiation) sensor at 10 cm depths from 10 cm to about 100 cm. These data were fit to an exponential decline function using natural logs. The resulting coefficient of this function was the light attenuation coefficient. Surface light readings were taken before and after the light depth profile to ensure that ambient light did not change during the profiling.

A submersible pump attached to a clear plastic hose was used to collect whole water samples from desired depths for laboratory analysis. In April, a depth-integrated sample was constructed from water collected at half-meter intervals (0.3, 1.0, 2.0 at Pond P and 0.3, 1.0 at Pond Q). In May separate surface and near bottom samples were collected and analyzed separately. From June through November samples were collected at 0.5 m intervals and analyzed individually for both Ponds P and Q. Aliquots (125 mL) of the water from each depth were filtered through Whatman GF/F glass fiber filters in the field. The filtrate was collected for laboratory analysis of ammonia-nitrogen, nitrate-nitrogen, and soluble reactive phosphorus. Unfiltered water samples were retained for ammonia-nitrogen, nitrate-nitrogen, total phosphorus, soluble reactive phosphorus, chlorophyll a, pheopigment, total suspended solids, and volatile suspended solids using standard methods.

Soluble reactive phosphorus was determined using the ammonium molybdate-ascorbic acid method (Wetzel and Likens 1979). Total phosphorus was measured using the persulfate digestion followed by ammonium molybdate-ascorbic acid (Wetzel and Likens 1979). The phenol hypochlorite method was employed for ammonia nitrogen determination (Solorzano 1969). Nitrate nitrogen was determined by a cadmium reduction method employing Hach NitraVer5 reagent powder pillows. Total suspended solids was the dry weight (80ºC) of residue from suspended materials that was collected by GF/F filters. Volatile suspended solids (organic weight) represented that portion of the dry residue that volatilized at 500ºC within 1 hr.

Chlorophyll was measured by filtering the phytoplankton from the whole water samples onto a 0.45 um membrane filters (Gelman GN-6) at a vacuum of less than 10 lbs/in^2. During the final phases of filtration, 0.1 mL of $MgCO_3$ suspension (1 g/100 mL water) was added to the filter to prevent premature acidification. Filters were stored in 20 mL plastic scintillation vials in the lab freezer for later analysis. Chlorophyll samples were extracted in a ground glass tissue grinder to which 4 mL of dimethyl sulfoxide (DMSO) was added. The filter disintegrated in the DMSO and was ground for about 1 minute by rotating the grinder under moderate hand pressure. The ground suspension was transferred back to its scintillation vial and stored in the refrigerator overnight. Samples were removed from the refrigerator and centrifuged for 5 minutes to remove residual particles. Chlorophyll concentration in the extracts was determined fluorometricaly using a Turner Designs Model 10 field fluorometer configured for chlorophyll analysis as specified by the manufacturer. The instrument was calibrated using standards obtained from Turner Designs. Fluorescence was determined before and after acidification with 2 drops of 10% HCl. Chlorophyll a was determined by the following equation which corrects for pheophytin interference:

Chlorophyll a (:g/L) = $F_sR_s(R_b-R_a)/(R_s-1)$

where F_s=concentration per unit fluorescence for pure chlorophyll
R_s=fluorescence before acid /fluorescence after acid for pure chlorophyll
R_b=fluorescence of sample before acid
R_a=fluorescence of sample after acid

Phytoplankton counts were also conducted on the whole water samples. Phytoplankton species composition and abundance was determined using the inverted microscope-settling chamber technique (Lund et al. 1958). Ten milliliters of well-mixed algal sample were added to a settling chamber and allowed to stand for several hours. The chamber was then placed on an inverted microscope and random fields were enumerated. At least two hundred cells were identified to species and enumerated on each slide. Counts were converted to number per mL by dividing number counted by the volume counted.

Zooplankton was collected using two methods. Microzooplankton (44 um – 202 um) was collected by pumping water from representative depths through a 44 um sieve bucket. From April through July a total of 96 L of water were pumped from 2-3 depths and composited into a single sample. From August through November 48 L of water were pumped at either 3 depths (Pond P) or 2 depths (Pond R) and analyzed separately. At Pond Q a single sample was collected by filtering 96 L from one depth. The resulting samples were preserved with formalin to a concentration of 10%.

Macrozooplankton (> 202 um) were collected at Pond P by towing a net vertically from a depth of 2.0 m to the surface. This vertical tow was done twice and composited into a single jar. Qualitative tows were collected at Pond R and no tows were collected at Pond Q due to its shallow depth. Horizontal towing was not practical with the canoe. The resulting samples were preserved with formalin to a concentration of 10%.

Microzooplankton and macrozooplankton samples were rinsed by sieving a well-mixed subsample of known volume and resuspending it in tap water. This allowed subsample volume to be adjusted to obtain an appropriate number of organisms for counting and for formalin preservative to be purged to avoid fume inhalation during counting. A one mL subsample was placed in a Sedgewick-Rafter counting cell and whole slides were analyzed until at least 200 animals had been identified and enumerated. A minimum of two slides was examined for each sample. References for identification were: Ward and Whipple (1959), Pennak (1978), and Rutner-Kolisko (1974). Zooplankton counts were converted to number per liter or cubic meter with the following formula:

Zooplankton = (N)(V_s)/(V_c)/(V_f)

where N=number of individuals counted
V_s=volume of reconstituted sample, (mL)
V_c=volume of reconstituted sample counted sample counted, (mL)
V_f=volume of water sieved, (L, for microzoo, or m^3, for macrozoo)

E. Data Analysis

Data for each parameter were entered into Excel spreadsheets for graphing of temporal and spatial patterns. SYSTAT statistical software was used to construct graphs to examine spatial and temporal patterns in the stream data. LOWESS trend lines were fit to the seasonal graphs. Patterns among sites were assessed by box plots. Seasonal and depth patterns in the ponds were depicted by graphs constructed in Quattro Pro. Arcview was used to plot average values for each parameter and to determine watershed and pond surface areas.

RESULTS

I. Climatic Summary for 2002

Air temperature was well above average for the period from January through April in 2002 (Table 3). May was slightly below normal and June about normal. Above normal temperatures returned again in July and lasted through September. August was 1.5^{o}C above normal in 2002. The remainder of the year was below normal. There were 56 days with maxima above 32.2^{o}C (90^{o}F) during 2002, an usually high number.

Precipitation was generally well below normal for the period from January through September (Table 3). The period July through September registered barely half of its normal rainfall. Precipitation was well above normal during October through December. As a result of the prolonged drought, no sampling was possible at Site E from June through October and at Site B on the May, June and September sampling dates due to lack of water in the stream. On several additional dates Site B was only represented by water in pools; there was no surface flow.

II. Stream Sites

A. Water Quality

Temperature exhibited a pronounced seasonal pattern (Figure 3). Values in spring were in the 10-16^{o}C range and increased to well above 20^{o}C at most sites in the summer.

Highest temperature was observed at Site A in August of nearly 30°C. This high value was probably due to solar heating of the surface layer of Pond P which supplies the water to Site A. Temperatures declined rapidly in the fall dropping below 10°C by November. A comparison of temperature patterns among sites (Figure 4) revealed that within site variation was quite large, mainly due to seasonal changes. Differences between sites were less marked. Site A had the highest median temperature and Site E had the lowest median temperature. However, this was mainly due to the fact that no measurements could be made at Site E from June to October due to lack of water.

Conductivity exhibited a weak seasonal pattern with a decline from March into the summer and a slight fall increase (Figure 5). This pattern was most clearly observed at Sites A and G. Site F remained high all year. Site C showed very little change. The consistently high values observed at Site F are clearly depicted in Figure 6. Sites C, D, and E were consistently the lowest. These spatial patterns can be related to land use or point sources. The high values observed at Site F were originally thought to be due to discharge from a heating/air conditioning unit upstream; however, Goddard personnel have questioned whether this any discharge from this unit would actually drain to Site F. Sites A, B, and G have a high incidence of developed land upstream and exhibited intermediate values which were generally highest in the spring, perhaps due to road salt runoff. Sites C, D, and E have a higher density of forest in their respective watersheds which tends to result in lower conductivities.

Dissolved oxygen (mg/L) exhibited a clear seasonal pattern declining from March through August and increasing somewhat in the fall (Figure 7). This basic seasonal pattern is at least partially linked to the seasonal temperature pattern since warm water holds less oxygen than cooler water. There were some clear differences in DO among stations (Figure 8). Sites A, B, C, and F had median values well above 5 mg/L and did not experience any readings below 2.5 mg/L. Sites D and E had median values below 5 mg/L and some values below 1 mg/L. Site G had a broad range of values median above 5 mg/L, but a significant number of low values.

To remove the temperature component from seasonal changes in dissolved oxygen, the DO values can be recalculated as “percent saturation”. When this was done a general seasonal pattern was still observed with highest values in the spring and a minimum in late summer (Figure 9). The spatial patterns were also similar with Sites A, B, C, and F having the highest readings while D and E were consistently low (Figure 10). Sites D and E are located on the same tributary which drains some forested wetland areas which may contribute to the low oxygen concentrations due to enhanced rates of decomposition.

pH exhibited a muted seasonal pattern (Figure 11) with a slight peak in late summer. Major differences were observed among sites. Sites A, B, F, and G were always circumneutral with median values of 6-7 (Figure 12). Site D was consistently acidic with a median value of about 4 and highest pH of about 5.5. Sites C and E were only slightly acidic.

Ammonia nitrogen readings fell into one of two groups: low values that were below the detection limit (20 ug/L) and higher values ranging as high as 1000 ug/L (Figure 13). [The values below detection limit are graphed as ½ of the detection limit, which in this case is 10 ug/L.] There was a slight trend toward increased readings in the summer at those stations which experienced values above the detection limit. Two stations had consistently elevated values: Site A below Pond P and Site G upstream of Pond Q (Figure 14). Sites B and F were consistently below detection. Stations C, D, and E were sometimes below detection, but other times moderately high.

Nitrate values exhibited a general increase in the spring followed by a clear decrease through the summer and then a leveling off (Figure 15). Some substantial differences were observed among stations. Sites C and E were consistently highest in nitrate with median values above 1 mg/L and highs above 2 mg/L (Figure 16). Stations A, B, D and F were consistently below 1 mg/L.

Soluble reactive phosphorus was generally below detection limit (2 ug/L) in most samples (Figure 17). In fact, the median value at Sites A, B, C, and D was the detection limit indicating the majority of the readings were less than 2 ug/L (Figure 18). Site F was the location with the highest values and the greatest frequency of values above the detection limit.

Total phosphorus was generally above the detection limit (5 ug/L). Values tended to be slightly higher in the summer (Figure 19). Sites A, F, and G consistently recorded the highest total phosphorus readings with a median of about 40 ug/L (Figure 20). Sites B, C, D, and E were consistently lowest with medians of about 20 ug/L.

Total suspended solids measures the amount of particulate matter per unit volume of water. TSS exhibited a general seasonal increase in the spring, a decline in midsummer and a leveling in the fall (Figure 21). Median values of 10-20 mg/L were found at Sites A, B, C, and D, while sites E, F, and G had lower medians (Figure 22).

Volatile suspended solids measures the amount of particulate organic matter per unit volume of water. VSS typically represented about 25-40% of TSS. VSS exhibited a similar seasonal pattern as TSS (Figure 23). Differences among sites were similar, but not identical. Site A had the highest median VSS value, while Site F had the lowest (Figure 24).

Turbidity was somewhat variable among dates with lowest values in March and September (Figure 25). Median turbidity was highest at Sites A, B, and D (about 20 units). Turbidity was consistently lower at Sites C, F, and G (Figure 26).

III. Pond Sites

A. Water Quality

1. Comparison of Average Values among Sites

Temperature exhibited a generally typical seasonal pattern with highest values in the summer (Figure 27). Highest average temperatures were observed at Pond Q in July exceeding 31^{o}C. Temperatures were generally higher and more variable in Pond Q. Both of these trends are related to the shallow depth of Pond Q. Solar radiation reached the bottom in this shallow pond discouraging thermal stratification and the shallow water column adjusted more readily to changes in ambient air temperature. Average temperatures in Ponds P and R exhibited more gradual seasonal patterns with a slow rise from about 20^{o}C in April to just over 25^{o}C in July and August. (Depth-profiles of temperature and other parameters at Ponds P and R will be presented in Section II.A.2. of the Results). The unusually high April pond temperatures were a reflection of the above normal April air temperatures. The large drop at all sites in November of 2002 corresponded with the onset of much cooler temperatures in the late fall.

Specific conductance measures the rate at which electricity is conducted through the water and is roughly proportional to the number of ions in the water. Conductance exhibited a decline through most of the period at all pond sites (Figure 28). Ponds Q and R had the highest conductance in the spring reaching nearly 800 uS/cm while Pond P was at about 525 in April. At Ponds P and R conductance declined steadily throughout the year reaching a low of about 150 uS/cm in November. Pond Q exhibited a similar decline through August reaching a minimum of about 125 uS/cm, but then increased through November to nearly 500 uS/cm. The high values in spring are probably related to runoff from road salt applications to Goddard roads and parking lots. The decline is due to progressive dilutions of this input during the spring, summer, and fall. Cause of the fall increase at Pond Q is not clear.

Dissolved oxygen (DO) is necessary to support the activities of animals in the aquatic environment. It enters the water through aeration from the atmosphere and photosynthesis by aquatic plants and algae. State water quality standards generally specify a minimum of 5 mg/L to support a balanced aquatic community. Depth-averaged DO levels exhibited differing seasonal patterns among the three ponds (Figure 29). Pond P started out with very high DO levels in the spring of nearly 12 mg/L. These levels dropped through June to below 6 mg/L. A peak in July back to 8 mg/L was followed by another decline to near 5 mg/L in August and then an increase reaching about 8 mg/L in November. In Pond Q DO was lowest in April and climbed fairly consistently through November reading nearly 12 mg/L. In Pond R DO was relatively low for most of the year (4-6 mg/L), but climbed to about 9 mg/L in November.

The amount of DO that water can retain when in contact with the atmosphere varies greatly with temperature. Since temperature varies seasonally over the study area, it is often instructive to examine not only the DO concentration, but also the DO as expressed as percent saturation. Values of about 100 percent (say, 80-120%) indicate that water is about in equilibrium with the atmosphere whereas much lower values indicate that water has been greatly impacted by respiration or decomposition and much higher values indicate that photosynthesis has been a dominant process. Using this standard, photosynthesis seems to have dominated Pond P in April and Pond Q from July through September (Figure 30). Respiration was dominant in Pond R for most of the year and in Pond P in June, August, September, and October. It is important to remember these are depth-averaged values (See next section for detailed presentation of depth profile data).

pH values reflect the soil and mineral content of the watershed as well as the activities of photosynthesis and respiration. The general trend of pH's near 7 at the three pond sites reflect a near neutral input from the watershed (Figure 31). Under these conditions, pH above 7.5 may indicate the dominance of photosynthesis whereas values below 6.5 could represent a dominance of respiration and decomposition in this area. Values indicative of photosynthetic activity were found in Pond P in April, May, and July and in Pond Q in August and September (Figure 31). These trends are similar to those found in the DO data. There were no pH's below 6.5, but Pond R had the lowest pH during much of the year consistent with dominance by respiration there.

Secchi depth reflects the clarity of the water. It is the depth at which a circular disk with alternating black and white quadrants disappears when viewed from the surface. Due to its shallow depth Secchi disk readings were not possible at Pond Q because the disk was usually visible on the bottom of the pond. At Pond P the water was very clear in April with a secchi depth of over 2 m observed (Figure 32). By May this had decreased to about 60 cm and remained between 60 and 80 cm for the remainder of the year. At Pond R the water clarity was much less with secchi depth ranging from 20 to about 40 cm throughout the year.

Light attenuation coefficient is another way of measuring water clarity. This coefficient is calculated using a profile of ambient light readings. These readings are more difficult to collect than the secchi depth readings and were successful on only four of the sampling dates (Figure 33). The values obtained were quite variable, but generally consistent with the range of secchi depths observed.

Turbidity is another measure of water clarity. It is a measure of the degree of light scattering by particles in the water. Seasonal turbidity patterns showed a minimum in spring and fall and higher turbidity in the summer (Figure 34). It also showed that turbidity was higher at Ponds Q and R than at Pond P. The observed differences in turbidity between Pond P and R are consistent with the Secchi depth measurements.

Ammonia nitrogen was consistently much higher at Pond P than in Ponds Q and R (Figure 35). In fact, in the later two ponds ammonia nitrogen was typically below or just above the detection limit of 0.02 mg/L. This great difference was not observed in April, but appeared very strong in May when Pond P ammonia nitrogen increased to over

0.6 mg/L. Pond P values gradually declined through the summer and into the fall reaching 0.1 mg/L in September before rebounding slightly in November.

Values of nitrate nitrogen were more similar among ponds (Figure 36). Interestingly, Ponds Q and R were typically somewhat higher than Pond P. All ponds exhibited a general trend of strong increase in May and gradual decline through the rest of the year. Peak values were above 2 mg/L in Ponds Q and R and 1.5 mg/L in Pond P. Thus, the total inorganic nitrogen (ammonia nitrogen + nitrate nitrogen) was actually nearly the same at all ponds in the peak of month of May. The exact cause of the May peak is unclear, but the decline through the year was probably due to plant and algal uptake during the growing season.

Soluble reactive phosphorus (SRP) measures the concentration of phosphate ion in the water. Phosphate ion is the form in which phosphorus is taken up by plants and algae and the form in which it is released by decomposition. SRP is generally higher when the supply of phosphorus from decomposition exceeds the demand from plant and algal growth. SRP was consistently highest in Pond Q (Figure 37) where values were above detection limits on all dates. SRP values were often below detection limits at Ponds P and R; in both cases only 3 of 7 dates had SRP values above the detection limit.

Total phosphorus reflects the total amount of phosphorus in the water column including both dissolved phosphate ion and phosphorus tied up in biological tissue, detritus, or inorganic particles. Total phosphorus levels are generally substantially higher than SRP values since phosphate is taken up very rapidly from the water. Depth-averaged total phosphorus was generally highest in Pond Q being particularly high in September (Figure 38). Pond R was generally slightly higher than Pond Q. Pond P values exhibited a clear seasonal pattern with a steady increase from May through August followed by a decline in September and October.

Total suspended solids (TSS) is a measure of the particulate matter in the water column as determined by its dry weight. The particulate matter could be phytoplankton algae, zooplankton, detritus, or inorganics such as clay particles. Total suspended solids were generally greater in Ponds Q and R than in Pond P (Figure 39). TSS exceeding 100 mg/L were observed only at Pond Q while values exceeding 50 mg/L were observed at both Ponds Q and R. Elevated levels in Pond R were probably the result of stormwater runoff since this pond receives substantial parking lot and road runoff. The elevated values in Pond Q were at least partially attributable to detritus and periphyton (attached algae and other microbes) that easily sloughed off in to the water. Pond P levels were quite consistent at about 20 mg/L except in early April when very low TSS was found.

Volatile suspended solids (VSS) was more similar among the three pond sites for most of the year (Figure 40). At Pond P VSS exhibited a gradual rise from less than 2 mg/L in April to about 12 mg/L in July and August. In Pond R an increase was observed in the spring to about 13 mg/L and then a gradual decline to about 5 mg/L in November. VSS was variable at Pond Q, but generally increased over time reaching a maximum of over 30 mg/L in September. Generally dates with high TSS at Pond Q had relatively high VSS. VSS was over half of TSS at Pond P indicating an organic matter source (like phytoplankton) predominated whereas at Pond R VSS was generally less than 25% of TSS indicating that inorganic solids (like clay particles) were more important in Pond R.

2. Depth Profiles in Ponds P and R

Temperature profiles reflect a basic property of freshwater lakes and ponds--their physical structure. Temperature is the main determinant of density in freshwater with cold water being the most dense and warm water the least. The temperature profile in Pond P reveals a stable depth stratification throughout the period of observation (Figure 41) with colder, more dense water on the bottom and a progression to warmer, less dense water on the surface on each date. Surface temperatures increased from 24^{o}C in April to 28^{o}C in July and then declined to 25^{o}C in September. Bottom temperatures increased from 15^{o}C in April to 22^{o}C in September. The progressive increase in bottom temperatures resulted in a decrease in stability of stratification by September. Pond P probably destratified (turned over) in October because by November temperatures were nearly isothermal at about 9^{o}C [November data not shown in Figure 41].

Though shallower, Pond R exhibited a similar pattern in vertical temperature structure (Figure 42). Surface temperatures increased from 26^{o}C in April to 30^{o}C in July and then declined to 27^{o}C in September. Bottom temperatures increased steadily from April through August rising from 12^{o}C to 25^{o}C. As in Pond P, destratification in Pond R probably occurred in October.

Specific conductance exhibited distinct seasonal changes, but little vertical structure in Pond P (Figure 43). In April there was little variation in conductance with depth with all values 500-550 uS/cm. By May conductance in most of the pond had decreased to about 325 uS/cm although higher values were found near the bottom. This change occurring throughout the pond suggested at least some mixing between the April and May samplings even though the temperature depth profile was almost identical between the two months. For the remainder of the year there was a gradual decline in conductance reaching 200 uS/cm by September and 150 uS/cm by November.

In Pond R similar patterns were found (Figure 44). In April values increased from 700 at the surface to 800 uS/cm near the bottom. By May conductance was 300 at the surface increasing to 400 uS/cm at the bottom. Values continued to decline through September approaching 160 uS/cm at all depths.

Dissolved oxygen (DO) concentrations reflect the interaction between photosynthesis which increases DO concentrations and respiration which decreases DO concentrations. Temperature is also a factor since cold water can hold more oxygen than warmer water. In April Pond P exhibited a highly unusual pattern in which DO increased dramatically with depth from about 9 mg/L near the surface to over 15 mg/L at the bottom (Figure 45). This pattern was the product of a clear water column which allowed lush growth of an algal mat on the bottom of the pond. Active photosynthesis of this mat pumped DO into the bottom waters of the pond. By May this mat had disappeared probably due to declines in water clarity and the oxygen profile reversed itself so that concentrations were low near the bottom and much higher near the surface. This reflected the predominance of respiration near the bottom and photosynthesis near the surface.

In Pond R the vertical pattern in dissolved oxygen was consistent throughout the year (Figure 46). Values were highest near the surface, rapidly declined between 0.5 m and 1.0 m depth and approached 0 near the bottom. Concentrations at the surface varied from 7.8 to 10.2 mg/L.

When expressed as percent saturation, temperature effects are removed and the focus can be on photosynthesis vs. respiration (Figures 47 & 48). These graphs reinforce the earlier analysis indicating that the balance between these two processes is responsible for the observed trends in DO. There were, however, some subtle differences. Whereas the highest DO concentrations in Pond P were observed in April near the bottom, the highest percent saturation DO values were observed in July in the upper layers.

Depth profiles in pH reflected a similar balance between photosynthesis and respiration as those in DO. Photosynthesis tends to increase pH and respiration decreases pH. The vertical pattern in Pond P in April showed higher pH near the bottom and lower near the surface, consistent with the observation of intense photosynthesis by a benthic algal mat (Figure 49). pH at the bottom was 8.5 while pH was 7 at the surface. By May the pH at the bottom was 6.8 and on the surface had increased to above 9. The reversal of this pattern was consistent with the disappearance of the benthic algal mat and development of phytoplankton nearer the surface. Values at 0.1 and 0.5 m were very similar indicating that the top 0.5 m of the water column was well-mixed. However, a distinct decline occurred at 1.0 m as a result of lower photosynthetic activity at this depth and a stratification of the water column. pH continued to decline through the water column as light levels decreased and respiration came to dominate over photosynthesis. The May pattern continued through the rest of the year with surface pH being generally lower on the other dates.

In Pond R pH was higher near the surface than at the bottom for the entire year (Figure 50). pH in surface waters increased strongly from April through September from about 7 to 8.4 consistent with continuing photosynthetic activity during the growing season. At the bottom there was also an increase from about 6.2 in the spring to 6.7 in August. This may have been due to some mixing or diffusion of surface waters since photosynthesis was improbable to due lack of light in the lower levels. A strong gradient was seen even between 0.1 and 0.5 m indicating rapid light attenuation and little mixing in the top 0.5 m.

Ammonia nitrogen was generally below detection limit of 0.02 mg/L in most samples (Figures 51 & 52). The exception to this was values in the deeper layers in Pond P. Ammonia nitrogen levels exceeded 1 mg/L at 2 m from May through August. In September and October ammonia nitrogen levels at 2 m declined to about 0.2 mg/L. These high levels are probably due to active decomposition of organic matter in the lower layers of Pond P.

In Pond P nitrate nitrogen was less strongly stratified than ammonia nitrogen (Figure 53). Nitrate values tended to decrease as the year progressed at all depths. In May they were about 1.5 mg/L at surface and bottom and by September they were below the detection limit at all depths. This decline was at least partially due to the uptake of nitrogen by phytoplankton during the growing season. The higher values in deep water was somewhat puzzling given the lack of phytoplankton production at greater depths and the near 0 dissolved oxygen levels observed. If DO had been used up, then nitrate should have been converted to ammonia by bacteria scavenging the oxygen from the nitrate. In Pond R a similar trend was seen with a progressive decline seasonally in nitrate and little depth stratification (Figure 54).

Soluble reactive phosphorus values were below detection limit in almost all samples from Pond P (Figure 55). This suggests that phosphorus is the limiting nutrient and is rapidly removed from the water for plant and algal growth. The greater frequency of detectable levels in Pond R (Figure 56) suggests that algal activity is placing less demand on phosphorus. It is probably that Pond R is more light limited.

Total phosphorus generally increased with depth in Pond P (Figure 57). Values generally increased from May through July and decreased in August and September. This seasonal pattern was generally consistent with VSS trends suggesting that total phosphorus and VSS are related. In Pond Q there was an increasing gradient with depth in July, August and September, but little depth variation in May and June (Figure 58).

Total suspended solids (TSS) exhibited a slight trend of increase with depth on most dates in Pond P (Figure 59). Values were generally in the 10-20 mg/L range. In Pond R there was a much more consistent and pronounced gradient in TSS with values at 1 m depth being 20-70 mg/L greater than at the surface (Figure 60). This is consistent with higher sediment loads entering Pond R and settling into the lower layers.

Volatile suspended solids (VSS) showed little consistent depth variation in Pond P (Figure 61). Values were generally 5-15 mg/L. In Pond R there was a general trend toward higher levels at greater depths, but this was not as consistent as with TSS (Figure 62). Values in Pond R were similar to those in Pond P. Since TSS was generally much greater in Pond R, this indicates a lower organic matter content to the suspended solids in Pond R consistent with more clay particles and less phytoplankton in Pond R than in Pond P.

B. Phytoplankton

Phytoplankton are the microscopic algae and cyanobacteria which form the base of the food web in open waters. They may be single-celled, filamentous, or colonial. They belong to a variety of groups including the eukaryotic green algae, diatoms, and cryptophytes as well as the prokaryotic cyanobacteria (formerly called blue-green algae). Total phytoplankton biomass is often assessed by measuring chlorophyll *a* concentration in the water. Further information can be obtained by microscopic examination and enumeration of whole water samples.

Chlorophyll *a* is a measure of the concentration of phytoplankton algae in the water column. Fragments of macrophyte or periphyton tissue that have broken loose and are suspended in the water column may also contribute. Chlorophyll levels were similar at all three ponds on many dates (Figure 63). In general, chlorophyll values increased from April through August and then declined in September. On some dates there were notable divergences from uniformity among sites. In April, chlorophyll *a* at Pond P was very low. This was the date on which large amounts of benthic algae were observed. In July both Pond Q and Pond R were substantially lower than Pond P. In September and November Pond Q was much higher than either of the other two ponds. Pond P exhibited the most consistent seasonal trend with a consistent rise from April through August reaching 60 ug/L before declining in September.

Pheopigment is a measure of chlorophyll breakdown products and results from dead or dying algal and plant material in the water column. Pheopigment levels in the three ponds generally mirrored chlorophyll levels at a reduced magnitude of approximately half (Figure 64). An exception to this was the reading of 50 ug/L in August at Pond Q which was actually greater than the chlorophyll reading of 40 ug/L. This probably reflects a greater proportion of detritus in the sample and presumably in the pond than was observed on other dates.

Depth patterns in chlorophyll a indicated that near surface values (0.1 and 0.5 m) were generally near 40 ug/L during the period May through September (Figure 65). The exception was June in which near surface values were about 20 ug/L. At 1 m depth chlorophyll was generally slightly higher at about 60 ug/L from June through August. Chlorophyll levels at the greater depths were more variable ranging from 20 to 70 ug/L on most dates, but exceeding 120 ug/L in August. Chlorophyll levels in Pond P were generally in the 20-40 ug/L range in June increasing to 40-60 ug/L in July, August, and September and dropping back to 20-40 ug/L in October. In Pond R chlorophyll showed a general pattern of increase with depth on many dates (Figure 66) . For example, in July surface chlorophyll was about 10 ug/L at 0.1 m and 15 ug/L at 0.5 m, but increased to nearly 40 ug/L at 1.0 m. A similar pattern was observed on other dates. Excepting the elevated reading at 0.5 m in August this pattern was consistent throughout the summer in Pond R.

Pheopigment concentrations followed patterns similar to those in chlorophyll a, but at reduced levels in both ponds (Figures 67 & 68).

Total phytoplankton cell density (number of cells per mL) exhibited a strong increase from April to May in Pond P followed by a steep decline in June (Figure 69). Values generally increased through the remainder of the summer and peaked in September at levels similar to those observed in May. Pond Q exhibited a similar pattern, but the September peak was much lower than the May peak. Pond R showed the September peak, but not the May maximum.

In Pond P cyanobacteria (TotBGA) were the dominant group in terms of cell density for the entire year (Figure 70). *Oscillatoria* was the dominant cyanobacterium in

May and June. *Microcystis* became dominant in July and shared dominance with *Aphanocapsa* in August. In July *Oscillatoria* was dominant again with Anabaena being codominant. Greens (TotGreens) and cryptophytes (TotCrypts) were of some importance in the summer.

Cyanobacteria were most important in the May peak in Pond Q (Figure 71). This was due to an unknown cyanobacterium. *Oscillatoria* was important in July and August and *Microcystis* was dominant in September. During much of the summer other groups, especially greens and cryptophytes, were equally important. From June through August *Chroomonas* was a dominant species. The green alga *Selenastrum* was important in July and *Staurastrum* made major contributions to phytoplankton cell density in September.

In Pond R cryptophytes were dominant in May led by *Cryptomonas* (Figure 72). The cyanobacterium *Oscillatoria* assumed major dominance in July and August. In September the green alga *Dictyosphaerium* and the cyanobacterium *Microcystis* were most abundant in terms of cell density.

Total biovolume in Pond P followed a pattern very similar to that in chlorophyll a (Figure 73). Very low biovolume in April was followed by a strong increase in May and then at gradual rise for through August. A slight decline in September preceded a stronger drop in November. A great deal more variation was observed at the other two ponds. Very high biovolume at Pond Q in September coincided with high chlorophyll levels, high DO saturation, and high pH. However, a significantly enhanced biovolume at Pond R in May was not reflected in the chlorophyll levels.

Biovolume dominance at Pond P was variable by season. In May and September cyanobacteria were dominant (Figure 74). *Oscillatoria* was the most abundant cyanobacterium in May, while in September Anabaena and *Oscillatoria* shared dominance. During most of the summer green algae had the highest biovolume, starting with *Scenedesmus* and *Chlamydomonas* in June followed by *Staurastrum* in July and August. Cryptophytes in the form of *Cryptomonas* were important in June and diatoms were of some significance in September represented by *Rhizosolenium*.

In Pond Q diatoms were dominant in June led by the filamentous genus *Melosira* (Figure 75). In July the euglenoid *Trachelomonas* (included in Other) and the green algae *Staurastrum* and *Pediastrum* were dominant. *Staurastrum* continued to dominate in August and become very abundant in September with at peak well off the graph.

Algae falling into the Other group were the most important during spring and summer in Pond R (Figure 76). In May *Euglena* was the dominant member of this group and from June through August it was *Trachelomonas*. In September the green algae *Dictyospherium* was dominant and in November *Cryptomonas* was most important.

C. Zooplankton

Zooplankton is the animal component of the open water plankton community. In freshwater the zooplankton is composed primarily of rotifers, cladocerans (water fleas), and copepods. These organisms feed on the phytoplankton and detritus in the water column and serve as food for fishes. Some larger zooplankton may feed on the smaller zooplankton. A healthy zooplankton community contains a mix of the three major groups including some large cladocerans. The large cladocerans such as *Daphnia* and *Diaphanosoma* are very active and efficient grazers. We sampled zooplankton in two ways. For the smaller taxa, those between 44 um and 202 um, we used a submersible pump to obtain water from several depths within the water column and filter it through a 44 um sieve. That is termed the microzooplankton sample and quantifies the rotifers and smaller crustacea like the cladoceran *Bosmina* and the nauplii (immature stages of copepods). The larger zooplankton (macrozooplankton) were quantified by net tow with a 202 um conical tow net.

Rotifer populations increased rapidly in Pond P from April through June peaking at over 10,000/L (Figure 77). A similarly rapid decline was observed from June to July followed by a more gentle decline to very low levels in November. Pond Q had generally lower levels than Pond P after April and declined for the entire year. Pond R had robust levels in April, showed a decline in May and then peaked for the year in June at 4,000/L. It showed a general decline after that through November. The peak levels observed at Ponds P and R are quite large in comparison with many aquatic systems.

In Pond P the most important genus of rotifers for most of the year was *Keratella* (Figure 78). However, the very large peak in June was due primarily to *Polyarthra*. In Pond Q, *Keratella* was dominant in spring and early summer along with *Synchaeta* (Figure 79). As populations declined generally in the late summer, *Brachionus* became more important. In Pond R *Keratella* was the most numerous in the spring and early summer along with *Filinia* (Figure 80). In late summer the three genera were co-dominant: *Filinia*, *Keratella* and *Brachionus*.

The small cladoceran *Bosmina* was quantified using the microzooplankton samples (Figure 81). In Pond P *Bosmina* declined in May and then increased consistently through August reaching a maximum of about 420/L. A steady decline followed in the fall. *Bosmina* densities were much lower in the other two ponds. *Bosmina* maximum in Pond Q was about 30/L in September and about 70/L in mid July.

Copepod nauplii were also counted from the microzooplankton samples. In Pond P a distinct maximum was observed in June at over 2000/L with a smaller peak of about 500/L in August (Figure 82). In Pond Q values generally remained between 500 and 1000/L except in August and November when densities were quite low. In Pond R there was a fairly steady increase in the spring and early summer culminating in a maximum of about 1600/L in July. A steady decline was observed in late summer and fall.

In Pond P it was possible on all dates to collect a vertical tow through the entire water column with the 202 um plankton net allowing an assessment of seasonal trends in macrozooplankton (Figure 83). The herbivorous cladoceran *Daphnia* was quite common throughout the year being found at densities between 50,000 and 100,000/m^3. The major spring and early summer peak in total macrozooplankton was due to *Ceriodaphnia* in May and then cyclopoid copepods in June. In July *Diaphanosoma* became dominant. In the remainder of the year several groups shared dominance as densities slowly declined. The concentrations of macrozooplankton in Pond P were quite large by comparison with other representative freshwater systems.

Collection was more difficult in Pond R due to its shallow depth and only two months were sampled (April and May). Densities of macrozooplankton in Pond R were substantially lower than in Pond P (Figure 84). *Daphnia* was dominant in both months with much smaller contributions from other taxa.

IV. Spatial Relationships among Sampling Sites

The spatial relationships among sampling sites were investigated by plotting the annual average at each site with color symbolizing the range of values for each parameter. Temperature is shown in Figure 85. Triangles represent the pond sites and circles the stream sites. Average temperature at all pond sites was in the highest range (20-25ºC) due to the fact that these sites are more open and receive a lot of solar heating. Average stream temperatures at stream sites varied depending on two factors: seasonal distribution of records and proximity to pond or other heat source. Temperature at Stream Site A was in the mid-range (15-20ºC) due to the upstream heat source from the pond. A similar effect occurred at Site C which was in the midrange compared to the adjacent Site D (in the low range) which did not have a pond upstream and drained an entirely forested watershed. Site F was in the mid range and was more open and may have been influenced by effluent from an upstream heating/cooling plant. Sites B and E were both in the low range, primarily due to the fact that observations were less common in the summer due to drying of the stream bed at both sites.

Specific conductance was in the moderate range (300-500 uS/cm) at all pond sites (Figure 86). At the stream sites, results were more variable. At stream site A, immediately below Pond P, conductivity was in the low range. It was also in the low range at sites C, D, and E. At site B which only had water during part of the year, conductivity was in the moderate range. High conductivity was observed at site F.

Dissolved oxygen averaged above saturation at Pond Q and slightly below saturation at Pond P (Figure 87). In contrast at Pond R annual average dissolved oxygen was well below saturation. The stream sites were quite variable. At Site A, immediately below Pond P, annual average DO was well below saturation. Site B was also in the same range. Site C, below Pond Q, averaged only slightly below saturation. Sites D and E, also on East Campus, were in the low range (25-50%) for the year indicating the major role of decomposition. Site F was only slightly below saturation.

Field pH averaged near neutral in all three ponds (Figure 88). At site A, just below Pond P, pH was slightly below 7. The same was true at nearby site B. At sites C and E on the East Campus, pH was acid (5-6 range) and at site D pH averaged even more acid (4-5). Sphagnum (peat moss) was growing in the area of these sites which is both indicative of and a cause of acidification. At site F the pH was near neutral.

Ammonia nitrogen was generally low at most sites (Figure 89). Pond R and stream site F both averaged below 20 ug/L as did stream site B. Slightly higher values were observed at Pond Q and stream sites C and E. Stream site D had clearly elevated values averaging 50-100 ug/L, possibly as a result of active decomposition in its vicinity. The highest average values of ammonia nitrogen were observed at Pond P and Site A where values averaged over 100 ug/L for the year. This appeared to be due to decomposition and nitrate reduction in the hypolimnion (lower layer) of Pond P.

In contrast, nitrate nitrogen was higher in Ponds Q and R and lower in Pond P (Figure 90). The highest average values were observed at stream sites C and E. Stream sites A and D were slightly elevated and stream sites B and F were in the lowest range of nitrate values.

Average total phosphorus was highest at Pond Q followed by Ponds P and R (Figure 91). Slightly elevated levels were observed at stream sites A and F and lowest values were found at the other stream sites B, C, D, and E.

Turbidity was quite high in Ponds Q and R (Figure 92). Intermediate values of average turbidity were characteristic of stream sites A, B, and D. Low average turbidity was found in Pond A and stream sites C, E, and F.

Average values of total suspended solids were clearly highest at Ponds Q and R (Figure 93). At Pond R this is principally due to inorganic particles washed in by stormwater runoff. At Pond Q it is a combination of runoff and the luxuriant growth of macrophytes which slough solids and attached algae into the water column. TSS values were intermediate at Pond P and stream sites A, B, C, and D. Lowest average TSS was found at stream sites E and F.

DISCUSSION

Part of the discussion will be a summary analysis of water quality patterns at each site with particular reference to Maryland water quality standards and guidelines from the Maryland Stream Survey. The discussion will conclude with an analysis of plankton communities in the ponds and comparison with similar systems.

The parameters measured in this study for which standards are available include temperature, dissolved oxygen, pH, and turbidity. The temperature standard specifies that the temperature may not exceed 32°C or the ambient temperature of natural surface waters whichever is greater. This standard is intended to apply to discharges of cooling water which are not an issue in this study. The dissolved oxygen standard states that the dissolved oxygen concentration may not be less than 5 mg/L at any time. There is a preliminary draft proposal to allow lower DO below thermoclines in lakes and ponds. Note that this standard does not leave any room for episodic values below the standard. The pH standard states that normal pH values may not be less than 6.5 or greater than 8.5. The turbidity standard is not directly applicable to our data due to differences in measurement units.

Water quality standards are under development by the state of Maryland for nitrogen and phosphorus, but are not currently available. However, the Maryland Biological Stream Survey has identified levels of N and P that represent thresholds above which anthropogenic impacts are likely. These levels are: total nitrogen – 2 mg/L, nitrate nitrogen – 1 mg/L, ammonia nitrogen – 0.02 mg/L, total phosphorus – 0.01 mg/L, orthophosphate (soluble reactive phosphorus) – 0.01 mg/L (Maryland Biological Stream Survey: 2000-2004, Volume II).

I. Stream Sites

Site A was located on the main campus at the boundary road on the stream that formed the outflow from Pond P. Site A exhibited a typical seasonal temperature pattern with summer values normally 2-3°C higher than other stream sites, but never exceeded 30°C. The elevated values are attributable to the summer heating of upstream Pond P waters which formed the major component of flow at this site. Dissolved oxygen values dropped below 5 mg/L in May, June, August, and September. This may also be related to upstream DO conditions in Pond P. The temperature and DO levels observed at site A during the summer are consistent with Pond P water from a depth of 1.5 m. At this depth in the pond oxygen became depleted during the summer yielding decreased DO. At site A pH was generally 6.5-7, slightly lower than the 1.5 m readings in Pond P and generally within water quality standards. Median ammonia nitrogen levels were about 0.12 mg/L, well above the guidelines for anthropological influence. Values at site A were distinctly higher during the summer period of intense stratification in Pond P, another indicator that the high levels were related to pond processes. Nitrate nitrogen at site A was highest in April and May and then declined for the rest of the period. This was generally similar to patterns in Pond P. Levels of nitrate nitrogen exceeded the threshold for anthropogenic influence only in May. Soluble reactive phosphorus never exceeded the limit of detection which was 5 ug/L, well below the anthropogenic threshold. Total phosphorus was higher and did exceed the threshold. In fact stream site A consistently had the highest phosphorus of any stream site with values slightly lower than observed in Pond P. Total suspended solids at site A was among the highest of the stream sites with a median of about 15 mg/L. Values were slightly higher than observed in Pond P at 1.5 m and more similar to those observed at 2.0 m. However, there could be some additional solids picked up between the pond and site A due to resuspension. Site A had the highest volatile suspended solids of any stream site, slightly lower than Pond P values.

Site B had an interrupted period of record since it was dry in May, June and September. This site had a small watershed on the main campus that consisted of a mixture of disturbed forest and parking lots leading to its intermittency. Temperature was well within standards. Specific conductance was somewhat higher than most other stream sites, mainly due to the paucity of summer data which tended to have lower conductance. Conductance was well within the range typical of freshwater. Dissolved oxygen was normally above 5 mg/L, but fell below in July and especially in November. The low value in November could be due to leaf decomposition in the stream. pH was well within water quality standards. Ammonia nitrogen was never above the detection limit of 0.01 mg/L and nitrate nitrogen was always less than 1 mg/L, the indicator value for anthropogenic influence. Soluble reactive phosphorus was always less than 0.01 mg/L while total phosphorus was slightly higher, but still quite low at a median of about 0.02 mg/L. Total and suspended solids were not excessive being similar to other stream sites.

Site C was located on the east campus and received part of its flow from Pond Q. In addition to Pond Q the watershed contained a large amount of bottomland forest and some buildings and parking lots. Temperature at site C followed a typical seasonal pattern with values consistently below site A. Conductivity was low (<300 uS/cm) all year and showed only a slight seasonal pattern. Dissolved oxygen was fairly constant through the year and always exceeded 6 mg/L. These DO's that were somewhat higher than most stream sites were probably attributable to the high DO's coming from Pond Q. Stream pH was somewhat depressed and usually was below the water quality standard of 6.5, but was much higher than at Site D. Median ammonia nitrogen was about 0.04 mg/L, somewhat above the threshold for anthropogenic influence, but much lower than at site A. Nitrate nitrogen was quite high at Site C with all but one month exceeding the 1.0 mg/L threshold. This was consistent with the high values observed upstream in Pond Q. Soluble reactive phosphorus never exceeded 0.01 mg/L and total phosphorus was consistently below 0.03 mg/L. This was interesting since SRP and TP were often elevated in Pond Q. TSS and VSS were consistently below 12 and 5 mg/L respectively.

Site D is located on the east campus and receives flow from a predominantly forested and wetland watershed which actually extends off site to the east. Just downstream of sites C and D their respective streams converge and leave the Goddard east campus flowing north. Site D had among the lowest temperature regimes of any site with median less than 15°C and no values above 22°C. Conductivity was quite low at this site and showed little seasonal fluctuation. Dissolved oxygen was chronically less than 5 mg/L at site D. This is apparently due to decomposition of organic matter in both the forested and wetland sections of this stream. It may also be due to inputs of anoxic groundwater. pH was consistently very low at this site (median of about 4), apparently due to decomposition as well as cation exchange by the abundant *Sphagnum* moss found in the wetland just upstream. Ammonia nitrogen was less than site A, but slightly higher than at site C with a median of about 0.06 mg/L. While this value is above that indicating anthropogenic influence, it is likely that this value is due to natural processes. Nitrate nitrogen was generally less than 1.0 mg/L. Soluble reactive phosphorus was typically below 0.01 mg/L, but on two occasions slightly exceeded this value. Total phosphorus was usually below 0.03 mg/L, but in June a value twice that was observed. TSS and VSS were somewhat higher than at site C, possibly due to detritus coming from the wetland.

Site E was located upstream of Site D on the same tributary. It was dry in June, July, and September. This contributed to the low range in temperature values observed since measurements were not made in the warmer months when the stream was dry. DO trends were similar to those at site D, but slightly higher. PH, while consistently below 6, was much higher than at site D suggesting that the wetlands downstream from site E are mainly responsible for the extremely low pH at site D. Median ammonia nitrogen was 0.02 mg/L, lower than sites C and D. Nitrate nitrogen was elevated at site E compared with site D suggesting that the higher values at site D could be due to reduction of nitrate to ammonia. At site E all values were greater than 1.0 mg/L suggestive human influences. Soluble reactive phosphorus and total phosphorus were low. TSS and VSS at site E were generally low; the high values observed probably represent inadvertent resuspension of fine particles in the shallow water often found at this site.

Site F was on the east campus on one of the few tributaries flowing out of Goddard toward the south. There was a low, but rather constant flow of water at this site. Water from this site entered Pond R about 100 m downstream. Temperature at site F was typical of most Goddard streams with median of about 16°C and no values above 26°C. Site F had by far the greatest specific conductance with median of over 1000 uS/cm. Dissolved oxygen was consistently well above 5 mg/L. Site F had consistently the highest pH of all stream sites at about 7 mg/L. Ammonia nitrogen was always below detection limit of 0.02 mg/L. Nitrate was almost always below 1.0 mg/L. Soluble reactive phosphorus was consistently slightly elevated, but the median was still less than 0.01 mg/L. The median total phosphorus was about 0.04 mg/L. Median TSS and VSS values at Site F were the lowest of any stream site.

Site G was located on the east campus above Pond Q and about 50 m downstream of a cluster of buildings. Temperature at Site G was similar to other stream sites on East campus. Specific conductance was higher than sites C, D, and E, but lower than site F. DO at site G was very variable with a median above 5 mg/L, but several observations below that value. PH at Site G was consistently between 6 and 7. Ammonia nitrogen was quite elevated similar to site A. The exact cause for this is unknown, but perhaps related to the buildings and parking lots immediately upstream. Nitrate nitrogen was generally below the 1.0 mg/L threshold. Soluble reactive phosphorus hovered around 0.01 mg/L; median total phosphorus was about 0.04 mg/L. Both TSS and VSS were low compared with other sites.

II. Pond Sites: Water Quality

Pond sites were characterized in two ways: depth-integrated averages and depth profiles. Temperature exhibited a distinct pattern of stratification in both Ponds P and R. Stratification was in place by the first sampling in April and continued through September. By early November when sampling concluded temperatures had declined and both ponds had mixed and become isothermal (constant temperature) with depth.

Thermal stratification had a major impact on the depth distribution of numerous parameters. Dissolved oxygen was distinctly depth-stratified in both ponds. In Pond P dissolved oxygen exhibited a distinct inverse stratification (higher DO at the bottom) in April due to an extensive development of benthic algae and resulting photosynthesis. In fact, DO concentrations reached 150% of saturation due to the rapid production of oxygen by the benthic algae and the lack of circulation of the water column. This benthic algal development was probably facilitated by the drought which allowed the water column to become very clear in the early spring. On the April sampling of Pond P the secchi disk depth was 2.1 m indicating that the bottom of Pond P at about 2.5 m had abundant light for photosynthesis. The April secchi depth was about 3 times the secchi disk depth for the remainder of the year. By May the oxygen profile had reversed itself in Pond P forming a clinograde curve, i.e. the surface (epilimnion) became supersaturated while the bottom waters (hypoliminion) became virtually devoid of water (hypoxic). While the Hydrolab data indicated that there was still some oxygen in the bottom waters (~1 mg/L), the real value may have been 0 (anoxic) because oxygen sensors generally are not accurate in this range. By November oxygen concentrations in Pond P had become nearly constant with depth consistent with pond mixing. In Pond R the clinograde type oxygen curve was seen from the entire period April to September with concentrations at or slightly above 100% in the surface layers and near 0 mg/L at the bottom. In November, while temperature showed only slight stratification, DO was substantially higher at the bottom indicating that limited stratification was again occurring.

The DO values observed in the lower portion of both ponds were below the water quality standard of 5 mg/L for a large portion of the year. As currently formulated these hypoxic water masses would clearly violate Maryland water quality standards. However, it is common for the hypolimnion of lakes and ponds to go anaerobic during periods of stratification. Proposed changes in the Maryland water quality standards would allow for exceptions to DO standards during stratification and these areas would no longer be in violation. While hypoxia is common in the lower layers of lakes and ponds it is by no means desireable. It removes these areas as habitat for vertebrate and most invertebrate animals. An important consideration should be the rate of hypoxia formation and thus the length of time during which hypoxia exists. In the case of Ponds P and R hypoxia set up early in the spring and remained throughout the summer and into the fall.

Due to its shallow nature light penetrated to the bottom of Pond Q throughout the year. In fact secchi depth readings were not possible for most of the year due to the fact that the disk was still visible at the bottom of the pond. This shallow and clear water column limited stratification and resulted in high oxygen concentrations throughout the year.

pH generally followed a pattern similar to that in dissolved oxygen due to the fact that both parameters are governed in large part by the interaction of photosynthesis and respiration. Photosynthesis will increase pH and respiration will decrease pH. Thus, the high pH's observed in the hypolimnion of Pond P in April under high photosynthesis reverted to lower values in May when light decreased and respiration dominated the hypolimnion. Likewise, in Pond R the clinograde DO curves were replicated in clinograde pH curves again reflecting the differential balance in photosynthesis and respiration. In both ponds surface pH was more variable during the clinograde period owing to the greater temporal variability in the photosynthetic processes than in the hypolimnion where respiration was more constant. In Pond Q pH values were similar to those observed in the epilimnion of the other two ponds reflecting the dominance of photosynthesis in this pond.

Ammonia nitrogen values were generally quite low, at or below the detection limit of 0.02 mg/L, in the epilimnion of both ponds. However, the hypolimnion of Pond P showed many readings much higher than this. Typically, the greater the depth, the greater the ammonia level. For example at 2.0 m ammonia was typically 1-2 mg/L whereas at 1.5 m values of 0.1-0.7 mg/L were the rule. This pattern can be explained in two ways: (1) ammonia is being created by reducing bacteria stripping oxygen from nitrate to use as a terminal electron acceptor in the absence of oxygen and/or (2) ammonia is being released from sediments. The latter process may be dominant because it is observed that nitrate also increases in the hypolimnion in Pond P. In Pond R only one hypolimnion measurement was above detection limits

Nitrate nitrogen values also followed a seasonal pattern with a maximum in May and a decline through the rest of the year. Nitrate was generally higher in Ponds Q and R than in Pond P. Nitrate typically demonstrated an increase with depth in Pond P. In the spring surface values were as high as 1.5 mg/L decreasing to about 0.5 mg/L in June and July and then to 0.2 mg/L by August. At depth values also decreased steadily during the summer and early fall. In Pond R there was little consistent depth stratification, but a similar decline seasonally from over 1.5 mg/L in May to less than 0.5 mg/L in August.

Soluble reactive phosphorus was quite low in Pond P at all times reflecting rapid uptake. While this was certainly to be expected in the upper layers, it was a little surprising that SRP did not accumulate in the lower layers where decomposition was active which would have released SRP into the water column. Values in Pond R were also low and did not follow a consistent depth profile. Pond Q consistently had the highest SRP values.

Total phosphorus showed a general increase with depth in both Ponds P and R at most depths. Total phosphorus values were generally well above the 0.01 mg/L threshold for anthropogenic influence.

Total suspended solids were substantially higher in Pond R than in Pond P reflecting a higher degree of stormwater input into Pond R which was designed as a stormwater treatment pond. Volatile suspended solids, the organic component of suspended solids, was more similar between the two ponds suggesting that Pond P was more dominated by organic solids such as phytoplankton and detritus whereas Pond R was more dominated by inorganic solids such as sediment and clay particles.

III. Pond Sites: Plankton

Chlorophyll concentrations indicated that all of the ponds would be classified as eutrophic or highly enriched. Table 4 compiles average values of Carlson's trophic state index (TSI) as determined by chlorophyll, secchi depth, and total phosphorus (Carlson 1977). Chlorophyll is the most direct measure of trophic status or productivity since it measures the actual standing crop of phytoplankton in the pond. Secchi depth is an indirect measure based on the fact that the more phytoplankton, the lower the light transparency. Total phosphorus is an indirect measure based on the fact that phosphorus is normally the limiting nutrient and sets an upper bound on productivity by phytoplankton. In Pond P the three measures of TSI are quite similar indicating that phytoplankton have utilized all of the total phosphorus and are the main factors responsible for light attenuation. In Pond R the total phosphorus TSI was slightly higher than the chlorophyll number suggesting that not all of the phosphorus has been incorporated in phytoplankton. The Secchi TSI was substantially greater than the chlorophyll number indicating that there was a substantial amount of light attenuation not attributable to chlorophyll, presumably due to suspended clay particles from stormwater runoff. This suggests that phytoplankton in Pond R are limited as much by light at by phosphorus.

The types of phytoplankton observed in the Goddard ponds are typical of freshwater pond ecosystems in the mid-Atlantic. Since cyanobacteria are smaller, they often dominate phytoplankton cell numbers. Using biovolume the predominance of green algae along with the cyanobacteria is noteworthy in Pond P. The spring dominance of diatoms in Pond Q is a common occurrence in freshwater ponds. The dominance by the euglenoids *Euglena* and *Trachelomonas* in Pond R for much of the year differentiated it from Ponds P and Q. Ponds P and Q experienced significant abundances of the bloom forming cyanobacterium *Microcystis*, but it did not reach nuisance proportions. The green alga *Staurastrum* was common in both Ponds P and Q.

Total rotifer densities in Pond P were on the high side. By comparison, annual maximum rotifer densities in the tidal freshwater Potomac River are generally in the range 2,000-10,000/L (Jones and Kelso 2003). The values observed in the other two ponds were on the lower end of this range. A midsummer peak in total rotifers is typical in the tidal freshwater Potomac. The peak in Ponds P and R was early summer and Pond Q actually had highest rotifer densities in April. In the tidal freshwater Potomac River *Brachionus* is the dominant genus, with *Keratella* and *Polyarthra* subdominant. In Pond P *Polyarthra* was exceedingly abundant in June and *Keratella* was dominant most of the remainder of the year. In Pond Q *Keratella* was dominant on most early dates with *Synchaeta* most abundant in May. *Brachionus* was the most numerous rotifer in July after other genera had declined. In Pond R *Keratella* again attained the highest levels in spring and early summer. *Filinia* was codominant in June. *Brachionus* was codominant with the other two genera in August. Thus, the Goddard ponds exhibited relatively high rotifer densities and a shift toward *Keratella* and away from *Brachionus*, but otherwise were similar to rotifer dynamics in the tidal freshwater Potomac River.

The small bodies cladoceran *Bosmina* attained densities in Pond P similar to those observed in the tidal freshwater Potomac River. Densities of *Bosmina* were consistently lower in Ponds Q and R. Other cladocerans were much more abundant in Pond P than in the tidal freshwater Potomac River. Maximum densities for *Daphnia* of 150,000/m3 compare with a twenty year maximum of 20,000/m3 and typical maxima of 1,000/m3 in the tidal freshwater Potomac River. *Diaphanosoma* maximum density in Pond P of about 300,000/m3 also exceeded the twenty year maximum of 100,000/m3 in the tidal freshwater Potomac River. The *Ceriodaphnia* density of 450,000/m3 in May was several orders of magnitude greater than ever observed in the Potomac study. Limited data for Pond R indicate lesser, but still quite significant number of cladocerans.

Copepod nauplii were more similar in abundance in all three ponds. Values were again generally somewhat above those observed in the tidal freshwater Potomac River. Adult copepods such as cyclopoids and *Diaptomus* were found at levels somewhat greater than those observed in the tidal freshwater Potomac River.

CONCLUSIONS

The aquatic survey of NASA's Goddard Space Flight Center undertaken in 2002 sampled streams at seven sites and sampled three ponds on a monthly basis from April through November. Results were analyzed and compared with water quality standards and guidelines as well as other studies in the area.

In general the water quality in the streams at Goddard was consistent with expectations for Maryland coastal plain streams and in compliance with state water quality standards and general guidelines. There were, however, some notable exceptions:

- Dissolved oxygen and pH were below state water quality standards at Site D and to a lesser extent at Site E, both on the same tributary on the East Campus. There was no obvious anthropogenic source for these conditions and they may be related to natural conditions in the watershed such as decomposition and significant peat moss populations in wetlands along the stream. Dissolved oxygen was also somewhat lower than expected at other sites.
- Conductivity was elevated at Site F which was originally attributed to discharge from a heating/cooling plant, but this has been ruled out by NASA personnel. The cause of this elevation is unknown.
- Ammonia nitrogen and total phosphorus were elevated at Site A due to discharge of water rich in these compounds from Pond P. Pond discharge was probably also responsible for depressed oxygen at Site A.

It is important to note that sampling of streams at Goddard was restricted to base flow conditions and that 2002 was an exceptionally dry year.

The ponds at Goddard exhibited generally eutrophic, or highly enriched, conditions. The two larger ponds P and R both underwent seasonal stratification which, together with the eutrophic status, resulted in severe oxygen depletion in the lower layers of both ponds for most of the year. This resulted in elevated values of ammonia nitrogen, especially in Pond P. Nitrogen and phosphorus values were generally in the range considered indicative of anthropogenic inputs as would be expected from the fact that much of the Goddard site is developed. Pond R appeared to be the most heavily impacted by stormwater runoff consistent with its role as a stormwater treatment pond. The phytoplankton in all ponds were characterized by typical freshwater species. The zooplankton in Pond P was composed of forms characteristic of mid-Atlantic freshwater habitats at densities that indicated robust populations.

Studies of the streams continued in 2003 with further water quality measurements as well as benthic macroinvertebrate sampling. Pond P was the subject of a detailed stratification-nutrient-phytoplankton study in 2003. These will be reported at a later date.

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Table 1. Stream Site Characteristics. NASA Goddard Aquatic Study. 2002.

Site	Campus	Drainage Basin	Watershed Area (ha)	Land Uses	Sampling Site Latitude Longitude
A	Main	Beaverdam	75.1	Pond P, Buildings, Roads, Lawns	39°00’08.11315”N 76°51’33.97626”W
B	Main	Beaverdam	9.9	Forest, Buildings, Roads	39°00’07.23503”N 76°51’10.30973”W
C	East	Beaverdam	80.4	Fields, Forest, Buildings, Roads, Pond Q	39°00’16.47819”N 76°50’33.82680”W
D	East	Beaverdam	50.2	Forest, Wetlands	39°00’16.39941”N 76°50’33.29803”W
E	East	Beaverdam	20.0	Forest, Wetlands	39°00’03.28041”N 76°50’04.25953”W
F	East	Patuxent	8.5	Buildings, Roads, Lawns, Forest	38°59’37.01768”N 76°50’28.79477”W
G	East	Beaverdam	39.3	Buildings, Roads, Forest	39°00’00.37569”N 76°50’32.28952”W

Note: 1 ha = 2.47 acres, 1 ha = 0.00386 mi^2, 1 ha = 0.01 km^2

Table 2. Pond Site Characteristics. NASA Goddard Aquatic Study. 2002.

Pond	Campus	Drainage Basin	Maximum Depth (m)	Pond Area (ha)	Watershed Area (ha)	Land Use	Sampling Site Latitude Longitude
P	Main	Beaverdam	2.5	2.37	74.0	Buildings Fields Roads	39°00’03.37312”N 76°51’32.98417”W
Q	East	Beaverdam	0.5	0.18*	60.7	Fields Buildings Roads	39°00’07.26986”N 76°50’37.05616”W
R	East	Patuxent	1.5	0.60	16.3	Buildings Roads Forest	38°59’33.57837”N 76°50’31.08120”W

*Includes inundated emergent wetland area.
Note: 1 ha = 2.47 acres, 1 ha = 0.00386 mi^2, 1 ha = 0.01 km^2

Table 3. Meteorological Data for 2002. National Airport. Monthly Summary.

MONTH	Air Temp (^{o}C)		Precipitation (cm)	
January	5.3	(1.4)	3.4	(6.9)
February	5.9	(3.0)	1.2	(6.9)
March	8.7	(8.4)	8.6	(8.0)
April	15.6	(13.6)	8.8	(6.9)
May	18.4	(19.1)	5.5	(9.3)
June	24.5	(24.4)	9.7	(8.6)
July	27.2	(26.7)	5.6	(9.6)
August	27.3	(25.8)	4.1	(9.9)
September	22.8	(21.8)	5.3	(8.4)
October	14.8	(15.4)	12.7	(7.7)
November	8.4	(9.9)	11.0	(7.9)
December	2.9	(4.1)	11.3	(7.9)

Note: 2002 monthly averages or totals are shown accompanied by long-term monthly averages (1961-1990).

Source: National Climatic Data Center, National Oceanic and Atmospheric Administration for temperature and precipitation.

Table 4. Carlson Trophic State Index (TSI) for Goddard Ponds.

	Predictor		
Pond	Secchi	Chlorophyll a	Total Phosphorus
P	64.0	62.7	61.2
Q	-----	63.3	74.8*
R	79.7	62.8	66.9

Values shown are averages for the period April – November 2002.
Values 60-80 indicate eutrophic conditions. Values >80 indicate hypereutrophic conditions.
*If high September total phosphorus value is excluded, the total phosphorus TSI is reduced to 70.6 at Pond Q.