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Impact of Aquatic Vegetation on Water Quality of the Delaware River Estuary

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Abstract

The objective of this study was to evaluate the impact of aquatic vegetation, emergent and submergent, on water quality of designated sections of the tidal freshwater Delaware Estuary. For the purpose of this study, we define water quality as dissolved oxygen, total phosphorus and nitrogen and determine flux rates for these parameters from emergent aquatic plants and wetlands in the tidal freshwater area of the estuary. The breadth of the survey region encompasses two study segments within the tidal freshwater estuary from the head of tide near Trenton, NJ (Morrisville-Trenton Bridge) to just downstream of the confluence with Chester Creek, PA (Commodore Barry Bridge). Our preliminary estimates are presented in a form that can be incorporated into the Delaware Estuary Model.

Specific tasks that were investigated for the two study areas include: (1) estimate aerial extent of emergent aquatic vegetation and the extent of each of the dominant species; (2) determine the biomass of the dominant species and productivity; (3) estimate the daily-average, net dissolved oxygen (DO), phosphorus and nitrogen fluxes within the water column, per estuary river-mile and over the range of the study time period based on a comprehensive literature survey; and (4) evaluate the potential impact of submerged aquatic vegetation on DO, phosphorus, and nitrogen net fluxes within the water column. Finally, we recommend a plan for potential future evaluation of emergent and submerged aquatic vegetation beds, including periphyton and epiphytes for the entire estuary, in terms of their ecological importance upon the tidal estuary in relation to water quality, productivity, and habitat.

We assessed the extent of tidal freshwater wetlands in the upper tidal Delaware River. To accomplish this task we used historical aerial photography, current aerial photography, and ground truth information. We identified a total of just over 1,400 ha of tidal freshwater wetlands in the lower and upper study areas with slightly more area in the upper estuarine zone (56% of total). Mudflat estimates, while not a major focus of this research, were taken from NWI estimates. Although the upper study area has a higher total area of marsh, on a per river-mile basis the lower study area (10 river-miles) has nearly 62 ha of marsh per river-mile while the upper study area (26 river-miles) has only 30 ha of marsh per river-mile. In both study areas, we found much more high than low marsh habitat (65% and 75% for the lower and upper study areas, respectively). The majority of the marsh areas were located in a small number of tributaries, with very little marsh along the mainstem of the Delaware River. The largest marsh areas were found in Mantua Creek (256 ha), Darby Creek (120 ha), Woodbury Creek (103 ha), Rancocas Creek (466 ha), and Crosswicks Creek (206 ha). However, there does appear to be a significant number of polygons and areas which were dominated by high marsh vegetation in 1977/78 and are dominated by low marsh vegetation today.

Our estimates are based on historical data with verification using an aerial overflight and ground-based assessments. Twenty-six sample polygons were identified and classified in 1997, either from ground or aerial reconnaissance. Changes in specific polygons from high marsh to low marsh indicated real changes and not errors in our photointerpretation. We interpret this apparent trend from high to low marsh as being consistent with previous studies. In addition, relative sea level has been rising at a rate 0.27 mm/yr for the last 50 years, which would produce a rise of about 60 mm between 1977 and 1997. If these vegetation trends can be substantiated, it implies that relative sea water level is rising faster than the rate of marsh accretion in these areas. Further study is needed to substantiate this possibility and to verify these changes.

As part of this study, we sampled numerous quadrants in the upper and lower emergent aquatic vegetation areas (EAVA) of the Rancocas and Mantua creeks. Considerable variability exists along the shores of these tributary creeks in the elevation profile of the creek banks. The broad-leaved *Nuphar advena* (=*luteum*, =*variegatum*) occurred in uniform stands along the edge of the channel and in ponded and pond-like areas. These areas typically contained at least 95% of the biomass and percent cover in the form of *N. advena*, although both arrow arum (*Peltandra virginica*) and pickerelweed (*Pontederia cordata*) also contributed to the biomass at somewhat less depth of inundation. From low to high marsh, the second prominent emergent aquatic vegetation (EAV) type was wild rice (*Zizania aquatica*), which typically formed a narrow to wide band of tall grass just landward from the low marsh. Wild rice was widespread at both upstream and downstream locations and composed 66-100% of the total dry EAV biomass, and became nearly monospecific in the bulk of its range. High marsh habitats contained much greater EAV species diversity, but certain species formed patches where they dominated EAV biomass. Cattail (*Typha angustifolia*) and common reed (*Phragmites australis*) also formed nearly monospecific stands having tremendous biomass. These tall plants were very patchy in occurrence, however, often being found only on high marsh levees. No measurements of *P. australis* biomass were made. Numerous other species were found during this study, and their spatial distribution in the tidal marsh was nearly always associated with specific flooding conditions that they are specially adapted for. Based on these flooding characteristics, high marsh vegetation will typically be underwater for up to 4 h per 12-h tidal cycle. The depth of flooding during this time may typically reach 0.3 m. Conversely, the vegetation typically will be out of the water for 2/3 of that tidal cycle. The various low marsh habitat types will be underwater from 3/4 to all of a 12-h tidal cycle up to a maximum depth of 1 or even 2 m. Low marsh vegetated with *N. advena* and *P. virginica* is predominantly within the stream bank to pond-like hydrologic regime, and is drained at normal low tide.

EAV biomass was high in all zones, as might be expected during August at the late peak of the growing season. The low marsh areas dominated by *Nuphar advena* contained 241 g dry weight m^{-2} and 811 g dry weight m^{-2} at Rancocas and Mantua creeks, respectively. Although low marsh biomass at Mantua Creek appeared higher than at Rancocas Creek, these means were not significantly (t-test, $p>0.05$) different from each other because of high variability among replicates at Mantua. The zone of EAV dominated by wild rice (*Zizania aquatica*) contained remarkably consistent biomass at both sites. At Rancocas Creek, the mean biomass in the wild rice zone was 323 g dry weight m^{-2}, and 382 g dry weight m^{-2} was measured at Mantua Creek. These values were not significantly (t-test, $p>0.05$) different between sites. EAV biomass was found to be greatest in high marsh patches of cattails (*Typha angustifolia*).

For the purposes of this study, we sought to establish estimates of EAV biomass in either the low or high marsh because these two zones were most easily identified and quantified by remote sensing (see above). Statistical procedures were therefore needed to determine whether EAV biomass was similar among marsh areas to permit derivation of a single, characteristic value for high marsh EAV biomass. To estimate the geographical extent of EAV biomass within the designated upstream and downstream river segments of the Delaware Estuary, we sought to calculate a universally applicable average or median EAV biomass for either the low marsh or high marsh, since the remote sensing data were limited to that level of sensitivity. The median low marsh EAV biomass at the upstream segment (e.g., Rancocas Creek) and downstream segment (e.g., Mantua Creek) was calculated to be 232 and 731 g dry weight m^{-2}, respectively. To calculate a similar figure for the high marsh, we pooled replicate data for different marsh areas and disregarded data for cattails due to its patchiness. Therefore, our estimates for the median EAV biomass in the high marsh of the upstream (e.g., Rancocas Creek) and downstream (e.g., Mantua Creek) reaches were 402 and 327 g dry weight m^{-2}, respectively, and it is important to recognize that these figures are conservative estimates since high biomass patches of *Typha* (and *Phragmites*) were not considered (i.e., aerial photography only differentiated high marsh versus low marsh and not specific vegetation types within each elevation zone).

Overall, our ground surveys of EAV biomass and productivity for each study site produced values that were either consistent with or lower than comparable literature reports. Our productivity and biomass values are considered conservative (i.e., low, underestimated by up to 2- to 3-fold). This could be explained by either or both of the aforementioned sampling problems: 1) we had to assume that no annual EAV production occurred beyond what was measurable as standing biomass in late August (produces an underestimate), and 2) species of EAV known to have much higher biomass and productivity (e.g., cattail and common reed) were not considered herein due to their patchy nature and would underestimate the total biomass.

To determine the effect of EAV on nutrient balances, estimated areas of high marsh, low marsh and mudflat areas, in the tidal freshwater sections of the Delaware Estuary were integrated with: 1) estimates of the median biomass of emergent aquatic vegetation, 2) average biomass elemental concentrations, and 3) the amount and average uptake rate of nitrogen, carbon and phosphorus determinations for the tidal freshwater areas of the upper Delaware Estuary. Aboveground N, C, and P biomass incorporation rates (i.e., kg/river-mile-day) ranged from approximately -0.5 to -131 kg N/RM-day for N, -10 to -3,600 kg C/RM-day for C, and -0.1 to -19 kg P/RM-day for P. These rates are for the time period between 1 July and 15 September. The highest amount of plant biomass, N, C and P , and highest rates of production were recorded in Darby (RM 85), Mantua (RM 89), Rancocas (RM 111) and Crosswicks creeks (RM 128). Approximately similar amounts of N and P in the aboveground biomass were calculated in the upper and lower EAV areas. From our calculations, an estimated 80 metric tons of N (1000 kg = 1 metric ton) and 13 metric tons of P are retained in the living aboveground biomass in the two tidal freshwater wetland areas of the Delaware Estuary in 1997. Our overall estimate for these elements do not take into account within-season turnover of plant biomass by either leaf mortality or herbivory. This can be a significant portion of the annual productivity and produces and underestimate. An additional bias is the lack of belowground biomass storage which was not sampled and could also cause an underestimate.

Estimates for sediment-water exchange processes (i.e., advection/diffusion) were made based on literature exchange rates and wetland areas from the current study. Estimates for sediment-water exchange were made for denitrification, ammonium+ammonia, nitrate+ nitrite, ortho-phosphate, and dissolved oxygen (i.e., sediment oxygen demand). While there were substantial variations in the literature data, the results indicate that an active exchange of material occurs between the sediments and water column in the Delaware tidal freshwater wetlands. For nitrogen, on average for the entire area and time period, 30 metric tons of nitrate+nitrite-N moved into the sediments, while slightly more nitrogen (33 metric tons) was removed from the sediments (and system) via denitrification (i.e., $NO_3^- \div N_{2\,(g)}$). A portion of the nitrate-nitrogen needed for denitrification can be supplied from coupled nitrification-denitrification within the sediments. Also, on average the wetland systems appear to be a large source of dissolved ammonium+ ammonia (75 metric tons) to the overlying water, while 77 metric tons of nitrogen was potentially-buried in the wetland areas. On average, approximately 17 metric tons of ortho-phosphate was removed from the wetland sediments to the overlying water while approximately 12 metric tons of phosphorus is permanently buried within the sediments. There was substantial variability in these estimates. Sediment oxygen demand estimates also were variable with approximately 3,000 metric tons of oxygen moving into the sediments from the overlying water.

Another objective of this project was to estimate the effect that emergent aquatic plants within the two study areas have on the oxygen concentration within the water column. Within the scope of this study, we utilized the carbon incorporation rate (i.e., estimated plant productivity on a carbon basis) and production of oxygen via photosynthesis. The amount of oxygen produced per river-mile per day was calculated using a 170-day growing season with linear rates over this time period. Total gross oxygen production ranged from 26 kg O_2/RM-day at river-mile 109 to approximately 9,500 kg O_2/RM-day at river-mile 111. As with the nitrogen, carbon and phosphorus calculations, the higher rates were calculated in sections of the river with highest areas of high and low marsh area (e.g., Rancocas Creek, Darby Creek and Mantua Creek). It is unclear how much of this oxygen actually gets into the water column and this would be an important area for future study. Plant respiration could account for a substantial percentage of the oxygen produced, while the flux of oxygen out of the plants through the roots could be consumed by sediment microbial processes. This could be especially important in the warmer summer months when microbial processes are maximal and sediment oxygen demand highest. Therefore, little oxygen may be released from emergent plants into the overlying water.

Our evaluation of information concerning the area of submerged aquatic vegetation (SAV) in the study areas was hampered by the paucity of available data and estimates for nutrient uptake, biomass, or oxygen dynamics. Hence, we were unable to provide estimates for these processes. Literature information, however, does suggest that SAVs can have a substantial impact on water quality and sediment dynamics. As with emergent plants, SAV can oxygenate the sediments affecting nitrification-denitrification dynamics. Additionally, nutrient uptake from both the roots and shoots can have an impact on concentrations within the water column. Future studies should be directed towards determining the extent of SAVs in the tidal freshwater areas, and calculating area-specific uptake rates.

The current study presents a first cut at the ranges in many biogeochemical fluxes within the tidal freshwater section of the Delaware Estuary. Importantly, all parameters and rates are strongly influenced by the extent (i.e., area) of tidal wetlands by river mile. Our results quantified the extent of emergent aquatic vegetation and wetland areas within the tidal freshwater section of the Delaware Estuary. As expected, there is active movement of bioactive elements (e.g., nitrogen, phosphorus, etc) throughout and within the tidal freshwater areas. The comparison between the rates derived from this study, and estimated inputs to the tidal freshwater portion of the Delaware suggests that removal of nitrogen via plant uptake, denitrification, and burial are minor, while on average, approximately 10% of the watershed sources of phosphorus can be buried within the marsh system. Although the current area of tidal freshwater wetlands is greatly reduced compared with their historical extent, the remaining wetlands may have an impact on the chemical form of nitrogen and phosphorus (e.g., oxidized forms versus organic forms) to the more saline portion of the estuary and the eventually the coastal ocean. Compared to salt marshes in the lower estuary, tidal freshwater wetlands are some of the first aquatic habitats to be affected by runoff from the upper Delaware Basin watershed and large urban areas, and hence they could have a greater per-area impact on estuarine water quality. However, to fully evaluate these relationships numerous unknowns need to be addressed. We need improved estimates of wetland areas, understanding the roles of benthic and epiphytic algae, as well as the importance of secondary producers on oxygen and nutrient cycles. The present study relied heavily on published data from other locales, and we recommend site-specific rate data be collected in an integrated manner to enable the complex biogeochemical relationships to be comprehensively modeled for the entire estuary.

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1.0 Introduction

The upper reach of the Delaware Estuary contains unique and extensive freshwater tidal wetlands with a diverse assemblage of emergent aquatic vegetation. The transport and transformation of chemicals through such systems involves a great number of interrelated chemical, biological and physical processes. Hydrologic interactions between river water and adjacent wetland environments enable specific biogeochemical processes to potentially have an impact on the water quality of the river. These processes (e.g. nutrient uptake/regeneration, denitrification, oxygen consumption/production) can alter both the chemical form and total load of material flowing through the freshwater tidal region of the estuary and affect material budgets within the more saline portions of the estuary.

Patrick et al. (1973) surveyed the entire Delaware Estuary for fresh and marine wetlands and estimated that there was approximately 2,310 ha (1 ha = 1 X 10^4 m^2) of tidal wetlands in the freshwater portion of the estuary. They found that tidal freshwater wetlands ranged in size from 30 ha to over 810 ha and consisted of a diverse population of emergent plants such as *Scirpus americanus*, *Polygonum punctatum*, *Zizania aquatica*, and *Nuphar advena*. In more disturbed areas (i.e., channelized, ditched or filled), *Phragmites communis* was a common species. They concluded that there were more disturbance and impact in the upper estuary's freshwater wetlands than in salt marshes located further down the estuary, with the greatest degradation and loss being in the Philadelphia region. Whigham and Simpson (1976) estimated that roughly 3,200 ha of tidal freshwater wetlands exist in the Delaware Estuary, while Tiner and Wilen (1988) estimated that approximately 1,000 ha remain in the upper estuary. Although greatly reduced in size, these wetlands are still thought to play a key role in the overall health of the river by buffering or altering the amount of nitrogen and phosphorus being transported from the watershed to the lower estuary and possibly affecting oxygen concentrations in the adjacent river water.

Few studies have examined the effects emergent aquatic plants have on water quality within the tidal freshwater portion of the Delaware Estuary. Whigham and Simpson (1976) examined the aerial extent and aboveground productivity of dominant emergent plants in the Hamilton Marsh system (486 ha) on the Delaware River. They estimated that nearly 32 metric tons (1 metric ton = 1000 kg) of nitrogen are incorporated into this wetland during the growing season.

They also recognized the importance of edaphic algae in the nutrient cycles of the wetland. Early studies by Grant and Patrick (1970) and those of Simpson et al. (1983a,b) demonstrated that tidal freshwater marshes in the Delaware River can retain specific forms of nitrogen and phosphorus in the spring/summer, but can release other forms of nitrogen and phosphorus during the fall.

Additional investigations into the role of freshwater tidal wetlands in estuarine biogeochemical cycles have been conducted within other areas along the mid-Atlantic region. For example, Ziegler (1993) and Ziegler et al. (1993; 1998) showed that substantial fluxes of inorganic nitrogen can occur in various sections (i.e., high versus low marsh) of a tidal freshwater marsh in the Patuxent River. In the same wetland, Groszkowski (1995) measured substantial denitrification rates that could have an impact on the inorganic nitrogen concentration of the mainstem river. These and other published studies indicate that there is substantial seasonality in the import/export flux of nitrogen and phosphorus in tidal freshwater wetlands.

In freshwater tidal wetlands, the fate of nutrients varies by topographical region and vegetational zone. Plant debris in the low and high marshes degrade at different rates, and consequently, have different influences on the flux of nitrogen in and out of marsh sediments (Odum et al., 1984; Whigham et al., 1989). The low marsh and newly developed marsh inundated by tidal waters for the majority of the tidal cycle, has only a small amount of litter, and hence stores very little nutrients. The high marsh, which is more mature, typically has more abundant litter and tends to store a greater amount of nutrients (Mitsch and Gosselink, 1993). Freshwater tidal wetlands contain a complex variety of microenvironments with regard to vegetation structure and composition, litter accumulation, and soil conditions (Odum et al., 1984; Simpson et al., 1983a,b,c).

The areal abundance and potential effects of submerged aquatic vegetation (SAV) on water quality of the upper Delaware Estuary are largely unknown. In the Chesapeake Bay, the importance of SAV in the overall ecosystem is well documented (for example, Carter et al., 1991; Caffery and Kemp, 1992 and others) and SAV protection is rapidly escalating (Chesapeake Bay Program, 1995; Bay Journal, 1998). However, Schuyler (1988) stated that virtually no efforts have been made to document the extent of plants that grow underwater (submergents) or have floating leaves (planmergents) in the tidal freshwater portion of the Delaware River. Submergents and planmergents (collectively grouped as SAV) are generally found in the lower portion of the intertidal zone where they are attached to specific substrates or rooted in the sediment. As with the Chesapeake Bay, the decline and absence of SAV in the freshwater section of the Delaware Estuary may be associated with anthropogenic inputs such as nutrients, suspended matter, as well as specific contaminants. In the Delaware Estuary, approximately 12 species of SAV have been observed in the tidal river since 1970, including: *Vallisneria americana*, *Myriophyllum spicatum*, *Elodea nuttallii*, *Najas flexillis*, *Potamogeton* sp. and others (Schuyler, 1988). Submerged aquatic vegetation can take up nutrients from both the water column and sediments as well as alter oxygen concentrations of the overlying water (Caffrey and Kemp, 1991; 1992; Short and McRoy, 1984; Sand-Jensen et al., 1982).

We also have a poor understanding of the biogeochemical role of edaphic and benthic microalgae. There is a growing body of evidence suggesting that annual production by these plants might be comparable to emergent aquatic vegetation (EAV) and SAV (Whigham and Simpson, 1976; Zedler, 1980; MacIntyre and Cullen, 1995) and this production can be an important food source for marsh consumers (Kreeger and Newell, 1998a,b in preparation).

Natural wetlands and aquatic systems have been used to help treat wastewater discharges since at least the beginning of this century (Kadlec and Knight, 1996). Due to a growing interest in the use of wetlands (tidal and non-tidal) as systems to treat nutrient inputs from various municipal and industrial sources, a better understanding and quantification of the actual benefits is developing. Because wetlands have a higher rate of biological activity than most systems and there is a near constant innundation and mixing of water, they can transform or sequester many pollutants over specific time scales. While many studies indicate improvements of water quality in relatively small systems and wetlands, it is the objective of this study to provide an estimate of the potential impact that tidal freshwater wetlands have on nutrient and oxygen cycling in the entire upper Delaware River. In addition, we did a preliminary review of the possible role of SAV in the tidal freshwater river of the Delaware Estuary.

1.1 Objective and Scope Statement

The overall objective of this study was to evaluate the impact of aquatic vegetation, emergent and submergent, on water quality of designated sections of the tidal freshwater Delaware Estuary. For the purpose of this study, we define water quality as the levels of dissolved oxygen, total phosphorus and nitrogen. The breadth of the survey region is described below and encompasses two study segments within the tidal freshwater estuary from the head of tide near Trenton, NJ (Morrisville-Trenton Bridge) to just downstream of the confluence with Chester Creek, PA (Commodore Barry Bridge). Our preliminary estimates of the role of aquatic vegetation in modifying water quality in this reach are presented in a form that can be incorporated into the Delaware Estuary Model.

Specific tasks that were investigated for the two study areas include: (1) estimate aerial extent of emergent aquatic vegetation and the extent of each of the dominant species; (2) determine the biomass of the dominant species and productivity; (3) estimate the daily-average, net dissolved oxygen (DO), phosphorus and nitrogen fluxes within the water column, per estuary river-mile and over the range of the study time period based on a comprehensive literature survey; and (4) evaluate the potential impact of submerged aquatic vegetation on DO, phosphorus, and nitrogen net fluxes within the water column. In addition, we recommend a plan for potential future evaluation of emergent and submerged aquatic vegetation beds, including periphyton and epiphytes for the entire estuary, in terms of their ecological importance upon the tidal estuary in relation to water quality, productivity, and habitat.

1.2 Approach

Our approach was to determine the areas and types of tidal freshwater wetlands along with biogeochemical measures to determine the potential impact these tidal wetlands have on water quality in the tidal river. Historic remote sensing data of wetland areas was augmented with field observations to estimate extent and marsh type. In addition, one overflight with a small plane was used to help verify remote sensing and field observations. Field measurements and remote sensing data were not used to estimate extent of submerged aquatic vegetation (SAV). Literature information, if available, was used to acquire data on SAV extent and dynamics. Since many of our estimates were based on existing literature data, which is very limited for the Delaware Estuary and tidal freshwater wetlands in general, our integrated assessment must be treated as a preliminary investigation requiring additional research efforts to quantify more accurately.

The time period for these estimates was the summer period between 1 July and 15 September. This 77-day period is generally the warmest time of the year in the Philadelphia region and plant and microbial processes should be at or near their maximal extent. However, in doing a literature study, not all published studies occurred during this time period. In most cases, data and rates were not corrected for temperature if within 5°C of the average temperature in this region (approximately 22°C).

2.0 Study Area

The mainstem Delaware River extends 530 km (330 mi) from the confluence of the East and West branches to the mouth of Delaware Bay at Cape Henlopen (Fig. 2-1). The drainage basin covers over 33,040 km^2 (12,757 mi^2) and includes areas in Pennsylvania, New Jersey, New York and Delaware. The water surface area of the estuarine portion is approximately 1,773 km^2, ranking among the largest on the east coast, surpassed by only the Chesapeake Bay, Long Island Sound and the Albermale/Pamlico Sounds (Sutton et al., 1996). To help locate areas on the river, the DRBC set up a river-mile system as a reference for the river-estuary. The Cape May-Henlopen transect at the bottom of Delaware Bay is designated as river-mile (RM) 0, and increases upstream to Trenton, NJ at RM 133.

The estuary can be divided into three distinct regions: 1) the tidal river, 2) transition zone, and 3) the lower region or Delaware Bay (Fig. 2-1; Sutton et al., 1996). This study is confined to the tidal freshwater portion of the river from the "falls" at Trenton (NJ) to the Delaware-Pennsylvania border, and is approximately 85 km (53 mi) long (RM 133 to RM 70). The limit of salt intrusion during high flow conditions is generally near Wilmington, DE, but can approach the Philadelphia area under low flow conditions. The tidal range varies in this reach of the river and is approximately 1.8 m to 2.5 m. This section of the river is the most heavily impacted of the three regions and the land use that borders this portion of the river is suburban, urban, and industrial. Approximately 40 industrial and municipal wastewater facilities discharge into the numerous tidal streams as well as the mainstem of the Delaware River.

Our study area included two distinct reaches in the tidal freshwater portion of the Delaware Estuary (Fig. 2-2). These regions encompass DRBC modeling sections of Zones 2, 3 and 4 (Sutton et al., 1996). The "upstream" study segment encompasses RM 133 to RM 108 and includes the estuary from the head of tide at the Trenton-Morrisville bridge at US 1 to approximately 0.9 km (1.3 mi) below Pennypack Creek in PA. This segment includes the estuary shoreline from the head of tide at the Trenton-Morrisville bridge at US 1, to 1.3 mi (0.9 km) below Pennypack Creek. This includes the tidal areas of the following creeks or rivers: Biles, Duck, Crosswicks, Blacks, Crafts, Scotts, Martins, Bustleton, Otter, Assiscunk, Neshaminy, Poquessing, Rancocas, Swede Run, Pennypack, and Pompeston. The lower "downstream" segment encompasses RM 92 to RM 82 and includes the area downstream of the Schuylkill

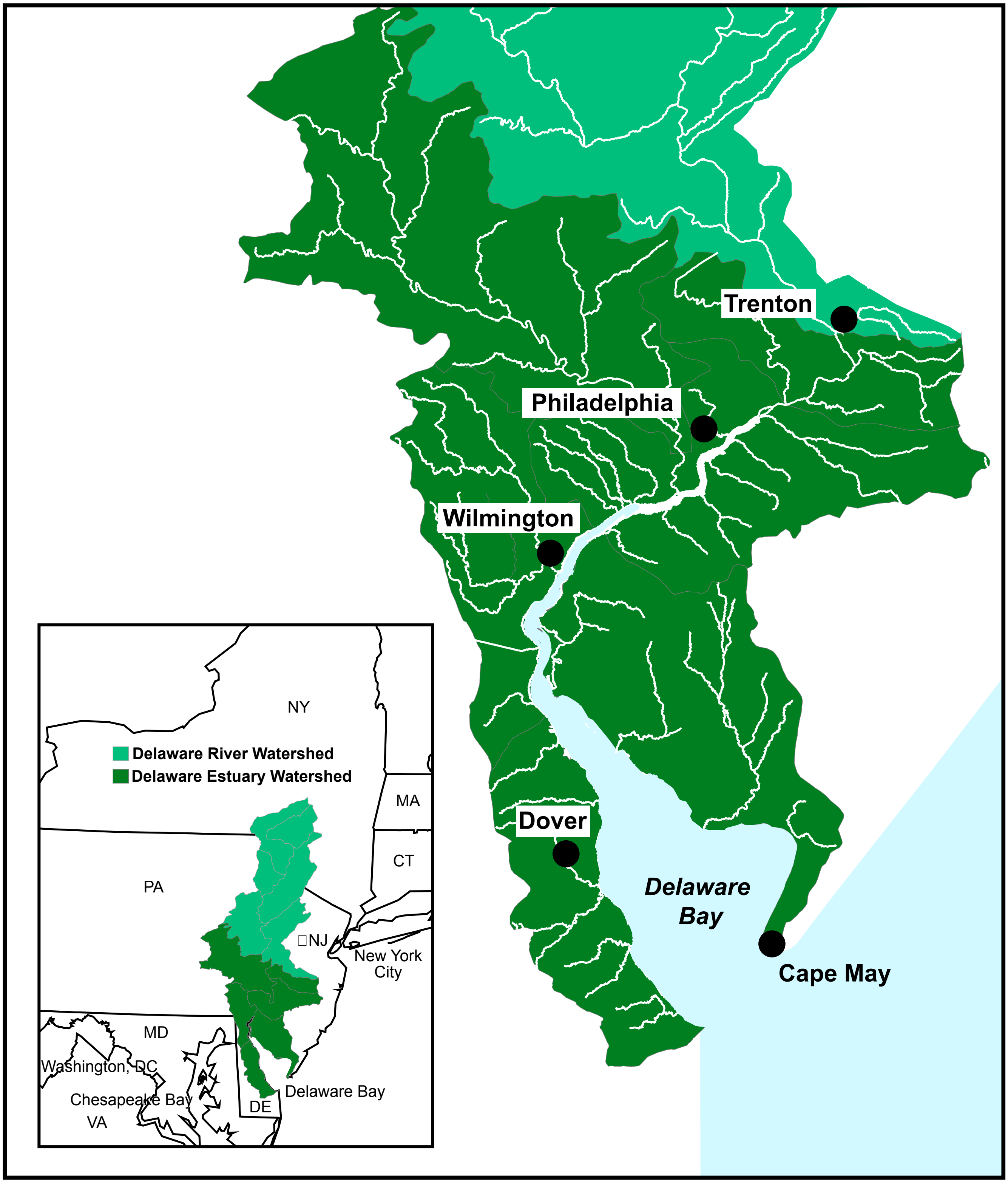

Figure 2-1. The mainstem Delaware River extends 530 km (330 mi) from the confluence of the East and West branches to the mouth of Delaware Bay at Cape Henlopen.

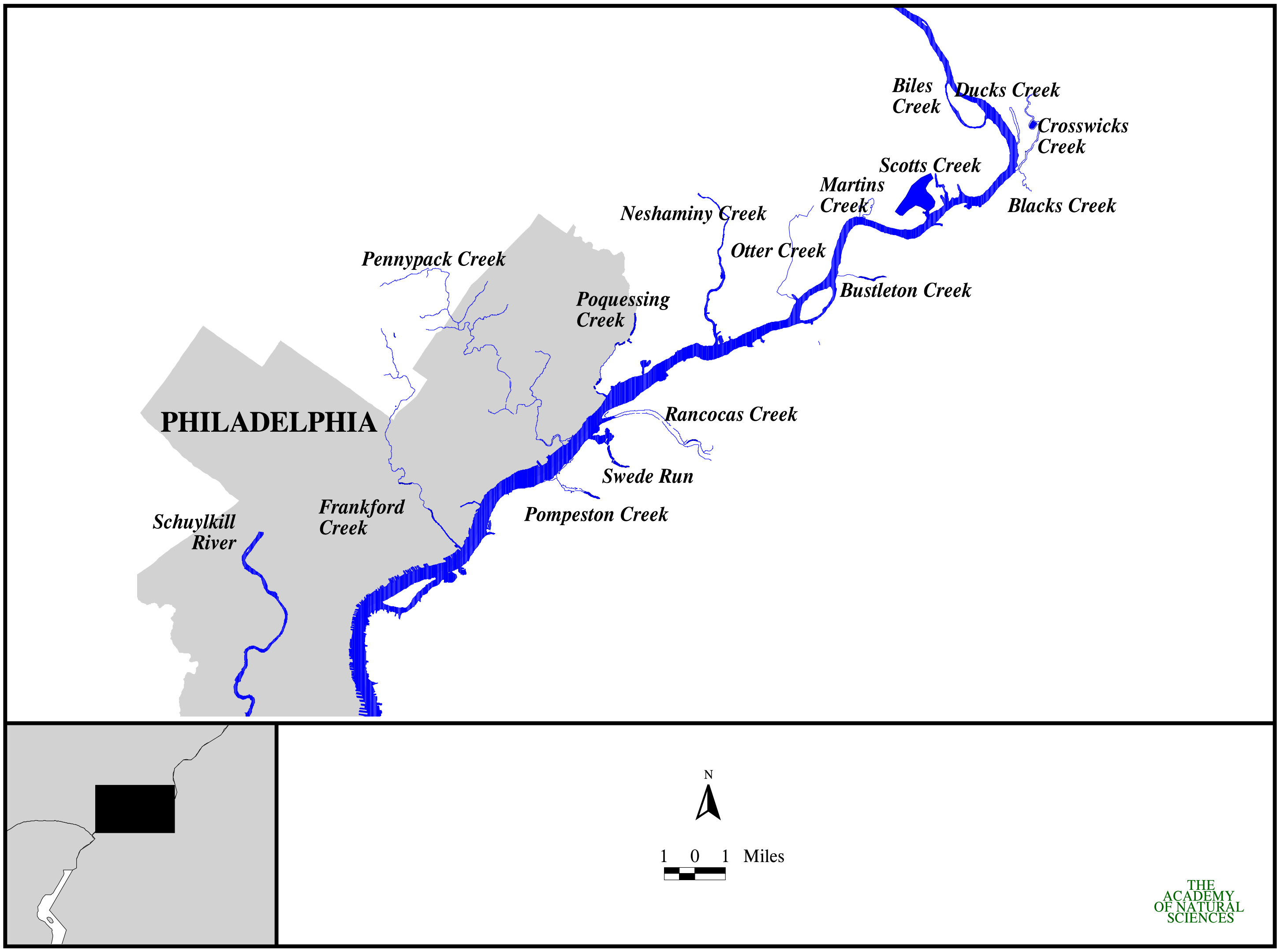

Figure 2-2a. The study area includes two distinct regions of the tidal freshwater portion of the Delaware Estuary: Upper Zone.

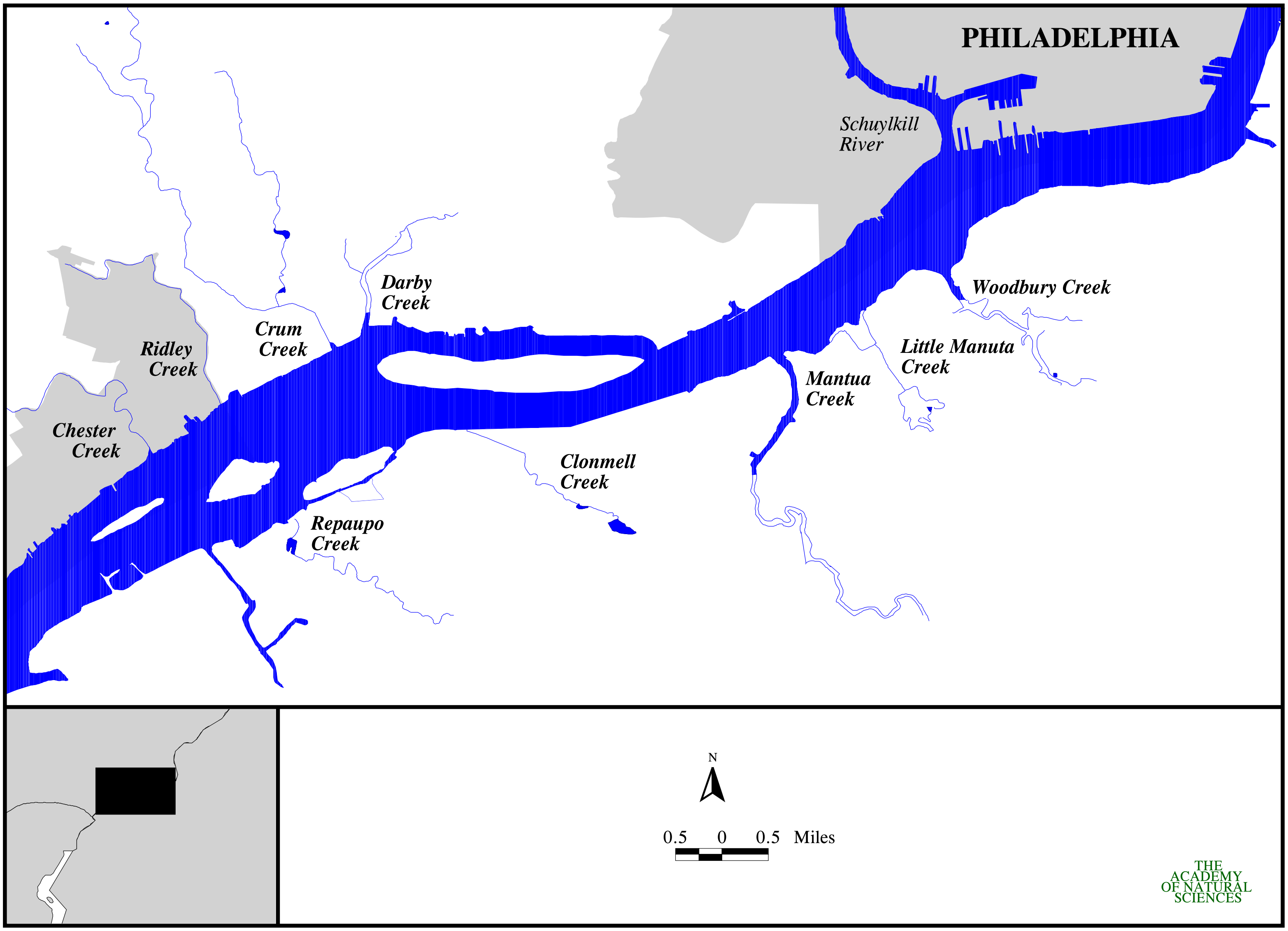

Figure 2-2b. The study area includes two distinct regions of the tidal freshwater portion of the Delaware Estuary: Lower Zone.

River to just below Chester Creek. This includes the tidal segments and shorelines of Woodbury, Main Ditch, Little Mantua, Mantua, Clonmell, Darby, Crum, Ridley, Repaupo, and Chester creeks. Our study is confined to emergent aquatic vegetation, above mean low tide, within the reaches of the two designated study areas. In addition, we sought preliminary data for submerged aquatic vegetation within the two study areas of the freshwater tidal Delaware Estuary.

3.0 Field and Laboratory Methods

3.1 Areal Extent of Emergent Aquatic Vegetation

3.1.1 Wetland Classification Methodology

Although this is not a mapping exercise, we selected a geographic information system (GIS) approach to allow for effective data capture, organization, manipulation, and presentation of the important spatial information pertaining to this investigation. A GIS also allows for efficient modification, extension, and update of the wetland delineation and classification with current aerial photography or to extend the classification to the entire upper estuary. However, it is important to keep in mind that we have not attempted to provide accurate mapping data. Rather, we have employed a generalized mapping scheme designed solely to support this investigation. The resulting digital data layers within the GIS are useful at a scale of 1:24000 or smaller.

We did not utilize the existing National Wetlands Inventory (NWI) classification since it is not consistent among the quadrangles within our study area and, therefore, is not satisfactory for making estimates of classified marsh areas. However, to provide a base map of the study area, we acquired and created a mosaic of the National Wetlands Inventory Digital Line Graph (DLG) files in a GIS base data layer for the 7.5-minute quadrangles listed in Table 3-1 (at the end of Section 3).

The areal extent of the resulting base coverage was "clipped" to exclude all but the upstream and downstream study areas of the Delaware River. We next "clipped" the coverage to include only tidal streams to the head of tide. The New Jersey head of tide demarcation was available as a digital data layer in Volumes 1 and 2 of The State of New Jersey Department of Environmental Protection GIS Resource Data set. The head of tide for Pennsylvania streams was estimated from the USGS 7.5-minute topographic maps to be at a dam when applicable, or at the first topographic contour (10 ft or 20 ft depending on the map's contour interval). All data layers were projected into the Universal Transverse Mercator (UTM) coordinate system for this study.

We searched federal, state and private sources for archival leaf-on color infrared photography of the study area. The only available leaf-on photography of the tidal zone was 1:12000-scale imagery taken during low tide in the late summers of 1977 and 1978 from the archives of the State of New Jersey Department of Environmental Protection, Division of Coastal Resources. This photography included some portions of Pennsylvania tidal wetlands along the Delaware

River, but did not include the tidal portions of the Darby Creek wetlands, the only substantial area of freshwater tidal wetland vegetation remaining in the Pennsylvania portion of our study area. A significant portion of the Darby Creek wetlands are controlled by tide gates, thereby disqualifying them from consideration in our study. However, as described below, we utilized NWI data to “fill in” the gaps for the appropriate wetland categories of Darby Creek.

The State of New Jersey has produced composite maps of the freshwater wetlands at a minimum mapping unit of 0.4 ha (1 ac) for the entire state based on March 1986 color infrared leaf-off photography. These maps were available as diazo prints at a scale of 1:12000. Unfortunately, the images used to develop these maps were taken at variable tide heights. In addition, all tidal emergent vegetation was mapped as a lumped category. We used the upper limit of wetlands, demarcated as the Upper Wetland Boundary (UWB), to bound our classification of the tidal wetland vegetation.

An estimate of the areal extent, depth and frequency of tidal flooding of vegetation can be based on a generalized mapping of the vegetation habitat schema proposed by Simpson et al. (1983a,b; Fig. 3-1). Freshwater tidal wetland vegetation does not exhibit the well-defined areal zonation that characterizes salt marsh vegetation. There is a tendency for vegetation assemblages to correlate with hydroperiod and, especially in the middle and higher tidal heights, to be of rather small areal extent. A visual signature based on color and texture of vegetation structure, when viewed in aerial photography, can distinguish the broad leaf upstanding leaves arranged in large uniform stands of low marsh from the more narrow-leafed, tall, and patchy or heterogeneous canopy structure of vegetation in the middle and high marsh.

Low marsh is characterized by monocultural stands of the broad leaved *Nuphar advena*, or on slightly higher ground, mixed stands of *Nuphar advena* with pickerelweed *(Pontederia chordata).* These show broad leaf canopies that tend to be distinct from bare mud or open water on the low side and the clumpy, patchy canopy of the heterogeneous marsh vegetation found on higher ground. Low marsh vegetation may also be found in back water areas that do not fully drain during normal low tides. On the lower reaches of a number of the tidal tributaries, low marsh vegetation occurs in prominent expanses. The spectral signature of this broad leaf canopy structure is distinctive and easily discerned in color infrared photography. On higher ground, where flooding is shallower and of shorter duration, marsh vegetation tends to be more

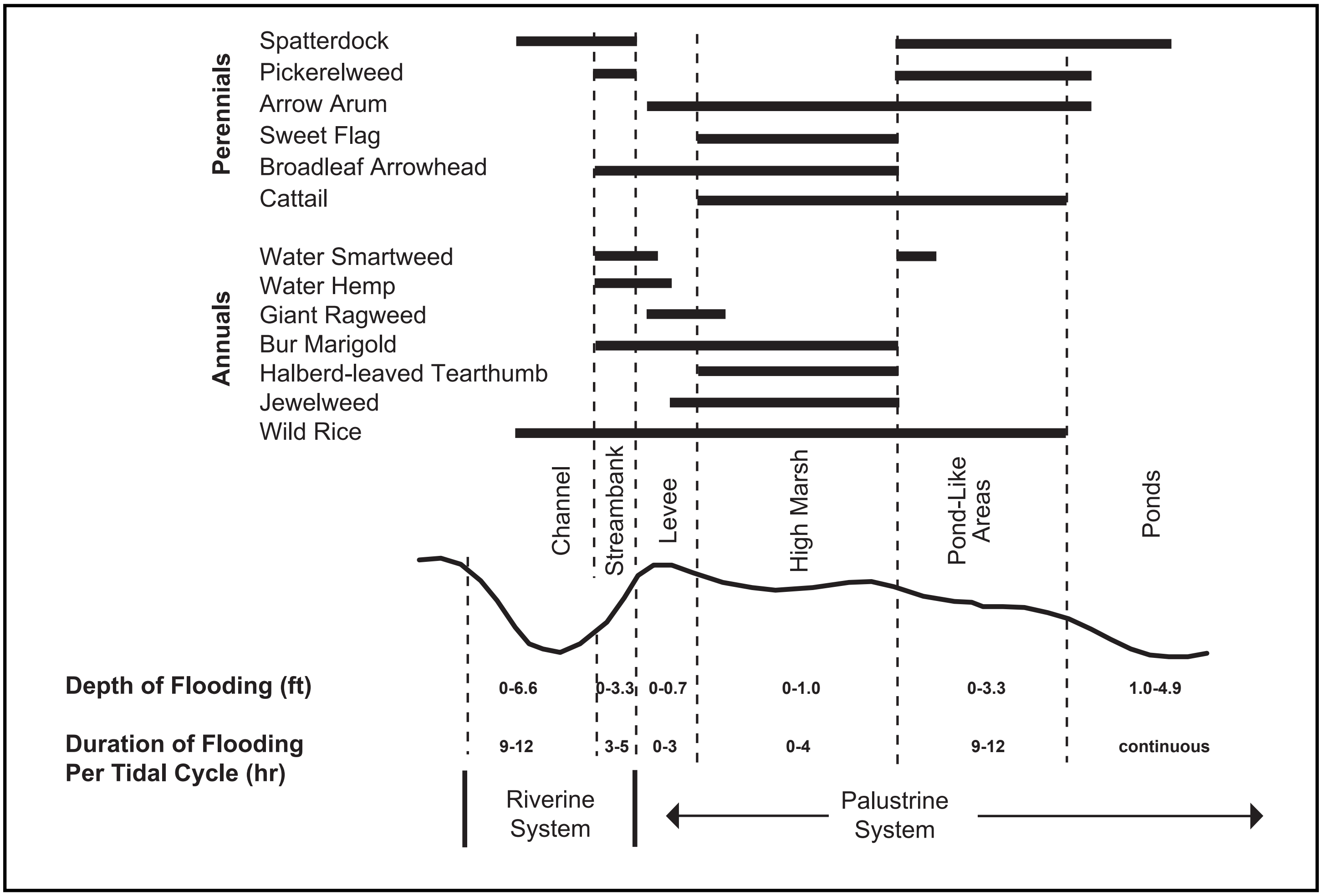

Figure 3-1. Vegetation habitat scheme modified from Simpson et al. (1983c).

heterogeneous. A number of the characteristic species are rather tall and have narrow leaves. The canopy tends to have a patchy or clumpy appearance which produces a visual texture that can be identified in color infrared photography as high marsh.

Initial examination of the 1977/78 color infrared aerial photography indicated that we could reliably identify and distinguish low and high marsh as well as bare mud at the photographic scale of 1:12000. The levee habitat (probably high marsh vegetation) and small clumps or bands of middle marsh vegetation cannot reliably be discerned at the scale of our analysis. We, therefore, classified vegetated polygons as either low marsh, high marsh or mud classes according to the dominant appearance of the canopy. A polygon was assigned to low or high marsh according to a judgement of the majority canopy signature. Thus, as an example, a vegetation polygon classified as high marsh might contain small patches of mud or low marsh vegetation.

Wild rice (Zizania aquatica) is middle marsh vegetation, but also occurs in both low and high marsh plant assemblages. Field visits to Mantua Creek revealed a tendency for wild rice under those conditions to become a monocultural dominant at an elevation of about 0.7 to 0.9 m above the mud/vegetation break. In some cases, a zone of mixed *Zizania aquatica* and other plants was found between the *Nuphar advena* and monocultural rice. In other cases the transition was directly from *Nuphar advena* to monocultural rice. We classified such a rice mix, identifiable by its spectral signature and position relative to *Nuphar advena*, as low marsh. Monocultural rice, also distinguishable by its spectral signature, was classified as high marsh. Rice also occurs in high marsh mixed with other species. Some of these other high marsh species include marigold (*Bidens laevis*), marsh smartweed (*Polygonum punctatum* and *Polygonum cannabina*), jewelweed (*Impatiens capensis*), and cattail (*Typha* sp.). We classed all such heterogeneous canopies into high marsh.

The photointerpretation was performed in the State of New Jersey Department of Environmental Protection Aerial Photo and Map Library with the assistance of Joseph R. Arsenault. The following diazo-digital Ortho photoquarter quadrangle maps of the 1986 New Jersey freshwater wetlands were purchased to draft the photo interpretation: Trenton East (southwest and southeast), Trenton West (southeast), Columbus (northwest), Bristol (northwest and northeast), Beverly (southwest and southeast), Mount Holly (northwest and northeast), Woodbury (northwest, northeast and southwest), and Bridgeport (northwest and northeast).

Spectra and texture were interpreted visually from the 1977/78 (9 in by 9 in) color infrared transparencies at the scale of 1:12000 to identify polygons of low marsh, high marsh and bare mud flats. Open water and upland polygons framing the marsh polygons were also noted. These polygons were hand drafted in pencil on diazo quarter quadrant 1986 freshwater wetland map sheets. Upland and other apparently stable features on the freshwater maps guided the drafting of the interpreted polygons. Interpreted features were traced from the 1977/78 transparencies when placement could not be interpolated from features available on the 1986 maps. Since these maps did not depict low tide conditions, high water obscured many important tidal landscape features and numerous details did require tracing from the 1977/78 photographs to the 1986 maps.

Subsequently, the polygons drafted on the diazo maps were transferred to acetate and digitized on a digitizing tablet into ArcView shape files using the UTM projection. Open water was digitized from photos, but supplemented when needed using NWI open water data layers. In addition, as mentioned previously, we utilized the NWI UWB to identify the upland boundaries.

The resulting data layers were proofed against the original pencil drafted diazo maps at a scale of 1:12000.

The resulting data layers in ArcView were: high marsh-aerial, low marsh-aerial, mud-aerial, upland-aerial, river-aerial, openwater-NWI, and uplands-NWI. The term "aerial" signifies the shape file was derived from our photo interpretation, while the other two shape files were based on NWI data. These two classes obtained from the NWI were employed to provide context and/or provide missing information. Although care has been taken in the photo interpretation and digitizing, the above procedures were intended to produce a generalized mapping suitable for estimating the areal extent of marsh surface type for the study areas. The product is not intended for use as a detailed map.

To compensate for the unavailability of aerial photography in the Darby Creek drainage we estimated area from the NWI to assign to general equivalents in our classification. Six tide-influenced NWI wetland classes are represented in the Darby Creek area. In addition, there are non-tidal wetlands in the area: diked wetland and lacustrine open water. We partitioned the NWI tidal wetland and mudflat categories into our high marsh, low marsh, and mud classification scheme as shown in Table 3-2. These equivalent categories were also used to compare the wetland areas based on our aerial photo interpretation and NWI areas later in the results section.

3.1.2 Ground- and Aerial-Truthing of Classification Results

We evaluated our classification of marsh type for current conditions since our inventory was based on historical aerial photos (1977/78). Discrepancies between classified marsh type and current conditions could be due to errors in our classification efforts or to change in marsh condition during the 20 years since the aerial photography. One approach to verification would be to view each wetland on the ground. This was not a feasible option within the limitations of this project. Furthermore, public access and/or adequate viewing vantage is not available for many wetland areas making it nearly impossible to conduct an exhaustive and definitive ground visitation verification. However, small aircraft provide a favorable vantage point for viewing and photographing the marshes and allows for viewing of wetland areas not accessible from the ground. Therefore, we used a combination of ground and aerial approaches to compare a sample of wetland polygons classified via our methodology with current conditions.

We selected a sub-sample of our classified marsh and mud polygons to compare our classification with current conditions. A simple scheme was devised to draw a random sample of the classified polygons such that the probability of selecting a polygon was proportional to its area. This was to ensure that each point of our classification space had an equal probability of being included in the sample. There were 276 polygons classified as high marsh, low marsh, and mud. The areas of all polygons in these three classes were summed to obtain a total marsh area. Next, the area of each polygon was converted to a fractional area by dividing by the total area of the 276 polygons. The polygons were then arranged in a series to form a cumulative sum of fractional areas which ranged from 0 to 1. Forty random numbers in the range 0 to 1 were then generated. Each random number was matched with the polygon whose contribution to the cumulative fractional area included that random number. Several of the 40 random numbers selected the same polygon, and there were 2 cases of 3 random numbers selecting the same polygon. By this procedure, 29 different polygons were selected for inclusion in our verification sample set. These 29 sample polygons were placed in an ArcView theme from which their class identification was withheld. A class-blind map of the sample polygons was printed. We then observed and classified each sample polygon either from the air or by ground visitation.

3.1.2.1 Aerial Reconnaissance

On 20 October 1997, atmospheric and tide conditions were suitable for our aerial photo reconnaissance. Although this was later in the season than desired due to contract delays, this was the first occasion when we were able to overfly the study area near low tide under cloud free conditions. The flight took place between 1330 and 1500 EDT. We directly overflew the Delaware River north of Philadelphia, Rancocas and Crosswicks creeks. Air traffic restrictions limited our view of the Woodbury-Mantua Creek area to an oblique view from the south and prevented any overflight of the Darby Creek area. During the flight, we obtained both hand held color video and 35-mm color transparency photographs.

The predicted low tide at Philadelphia on 20 October 1997 occurred at 1232 EDT. Predicted low tide at stations north of Philadelphia along the Delaware River and for locations upstream in the New Jersey tidal tributaries varied from 1 to 3 h after the predicted low tide at Philadelphia. Consequently, our overflight took place within 1 to 1.5 h of low tide at these wetland sites.

Visual inspection of tidal conditions in Rancocas Creek indicated that the tide was quite low. Our observations on Mantua Creek were within 2 h of low tide and appeared somewhat below the middle of the tidal range.

We were able to locate and identify 16 of the 29 sample polygons in the flight video imagery. Seven of the 29 sample polygons were photographed from the air on 35-mm film. Overall, 18 of the 29 sample polygons were found in either the video or 35-mm coverage. The 18 sample polygons were classified, without reference to our previous classification, as high marsh, low marsh, or mud through examination of the new photographs and video. The results of this classification of the sample polygons is given in Table 3-3.

Mud banks were evident in the video which were not revealed in the 1977/78 photo interpretation. This may perhaps be due to the fact that they were underwater and did not show up in the infrared photography, or perhaps they are new features. Naturally, these mud banks are not represented in the polygon sample. To estimate their current areal extent we carefully examined the video photography of the visible mud flats and low marsh along the upper reach of the Delaware River study. From this visual inspection, we drafted their approximate areal extent in a new ArcView shape file. In addition, the current photography revealed a new, created wetland on a formerly upland portion of Duck Island adjacent to interstate highway 295. This created wetland site is not considered in our aerial estimates; however, its position is estimated and sketched in an ArcView shape file.

3.1.2.2 Ground Reconnaissance

During a one day on-the-ground reconnaissance on 3 November 1997, we were able to visit, view, and videotape 15 of the 29 sample polygons. We could identify the vegetation species even though most of the high marsh vegetation had become senescent. This allowed for adequate marsh typing. However, we could not always see the entire the marsh. Road or railroad bridges provided excellent perspective on some polygons, but not all sample polygons nor all of any one polygon could be viewed from land. Some, bordered by private property, could be viewed only partially through fringing woods.

Our ground survey emphasized visiting sample polygons missed during the aerial reconnaissance. We were able to visit 12 of the 14 Woodbury-Mantua Creek area sample polygons. The remaining two proved to be inaccessible. Of the 12, the classification of 3 did not agree with our photointerpretation of the 1977/78 aerial photography. One of these was seen to be mixed high and low marsh. Two others, #43 and #44, were clearly low marsh. They appear to have changed from high to low marsh. We visited two polygons in Rancocas Creek which we were unable to locate from the air. One of them, #26, was observed to be the same class as mapped. For the other, #189, the ground observation was somewhat tentative because we could not obtain a clear view of the whole marsh. From our view, it appeared to be high marsh while we had mapped it as low marsh. We were able to visit only one of the two sample polygons we had scheduled to visit in the Crosswicks area #77. It was seen clearly to be high marsh, in agreement with our mapped classification. The results of the ground-based classification and a comparison with the 1977/78 aerial photo classification are given in Table 3-3.

3.1.3 Areal Extent of Wetlands by River-mile

Upon completion of the wetland classification, the ArcView shape files for the classification, as well as the NWI data, were transferred from the University of Delaware to the Patrick Center's Spatial Analysis/GIS Laboratory for further analysis. In order to conduct some of the more intensive spatial operations, we converted the ArcView files to ArcInfo GIS coverages on a Sun Workstation. ArcInfo GIS is a much more powerful and flexible GIS software package that allows for advanced spatial analyses not available within ArcView.

In order to determine marsh type areas by river-mile, we first created a "river-mile" coverage using existing digital data of the Delaware River and large scale maps showing river-mile locations (Tyrawski, 1979). River mile point locations were located using "heads-up" digitizing with existing digital data layers as a guide. Perpendicular lines were then drawn over the points, bisecting the river into river-mile segments. In addition, river-mile polygons were adjusted to ensure that the tributaries and their related marsh-type data were not split into separate river-mile segments. Once these bisecting lines were drawn, additional lines were added to extend the river-mile areas to the outer boundaries of the coverages. The resulting polygons were then tagged with a mile number attribute. Tributaries to the Delaware River were assigned to the river-mile containing their confluence with the mainstem river. For this study, the area of wetlands within

RM82 to RM83 is assigned to RM 82, up to RM 133 which contains wetlands between RM132 and RM133 the upper boundary.

As mentioned previously, we had to convert each of the ArcView shape files into ArcInfo data layers before we could perform various important operations (e.g., intersecting the river-mile polygons with wetland polygons). This was done using ArcInfo's *shapearc* and *regionpoly* commands. *Shapearc* converts an ArcView shape file to a coverage with region attributes and *regionpoly* converts the region attributes to polygon attributes. Although this is an indirect conversion method which results in minor area-size distortion, it is the most accurate shape file-coverage conversion method available within ArcInfo.

Once the necessary data layers were created, the classification coverages as well as the NWI coverages were intersected with the river-mile polygons. The resulting data layers contained all marsh-type attributes (area, river name), as well as the river-mile number attribute. The final step was to convert the coverage data tables into database files which were then imported into MS Access for summarization and presentation.

3.2 Collection and Analysis of Dominant Species of Emergent Aquatic Vegetation

Aboveground, living biomass of dominant species of emergent aquatic vegetation (EAV) was assessed at discrete sampling locations along tributary tidal creeks at each of the upstream and downstream river segments. Simpson et al. (1983a,b,c) described six habitat types that typically occur in these tidal marshes moving from the deepest point in the creek to the non-tidal, landward margin (see Fig. 3-1); channel, stream bank, levee, high marsh, pond-like, and pond. EAV can be found in all these habitats except the main channel; however, for the purposes of our study, EAV biomass was differentiated only as being in the low marsh (i.e., stream bank, pond-like, pond) or high marsh (i.e., levee, high marsh) because these broad, elevation-associated classifications were more readily discernable from aerial photography as compared to the detailed habitats described by Simpson et al. (1983a,b,c).

In addition to describing the six characteristic habitat types, Simpson et al. (1983a,b,c) also reported typical patterns of EAV that occur in these habitats. Generally, the low elevation creek banks and ponds are dominated by succulent species such as *Nuphar (Nuphar advena*). As elevation increases, wild rice (*Zizania aquatica*) mixes with *Nuphar*, but then rapidly becomes a uniform monoculture. The highest elevations (i.e., levees, high marsh) that still receive regular tidal inundation contain much greater diversity of EAV species, such as ragweed and marigold. However, in certain areas, monspecific patches of cattails (*Typha* sp.) and common reed (*Phragmites australis*) can replace the normal mixed community of high marsh species.

To accommodate limited resources and time, four vegetation associations (Table 3-4) were identified *a priori* based on our own reconnaissance and the report by Simpson et al. (1983a,b,c). Sampling stations were selected to ensure that each of these associations would be qualitatively and quantitatively sampled.

3.2.1 Field Sampling Locations

Tributary streams and general survey reaches within each tributary were first identified from aerial maps. These areas were selected as containing a representative mix of vegetated low and high marsh that is characteristic of the targeted river sections. The upstream tidal freshwater tributary was Rancocas Creek, and the downstream tributary was Mantua Creek. After selecting reaches having characteristic EAV assemblages from the aerial maps, the PI's performed several reconnaissance boat trips to establish specific sampling locations. During these reconnaissance trips, all four EAV Types were identified at each of the Rancocas and Mantua Creeks. We found no single location which would permit sampling all four EAV types along a single linear transect, mainly because Type 3 (*Typha* sp.) was only found infrequently in discrete patches. The following five sampling stations were established to ensure that each EAV type was sampled (Fig. 3-2**)**: Rancocas 1 (EAV Types 1, 2, 4), Rancocas 2 (EAV Type 3), Mantua 1 (EAV Type 1 and 2), Mantua 2 (EAV Type 3), Mantua 3 (EAV Type 4).

3.2.2 Field Sampling

Stations on Rancocas Creek were sampled during low tide on 18 August 1997. At Rancocas 1, EAV Types 1, 2 and 4 were sampled by orienting a linear 60-m transect perpendicular to the creek (Fig. 3-3), which had its origin at the boundary between open water and the edge of the low marsh/EAV community (i.e., at the edge of the *Nuphar* bed). The orientation and length of this transect graded first through an extensive, low marsh bed of *Nuphar* (Type 1), second through a band of *Zizania* at mid/high marsh elevations (Type 2), and ended with the high marsh,

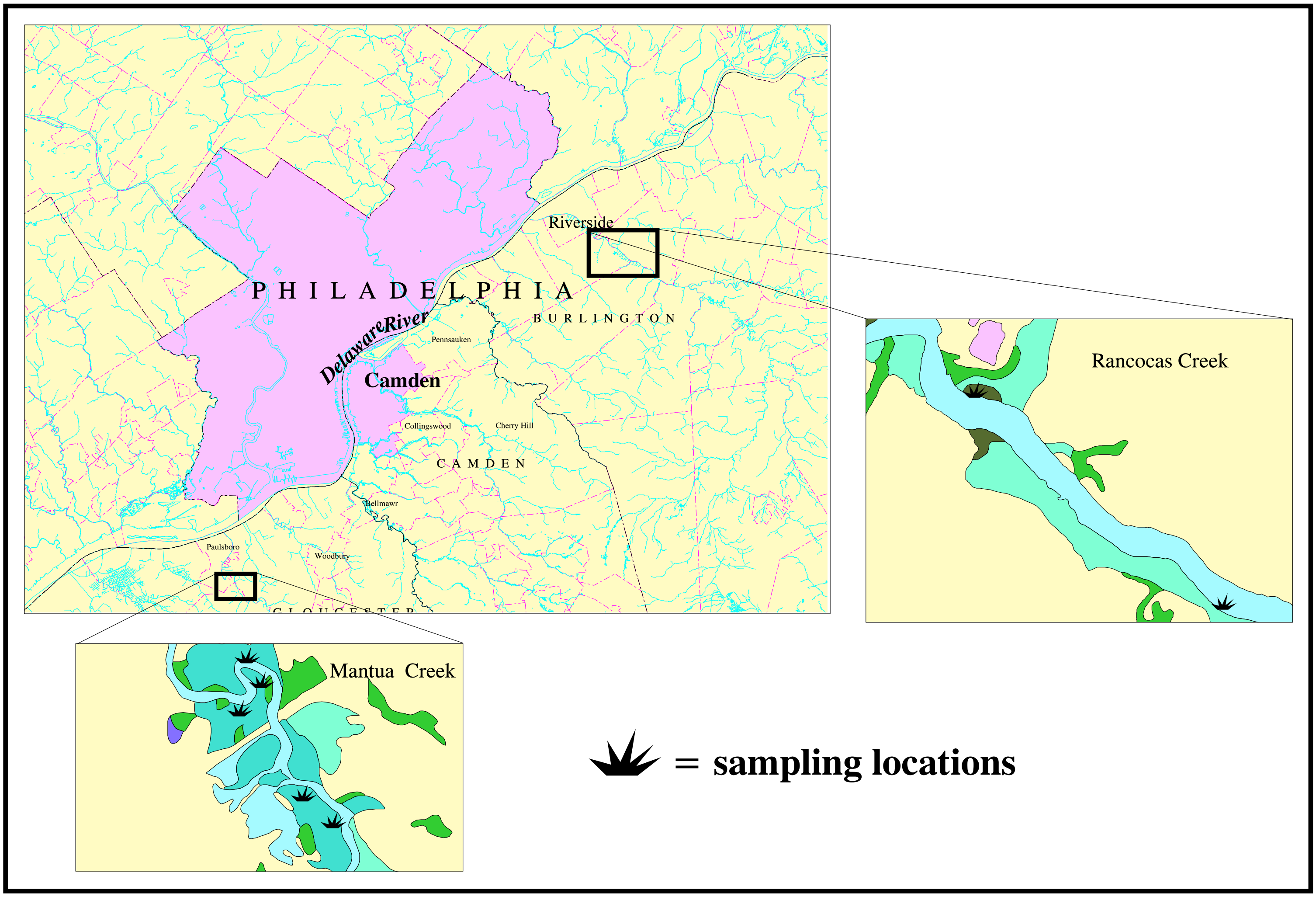

Figure 3-2. Location of sampling stations in the Rancocas and Mantua creeks, August 1997.

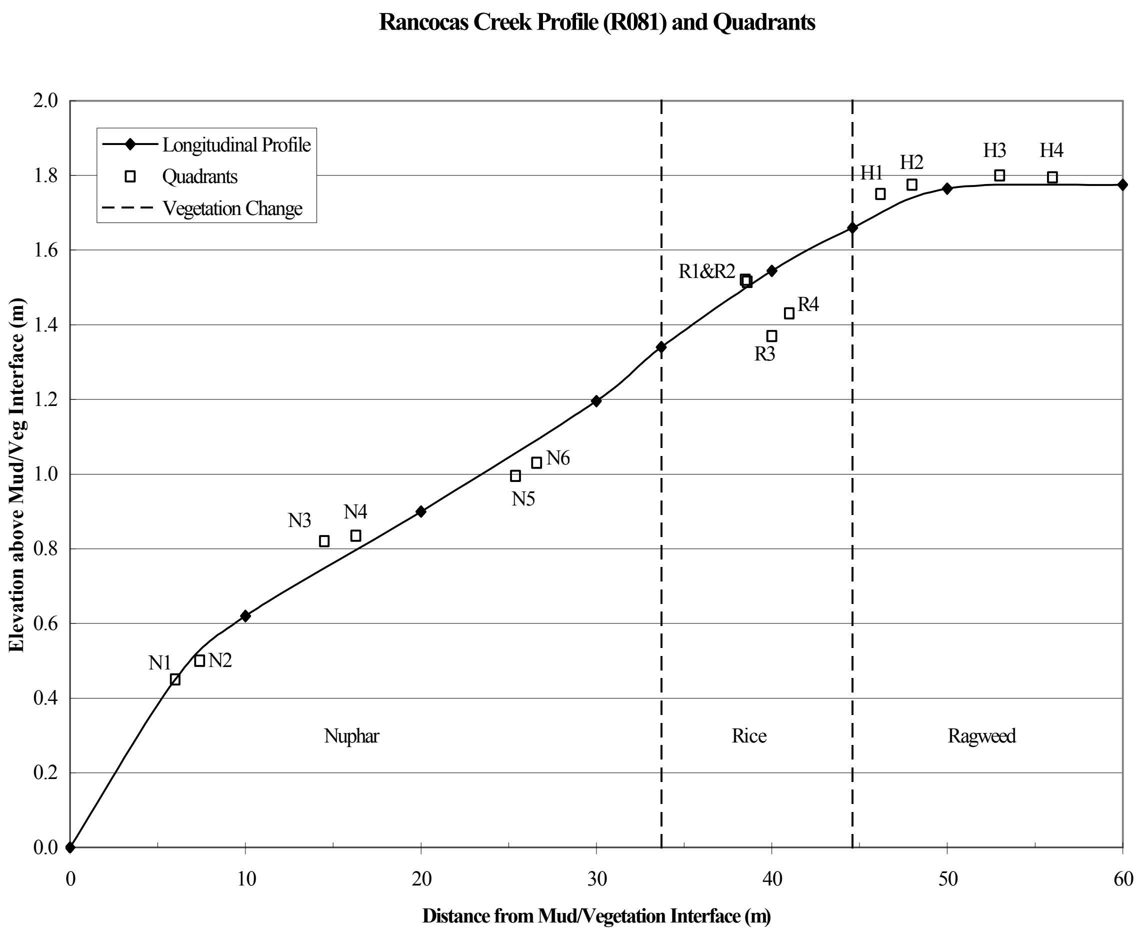

Figure 3-3. Elevation profile of sampling transect in the Rancocas Creek.

mixed community of EAV (Type 4). Fourteen 1-m^2 quadrats were situated in these different EAV communities as follows; Type 1 (n=6, denoted as N1 to N6 in Fig. 3-3), Type 2 (n=4, denoted as R1 to R4 in Fig. 3-3), Type 4 (n=4, denoted as H1 to H4 in Fig. 3-3). Although the replication per EAV type was systematic, the exact locations of the sampling quadrats within each vegetation zone were random. Sampling at Rancocas 2 was not along a linear transect; rather, four quadrats were randomly situated in a stand of *Typha* (EAV Type 3, see Figs. 3-2 and 3-3).

Mantua Creek was sampled on 20 August 1997. One 35 m linear transect was established at Mantua 1 that was oriented in the same way as that described above for Rancocas 1 (Figs. 3-2 and 3-4); however, only EAV Types 1 (n=6 quadrats) and 2 (n=4 quadrats) were represented at this site (*Nuphar* and *Zizania*). Types 3 and 4 (n=4 quadrats each) were sampled separately without a transect line (Fig. 3-4), as described above for Rancocas 2.

Quadrat sampling consisted of clipping aboveground EAV for biomass determination. A visual estimate of percent cover was used to determine which EAV species was dominant. All live (i.e., green), aboveground biomass was clipped to within 2 cm of the sediment surface within each quadrat. Plant material was placed in plastic sample bags, and transported to the laboratory at the Academy of Natural Sciences where it was stored at <5°C until it was analyzed.

Precise locations of both individual sampling quadrats as well as linear transects were recorded with a global positioning system (GPS; Trimble Geoexplorer) with differential correction. We also used surveying techniques (Laserplane Model L500 laser level and 100-m open reel fiberglass tape) to define elevation profiles along transects at Rancocas 1 and Mantua 1. Elevations were recorded every 10 m along a straight line from the mud/vegetation interface to a high point in the marsh. In addition, elevations and distances along the transect were recorded for all major changes in the vegetation community. Finally, quadrats were surveyed for their distances along the transect and their elevations. To confirm whether the elevation gradient along our sampling transects was characteristic of that usually encountered along these freshwater tributaries, the study sites were revisited in October and four additional 100-m transects were established, positioned by GPS, and the elevation gradients recorded for comparison.

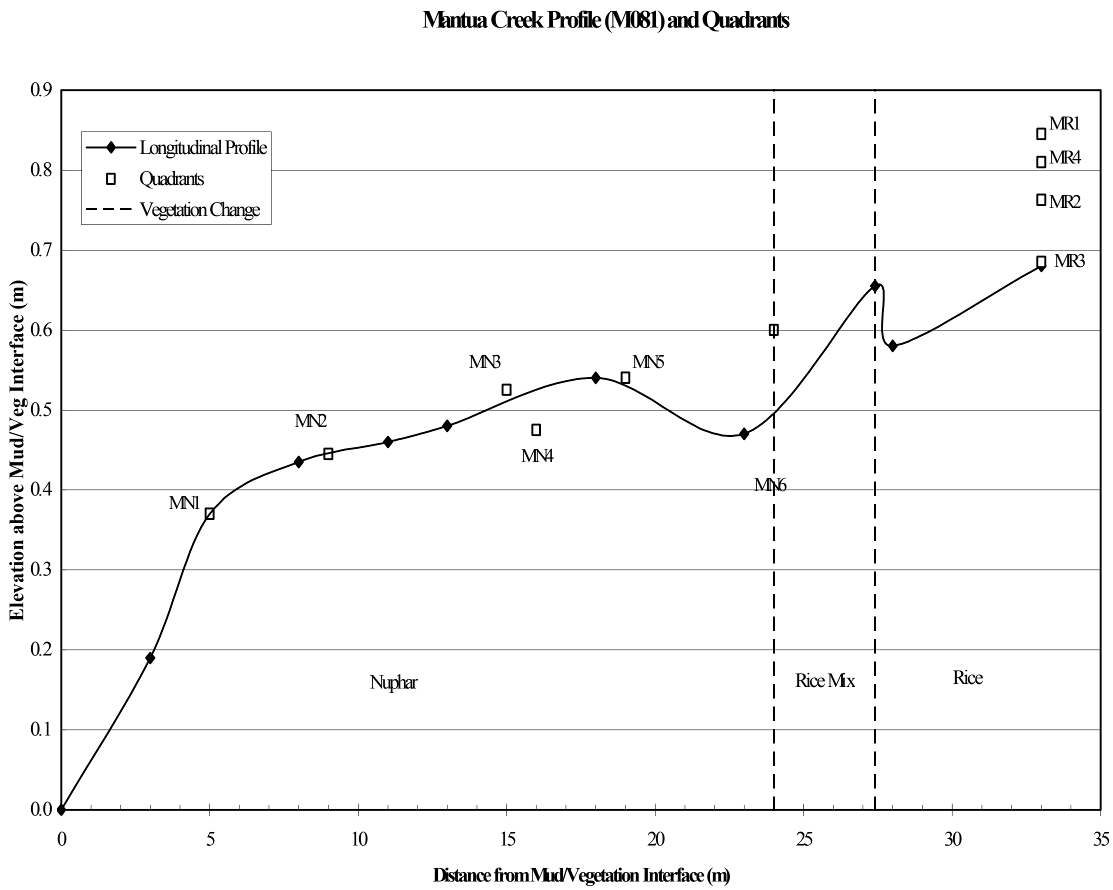

Figure 3-4. Elevation profile of sampling transect in the Mantua Creek.

3.2.3 Laboratory Preparation and Analysis

Upon return to the laboratory, blades of EAV sampled from each quadrat were sorted by species and repackaged as sub-samples that were thereafter analyzed separately. If any EAV blades appeared dead or senescent, they were removed and not counted. Each sorted group was then briefly sprayed with distilled water to remove sediment, and patted to surface dryness with absorbent toweling.

Wet weights of sub-samples were measured to the nearest mg, dried at 105°C for 24 h or until a constant weight was attained, and re-weighed to the nearest mg. Wet weights were recorded for potential use in describing the physiological condition of the EAV (e.g., dry/wet weight ratios). Dry weights were considered to be the primary unit of biomass. Dry weights for separate species clipped from the same quadrat were summed to calculate the EAV biomass. After drying and weighing, a portion of each sub-sample was ground in a mill (Wiley Model 4) and stored at room temperature for re-weighing and elemental analysis.

3.2.3.1 Elemental Analyses for Nitrogen, Carbon, and Phosphorus

Plant samples were ground and pulverized with an agate mortar and pestle and the powder was placed into pre-cleaned plastic vials. Total plant carbon and nitrogen were analyzed with a Carlo Erba 1106 Elemental analyzer. Samples, approximately 20 mg, were weighed to the nearest 0.1 µg into pre-cleaned tin boats and sealed. The samples were combusted at 1100°C under a He stream and the resultant CO_2 and N_2 were separated chromatographically and detected via a thermal conductivity detector. The signal was processed with an HP 3920 integrator and concentrations were calculated via an Excel spreadsheet. Total plant phosphorus was determined using a modified version of Aspila et al. (1976). Approximately 20 mg of dried plant material was placed into a glass tube and 0.2 ml of 50% (w/v) $MgNO_3$ (Ruttenberg, 1992) was added to wet the sample and aid the oxidation. The sample was capped with Al foil and combusted at 550°C for 90 min, then allowed to cool. At this point, 20 ml of 1N HCl was added and the sample was vigorously shaken for 2 min and allowed to stand for at least 16 h. The sample was then mixed, the remaining material allowed to settle, and the sample was analyzed with standard phosphate procedures using an Alpkem Rapid Flow Analyzer. Standard reference material (spinach leaves) and procedural blanks were analyzed periodically during this study. All concentrations are reported at mg/g on a dry weight basis.

3.3 Literature Search. Nutrient and Oxygen Dynamics of Emergent and Submergent Aquatic Vegetation in Tidal Freshwater Rivers

We undertook an extensive literature search to review relevant plant/wetland processes to estimate biogeochemical fluxes of N, P, and oxygen. We focused our search on previous studies conducted on tidal freshwater wetlands within the mid-Atlantic region so that the information would be comparable to those in the Delaware Estuary.

We completed four library searches for emergent plants and two searches for submerged aquatic plants in tidal freshwater environments. The following abstracting services were searched through DIALOG: BIOSIS, Aquatic Science Abstracts, Cambridge Science Abstracts, NTIS, Oceanic Abstracts, Enviroline, CAB Abstracts, Environmental Bibliography, Biological Abstracts, and Wilson Applied Science & Technology. During our preliminary search, using very specific keywords, we found only a limited number of publications. We then broadened our search to include any combination of the following key words: tidal freshwater wetlands. Approximately one hundred references were found, several including abstracts. From these references we were able to locate additional information, usually in the form of unpublished master's theses, dissertations and government technical reports. In addition, we searched the Aquatic and Wetland Plant Database from the University of Florida (http://aquat1.ifas.ufl.edu/database.html) for relevant citations. From these references we integrated relevant information for this study.

Table 3-1 National Wetland Inventory 7.5-minute quadrangles used as GIS base data layer.

Trenton West	Camden
Trenton East	Philadelphia
Columbus	Lansdowne
Bristol	Runnymead
Beverly	Woodbury
Frankfort	Bridgeport
Mount Holly	Marcus Hook
Moorestown	

Table 3-2 NWI tidal wetland/mud categories and equivalent categories used in this study.

NWI Symbol and NWI Name	Equivalent
P_EM_R (Palustrine emergent seasonal tidal)	high marsh
P_EM5_R (Palustrine emergent seasonal tidal)	high marsh
R1EM4/FL_N (Riverine tidal emergent organic/mudflat tidal regularly flooded)	low marsh
R1EM/AB4N (Riverine tidal emergent/aquatic)	low marsh
R1FL (Riverine tidal mudflat)	mudflat
R1FL_N (Riverine tidal mudflat regularly flooded)	mudflat

Table 3-3. Results of aerial and ground reconnaissance to verify our classification based on 1977/78 aerial photography[1].

	From Air		On Ground	Mapped	Classification Confirmed	
Polygon ID	From Video	35 mm Slide	Field Visit	Class	From Air	On Ground
Mantua-Woodbury Creek						
112				H		
110			H	H		yes
42			H	H		yes
202	L		L	L	yes	yes
207			L	L		yes
43	L	L	L	H	no	no
44	L	L	L	H	no	no
47			H	H		yes
49			H	H		yes
82			H	H		yes
93			H	H		yes
38				H		
94	L/H		H	H	yes	yes
106	L	L	Mixed	H	no	no
Rancocas Creek						
6	L/H			H	no	
147	H/L			H	yes	
1	L	L		H	no	
5	L/H			H	no	
184		L		L	yes	
185	L	L		L	yes	
12		H		H	yes	
26	L?		H	H	no	yes
189	H		H?	L	no	no
Newbold Island						
249	L			L	yes	
272	L			Mud	no	
Crosswicks Creek						
139	H/L					yes
62	L/H					no
77			H	H		yes
81				H		

[1] A blank indicates that no identification was made. H indicates high marsh; L low marsh; Mud for mud flat. H/L indicates a polygon judged to be high marsh with extensive areas of low marsh while L/H indicates low marsh with extensive areas of high marsh. Mixed indicates that high and low marsh contribute equally to the polygon and ? indicates a high level of uncertainty in the observation.

Table 3-4. Habitat and species characteristics of the four major types of emergent aquatic vegetation (EAV) associations examined in this study.

EAV Type	Habitats[1]	Relative Elevation	Dominant EAV
1	Creek banks, ponds	Low Marsh	Spatterdock (*N.avdena*)
2	High Marsh, levees	High Marsh	Wild rice (*Z. aquatica*)
3	High Marsh, levees	High Marsh	Cattail (*Typha* sp.)
4	High Marsh, levees	High Marsh	Ragweed, marigold

[1]Described in Simpson et al. (1983c).

4.0 Results and Discussion

4.1 Areal Extent of Wetlands in the Upper Delaware Estuary

Table 4-1 (at end of Section 4) presents the high marsh and low marsh areas resulting from our interpretation of aerial photos aggregated by Delaware River reach (upper and lower) and tributary creek. In all, tidal marsh areas was estimated from aerial photos for 11 tidal tributaries. The wetland areas for one tributary, Darby Creek, were estimated using NWI data since aerial photos were not available. It is important to note that we emphasized the classification and identification of high and low marsh in the photo interpretation process. For tributary creeks, we were able to delineate mud flats and, therefore, included them in the process. However, for the main stem of the Delaware River, we did not attempt to delineate mud flats. Therefore, we do not present our mud flat estimates, but utilize NWI estimates for mud flat areas later in our calculations of nutrient and oxygen fluxes.

We identified a total of 619 ha and 793 ha of marsh for the lower and upper study areas, respectively (Table 4-1). Although the upper study area has a higher total area of marsh, on a per river-mile basis the lower study area (10 river-miles) has nearly 62 ha of marsh per river-mile while the upper study area (26 river-miles) has only 30 ha of marsh per river-mile. In both the lower and upper study areas, we found much more high marsh than low marsh (65% and 75% for the lower and upper study areas, respectively).

The majority of the marsh areas were located in a small number of tributaries, with very little marsh along the mainstem of the Delaware River. In the lower study area, the largest marsh areas were found in Mantua Creek (256 ha), Darby Creek (120 ha), and Woodbury Creek (103 ha). In the upper study area, the majority of the marsh was found in Rancocas Creek (466 ha) and Crosswicks Creek (206 ha).

As described previously, we also summarized the marsh areas by Delaware River river-mile. We provide this context for the wetlands to allow for incorporation into the Delaware Estuary Model. Table 4-2 presents the high and low marsh areas as a result of our aerial photo interpretation by river-mile. In addition, we identify the tributary creeks assigned to each river-mile (only the tributaries found to have marsh are included). RM 111 had the greatest area of marsh of all the river-miles (467 ha), followed by RM 89 (256 ha) and RM 128 (223). Figures 4-

1 and 4-2 illustrate the marsh areas mapped using our aerial photo interpretation along with the river-mile polygons for the upper and lower study areas, respectively.

For comparison, we present the NWI-based estimates of high marsh, low marsh, and mud flat by river-mile in Table 4-3. We partitioned the NWI tidal wetland and mudflat categories into our high marsh, low marsh, and mud classification scheme as described earlier in Table 3-2. As with our aerial photo interpretation results, the largest areas of marsh (high plus low marsh) in the NWI are within RM 111, RM 89, and RM 128. The major difference between our marsh estimates and NWI is in the low marsh estimates. NWI has very little low marsh area within the two study areas of the Delaware River, while we identified 411 ha of low marsh in the two study areas combined. However, while there are large differences between our low marsh estimates and those of NWI (Table 4-4), the total marsh areas compare favorably with only a -20% and -1% difference for the lower and upper study areas, respectively. It is important to remember that the NWI areas for Darby Creek are included in both the aerial and NWI estimates since we were unable to locate aerial photo of these wetlands.

The mud flats from NWI are also provided in Table 4-3. We will use these areas along with our high and low marsh estimates by river-mile in Table 4-2 for estimating the impact of aquatic vegetation and mud flats on water quality of the Delaware River.

4.2 Emergent Aquatic Vegetation of the Freshwater Tidal Delaware Estuary

4.2.1 Species and Habitat

Elevation transects at both Rancocas and Mantua Creeks indicate that considerable variability exists along the shores of these tributary creeks in the elevation profile of the creek banks. The tidal range at Rancocas Creek is greater than that at Mantua Creek, which was evident by our measurements of the vertical height of the EAV band. The height of the EAV zone is only used as an indication of the relative differences between upstream and downstream tidal ranges, not absolute tidal heights. At Rancocas Creek, EAV extended 1.8 m above the mud/vegetate interface; whereas, at Mantua Creek the height of the EAV zone was less than 0.8 m. Furthermore, there often was not a gradual increase in shore height with distance away

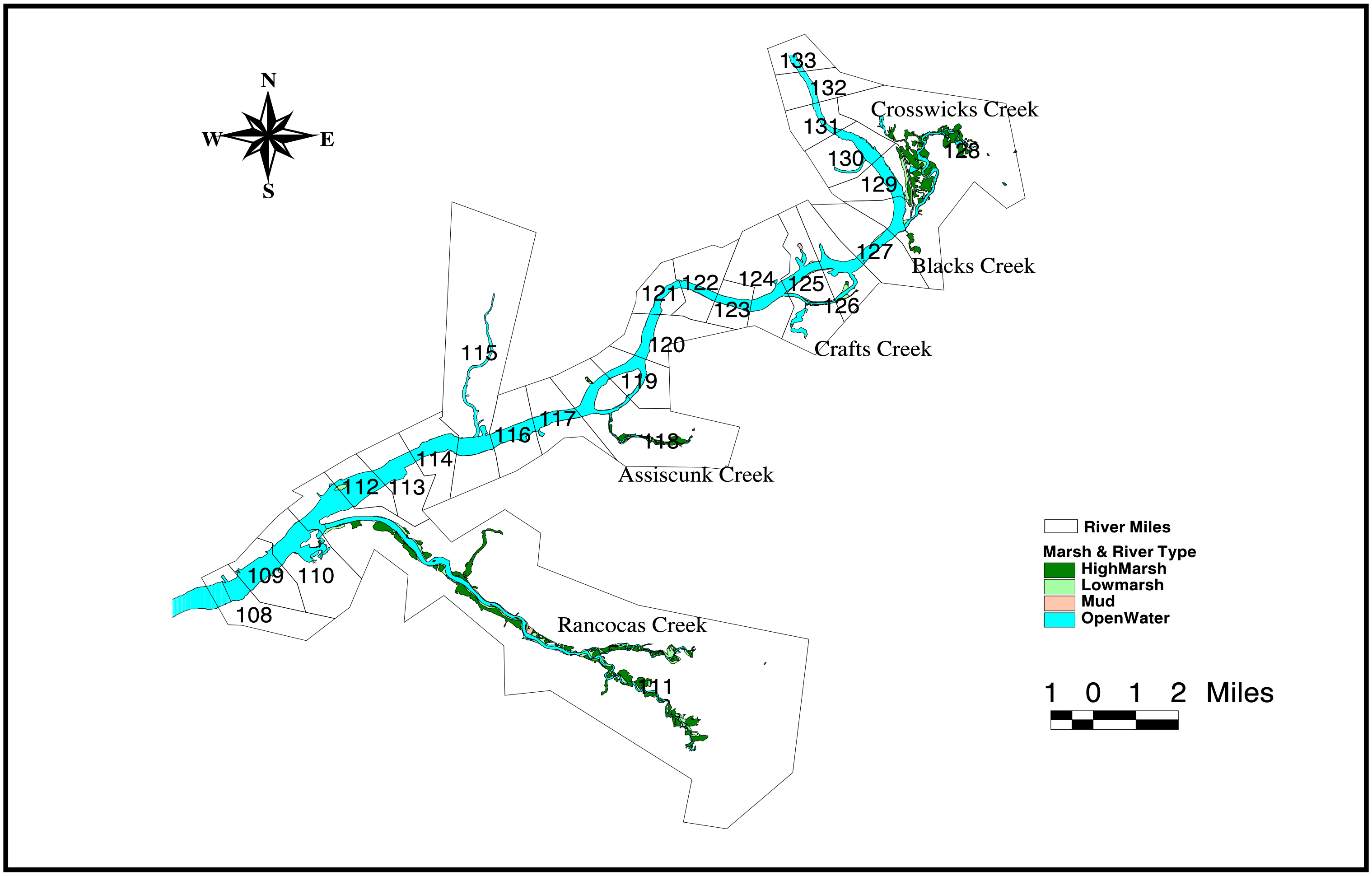

Figure 4-1. Low and high marsh identified through aerial photo interpretation for the upper Delaware River study area.

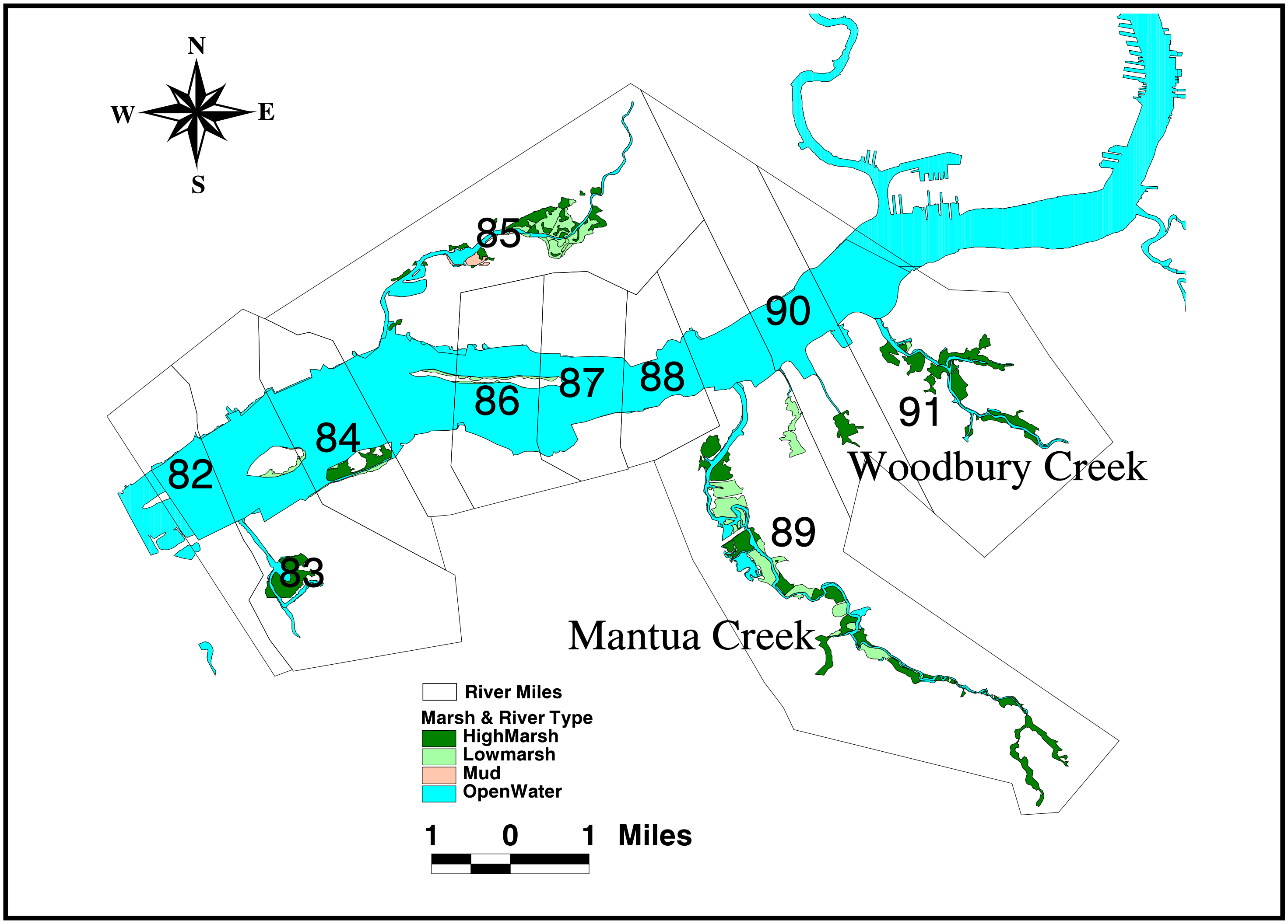

Figure 4-2. Low and high marsh identified through aerial photo interpretation for the lower Delaware River study area.

from mean low water; rather, side channels, troughs and small creeks are common on the landward side of creek bank levees. Nonetheless, in most areas, a gradation of EAV species occurred with distance from mean low water, and our repeated transect comparisons, although quite variable, indicate that our EAV sampling transects were not uncharacteristic (Fig. 4-3).

Characteristically, the broad-leaved *Nuphar advena* (=*luteum*, =*variegatum*) occurred in uniform stands along the edge of the channel (foreground in Fig. 4-4) and in ponded and pond-like areas (i.e., low marsh, Type 1 habitat). These areas typically contained at least 95% of the biomass and percent cover in the form of *N. advena*, although both arrow arum (*Peltandra virginica*) and pickerelweed (*Pontederia cordata*) also contributed to the biomass at somewhat less depth of inundation. From low to high marsh (i.e., Type 2), the second prominent EAV type was wild rice (*Zizania aquatica*), which typically formed a narrow to wide band of tall grass just landward from the Type 1 low marsh (background in Fig. 4-4). Wild rice was widespread at both upstream and downstream locations. Near the low marsh/high marsh boundary, stands of wild rice often intermixed with the succulent, low marsh species. Nonetheless, within the Type 2 area, wild rice composed 66-100% of the total dry EAV biomass, and became nearly monospecific in the bulk of its range.

High marsh habitats contained much greater EAV species diversity, but certain species formed patches where they dominated EAV biomass. Cattail (*Typha angustifolia*) and common reed (*Phragmites australis*) also formed nearly monospecific stands having tremendous biomass. These tall plants were very patchy in occurrence, however, often being found only on high marsh levees. We measured the actual EAV biomass within a cattail patch (Type 3) at both the upstream and downstream sites. No measurements of *P. australis* biomass were made.

Greatest EAV diversity was found in the most landward high marsh (Type 4). Common species in this zone were giant ragweed (*Ambrosia trifida*), bur marigold (*Bidens laevis*), and jewelweed (*Impatiens capensis*). Although these species composed the bulk of the EAV biomass in Type 4 habitat, numerous other species were also present; however, we did not systematically identify species since the focus of this investigation was on EAV biomass and not biodiversity.

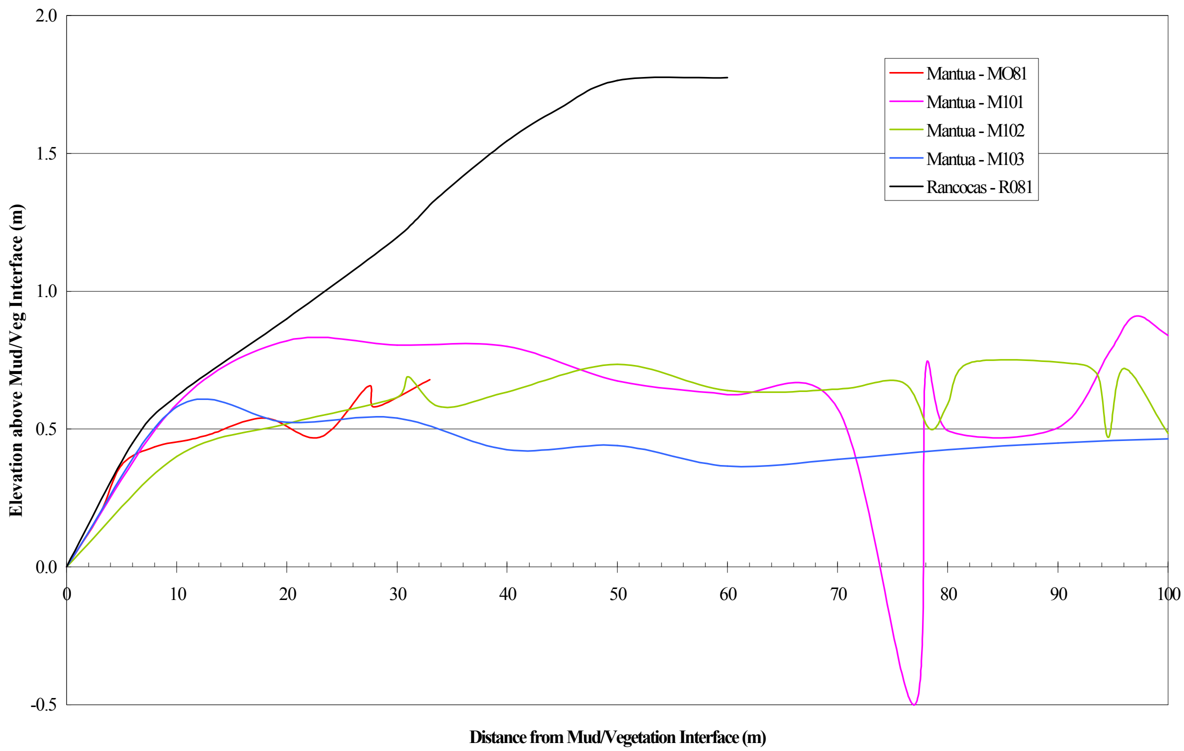

Figure 4-3. Comparison of elevation profiles of 5 linear transects from the Rancocas and Mantua creeks.

Figure 4-4. Photograph of Academy staff members surveying the elevation of the marsh at the boundary between the low marsh (foreground, dominated by broad-leaved, *Nuphar*) and high marsh (background, dominated by wild rice, *Zizania*).

Where the elevation gradient was particularly gradual, high and low marsh EAV often intermixed over wide areas. Similarly, there were rarely discrete boundaries between wild rice and ragweed/marigold high marsh areas (Types 2 and 4). In instances where EAV sampling quadrats contained a mixture of EAV types, the percent cover estimate for each dominant species was used to assign the collected sample to one of the four EAV types.

Numerous other species were found during this study, and their spatial distribution in the tidal marsh was nearly always associated with specific flooding conditions that they are specially adapted for growth. Thunhorst (1993) summarized typical EAV heights and species-specific tolerances to tidal inundation for the following EAV observed in our study:

- *Nuphar luteum* and *N. advena* (spatterdock, yellow water lily). Height to 0.4 m; tidal zone, below mean low water where average depth is 0.3 to 0.9 m.
- *Peltandra virginica* (arrow arum). Height to 0.6 m tall; freshwater up to approx 1-2 ppt salinity; tidal zone, mid-tide to spring tide elevation.
- *Polygonum hydropiperoides* (mild water pepper). Height to 1.5 m; tidal zone, upper 50% of intertidal zone.
- *Polygonum punctatum* (marsh smartweed). Height to 1.1 m; tidal zone, upper 50% of intertidal zone.
- *Pontederia cordata* (pickerelweed). Height to 1.1 m; tidal zone, upper 50% of intertidal zone.
- *Sagittaria latifolia* (buck potato or big-leaved arrowhead). Height to 1.2 m; tidal zone, upper 50% of intertidal zone; regularly to permanently inundated up to 0.6 m.
- *Sium suave* (water parsnip). Height to 2.1 m; regularly to permanently inundated up to 0.2 m.
- *Typha* sp. (cattail). Height to 3 m; tidal zone, upper 20% of intertidal zone to spring tide elevation.
- *Zizania aquatica* (wild rice). Height to 3 m; tolerates infrequent flooding by water containing some salt; tidal zone, middle 50% of tidal zone.

Based on these flooding characteristics, high marsh vegetation will typically be underwater for up to 4 h per 12-h tidal cycle. The depth of flooding during this time may typically reach 0.3 m. Conversely, the vegetation typically will be out of the water for 2/3 of that tidal cycle. The various low marsh habitat types will be underwater from 3/4 to all of a 12-h tidal cycle up to a maximum depth of 1 or even 2 m. Low marsh vegetated with *N. advena* and *P. virginica* is predominantly within the stream bank to pond-like hydrologic regime, and is drained at normal low tide (Fig. 3-1).

4.2.2 Biomass of Emergent Aquatic Vegetation in the Delaware River

EAV biomass was high in all zones, as might be expected during August at the late peak of the growing season. The low marsh areas dominated by spatterdock *(Nuphar advena)* (Type 1) contained 241 g dry weight m^{-2} and 811 g dry weight m^{-2} at Rancocas and Mantua Creeks, respectively (Table 4-5). Although Type 1 EAV biomass at Mantua Creek appeared higher than at Rancocas Creek, these means were not significantly (t-test, $p>0.05$) different from each other because of high variability among replicates at Mantua.

The zone of EAV dominated by wild rice (*Zizania aquatica*) contained remarkably consistent biomass at both sites. At Rancocas Creek, the mean biomass in the wild rice zone was 323 g dry weight m^{-2}, and 382 g dry weight m^{-2} was measured at Mantua Creek. These values were not significantly (t-test, $p>0.05$) different between sites.

EAV biomass was found to be greatest in high marsh patches of cattails (*Typha angustifolia*) (Type 3). Cattails contained an average of 1305 and 1191 g dry weight m^{-2} at Rancocas and Mantua Creeks, respectively. The high marsh community of mixed EAV (Type 4) contained similar biomass as that seen in Types 1 and 2; 524 and 282 g dry weight m^{-2} EAV at Rancocas and Mantua Creeks, respectively.

For the purposes of this study, we sought to establish estimates of EAV biomass in either the low (Type 1) or high marsh (Types 2, 3, 4) because these two zones were most easily identified and quantified by remote sensing (see above). Statistical procedures were therefore needed to determine whether EAV biomass was similar among Types 2, 3 and 4 to permit derivation of a single, characteristic value for high marsh EAV biomass (see below). To test whether these biomass values were significantly different than those for the other EAV communities, a one-way ANOVA was performed for Rancocas and Mantua Creeks separately. In both cases, the statistic was highly significant ($p<0.01$), and a Tukey's multiple range analysis indicated that for both sites, the cattail community (Type 3) was significantly higher in biomass compared to the other 3 EAV zones (Types 1, 2, 4).

In order to estimate the geographical extent of EAV biomass within the designated upstream and downstream river segments of the Delaware Estuary, we sought to calculate a universally applicable average or median EAV biomass for either the low marsh or high marsh, since the remote sensing data were limited to that level of sensitivity. Since only one vegetation type dominated the low marsh zone at Rancocas and Mantua Creeks, we will simply use the median values measured for the Type 1 habitat in either Rancocas (i.e., represents upstream low marsh EAV biomass) or Mantua (i.e., represents downstream low marsh EAV biomass) Creek. Hence, the median low marsh EAV biomass at the upstream segment (e.g., Rancocas Creek) and downstream segment (e.g., Mantua Creek) was calculated to be 232 and 731 g dry weight m^{-2}, respectively (Table 4-5).

To calculate a similar figure for the high marsh, we pooled replicate data for EAV Types 2 and 4, but disregarded data for EAV Type 3 for the following reasons. The Type 3 high marsh dominated by cattails (*Typha*) was very patchy and limited in spatial coverage at both Rancocas and Mantua Creeks; whereas, the Type 2 and 4 habitats were common all along each tributary and they were deemed more widespread and characteristic. Also, Type 3 high marsh contained approximately twice the EAV biomass typically found in the other high marsh zones. Since the biomass measured in high marsh Types 2 and 4 did not differ significantly ($p>0.05$) at either site, we were justified in lumping those data; whereas, as noted above, Type 3 data were significantly different and would not be able to be considered as part of the same normal distribution. Hence, our estimates for the median EAV biomass in the high marsh of the upstream (e.g., Rancocas Creek) and downstream (e.g., Mantua Creek) reaches were 402 and 327 g dry weight m^{-2}, respectively (Table 4-6). But it is important to recognize that these figures are conservative estimates since high biomass patches of *Typha* (and *Phragmites*) were not considered.

4.2.3 Annual Primary Production by EAV

Typically, to estimate rates of primary production, biomass (i.e., standing stock) measurements are made at both the beginning of the growing season (e.g., early May in the mid-Atlantic region) and soon after the peak of the growing season (e.g., late August/early September). By difference, live, aboveground biomass can be used to calculate production using the method of Smalley (1958 as cited in Roman and Daiber, 1984; Good et al., 1982), whereby the biomass difference between the end and start of the growing season is divided by the time of growth to yield rates of primary production by EAV.

Due to limited resources and a late start for this project, only one determination of EAV biomass was feasible. We timed our biomass collection for the late peak growing season (late August) when biomass was maximal, and only biomass from the current growing season (i.e., assumed to be from 1 March to 18 August; 170 days) was included in the collection. By assuming that little EAV biomass had already senesced, little leaf turnover (which was evident), and that no further production would occur, we obtained a simple estimate of annual EAV production using the Smalley Method by simply assuming the EAV biomass at the start of the growing season was zero. Hence, this approach yielded productivity (g m^{-2} yr^{-1}) values that were identical to our measures of peak end-of-season biomass (g m^{-2}), which were reported in Tables 4-5 and 4-6. These assumptions result in a conservative production estimate (i.e., production could be greater, but is not likely to be smaller).

Our estimates of annual EAV production are generally similar to earlier findings in the literature. In their studies of tidal marshes on the New Jersey side of the Delaware River, McCormick (1977) reported rates of production of 863 g m^{-2} yr^{-1} and Whigham and Simpson (1975) reported 780 g m^{-2} yr^{-1} for low marsh areas dominated by *Nuphar*. This is in close agreement with our median of 731 g m^{-2} yr^{-1} for Mantua Creek, but is greater than our low marsh estimate for Rancocas Creek (232 g m^{-2} yr^{-1}). For the high marsh, our production estimates for Rancocas and Mantua Creeks (402 and 327 g m^{-2} yr^{-1}, respectively) are markedly lower than literature reports for the same EAV and systems. For example, previous estimates of wild rice (*Zizania aquatica*) production values for New Jersey tidal wetlands range from 940 g m^{-2} yr^{-1} (Whigham and Simpson 1976) to 1390 g m^{-2} yr^{-1} (McCormick and Ashbaugh, 1972) to 1520 g m^{-2} yr^{-1} (McCormick, 1977) to 1589 g m^{-2} yr^{-1} (Whigham and Simpson, 1975) to 1600 g m^{-2} yr^{-1} (Good and Good, 1975). High values have also been reported for other high marsh EAV such as giant ragweed (*Ambrosia trifida*, 1160 g m^{-2} yr^{-1}; Whigham and Simpson, 1975).

In summary, our ground surveys of EAV biomass and productivity for each study site produced values that were either consistent with or lower than comparable literature reports. Due to the consistency of the values reported in earlier studies, particularly for the high marsh, we must consider our productivity values to be conservative (i.e., low, underestimated by up to 2- to

3-fold). This could be explained by either or both of the aforementioned sampling problems: 1) we had to assume that no annual EAV production occurred beyond what was measurable as standing biomass in late August and 2) species of EAV known to have much higher biomass and productivity (e.g., cattail and common reed) were not considered herein due to their patchy nature (produces an underestimate). Suggestions for improving the accuracy of these measures of EAV production rates are provided in Section 5.

4.3 Nitrogen, Carbon, Phosphorus and Oxygen within Emergent Vegetation

Plant nutrients, such as nitrogen and phosphorus, are introduced into the Delaware Estuary from many sources. Sources to the tidal river include point sources (e.g., municipal and industrial wastewater), non-point inputs (e.g., urban and agricultural runoff, atmospheric deposition), and upstream runoff. Upstream sources would also include many of these same sources and can be the dominant input in many cases. Once these nutrients are in the tidal freshwater portion of the estuary they can removed from the water column by processes such as plant uptake of nitrogen (N) and phosphorus (P) (e.g., emergent plants and others), sediment-water exchange of nutrients, denitrification, transport into the more saline portion of the lower estuary, and sedimentation. As a consequence of photosynthesis, aquatic emergent plants not only take up nutrients (via photosynthesis), but also produce dissolved oxygen. The oxygen that is produced from plants can possibly leave the plant via advection/diffusion into the sediments and possibly the water column. In the following sections, we will estimate the amount of nitrogen, carbon, and phosphorus that is bound into living aboveground biomass of emergent tidal freshwater plants in the Delaware Estuary. This estimate will then be used to discuss the production and fate of dissolved oxygen in these systems. In addition, estimates of the average flux for nitrogen, phosphorus, and dissolved oxygen between the water column and sediments will be made.

4.3.1 Emergent Plant Incorporation of Nitrogen, Carbon and Phosphorus

As part of the field sampling effort, plant samples from low and high marsh areas were obtained and analyzed for the amount nitrogen, carbon and phosphorus (NCP) (see Section 3). Carbon was also included due its use in determining oxygen fluxes. Plants were sorted to provide samples of distinct species in most cases. Concentrations of nitrogen, carbon and phosphorus showed substantial variations among species (Tables 4-7 and 4-8). For example, *Nuphar advena* (spatterdock) and *Peltandra virginica* (arrow arum), typical low and mid marsh species, contained greater amounts nitrogen and phosphorus than high marsh plant species in both the Rancocas and Mantua Creek systems. A one-way analysis of variance (ANOVA), indicated that both spatterdock and arrow arum were significantly different from all other species for both N and P ($p < 0.0001$). Average N concentrations in both marshes were 19.0 ± 3.5 mg N/g dry weight(dw) (n=27) for spatterdock and 23 ± 9.8 mg N/g dw (n=6) for arrow arum (Tables 4-7 and 4-8). Total plant phosphorus concentrations from both marshes averaged 3.1 ± 0.5 mg P/g dw for *Nuphar* and averaged 3.1 ± 0.4 mg N/g dw for arrow arum. Plants of the high marsh (e.g., *Typha latifolia* [cattail], *Zizania aquatica* [wild rice], and *Ambrosai trifida* [giant ragweed]) contained much lower amounts of N and P (Tables 4-7 and 4-8). Concentrations of total carbon in the various plants were variable and generally lowest in spatterdock and arrow arum (average range: 385 to 391 mg C/g dw) and highest in cattail (433 to 462 mg C/g dw) for both marsh systems. These concentrations are in general agreement to those measured in other tidal freshwater marshes of the Delaware Estuary by Whigham and Simpson (1976) and Simpson et al. (1983a,b,c), and other wetlands (Boyd, 1978; Kadlec and Knight, 1996).

Using the NCP contents of the various plants, molar elemental ratios were calculated (i.e., C:N, C:P, and N:P; Tables 4-7 and 4-8). These ratios reveal interesting aspects of the plant community of the tidal freshwater marshes in general and within the Delaware Estuary. For example, both spatterdock and arrow arum have lower C:N ratios (e.g., 22 to 26) compared to other species. Ragweed and cattail have higher C:N of between 59 to 64, while wild rice ranges from 35 to 53. This is most likely due to the lower amounts of structural carbon compounds within the tissue of low marsh plants. High marsh species, such as wild rice and cattail, require more supporting tissue (e.g., C-rich cellulose), compared to either smaller plants or plants that are inundated during a typical tidal cycle and contain more protoplasmic material. In this regard, Polisini and Boyd (1972) showed a negative correlation between N content and percentage of cell wall material.

Limitation by micro-nutrients such as N or P can potentially alter the species composition of marsh vegetation (Valielia and Teal, 1974; Vermeer, 1986; Vitousek and Walker, 1987), and nutrient limitation by either N or P may be assessed by following the distribution of N to P in the wetland plants. In this regard, some species (e.g., *Phragmites* sp.) may be at an ecological advantage under high phosphorus conditions. Koerselman and Meuleman (1996) showed that nutrient limitation can be established from the N:P ratio in plant tissue. Their study suggests that N:P ratios > 35 indicate P limitation on a community level, while ratios < 30 indicate N limitation. Figure 4-5 shows the distribution of N and P (molar basis) for all plants at the Rancocas and Mantua Creek marsh sites. In the Rancocas wetlands, the average N to P ratio ranged from 11.9 ± 0.5 for wild rice to 19.5 ± 3.7 for ragweed, with an overall average of 15.8 ± 4.0 (%relative standard deviation [%RSD] = 25). There was no clear difference among species, but on average *Nuphar* had the lowest N:P value (Fig. 4-5). In Mantua Creek, the N:P ratio ranged from 9.7 ± 1.9 for ragweed to 17.7 ± 3.9 for cattail with an overall average of 13.2 ±4.0 (%RSD = 30). In both rivers, the highest plant sample had an N:P ratio of 28. Hence, in both Rancocas and Mantua Creeks, our N:P ratios suggest that emergent plant production is limited by available N and not P. It is unclear as to the cause of this limitation given the high nutrient levels in the tidal freshwater zone of the Delaware Estuary. It is possible that the retention of P in the freshwater zone, as proposed in the Chesapeake Bay, may provide the relative surplus of available P compared to N or the removal of N via denitrification. In either case, further work is needed to determine whether tidal marsh production is indeed N limited and how the resource availability can be a driving force for plant production and species composition.

From the available data, estimates were made as to the total amount of NCP contained in the two tidal freshwater wetland areas of the river. For the upper section of the river (RM 108 to 133), the data from Rancocas wetlands are used for these calculations along with the areal coverage, while the Mantua wetlands are used for the lower study area (RM 82 to 90). For these calculations, median or average values were used throughout. However, in some cases there were substantial variations in the data. As described above, Table 4-6 presents the median biomass for the high and low marshes of both creeks. Using the median areal biomass (g/m^2) and the wetland areas for high and low wetlands, the total aboveground emergent plant biomass in each river-mile is obtained. With these data and the average concentration of NCP in emergent plants (Table 4-9), the total amount of NCP was derived (Table 4-10). By dividing the total

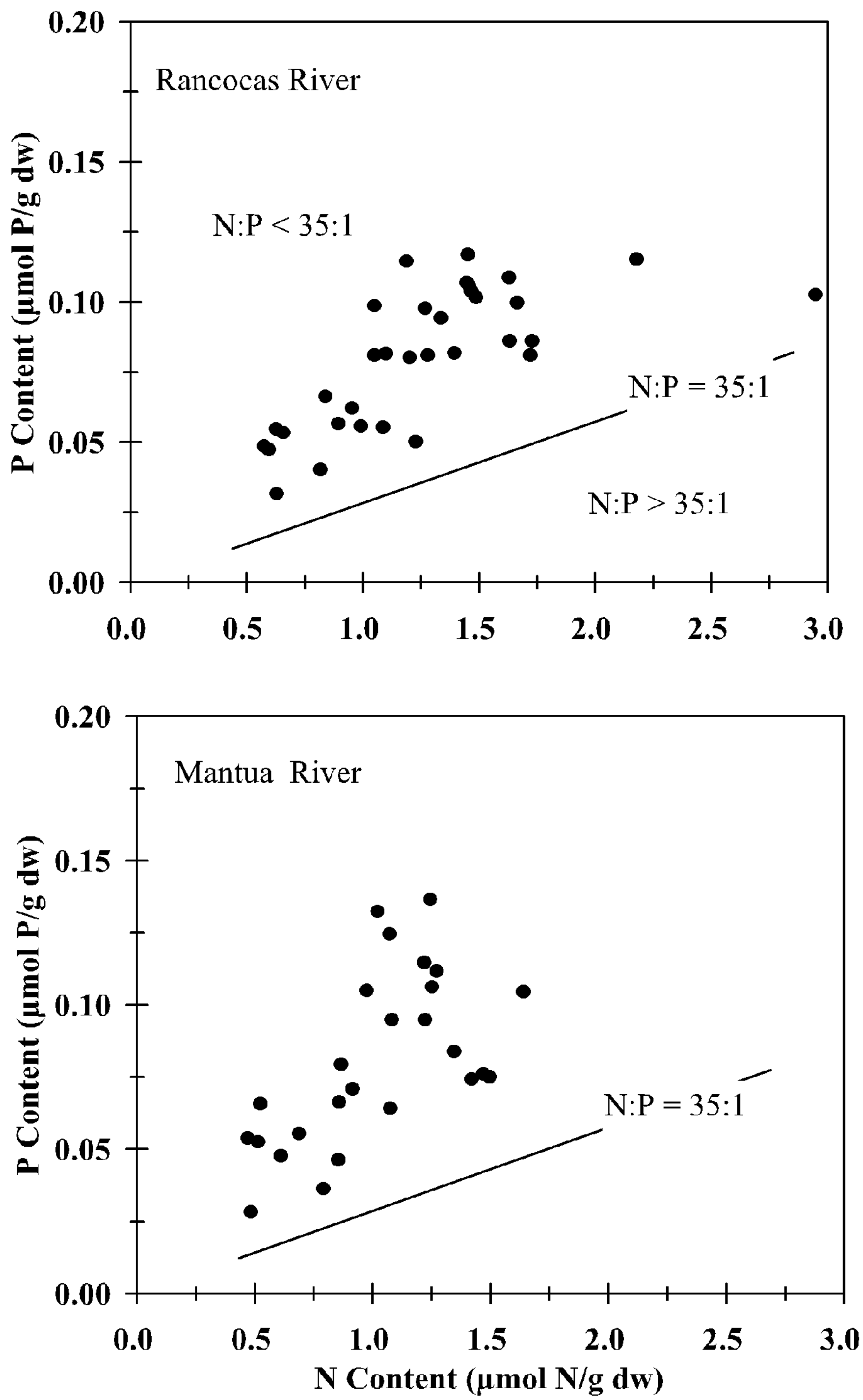

Figure 4-5 Relationship between nitrogen and phosphorus in plant material from Rancocas and Mantua creeks.

amounts by the time period (170 days), the average daily production per river-mile was obtained for NCP (Table 4-11; Note: (-) rates indicate incorporation into plant biomass).

There was a substantial amount of spatial variation in these results which is most likely due to the amount of high/low marsh area in each zone. Aboveground N, C, and P biomass incorporation rates (i.e., kg/river-mile (RM)-day) ranged from approximately -0.5 to -131 kg N/RM-day for N, -10 to -3,600 kg C/RM-day for C, and -0.1 to -19 kg P/RM-day for P (Table 4-11). The highest amount of plant biomass and productivity are in the Darby Creek (RM 85), Mantua (RM 89) and Rancocas (RM 111) creeks, and Crosswicks Creek (RM 128) systems (Table 4-11). Another source of potential variation is the amount of biomass assumed (from our average) for each marsh system. Previously, Whigham and Simpson (1976) estimated the amount of N in aboveground biomass within the Hamilton marsh system (Crosswick Creek). Their study analyzed plant biomass data over a full growing season and determined that approximately 32 metric tons (1000 kg = 1 metric ton) of N was stored in the marsh compared to 11 metric tons from our estimate. This difference may reflect better site-specific aboveground biomass information compared to our estimates. In this regard, N contents and areal amounts of emergent plants are similar, while their biomass estimate was substantially higher than our *average* estimate derived from sites on the Rancocas Creek. Additionally, our overall estimate for these elements do not take into account within-season turnover of plant biomass by either leaf mortality or herbivory. This can be a significant portion of the annual productivity and would make our estimates low. An additional bias is due to the fact that we did not sample belowground biomass which would also make our estimates low. The total amount of N and P in the aboveground biomass of the upper and lower EAV areas is given in Table 4-12. Similar amounts were estimated in both the upper and lower areas, with slightly more N and P stored in the upper EAVA. We estimated that 80 metric tons of N and 13 metric tons of P were retained in the living aboveground biomass in the two tidal freshwater wetland areas of the Delaware Estuary in 1997.

4.3.2 Marsh Sediment Fluxes of Nitrogen, Phosphorus and Oxygen

Background information into the mechanisms and biogeochemical processes that control the interactions of nitrogen, phosphorus, and oxygen between wetland sediments and the water column are beyond the scope of this study. However, Mitsch and Gosselink (1993) and references cited within, provide some information regarding the microbial and physical reactions affecting the these elements in wetlands (see also Klump and Martens, 1983; Bowden, 1987). In this section we explore the magnitude and direction of specific fluxes between the water column and the wetland sediments. Denitrification is a microbial process that occurs under anoxic conditions in which dissolved nitrate is reduced to nitrogen gas (Seitzinger, 1988a). The nitrate for this process is supplied from the overlying water column or decomposition reactions within the sediments. This process can be a major removal mechanism for nitrogen from aquatic systems. Other reactions evaluated include the movement (via advection/diffusion) of dissolved ammonium, nitrate+nitrate, and phosphate between the sediment and water column. In addition, to make preliminary estimates of oxygen dynamics, rates of sediment oxygen demand (SOD) were also obtained. While very limited, only literature rates and information were obtained for tidal freshwater wetlands in the mid-Atlantic region.

Most of the studies obtained for this project used core incubations to measure fluxes between wetland sediments and overlying water. Physical manipulations of the sediment can alter the natural conditions of in-place sediments and modify the "real" flux of many elements. Also, rates are generally given as mass per area per unit time (i.e., g of an element/m^2-day) and do not take into account the time that the wetland area is inundated by tidal water. For this study, flux rates were not corrected for tidal period innundation. Some studies such as those of Simpson et al. (1983a,b,c) and Velinsky et al. (unpublished data), measure the amount of nutrient coming into and leaving a specific marsh area over multiple tidal cycles. This information can then be used to calculate the flux of a nutrient between the wetland and tidal river. While some information is lost with this type of study in that potential difference between high and low areas and vegetated and unvegetated are averaged together, they do provide a whole marsh information regarding the interactions between the tidal river and wetlands. Also, a whole marsh study can be augmented with core incubations to examine both specific areas and biogeochemical processes.

4.3.2.1 Sediment-Water Exchange of Nitrogen

4.3.2.1.1 Denitrification

Denitrification is carried out by specific microorganisms in anoxic conditions with dissolved nitrate as the terminal electron acceptor. This process results in the formation of nitrous oxide and nitrogen gas in which the nitrogen gas can then diffuse into the air or overlying water. Seitzinger (1988a) discussed many of the controlling factors governing this reaction and its extent in many aquatic environments. Controlling factors for denitrification include ample supply of labile organic matter, dissolved nitrate, temperature, and anoxic conditions (Seitzinger, 1988a,b). The source of nitrate for denitrification can be from the water column (i.e., flux of nitrate into the sediments) or from nitrification of porewater ammonium (i.e., coupled nitrification-denitrification). For the purposes of this study, the actual source of nitrogen for denitrification is not considered.

Denitrification rates, both published and unpublished, were obtained and summarized in Table 4-13. These rates are for tidal freshwater wetlands and were not corrected for temperature differences between studies. Average rates ranged from -0.088 to 0.088 g N/m^2-day with no clear trend in the limited data between the three wetland zones (i.e., high and low marsh and mudflats). Negative values indicate possible N_2 fixation within the sediment and could be a possible new nitrogen addition to the marsh and water column, especially during the summer when nitrogen levels could be at a low (Zelenke, 1997), while positive values indicate denitrification with the flux of nitrogen gas from the sediments. Due to the extreme variability in the data, the average rates for high marsh, low marsh, and mudflat marsh areas were used. These rates were used in conjunction with the wetland areas from Table 4-2 and 4-3 (mudflats) to calculate a total daily removal of nitrogen per river-mile (kg N/RM-day) for the study area (Table 4-14).

These calculations suggest rates of denitrification for the emergent wetland areas of the Delaware Estuary ranged from -0.04 to 157 kg N/RM-day with the highest rates in river-mile sections with the highest wetland area (e.g., Rancocas Creek, Crosswicks Creek). Over the time period assumed for this study (77 days; July 1 to September 15), approximately 33 metric tons of N is removed from the tidal freshwater river system of the Delaware Estuary (Table 4-14). It should be recognized that there is substantial variation in these rates (Table 4-13). For example, using the average range for high marsh, low marsh, and mudflat areas from Table 4-15, the range in the amount of nitrate that is reduced via denitrification is -101 to 133 metric tons. It is apparent that the wide range in rates from the various studies indicates that site specific studies are essential to better assess this reaction within the emergent wetlands of the Delaware River.

4.3.2.1.2 Sediment-Water Exchange of Ammonia+ammonium and Nitrate+nitrite

Studies were sought that measured the areal flux (e.g., g N/m^2-day) of ammonia+ammonium and nitrate+nitrite between the water column and tidal freshwater wetlands. We focused our search for wetland studies within the mid-Atlantic region as being most similar to the wetlands in the upper Delaware Estuary. These data are summarized in Table 4-16. Average ammonia+ammonium and nitrate+nitrite rates ranged from -0.071 to 0.15 g N/m^2-day and from !0.093 to 0.025 g N/m^2-day, respectively. Negative rates indicate a flux of the chemical into the marsh sediment, while positive rates indicate a measurable flux from the sediment to the overlying water. Generally, summertime fluxes of ammonia+ammonium are directed out of the marsh system due to high rates of organic matter decomposition and a build up of dissolved ammonia+ammonium in the pore fluids relative to the overlying water. It is probable that a portion of the porewater ammonia+ammonium is oxidized to nitrate+nitrite within the sediments via nitrification. Fluxes of dissolved nitrate+nitrite are generally directed from the water column into the wetland sediments due to either the uptake of nitrogen by emergent plants and benthic algae, or more likely, to help supply the nitrate for denitrification. There was no clear trend between fluxes from within the high marsh, low marsh or mudflat areas, therefore average flux rates were used for these calculations. These rates were used in conjunction with the wetland areas from Table 4-2 and mudflat areas from Table 4-3 to calculate a total daily removal (or addition) of nitrogen per river-mile (kg N/RM-day) for the study area (Tables 4-17 and 4-18).

These calculations suggest that the average flux of ammonia+ammonium is positive (out of the marsh) while nitrate+nitrite fluxes are directed into the marsh (Tables 4-17 and 4-18). For the emergent wetland areas of the Delaware Estuary ammonia+ammonium average fluxes per river-mile ranged from 0.5 to 208 kg N/RM-day, while nitrate+nitrite fluxes ranged from !82 to !0.2 kg N/RM-day. As noted earlier, the highest rates occurred in the river-miles with the greatest amounts of wetland area. Over the time period assumed for this study (77 days), on average 75 metric tons of N as ammonia+ammonium is added to the water column of the tidal river, while approximately 30 metric tons of N as nitrate+nitrite is removed from the tidal river in our study area (Tables 4-17 and 4-18); thus, a net of 45 metric tons of N is added to the water column. The ammonia+ammonium can then be transported down river, taken up by phytoplankton, or oxidized to nitrate+nitrite via nitrification.

While the rates used for these calculations are from studies conducted mainly during the summer season, there was substantial variability among studies from the various locations. This variability can be due to natural variation within each marsh (e.g., amount and type of organic matter), ambient nutrient concentrations and loads, and tidal influences. To illustrate the amount of variability in these estimates, the average low and high flux rates from all areas were used to estimate the amount of N moved in the wetlands of the river (Table 4-15). The range in the amount of ammonia+ammonium that diffuses out of the sediments of the wetlands ranged from approximately 17 to 161 metric tons of N, while for nitrate+nitrite the range was from -106 to -9 metric tons of N. From these results, it is apparent that ammonia+ammonium generally was released from the wetland sediments during the summer months, while nitrate+nitrite was, on average, taken up by the wetland sediments. Also, the wide range in these estimates make it apparent that site-specific studies within the tidal Delaware River are needed to better assess this reaction within the emergent wetlands and their interactions with the river.

4.3.2.2 Sediment-Water Exchange of Phosphate

Sediment-water column exchange rates for dissolved phosphate in tidal freshwater wetlands are summarized in Table 4-19. These studies focused on tidal freshwater wetlands in the mid-Atlantic area so that they would be comparable to Delaware Estuary wetlands. Fluxes ranged from -0.003 to 0.24 g P/m^2-day and, on average, were directed out of the marsh into the adjacent water column. These data were mostly from a study in Jug Bay, a tidal freshwater wetland on the Patuxent River in Maryland. Using the average literature value and wetland areas (per river-mile), rates varied from 0.1 to 47 kg P/m^2-day from RM 82 to RM 133 (Table 4-20). Generally, the phosphate was released from the wetlands and, over the entire length of this study area, 17 metric tons (with a range of -6 to 67 metric tons; Table 4-15) of phosphate was introduced into the water column.

4.3.2.3 Sediment-Water Exchange of Dissolved Oxygen (Sediment Oxygen Demand)

Freshwater tidal wetland studies on sediment oxygen demand are limited. The studies and results obtained were focused on studies in tidal wetlands from the Atlantic region (i.e., Hudson River and Patuxent River; Table 4-19). Average summertime flux rates ranged from !0.70 to !3.1 g O_2/m^2-day (Note: negative fluxes indicate that dissolved oxygen is moving into the sediments) and, due to the small data set, there was no clear difference between high marsh, low marsh and mudflat wetland areas. Using the average rate and the wetland areas per river-mile (Table 4-2 and 4-3), sediment oxygen consumption rates varied from -18 to -8,400 kg O_2/m^2-day from RM 82 to RM 133, respectively (Table 4-21). The greatest rates, as with the other parameters, occurred in the sections of the river with the greatest wetland areas (e.g., Rancocas River, Crosswicks and Mantua Creeks). Estimates of oxygen consumption from the sediments ranged from -3,800 to -1,800 metric tons of oxygen removed from the water column by sediment respiration processes (Table 4-15).

4.3.3 Burial of N and P in Tidal Freshwater Wetlands of the Delaware Estuary

Estimates of burial rates for nitrogen and phosphorus were obtained for tidal freshwater wetlands (Table 4-22; Note (-) indicates burial into sediments). Average rates for N ranged from -9 to -23 g N/m^2-yr and for P from -1.3 to -3.5 g P/m^2-yr. These rates are for locations in the mid-Atlantic region and the Mississippi River. Using the average low and high burial rate, the amount of total N and P buried in the sediments per river-mile was derived for the time period of this study (Table 4-23 and 4-24). The burial rates were applied to both high, low and mudflat wetland areas and summed to provide a total amount over the entire area. The mass of material was apportioned for the 77-day active period and provided as a daily rate (kg/RM-day) for comparison to other processes. Annual burial rates were assumed to be a linear accumulation over the year. From these data, approximately 77 metic tons of N and 12 metric tons of P are buried in this area during the study period (Table 4-23 and 4-24). The river-miles with the greatest burial are those with the greatest amount of wetland area (i.e., high, low and mudflat) such as the Rancocas, Mantua, and Crosswick creeks. Also, due to the range in literature rates in burial, there was a substantial range in the estimates of nitrogen (24 to 105 metic tons) and phosphorus (3 to 15 metric tons) buried in the sediments (Table 4-15).

One study that is not included in Table 4-22 was conducted in the tidal freshwater wetlands of the Delaware Estuary (Orson et al., 1992). The focus of their extensive study was to determine sedimentation rates and trace metal fluxes using cores from marshes in Hamilton, Rancocas, Woodbury, and Oldmans creeks in the Delaware Estuary. They measured sedimentation rates with both ^{210}Pb and ^{137}Cs. Average rates using ^{210}Pb varied from 0.7 cm/yr at Hamilton Marsh to 1.7 cm/yr at Rancocas Creek with good agreement between both radiometric methods. While this study was comprehensive, it did not include the needed background data for the present study. However, using their calculated linear sedimentation rates (i.e., cm/yr) and assuming a water content of 70% from the Woodbury Creek marsh (Orson et al., 1990) and a dry sediment density of 1.8 g/cm^3, a mass sedimentation rate can be derived (i.e., g sediment/cm^3-yr). To derive N and P sedimentation rates, total N and P concentration data were used from three cores taken in Mantua Creek wetlands (Velinsky, unpublished data). Concentrations of total nitrogen and phosphorus at a depth of approximately 23 cm (i.e., below a depth of intense diagenesis) averaged 5.1 ± 1.1 mg N/g dw for nitrogen (n=3) and 1.3 mg P/g dw for phosphorus (n=3). From these data, we estimated rates of nitrogen and phosphorus accumulation from -13 g N/m^2-yr and -3.2 g P/m^2-yr in the Hamilton marsh to -32 g N/m^2-yr and -8.0 g P/m^2-yr in the Rancocas Creek wetlands.

There is good agreement between the rates calculated from the data in Orson et al. (1992) and those obtained from the literature. Considering the assumptions made, it appears that phosphorus deposition is higher in the tidal freshwater region of the Delaware Estuary than other tidal freshwater wetlands in the mid-Atlantic region, while nitrogen accumulation is comparable. These data and calculations indicate that burial of nitrogen and phosphorus is a major removal mechanism in the tidal freshwater portion of the estuary, but further studies are needed to determine both spatial variability in the deposition rate between wetlands, as well as more accurate rates for a mass balance model.

4.3.4 Dissolved Oxygen Production During Emergent Plant Production

One aspect of this study was to estimate the input of dissolved oxygen from emergent plant production within wetland systems. Oxygen is formed as a result of photosynthesis in plants, both emergent plants and phytoplankton. When photosynthesis takes place within the water column by phytoplankton or submerged plants, oxygen is added to the water directly and levels can be substantially higher than saturation during parts of the day (Kadlec and Knight, 1996). Once in the water, changes in the dissolved oxygen concentration of a given volume of water are caused by 1) air-water exchange; 2) consumption of oxygen through animal and microbial respiration both in the water and sediments; and 3) oxidation of reduced chemical species such as hydrogen sulfide and ammonia+ammonium (i.e., nitrification). In non-shaded areas where nutrients are abundant, water column photosynthesis and oxygen production can be observed to change on a daily basis following light levels. The situation can be much more complex within tidal freshwater wetlands with an abundant cover of emergent plants. Kadlec and Knight (1996) suggest that as a result of the dense coverage by emergent plants, average dissolved oxygen concentrations can be lower in wetlands in conjunction with a decrease in the amplitude of the diurnal oxygen cycle. This may be a result of the large amount of plant respiration by macrophytes, as they need to keep aerobic conditions around the root system that are within anaerobic sediments. In many cases, the diurnal movement of water via the tide can resupply the wetland with oxygenated water.

In many wetlands during the warmer summer months, sediments can turn anoxic as a result of an inbalance between the rate of sediment microbial oxygen consumption and diffusion of oxygen from the overlying water. While in tidal wetlands, during low tide, atmospheric oxygen can enter at a faster rate (i.e., mass transfer of oxygen is faster in air versus water) and in general, sediments will remain anoxic below a thin and variable layer of oxic conditions. In common with other rooted plants, wetland macrophytes such as *Nuphar* need oxygen within the root zone for respiration. In some cases, photosynthetically-produced oxygen is not sufficient for plant respiration in part due to the anoxic conditions around the root zone. Therefore, many wetland plants such as *Nuphar*, *Phragmites*, and *Zizania* (Dacey, 1981; Raskin and Kende, 1985; Armstrong et al., 1988; Brix et al., 1992; 1996) developed aerenchyma (i.e., air ducts) that help transport oxygen from the atmosphere to the root zone (Grosse et al., 1996). Internal movement of oxygen in wetland plants can occur by passive diffusion following concentration gradients within the lacunae system, and by convective flow of air through the internal gas spaces of the plants. The oxygen is transported to the roots and into the rhizosphere (Brix, 1993). Brix (1990; 1993) showed that substantial quantities of oxygen are passed through these air ducts to help maintain respiration rates within the root zones, while transporting carbon dioxide and methane to the atmosphere (Dacey and Klug, 1979). The excess supply of oxygen over that which is required for plant respiration has been termed the plant aeration flux (PAF) and is a difficult process to measure (Brix, 1990; Armstrong et al., 1990, and others). For example, PAF rates for *Phragmites* ranged from 0.02 to as high as 12 g O_2/m^2 day (see Kadlec and Knight, 1996) using various direct and indirect measurement methods.

Kadlec and Knight (1996) derived equations to determine the dissolved oxygen balance within wetland systems. While a discussion of these equations is beyond the scope of this study, they derived an equation regarding wetland oxygen demand (R_{WOD}):

$$R_{WOD} = R_{DOD} + R_{RES} - R_{PAF} - R_{PHOTO} \qquad (1)$$

where R is the rate of: oxygen consumption via decomposition (DOD); consumption via respiration (RES); addition via plant aeration (PAF); and addition via plant photosynthesis (PHOTO). All rates are in units of mass of oxygen per area per day (e.g., g O_2/m^2-day). Additionally, they derived the wetland net oxygen supply rate (R_{NOSR}) which is the balance between physical inputs (i.e., air-water exchange and hydraulic inputs) and R_{WOD}; where R_{NOSR} can be positive (supply), negative (consumption), or zero. Overall, it is difficult to predict how tidal freshwater emergent plants affect water column dissolved oxygen with the present effort due to many unknowns. The oxygen demand from sediments (SOD), water column (BOD_C and BOD_N), and plant respiration can impact the oxygen concentration in overlying water, and in many cases, these consumption terms may consume a substantial amount of water column oxygen above that produced via photosynthesis (Kadlec and Knight, 1996). Further field and modeling studies need to be conducted to more fully assess the impact of tidal freshwater plant production on the oxygen balance in the water column.

To provide a first-order estimate of the input of oxygen from wetland plant production, we used our estimates of biomass production and stoichiometric relationship between carbon fixation and oxygen production. Photosynthetic fixation of a carbon source (i.e., CO_2 or HCO_3^{2-}), using the conventional photosynthetic equation for a hexose product, yields an equivalent amount of oxygen. If the photosynthetic quotient (PQ) is unity, then for every 1 g of carbon fixed, 2.67 g of oxygen is produced. However, it should be noted that the PQ varies considerably depending on what is biochemically produced within the plant (e.g., fats, proteins, etc.) and can change substantially over the season. Westlake (1963) suggested that a more realistic average quotient would be approximately 1.2 for temperate plant communities (1 g C per 3.2 g O_2). From this relationship, it is possible to take our carbon biomass estimate and calculate the amount of oxygen produced.

Using the carbon biomass values calculated from our data (Table 4-10) and assuming that 2.67g O_2/g C, the total gross amount of oxygen produced was calculated (Table 4-25). The amount of oxygen produced per river-mile per day was calculated using a 170-day growing season (assume linear growth). Total gross oxygen production ranged from 26 kg O_2/RM-day at RM 109 to approximately 9,500 kg O_2/RM-day within RM111 (Table 4-25). As with the nitrogen, carbon and phosphorus calculations, the higher rates were calculated in sections of the river with highest areas of high and low marsh (e.g., Rancocas Creek, Darby Creek, and Mantua Creek). Sculthorpe (1967) and Fogel (personal communication) suggested that freshwater emergent plants respire approximately 70% of the gross oxygen production (P/R = 1.4). Using this value the rates were adjusted for plant respiration to yield net oxygen production (Table 4-25). From this calculation and the total high and low marsh area (1,410 ha), the net oxygen production over the study area for the growing season is approximately 0.7 g O_2/m^2-day. This average rate, while tentative, is close to that calculated for emergent plants in River Ivel by Edwards and Owens (1962, as cited in Sculthorpe, 1967). Their net oxygen production estimate ranged from 0.2 to 5.2 g O_2/m^2-day.

It is unclear as to how much of the potential net oxygen formation would move into the water. Estimates of net oxygen production derived from organic carbon productivities assume: 1) an unchanging PQ ratio, 2) that all oxygen produced during photosynthesis is liberated from the plant body, and importantly 3) that all the oxygen is liberated to the water column and not to the atmosphere. Passive and convective gas transport and oxygen solubility affect the amount of oxygen that might dissolve into the water. Also, the rates are average daily rates while photosynthesis is restricted to daylight hours and is related to the intensity of solar radiation. These factors and those stated above result in substantial uncertainty of these estimates.

4.3.5 Nitrogen, Phosphorus, and Oxygen Within Submergent Aquatic Vegetation

Submerged aquatic vegetation (SAV) has often been cited as an important component of the aquatic environment, both marine and freshwater, in relation to habitat, nutrient cycling and oxygen dynamics (Carpenter and Lodge, 1986; Kemp et al., 1984). In the Chesapeake Bay, the extent and importance of SAV in the overall ecosystem is well studied (for example, Orth et al., 1994; Carter et al., 1991; Caffery and Kemp, 1992 and others) and SAV protection is rapidly escalating (Chesapeake Bay Program, 1995; Bay Journal, 1998). However, Schuyler (1988) stated that virtually no work has been documented on the plants that grow underwater (submergents) or have floating leaves (planmergents) in the tidal freshwater portion of the Delaware River. Submergents and planmergents (collectively grouped as SAV) are generally found in the lower portion of the intertidal zone where they are attached to specific substrates or rooted in the sediment. As with the Chesapeake Bay, the decline and absence of SAV in the freshwater section of the Delaware Estuary may be associated with anthropogenic inputs such as nutrients, suspended matter, as well as specific contaminants.

Submerged aquatic vegetation can take up nutrients from both the water column and sediments as well as alter oxygen concentrations of the overlying water (Caffery and Kemp, 1991; 1992; Short and McRoy, 1984; Sand-Jensen et al., 1982). Diel oxygen changes of as much as 8 mg/L have been documented in waters of dense SAV beds (Ondok et al., 1984). Oxygen release appears to be linked to photosynthetic O_2 production occurring only when plants are illuminated (Caffrey and Kemp, 1991; Carpenter and Lodge, 1986 and others). Between 1 to 23% of the gross O_2 production may be released by various species of SAV (Kemp and Murray, 1986; Caffery and Kemp, 1991). Caffrey and Kemp (1991) reported an O_2 flux for an estuarine SAV of approximately 120 µmol g^{-1} dry weight biomass h^{-1} in the Choptank River (MD). Biomass of *Potamogeton perfoliatus* ranged from 56 to 364 g dw m^{-2} from April to September with the lowest biomass in April and the highest in July. Kemp and Murray (1986) measured the release of O_2 from the roots of *Potamogeton perfoliatus* of approximately 0.55 mmol O_2 m^{-2} h^{-1} in the Chesapeake Bay.

As with emergent aquatic vegetation, the source of nutrients, either to sediments or open water, for SAV is not fully understood. Carignan and Kalff (1980) used radio-labeled phosphorus (^{32}P), and determined that on average 72% of the phosphorus was obtained from the sediments for various freshwater submerged macrophytes (e.g., *Najas flexillis*, *Myriophyllum alterniflorum* and others). The phosphorus that was taken into the plants was then translocated into the shoots of the plants. Similarly, Barko and Smart (1981) showed that both nitrogen and phosphorus for plant growth were taken up by the roots for a variety of marine and freshwater species. Caffery and Kemp (1992) showed that for the species *Potamogeton perfoliatus* root uptake was generally greater than shoot uptake of nitrogen with total nitrogen incorporation rates of between 142 and 148 nmol N/cm^{-2} d^{-1} (based on plant surface area). They suggested that changes in the root:shoot biomass may control the relative importance between root and shoot uptake.

In the tidal freshwater Delaware River, approximately 12 species of SAV have been observed in the tidal river since 1970, including: *Vallisneria americana*, *Myriophyllum spicatum*, *Elodea nuttallii*, *Najas flexillis*, *Potamogeton sp.* and others (Schuyler, 1988). Personal observations taken in August 1997 indicate extensive beds of SAV in the shallows of the Rancocas Creek. In this regard, Carter et al. (1988) reported SAV biomass in the tidal freshwater Potomac River of between 300 to 1000 g dw m^{-2} for a community that comprised species found in the Delaware River (Schuyler, 1988). They measured substantial increases of dissolved oxygen in SAV beds compared to control sites. However, as part of our literature search, we did not find any estimates for the areal extent and biomass for SAV in the tidal freshwater Delaware River Estuary. A similar finding was reported by Sullivan et al. (1991) in their study of the overall habitat of the Delaware Estuary. Therefore, we were unable to quantitatively determine the importance of SAV production and nutrient uptake on the water quality of the river. Future studies should be directed towards determining the extent and biomass of SAV in the tidal freshwater areas as well as mesocosm experiments to determine nutrient and oxygen fluxes.

4.4 Inputs and outputs of N and P in TFW of the Delaware Estuary

Literature data obtained for rates of denitrification, sediment-water exchange of ammonium+ammonia, nitrate+nitrite and ortho-phosphate, and burial of nitrogen and phosphorus were applied to the area of tidal freshwater wetlands of the Delaware. As stated previously, only studies from the warmer summer period were used in our analysis. The various rates were summarized and extrapolated to measured marsh areas (Table 4-1) to produce fluxes on a mass per time basis (i.e., kg/day). While there were substantial variations in many rates, there are interesting features in the calculated fluxes (Table 4-26). Plant uptake accounted for a substantial amount of nitrogen and phosphorus removal from the river with incorporation rates of 480 kg/yr and 80 kg/yr, respectively. By August, plants incorporated 80 metric tons (MT) of nitrogen and 13 MT of phosphorus in their aboveground biomass. Denitrification (NO_3^- --> N_2) also removed a substantial amount of nitrogen from the river and the rate was, on average, balanced by the flux of NO_3^-+NO_2^- into the sediments (i.e., +430 versus -390 kg N/yr). Using literature-derived summer flux data, a large portion of the sedimentary nitrogen and phosphorus was exchanged into the overlying water column. The large NH_4^++NH_3-N flux out of the sediment can account for all N taken up by the plants as well as a portion of the NO_3^-+NO_2^--N flowing into the sediments. Burial rates of N and P were large, and for nitrogen were an important removal term.

These average rates were compared to loads of nitrogen and phosphorus from upstream sources (Delaware River and tributaries), waste-water treatment plants (WWTP), combined sewer overflows, and atmospheric deposition (Table 4-27). Atmospheric deposition rates from Scudlark and Church (1993) were applied to the surface area of the wetlands only, as watershed inputs are accounted for in the upstream sources. It is evident that discharge of nitrogen from WWTP account for a substantial portion of the total load to this area with slightly lesser amounts from upstream sources. For phosphorus, upstream inputs at Trenton and other smaller tributary streams/creeks account for approximately 70% of the total load of phosphorus to upper tidal Delaware Estuary. No estimates were given for combined sewer or atmospheric deposition sources for phosphorus.

In comparison, the amount of nitrogen and phosphorus added to the tidal freshwater area of the Delaware River from the different sources overshadows the amount of nitrogen cycled through the wetlands in this region. For phosphorus, wetland burial and plant biomass can account for approximately 10% of the total load to the area. However, the flux of o-PO_4 apparently returns a similar amount back to the river during this time period. Therefore, we estimate that the remaining tidal freshwater wetlands in the Delaware are not presently extensive enough to have a major impact on the flow of nitrogen to the more saline portion of the estuary and the coastal ocean (although phosphorus fluxes could be somewhat altered). The reason for this is due to the high loading rates of nitrogen and to a lesser extent phosphorus from upstream sources and WWTPs compared to the fluxes of nitrogen and phosphorus into or out of these wetlands. We caution, however, that our calculated nutrient fluxes through these wetlands could be underestimated if other biological processes are sizeable, such as belowground production by EAV, production by other primary producers like benthic algae, and secondary production. It should also be pointed out that the importance of these wetlands on a smaller scale (i.e., within a specific tidal creek) may have some impact on the local water quality. Our conclusions were also based in part on literature data which have a large error range and are from other geographic regions. Hence, we suggest that further studies are needed to improve the robustness of our findings.

Table 4.1 Marsh area by study area and tributaries based on aerial photo interpretation.

Lower Study Area (RM 82 to 91)	High Marsh (ha)	Low Marsh (ha)	Total (ha)
Delaware River Mainstem - Lower	28	29	57
Cedar Swamp	44	0	44
Chester Creek:	0	0	0
Repaupo Creek	0	0	0
Ridley Creek	0	0	0
Crum Creek	0	0	0
Darby Creek[1]	59	62	120
Clomnell Creek	0	0	0
Mantua Creek	156	100	256
Little Mantua Creek	0	21	21
Main Ditch	18	0	18
Woodbury Creek	101	2	103
Total Lower Study Area	405	214	619
Upper Study Area (RM 108 to 133)	**High Marsh**	**Low Marsh**	**Total**
Delaware River Mainstem - Upper	3	24	27
Pompston Creek	0	0	0
Swede Run	0	0	0
Rancocas Creek	340	126	466
Poquessing Creek	0	0	0
Neshaminy Creek	0	0	0
Assiscunk Creek	37	7	43
Otter Creek	2	1	4
Bustleton Creek	0	0	0
Martins Creek[2]	*	*	*
Crafts Creek	0	0	0
Scotts Creek	0	0	0
Blacks Creek	14	0	14
Crosswicks Creek	187	23	209
Duck Creek	14	16	30
Biles Creek	0	0	0
Total Upper Study Area	596	197	793

[1] Darby Creek marsh/mud areas were obtained from NWI data

[2] No data available (*)

Table 4.2 Marsh area by river-mile based on aerial photo interpretation/classification.

Rivermile	High Marsh (ha)	Low Marsh (ha)	Total (ha)	Rivers/Creeks Included[1]
82	0	0	0	DE Riv-Lower
83	44	7	51	DE Riv-Upper, Cedar Swamp
84	28	8	36	DE Riv-Lower
85	58	65	123	DE Riv-Lower, Darby[2]
86	0	9	9	DE Riv-Lower
87	0	2	2	DE Riv-Lower
88	0	0	0	DE Riv-Lower
89	156	100	256	DE Riv-Lower, Mantua
90	18	21	40	DE Riv-Lower, Little Mantua, Main Ditch
91	101	2	103	DE Riv-Lower, Woodbury
108	0	0	0	DE Riv-Upper
109	0	2	2	DE Riv-Upper
110	1	1	3	DE Riv-Upper
111	340	128	467	DE Riv-Upper, Rancocas
112	0	6	6	DE Riv-Upper
113	0	0	0	DE Riv-Upper
114	0	0	0	DE Riv-Upper
115	0	0	0	DE Riv-Upper
116	0	0	0	DE Riv-Upper
117	0	0	0	DE Riv-Upper
118	39	8	47	DE Riv-Upper, Assiscunk, Otter
119	0	0	0	DE Riv-Upper
120	0	0	0	DE Riv-Upper
121	0	0	0	DE Riv-Upper
122	0	0	0	DE Riv-Upper
123	0	0	0	DE Riv-Upper
124	0	0	0	DE Riv-Upper
125	0	4	4	DE Riv-Upper
126	1	7	9	DE Riv-Upper
127	0	3	3	DE Riv-Upper
128	201	23	223	DE Riv-Upper, Blacks, Crosswicks
129	14	16	30	DE Riv-Upper, Duck
130	0	0	0	DE Riv-Upper
131	0	0	0	DE Riv-Upper
132	0	0	0	DE Riv-Upper
133	0	0	0	DE Riv-Upper
Total Lower	405	214	619	
Total Upper	596	197	793	
Total All	1,001	411	1,412	

[1] Only rivers or creeks having marsh areas are included.

2 Darby Creek wetland areas within RM 85 are based on NWI since no aerial photos were available.

Table 4.3 Marsh and mud areas by river-mile based on National Wetlands Inventory (NWI)[1].

River-mile	High Marsh (ha)	Low Marsh (ha)	Mud (ha)	Total (ha)	Rivers/Creeks Included
82	0	0	26	26	DE Riv-Lower
83	46	0	76	122	DE Riv-Upper, Cedar Swamp
84	34	0	115	149	DE Riv-Lower
85	61	62	87	210	DE Riv-Lower, Darby
86	0	0	117	117	DE Riv-Lower
87	2	0	106	109	DE Riv-Lower
88	12	0	10	23	DE Riv-Lower
89	212	0	13	225	DE Riv-Lower, Mantua
90	22	0	40	61	DE Riv-Lower, Little Mantua, Main Ditch
91	44	0	45	88	DE Riv-Lower, Woodbury
108	8	0	8	16	DE Riv-Upper
109	0	0	24	24	DE Riv-Upper
110	0	0	43	43	DE Riv-Upper
111	438	0	55	493	DE Riv-Upper, Rancocas
112	7	0	9	16	DE Riv-Upper
113	0	0	32	32	DE Riv-Upper
114	0	0	1	1	DE Riv-Upper
115	5	0	0	5	DE Riv-Upper
116	0	0	5	5	DE Riv-Upper
117	0	0	3	3	DE Riv-Upper
118	55	0	1	56	DE Riv-Upper, Assiscunk, Otter
119	0	0	0	0	DE Riv-Upper
120	0	0	2	2	DE Riv-Upper
121	0	0	19	19	DE Riv-Upper
122	0	0	53	53	DE Riv-Upper
123	0	0	7	7	DE Riv-Upper
124	8	0	4	12	DE Riv-Upper
125	12	0	14	26	DE Riv-Upper
126	0	0	15	15	DE Riv-Upper
127	1	0	12	13	DE Riv-Upper
128	232	0	3	235	DE Riv-Upper, Blacks, Crosswicks
129	18	0	13	31	DE Riv-Upper, Duck
130	0	0	46	46	DE Riv-Upper
131	0	0	13	13	DE Riv-Upper
132	0	0	6	6	DE Riv-Upper
133	0	0	0	0	DE Riv-Upper
Total Lower	433	62	636	1,130	
Total Upper	786	0	388	1,174	
Total All	1,219	62	1,024	2,305	

[1] Categories shown were developed by grouping various NWI categories (see text Table 3-2).

Table 4.4 Comparison of the marsh areas based on aerial photography versus NWI.

Study Area	Marsh/Mud Type	Aerial[1] (ha)	NWI (ha)	% Difference
DE River-Lower	High Marsh	405	433	7%
	Low Marsh	214	62	-71%
	Total Marsh	619	495	-20%
DE River-Upper	High Marsh	596	786	32%
	Low Marsh	197	0	-100%
	Total Marsh	793	786	-1%

[1] Includes NWI areas for Darby Creek since no aerial photos were available.

Table 4-5. Mean (±SD, n) biomass (g dry weight m^{-2}) of emergent aquatic vegetation with Type 1 (low marsh, *Nuphar*), Type 2 (mid/high marsh, *Zizania*), Type 3 (high marsh, *Typha*) and Type 4 (high marsh, mixed species) zones at Rancocas and Mantua Creeks in August 1997.

Marsh Type	Rancocas Creek	Mantua Creek	t-test p-value
Type 1 Low Marsh	241	811	ns
(*Nuphar sp.*)	(±36, 6)	(±700, 6)	
Type 2 High Marsh	323	382	ns
(*Zizania sp.*)	(±168, 4)	(±83, 4)	
Type 3 High Marsh	1305	1191	ns
(*Typha sp.*)	(±268, 4)	(±831, 4)	
Type 4 High Marsh	524	282	ns
(mixed species)	(±159, 4)	(±82, 4)	

ns = not significant at a=0.05.

Table 4-6. Minimum, maximum and median biomass (g dry weight m^{-2}) of emergent aquatic vegetation in either the low or high marsh of Rancocas and Mantua Creeks in August 1997.

Study Site	Marsh Elevation	Minimum	Maximum	Median	Sample Size
Rancocas	Low	208	300	232	6
	High	146	666	402	8
Mantua	Low	156	1952	731	6
	High	190	502	327	8

Table 4-7. Concentrations of nitrogen, carbon and phosphorus from plants in the Rancocas Creek[1] .

		Nitrogen mg N/ g dw	Carbon mg C/ g dw	Phosphorus mg P/ g dw	C:N Molar	C:P Molar	N:P Molar
Rancocas Creek							
Nuphar sp.	Mean	20.4	391.1	3.1	23.0	337.4	14.9
	± 1s	3.9	15.6	0.4	3.9	53.6	2.6
	%RSD	19	4	13	17	16	17
	n = 13						
Wild Rice	Mean	9.2	413.8	1.7	53.4	636.0	11.9
	± 1s	1.5	11.2	0.2	7.2	68.3	0.5
	%RSD	16	3	12	13	11	4
	n = 6						
*Arrow Arrum**	Mean	23.0	385.4	3.1	21.6	324.0	16.3
	± 1s	9.8	36.7	0.4	6.5	58.7	7.2
	% RSD	43	10	13	30	18	44
	n = 6						
Ragweed*	Mean	17.1	401.5	2.0	28.4	547.2	19.5
	± 1s	4.1	6.2	0.5	6.2	111.3	3.7
	% RSD	24	2	25	22	20	19
	n = 4						
Typha sp.	Mean	12.7	432.6	1.6	42.0	772.6	17.8
	± 1s	3.0	33.6	0.6	12.5	334.0	2.5
	% RSD	23	8	36	30	43	14
	n = 7						

[1]Samples contained a small proportion of other species. (August 1997). %RSD - % relative standard deviation = (standard deviation/mean) x 100.

Table 4-8. Concentrations of nitrogen, carbon and phosphorus in plants in the Mantua Creek[1].

		Nitrogen mg N/ g dw	Carbon mg C/ g dw	Phosphorus mg P/ g dw	C:N Molar	C:P Molar	N:P Molar
Mantua Creek							
Nuphar sp.	Mean	17.7	386.6	3.2	26.0	326.8	13.0
	± 1s	2.7	14.8	0.7	3.5	77.1	4.2
	% RSD	15	4	20	13	24	33
	n = 15						
Wild Rice*	Mean	13.0	390.5	2.1	35.4	474.9	13.6
	± 1s	1.4	14.8	0.2	3.0	54.9	2.6
	% RSD	11	4	9	8	12	19
	n = 7						
Ragweed	Mean	7.7	413.0	1.8	64.2	608.0	9.7
	± 1s	1.3	12.4	0.2	9.6	43.3	1.9
	% RSD	17	3	11	15	7	20
	n = 4						
Typha sp.	Mean	9.6	461.9	1.2	59.1	1024.9	17.7
	± 1s	2.4	8.2	0.3	14.8	259.3	3.9
	% RSD	25	2	24	25	25	22
	n = 4						

[1]Samples contained a small proportion of other species. (August 1997). %RSD - % relative standard deviation = (standard deviation/mean) x 100.

Table 4-9. Weighted-average concentration of NCP in emergent plants from the upper and lower EAVA[1]

	Nitrogen mg N/ g dw	Carbon mg C/ g dw	Phosphorus mg P/ g dw
Upper EAVA - Rancocas			
Low Marsh	20.4	391.1	3.1
High Marsh	12.0	360.8	1.7
Lower EAVA - Mantua			
Low Marsh	17.7	386.6	3.2
High Marsh	11.5	429.3	1.8

[1]Low marsh concentrations are for *Nuphar*, while high marsh concentrations are a biomass-weighted average concentration.

Table 4-10. Amount of nitrogen, carbon and phosphorus per river mile in tidal freshwater wetlands of the upper Delaware estuary[1].

River Mile	River/Creek	Total N (kg)	Total C (kg)	Total P (kg)
82	DE Riv-Lower			
83	Cedar Swamp	2,601	82,472	430
84	DE Riv-Lower	2,075	61,453	350
85	DE Riv-Lower; Darby Creek	10,568	264,810	1,857
86	DE Riv-Lower	1,168	25,514	211
87	DE Riv-Lower	238	5,198	43
88	DE Riv-Lower			
89	DE Riv-Lower; Mantua Creek	18,815	501,691	3,259
90	Little Mantua Creek, Main Ditch	3,453	86,166	608
91	DE Riv-Lower; Woodbury	4,031	146,746	637
108	DE Riv-Upper			
109	DE Riv-Upper	86	1,653	13
110	DE Riv-Upper	138	3,424	20
111	DE Riv-Upper; Rancocas Creek	22,418	608,325	3,224
112	DE Riv-Upper	267	5,118	40
113	DE Riv-Upper			
114	DE Riv-Upper			
115	DE Riv-Upper			
116	DE Riv-Upper			
117	DE Riv-Upper			
118	DE Riv-Upper; Assiscunk and Otter Creeks	2,255	63,589	323
119	DE Riv-Upper			
120	DE Riv-Upper			
121	DE Riv-Upper			
122	DE Riv-Upper			
123	DE Riv-Upper			
124	DE Riv-Upper			
125	Crafts Creek			
126	DE Riv-Upper	409	8,618	61
127	DE Riv-Upper	121	2,326	18
128	Blacks Creek and Crosswicks Creek	10,760	311,879	1,533
129	DE Riv-Upper; Duck Creek	1,413	34,306	206
130	DE Riv-Upper; Biles Creek			
131	DE Riv-Upper			
132	DE Riv-Upper			
133	DE Riv-Upper			

[1]Amounts are for high and low areas.

Table 4-11. Daily uptake rate of nitrogen, carbon and phosphorus per river mile in tidal freshwater wetlands of the upper Delaware estuary[1].

River Mile	River/Creek	Total N kg/RM-day	Total C kg/RM-day	Total P kg/RM-day
82	DE Riv-Lower			
83	Cedar Swamp	-15.3	-485.1	-2.5
84	DE Riv-Lower	-12.2	-361.5	-2.1
85	DE Riv-Lower; Darby Creek	-62.2	-1,557.7	-10.9
86	DE Riv-Lower	-6.9	-150.1	-1.2
87	DE Riv-Lower	-1.4	-30.6	-0.3
88	DE Riv-Lower			
89	DE Riv-Lower; Mantua Creek	-110.7	-2,951.1	-19.2
90	Little Mantua Creek, Main Ditch	-20.3	-506.9	-3.6
91	DE Riv-Lower; Woodbury	-23.7	-863.2	-3.7
108	DE Riv-Upper			
109	DE Riv-Upper	-0.5	-9.7	-0.1
110	DE Riv-Upper	-0.8	-20.1	-0.1
111	DE Riv-Upper; Rancocas Creek	-131.9	-3,578.4	-19.0
112	DE Riv-Upper	-1.6	-30.1	-0.2
113	DE Riv-Upper			
114	DE Riv-Upper			
115	DE Riv-Upper			
116	DE Riv-Upper			
117	DE Riv-Upper			
118	DE Riv-Upper; Assiscunk and Otter Creeks	-13.3	-374.1	-1.9
119	DE Riv-Upper			
120	DE Riv-Upper			
121	DE Riv-Upper			
122	DE Riv-Upper			
123	DE Riv-Upper			
124	DE Riv-Upper			
125	Crafts Creek			
126	DE Riv-Upper	-2.4	-50.7	-0.4
127	DE Riv-Upper	-0.70	-13.7	-0.1
128	Blacks Creek and Crosswicks Creek	-63.3	-1,834.6	-9.0
129	DE Riv-Upper; Duck Creek	-8.3	-201.8	-1.2
130	DE Riv-Upper; Biles Creek			
131	DE Riv-Upper			
132	DE Riv-Upper			
133	DE Riv-Upper			

[1]Rate assumes a linear growing period from March 1 to August 18 (170 days) for both low and high marsh areas. (-) values indicate uptake of N, C, and P by the plants from all sources.

Table 4-12. Total amount of nitrogen, carbon and phosphorus in emergent wetland biomass in the tidal freshwater portion of the Delaware estuary (August 1997).

	Total N	Total C	Total P
Lower EAVA total (kg)	42,950	1,174,050	7,395
Upper EAVA total (kg)	37,868	1,039,238	5,437
Total Upper and Lower EAVA (Metric Tons)[1]	80.8	2,213	12.8

[1] 1 metric ton = 1000 kg

Table 4-13. Rates of denitrification for tidal freshwater wetlands[1].

Marsh Area	Location	Year	Low	High	Average	Method	Reference	Comments
Tivoli Bay: HM	Hudson River	1996	-0.011	0.076	0.033	Ar/N2	Zelenke, 1997	Core incubations (16 to 21 °C)
Tivoli Bay: LM	Hudson River	1996	-0.315	0.027	-0.088	Ar/N2		
Tivoli Bay: Mud	Hudson River	1996	-0.299	0.046	-0.073	Ar/N2		
Jug Bay Wetlands: HM	Patuxent River	1995	0.009	0.055	0.034	Ar/N2	Groszkowski, 1995	Core incubations (29 to 30 °C)
Jug Bay Wetlands: LM	Patuxent River	1995	0.000	0.050	0.020	Ar/N2		
Jug Bay Wetlands: Mud	Patuxent River	1995	0.028	0.037	0.033	Ar/N2		
Jug Bay Wetlands: HM	Patuxent River	1995	0.012	0.171	0.088	Ar/N2	Groszkowski, 1995	Continous incubations (29 to 30
Jug Bay Wetlands: LM	Patuxent River	1995	0.007	0.042	0.028	Ar/N2		
Jug Bay Wetlands: Mud	Patuxent River	1995	0.000	0.079	0.030	Ar/N2		
North River	North River	1991	0.008	0.024	0.016	Model	Bowden et al., 1991	
Barataria Basin	Mississippi River	1986	ND	ND	0.002	Acetylene	Smith and Patrick, 1982	
Other Wetlands; Palustrine								
Great Meadows	Massachusetts	1979	ND	ND	0.151		Bartlett et al., 1979	
Lake Wingra	Wisconsin	1978	ND	ND	0.001		Prentki et al., 1978	

[1]Rates are g N/m^2-day. HM - high marsh, LM - Low marsh, Mud - Mudflat. (+) values are efflux of N_2 gas out of the marsh and (-) are influx of N_2 into the marsh.

Table 4-14. Average amount of nitrogen gas removed or added from emergent marsh system via denitrification or N fixation[1].

River Mile	River/Creek	High Marsh kg N	Low Marsh kg N	Mudflat kg N	Total kg N	Total Daily Rate kg N/RM-day
82	DE Riv-Lower			-67	-67	-0.9
83	Cedar Swamp	1,754	-75	-192	1,487	19.3
84	DE Riv-Lower	1,103	-82	-293	728	9.5
85	DE Riv-Lower; Darby Creek	2,323	-663	-221	1,439	18.7
86	DE Riv-Lower		-92	-298	-390	-5.1
87	DE Riv-Lower		-19	-270	-289	-3.7
88	DE Riv-Lower			-26	-26	-0.3
89	DE Riv-Lower; Mantua Creek	6,201	-1,026	-34	5,141	66.8
90	Little Mantua Creek, Main Ditch	734	-218	-100	415	5.4
91	DE Riv-Lower; Woodbury Creek	4,012	-19	-113	3,879	50.4
108	DE Riv-Upper			-19	-19	-0.2
109	DE Riv-Upper		-19	-62	-80	-1.0
110	DE Riv-Upper	58	-15	-109	-66	-0.9
111	DE Riv-Upper; Rancocas Creek	13,518	-1,307	-139	12,072	156.8
112	DE Riv-Upper		-58	-23	-81	-1.1
113	DE Riv-Upper			-81	-81	-1.1
114	DE Riv-Upper			-3	-3	-0.04
115	DE Riv-Upper					
116	DE Riv-Upper			-12	-12	-0.2
117	DE Riv-Upper			-8	-8	-0.1
118	DE Riv-Upper; Assiscunk and Otter Creeks	1,542	-84	-2	1,456	18.9
119	DE Riv-Upper					
120	DE Riv-Upper			-5	-5	-0.1
121	DE Riv-Upper			-50	-50	-0.6
122	DE Riv-Upper			-135	-135	-1.7
123	DE Riv-Upper			-17	-17	-0.2
124	DE Riv-Upper			-9	-9	-0.1
125	Crafts Creek		-42	-36	-78	-1.0
126	DE Riv-Upper	59	-73	-37	-51	-0.7
127	DE Riv-Upper		-26	-31	-57	-0.7
128	Blacks Creek, Crosswicks Creek	7,997	-232	-9	7,757	100.7
129	DE Riv-Upper; Duck Creek	546	-163	-33	351	4.6
130	DE Riv-Upper; Biles Creek			-117	-117	-1.5
131	DE Riv-Upper			-34	-34	-0.4
132	DE Riv-Upper			-16	-16	-0.2
133	DE Riv-Upper					
	Total (Metric Tons)	**39.8**	**-4.2**	**-2.6**	**33.0**	

[1]Calculations based on average denitrification rates for tidal freshwater wetlands, uncorrected for temperature differences between studies. Positive fluxes are directed out of the marsh, while negative fluxes are directed into the marsh. Masses assume a 77-day active period in the wetland.

Table 4-15. Summary of various biogeochemical processes for the tidal freshwater wetlands of the upper Delaware Estuary.

	Low Estimate	High Estimate	Average Estimate
Denitrication			
High Marsh (kg)	2,543	77,611	39,846
Low Marsh (kg)	-32,520	12,571	-4,211
Mudflat (kg)	-70,973	42,584	-2,602
Total Areas (kg)	-100,950	132,766	33,032
Metric Tons	-100.9	132.8	33.0
Ammonium+ammonia			
High Marsh (kg)	6,936	66,281	30,828
Low Marsh (kg)	2,850	27,232	12,666
Mudflat (kg)	7,097	67,819	31,544
Total Areas (kg)	16884	161332	75,038
Metric Tons	16.9	161.3	75.0
Nitrate+nitrite			
High Marsh (kg)	-43,701	3,858	-12,169
Low Marsh (kg)	-17,955	1,585	-5,000
Mudflat (kg)	-44,715	3,947	-12,451
Total Areas (kg)	-106371.2	9389.9	-29,619
Metric Tons	-106.4	9.4	-29.6
Burial Nitrogen			
High Marsh (kg)	-9,774	-42,985	-31,792
Low Marsh (kg)	-4,016	-17,660	-13,062
Mudflat (kg)	-10,001	-43,982	-32,529
Total Areas (kg)	-23,791	-104,627	-77,383
Metric Tons	-23.8	-104.6	-77.4
Ortho-Phosphate			
High Marsh (kg)	-2,617	27,617	6,979
Low Marsh (kg)	-1,075	11,347	2,868
Mudflat (kg)	-2,678	28,258	7,141
Total Areas (kg)	-6370.1	67221.2	16,988
Metric Tons	-6.4	67.2	17.0
Burial Phosphorus			
High Marsh (kg)	-1,146	-6,106	-4,898
Low Marsh (kg)	-471	-2,509	-2,012
Mudflat (kg)	-1,172	-6,248	-5,011
Total Areas (kg)	-2788	-14,863	-11,921
Metric Tons	-2.8	-14.9	-11.9
Sediment Oxygen Demand			
High Marsh (kg)	-1,579,167	-751,415	-1,242,615
Low Marsh (kg)	-648,808	-308,722	-510,534
Mudflat (kg)	-1,615,806	-768,849	-1,271,445
Total Areas (kg)	-3843780.3	-1,828,986	-3,024,594
Metric Tons	-3,844	-1,829	-3,025

1 (-) values are directed into the wetland from the all potential sources. Amounts are for the 77-day time period of this study.

Table 4-16. Sediment-water exchange rates of ammonia and nitrate+nitrite for various tidal freshwater wetlands.

Marsh Area	Location	Year	Ammonia -----------------------			Nitrate+nitrite ----------			Method	Reference
			Low	High	Average	Low	High	Average		
Phillips Creek Marsh	James River, VA	1992	0.004	0.049	0.029				Chamber method	Chambers, 1992
Rhodes River	Chesapeake Bay	1982	-0.004	0.000		-0.001	-0.002		Hydrologic export model	Jordan et al. 1983
Gott's Marsh	Chesapeake Bay	1975	ND	ND	-0.001	ND	ND	-0.003		Heinle and Flemner, 1976
North River Marsh	Massachusectts	1985	-0.089	-0.016	-0.071	0.036	-0.302	-0.093	Flume	Bowden, 1986
Tivoli Bay Marsh: North Bay: LM	Hudson River	1997	ND	ND	0.051	0.000	0.051		Core incubation	Zelenke, 1997 (unpublished)
Tivoli Bay Marsh: North Bay: HM	Hudson River	1997	-0.007	0.009		ND	ND	ND	Core incubation	Zelenke, 1997 (unpublished)
Tivoli Bay Marsh: South Bay: LM	Hudson River	1997	-0.010	0.003		ND	ND	0.000	Core incubation	Zelenke, 1997 (unpublished)
Jug Bay Wetlands: HM	Patuxent River	1992	-0.086	0.006	-0.040	-0.010	0.061	0.025	Core incubation	Zeigler et al, Submitted
Jug Bay Wetlands: LM	Patuxent River	1992	0.002	0.062	0.027	-0.038	0.014	-0.013		
Jug Bay Wetlands: Mud	Patuxent River	1992	-0.004	0.070	0.033	-0.089	-0.005	-0.047		
Freshwater tidal marsh	Potomac River	1987	ND	ND	-0.057	ND	ND	0.001	Core incubation	Seitzinger, 1987
Jug Bay Wetlands: HM	Patuxent River	1995	0.000	0.111	0.038	-0.028	-0.001	-0.011	Core incubation: Batch	Groszkowski, 1995
Jug Bay Wetlands: LM	Patuxent River	1995	0.049	0.072	0.062	-0.018	0.000	-0.009	Core incubation: Batch	
Jug Bay Wetlands: Mud	Patuxent River	1995	0.026	0.121	0.074	0.000	0.000	0.000	Core incubation: Batch	
Jug Bay Wetlands: HM	Patuxent River	1995	0.081	0.366	0.150	-0.167	0.002	-0.078	Core incubation: Continuous	Groszkowski, 1995
Jug Bay Wetlands: LM	Patuxent River	1995	0.078	0.114	0.094	-0.162	0.000	-0.059	Core incubation: Continuous	
Jug Bay Wetlands: Mud	Patuxent River	1995	0.081	0.167	0.124	-0.171	-0.001	-0.065	Core incubation: Continuous	
Kenilworth Marsh: June - Sept.	Anacostia River	1995	0.017	0.164	0.089	-0.088	0.247	0.130	Whole marsh mass balance	Velinsky et al., unpublished

Notes: Sites with only averge indicates only single reported value. HM- High Marsh, LM - Low marsh, Mud - Mudflat

(-) = into marsh sediment or area

(+) = out of marsh sediment or area

Units: g N/m2-day for both nitrogen species

Table 4-17. Average ammonium-ammonia fluxes from tidal freshwater marshes in the upper Delaware estuary[1].

River Mile	River/Creek	High Marsh kg N	Low Marsh kg N	Mudflat kg N	Total kg N	Total Daily Rate kg N/RM-day
82	DE Riv-Lower			816	816	10.6
83	Cedar Swamp	1,357	225	2,330	3,912	50.8
84	DE Riv-Lower	853	246	3,552	4,651	60.4
85	DE Riv-Lower; Darby Creek	1,797	1,993	2,683	6,474	84.1
86	DE Riv-Lower		278	3,607	3,885	50.5
87	DE Riv-Lower		57	3,271	3,328	43.2
88	DE Riv-Lower			318	318	4.1
89	DE Riv-Lower; Mantua Creek	4,798	3,085	414	8,296	107.7
90	Little Mantua Creek, Main Ditch	568	657	1,218	2,443	31.7
91	DE Riv-Lower; Woodbury Creek	3,104	57	1,374	4,536	58.9
108	DE Riv-Upper			233	233	3.0
109	DE Riv-Upper		56	746	802	10.4
110	DE Riv-Upper	45	44	1,326	1,415	18.4
111	DE Riv-Upper; Rancocas Creek	10,459	3,931	1,684	16,074	208.7
112	DE Riv-Upper		174	282	456	5.9
113	DE Riv-Upper			988	988	12.8
114	DE Riv-Upper			35	35	0.5
115	DE Riv-Upper					
116	DE Riv-Upper			147	147	1.9
117	DE Riv-Upper			98	98	1.3
118	DE Riv-Upper; Assiscunk and Otter Creeks	1,193	252	26	1,471	19.1
119	DE Riv-Upper					
120	DE Riv-Upper			63	63	0.8
121	DE Riv-Upper			601	601	7.8
122	DE Riv-Upper			1,631	1,631	21.2
123	DE Riv-Upper			205	205	2.7
124	DE Riv-Upper			112	112	1.4
125	Crafts Creek		127	434	561	7.3
126	DE Riv-Upper	46	220	448	713	9.3
127	DE Riv-Upper		79	373	452	5.9
128	Blacks Creek, Crosswicks Creek	6,187	696	104	6,988	90.8
129	DE Riv-Upper; Duck Creek	422	489	398	1,309	17.0
130	DE Riv-Upper; Biles Creek			1,418	1,418	18.4
131	DE Riv-Upper			414	414	5.4
132	DE Riv-Upper			195	195	2.5
133	DE Riv-Upper					
	Total (Metric Tons)	**31**	**13**	**32**	**75**	

[1]Positive fluxes are directed out of the marsh sediment, while negative fluxes are directed into the marsh sediment. Rate assumes a 77-day active period in the wetlands.

Table 4-18. Average nitrate+nitrite fluxes from tidal freshwater marshes in the upper Delaware estuary[1].

River Mile	River/Creek	High Marsh kg N	Low Marsh kg N	Mudflat kg N	Total kg N	Total Daily Rate kg N/RM-day
82	DE Riv-Lower			-322	-322	-4.2
83	Cedar Swamp	-536	-89	-920	-1,544	-20.1
84	DE Riv-Lower	-337	-97	-1,402	-1,836	-23.8
85	DE Riv-Lower; Darby Creek	-709	-787	-1,059	-2,555	-33.2
86	DE Riv-Lower		-110	-1,424	-1,533	-19.9
87	DE Riv-Lower		-22	-1,291	-1,314	-17.1
88	DE Riv-Lower			-125	-125	-1.6
89	DE Riv-Lower; Mantua Creek	-1,894	-1,218	-163	-3,275	-42.5
90	Little Mantua Creek, Main Ditch	-224	-259	-481	-964	-12.5
91	DE Riv-Lower; Woodbury Creek	-1,225	-23	-542	-1,790	-23.3
108	DE Riv-Upper			-92	-92	-1.2
109	DE Riv-Upper		-22	-295	-317	-4.1
110	DE Riv-Upper	-18	-17	-524	-559	-7.3
111	DE Riv-Upper; Rancocas Creek	-4,128	-1,552	-665	-6,345	-82.4
112	DE Riv-Upper		-69	-111	-180	-2.3
113	DE Riv-Upper			-390	-390	-5.1
114	DE Riv-Upper			-14	-14	-0.2
115	DE Riv-Upper					
116	DE Riv-Upper			-58	-58	-0.8
117	DE Riv-Upper			-39	-39	-0.5
118	DE Riv-Upper; Assiscunk and Otter Creeks	-471	-99	-10	-581	-7.5
119	DE Riv-Upper					0.0
120	DE Riv-Upper			-25	-25	-0.3
121	DE Riv-Upper			-237	-237	-3.1
122	DE Riv-Upper			-644	-644	-8.4
123	DE Riv-Upper			-81	-81	-1.1
124	DE Riv-Upper			-44	-44	-0.6
125	Crafts Creek		-50	-171	-221	-2.9
126	DE Riv-Upper	-18	-87	-177	-281	-3.7
127	DE Riv-Upper		-31	-147	-179	-2.3
128	Blacks Creek, Crosswicks Creek	-2,442	-275	-41	-2,758	-35.8
129	DE Riv-Upper; Duck Creek	-167	-193	-157	-517	-6.7
130	DE Riv-Upper; Biles Creek			-560	-560	-7.3
131	DE Riv-Upper			-164	-164	-2.1
132	DE Riv-Upper			-77	-77	-1.0
133	DE Riv-Upper					
	Total (Metric Tons)	-12	-5	-12	-30	

[1]Positive fluxes are directed out of the marsh sediment, while negative fluxes are directed into the marsh sediment. Rate assumes a 77-day active period in the wetlands.

Table 4-19. Sediment-water exchange rates of ortho-phosphate and sediment oxygen demand for various tidal freshwater wetlands.

Marsh Area	Location	Year	Phosphate Low	High	Average	SOD Low	High	Average	Method	Reference
Phillips Creek Marsh	James River, VA	1992	-0.001	0.01	0.0013				Chamber method	Chambers, 1992
Rhodes River	Chesapeake Bay	1982							Hydrologic export model	Jordan et al. 1983
Gott's Marsh	Chesapeake Bay	1975								Heinle and Flemner, 1976
North River Marsh	Massachusectts	1985							Flume	Bowden, 1986
Tivoli Bay Marsh: North Bay: LM	Hudson River	1997				-2.634	-0.669	-1.343	Core incubation	Zelenke, 1997 (unpublished)
Tivoli Bay Marsh: North Bay: HM	Hudson River	1997		0.003		-1.582	-1.409		Core incubation	Zelenke, 1997 (unpublished)
Tivoli Bay Marsh: South Bay: LM	Hudson River	1997				-0.814	-0.745		Core incubation	Zelenke, 1997 (unpublished)
Jug Bay Wetlands: HM	Patuxent River	1992							Core incubation	Zeigler et al, Submitted
Jug Bay Wetlands: LM	Patuxent River	1992								
Jug Bay Wetlands: Mud	Patuxent River	1992								
Freshwater tidal marsh	Potomac River	1987							Core incubation	Seitzinger, 1987
Jug Bay Wetlands: HM	Patuxent River	1995	-0.002	0.050	0.020	-1.243	-0.348	-0.697	Core incubation: Batch	Groszkowski, 1995
Jug Bay Wetlands: LM	Patuxent River	1995	-0.002	0.000	0.000	-1.475	-0.575	-1.061	Core incubation: Batch	Groszkowski, 1995
Jug Bay Wetlands: Mud	Patuxent River	1995	0.000	0.002	0.001	-0.816	-0.401	-0.612	Core incubation: Batch	Groszkowski, 1995
Jug Bay Wetlands: HM	Patuxent River	1995	-0.002	0.239	0.055	-3.856	-2.358	-3.056	Core incubation: Continuous	Groszkowski, 1995
Jug Bay Wetlands: LM	Patuxent River	1995	-0.007	0.009	-0.001	-3.033	-1.083	-2.259	Core incubation: Continuous	Groszkowski, 1995
Jug Bay Wetlands: Mud	Patuxent River	1995	-0.005	0.007	0.000	-2.989	-1.187	-2.258	Core incubation: Continuous	Groszkowski, 1995
Kenilworth Marsh: June - Sept.	Anacostia River	1995	-0.008	0.006	-0.003				Whole marsh mass balance	Velinsky et al., unpublished

Notes: Sites with only averge indicates only single reported value. HM- High Marsh, LM - Low marsh, Mud - Mudflat

(-) = into marsh sediment or area

(+) = out of marsh sediment or area

Units: g P/m^2-day for phosphate and g O_2/m^2-day for sediment oxygen demand (SOD)

Table 4-20. Average ortho-phosphate fluxes from tidal freshwater marshes in the upper Delaware estuary[1].

River Mile	River/Creek	High Marsh kg P	Low Marsh kg P	Mudflat kg P	Total kg P	Total Daily Rate kg P/RM-day
82	DE Riv-Lower			185	185	2.4
83	Cedar Swamp	307	51	528	886	11.5
84	DE Riv-Lower	193	56	804	1,053	13.7
85	DE Riv-Lower; Darby Creek	407	451	607	1,466	19.0
86	DE Riv-Lower		63	817	880	11.4
87	DE Riv-Lower		13	741	753	9.8
88	DE Riv-Lower			72	72	0.9
89	DE Riv-Lower; Mantua Creek	1,086	698	94	1,878	24.4
90	Little Mantua Creek, Main Ditch	129	149	276	553	7.2
91	DE Riv-Lower; Woodbury Creek	703	13	311	1,027	13.3
108	DE Riv-Upper			53	53	0.7
109	DE Riv-Upper		13	169	182	2.4
110	DE Riv-Upper	10	10	300	320	4.2
111	DE Riv-Upper; Rancocas Creek	2,368	890	381	3,639	47.3
112	DE Riv-Upper		39	64	103	1.3
113	DE Riv-Upper			224	224	2.9
114	DE Riv-Upper			8	8	0.1
115	DE Riv-Upper					
116	DE Riv-Upper			33	33	0.4
117	DE Riv-Upper			22	22	0.3
118	DE Riv-Upper; Assiscunk and Otter Creeks	270	57	6	333	4.3
119	DE Riv-Upper					
120	DE Riv-Upper			14	14	0.2
121	DE Riv-Upper			136	136	1.8
122	DE Riv-Upper			369	369	4.8
123	DE Riv-Upper			46	46	0.6
124	DE Riv-Upper			25	25	0.3
125	Crafts Creek		29	98	127	1.6
126	DE Riv-Upper	10	50	101	161	2.1
127	DE Riv-Upper		18	85	102	1.3
128	Blacks Creek, Crosswicks Creek	1,401	158	24	1,582	20.5
129	DE Riv-Upper; Duck Creek	96	111	90	296	3.8
130	DE Riv-Upper; Biles Creek			321	321	4.2
131	DE Riv-Upper			94	94	1.2
132	DE Riv-Upper			44	44	0.6
133	DE Riv-Upper					
	Total (Metric Tons)	7.0	2.9	7.0	16.8	

[1]Positive fluxes are directed out of the marsh sediment, while negative fluxes are directed into the marsh sediment. Rate assumes a 77-day active period in the wetlands.

Table 4-21. Average sediment oxygen fluxes from tidal freshwater marshes in the upper Delaware estuary[1].

River Mile	River/Creek	High Marsh kg O_2	Low Marsh kg O_2	Mudflat kg O_2	Total kg O_2	Total Daily Rate kg O_2/RM-day
82	DE Riv-Lower			-32,872	-32,872	-427
83	Cedar Swamp	-54,693	-9,061	-93,922	-157,677	-2,048
84	DE Riv-Lower	-34,383	-9,916	-143,159	-187,459	-2,435
85	DE Riv-Lower; Darby Creek	-72,436	-80,348	-108,156	-260,940	-3,389
86	DE Riv-Lower		-11,208	-145,378	-156,586	-2,034
87	DE Riv-Lower		-2,283	-131,853	-134,136	-1,742
88	DE Riv-Lower			-12,806	-12,806	-166
89	DE Riv-Lower; Mantua Creek	-193,384	-124,329	-16,680	-334,392	-4,343
90	Little Mantua Creek, Main Ditch	-22,887	-26,483	-49,090	-98,461	-1,279
91	DE Riv-Lower; Woodbury Creek	-125,115	-2,315	-55,385	-182,815	-2,374
108	DE Riv-Upper			-9,398	-9,398	-122
109	DE Riv-Upper		-2,261	-30,073	-32,335	-420
110	DE Riv-Upper	-1,819	-1,777	-53,458	-57,054	-741
111	DE Riv-Upper; Rancocas Creek	-421,565	-158,452	-67,874	-647,891	-8,414
112	DE Riv-Upper		-7,003	-11,378	-18,382	-239
113	DE Riv-Upper			-39,809	-39,809	-517
114	DE Riv-Upper			-1,397	-1,397	-18
115	DE Riv-Upper					
116	DE Riv-Upper			-5,933	-5,933	-77
117	DE Riv-Upper			-3,958	-3,958	-51
118	DE Riv-Upper; Assiscunk and Otter Creeks	-48,079	-10,149	-1,055	-59,283	-770
119	DE Riv-Upper					
120	DE Riv-Upper			-2,549	-2,549	-33
121	DE Riv-Upper			-24,208	-24,208	-314
122	DE Riv-Upper			-65,756	-65,756	-854
123	DE Riv-Upper			-8,270	-8,270	-107
124	DE Riv-Upper			-4,497	-4,497	-58
125	Crafts Creek		-5,118	-17,483	-22,601	-294
126	DE Riv-Upper	-1,835	-8,859	-18,044	-28,737	-373
127	DE Riv-Upper		-3,182	-15,050	-18,232	-237
128	Blacks Creek, Crosswicks Creek	-249,390	-28,068	-4,206	-281,665	-3,658
129	DE Riv-Upper; Duck Creek	-17,027	-19,720	-16,031	-52,778	-685
130	DE Riv-Upper; Biles Creek			-57,156	-57,156	-742
131	DE Riv-Upper			-16,696	-16,696	-217
132	DE Riv-Upper			-7,863	-7,863	-102
133	DE Riv-Upper					
	Total (Metric Tons)	**-1,243**	**-511**	**-1,271**	**-3,025**	

[1]Positive fluxes are directed out of the marsh sediment, while negative fluxes are directed into the marsh sediment. Rate assumes a 77-day active period in the wetlands.

Table 4-22. Range of mass sedimentation rate for N and P in tidal freshwater wetlands.

		Rate (g N m^{-2} yr^{-1})			Rate (g P m^{-2} yr^{-1})				
Marsh Area	**Location**	**Low**	**High**	**Average**	**Low**	**High**	**Average**	**Method**	**Reference**
Tivoli Bay	Hudson River	-2.30	-15.90	-9.10	-0.69	-3.61	-2.15	^{210}PB	Zelenke, 1997
Jug Bay	Patuxent River	NR	NR	-23.40	NR	NR	-3.54	^{210}PB	Zelenke, 1997
Jug Bay	Patuxent River	-7.00	-25.00	-16.00	-0.40	-2.20	-1.30	Pollen	Khan and Brush, 1994
Barataria Basin	Mississippi River	NR	NR	-12.00	NR	NR	NR	^{137}Cs	Delaune et al., 1986

(-) rate s indicate burial from all sources.

Table 4-23. Burial estimate of nitrogen in tidal freshwater wetlands of the Delaware estuary.

River Mile	River/Creek	High Marsh kg N	Low Marsh kg N	Mudflat kg N	Total kg N	Total Daily Rate kg N/RM-day
82	DE Riv-Lower			-841	-841	-10.9
83	Cedar Swamp	-1,399	-232	-2,403	-4,034	-52.4
84	DE Riv-Lower	-880	-254	-3,663	-4,796	-62.3
85	DE Riv-Lower; Darby Creek	-1,853	-2,056	-2,767	-6,676	-86.7
86	DE Riv-Lower		-287	-3,719	-4,006	-52.0
87	DE Riv-Lower		-58	-3,373	-3,432	-44.6
88	DE Riv-Lower			-328	-328	-4.3
89	DE Riv-Lower; Mantua Creek	-4,948	-3,181	-427	-8,555	-111.1
90	Little Mantua Creek, Main Ditch	-586	-678	-1,256	-2,519	-32.7
91	DE Riv-Lower; Woodbury Creek	-3,201	-59	-1,417	-4,677	-60.7
108	DE Riv-Upper			-240	-240	-3.1
109	DE Riv-Upper		-58	-769	-827	-10.7
110	DE Riv-Upper	-47	-45	-1,368	-1,460	-19.0
111	DE Riv-Upper; Rancocas Creek	-10,786	-4,054	-1,737	-16,576	-215.3
112	DE Riv-Upper		-179	-291	-470	-6.1
113	DE Riv-Upper			-1,018	-1,018	-13.2
114	DE Riv-Upper			-36	-36	-0.5
115	DE Riv-Upper					
116	DE Riv-Upper			-152	-152	-2.0
117	DE Riv-Upper			-101	-101	-1.3
118	DE Riv-Upper; Assiscunk and Otter Creeks	-1,230	-260	-27	-1,517	-19.7
119	DE Riv-Upper					
120	DE Riv-Upper			-65	-65	-0.8
121	DE Riv-Upper			-619	-619	-8.0
122	DE Riv-Upper			-1,682	-1,682	-21.8
123	DE Riv-Upper			-212	-212	-2.7
124	DE Riv-Upper			-115	-115	-1.5
125	Crafts Creek		-131	-447	-578	-7.5
126	DE Riv-Upper	-47	-227	-462	-735	-9.5
127	DE Riv-Upper		-81	-385	-466	-6.1
128	Blacks Creek, Crosswicks Creek	-6,381	-718	-108	-7,206	-93.6
129	DE Riv-Upper; Duck Creek	-436	-505	-410	-1,350	-17.5
130	DE Riv-Upper; Biles Creek			-1,462	-1,462	-19.0
131	DE Riv-Upper			-427	-427	-5.5
132	DE Riv-Upper			-201	-201	-2.6
133	DE Riv-Upper					
	Total (Metric Tons)	**32**	**13**	**33**	**77**	

Rate assumes a 77-day active period in the wetlands. (-) rates indicates burial of N from all sources.

Table 4-24. Burial estimate of phosphorus in tidal freshwater wetlands of the Delaware estuary.

River Mile	River/Creek	High Marsh kg P	Low Marsh kg P	Mudflat kg P	Total kg P	Total Daily Rate kg P/RM-day
82	DE Riv-Lower			-130	-130	-1.7
83	Cedar Swamp	-216	-36	-370	-621	-8.1
84	DE Riv-Lower	-136	-39	-564	-739	-9.6
85	DE Riv-Lower; Darby Creek	-285	-317	-426	-1,028	-13.4
86	DE Riv-Lower		-44	-573	-617	-8.0
87	DE Riv-Lower		-9	-520	-529	-6.9
88	DE Riv-Lower			-50	-50	-0.7
89	DE Riv-Lower; Mantua Creek	-762	-490	-66	-1,318	-17.1
90	Little Mantua Creek, Main Ditch	-90	-104	-193	-388	-5.0
91	DE Riv-Lower; Woodbury Creek	-493	-9	-218	-721	-9.4
108	DE Riv-Upper			-37	-37	-0.5
109	DE Riv-Upper		-9	-119	-127	-1.7
110	DE Riv-Upper	-7	-7	-211	-225	-2.9
111	DE Riv-Upper; Rancocas Creek	-1,662	-625	-268	-2,554	-33.2
112	DE Riv-Upper		-28	-45	-72	-0.9
113	DE Riv-Upper			-157	-157	-2.0
114	DE Riv-Upper			-6	-6	-0.1
115	DE Riv-Upper					
116	DE Riv-Upper			-23	-23	-0.3
117	DE Riv-Upper			-16	-16	-0.2
118	DE Riv-Upper; Assiscunk and Otter Creeks	-189	-40	-4	-234	-3.0
119	DE Riv-Upper					
120	DE Riv-Upper			-10	-10	-0.1
121	DE Riv-Upper			-95	-95	-1.2
122	DE Riv-Upper			-259	-259	-3.4
123	DE Riv-Upper			-33	-33	-0.4
124	DE Riv-Upper			-18	-18	-0.2
125	Crafts Creek		-20	-69	-89	-1.2
126	DE Riv-Upper	-7	-35	-71	-113	-1.5
127	DE Riv-Upper		-13	-59	-72	-0.9
128	Blacks Creek, Crosswicks Creek	-983	-111	-17	-1,110	-14.4
129	DE Riv-Upper; Duck Creek	-67	-78	-63	-208	-2.7
130	DE Riv-Upper; Biles Creek			-225	-225	-2.9
131	DE Riv-Upper			-66	-66	-0.9
132	DE Riv-Upper			-31	-31	-0.4
133	DE Riv-Upper					
	Total (Metric Tons)	**4.9**	**2.0**	**5.0**	**11.9**	

Rate assumes a 77-day active period in the wetlands. (-) rates indicates burial of P from all sources.

Table 4-25. Oxygen production from high and low marsh areas in the tidal freshwater Delaware estuary.

River Mile	River/Creek	Total O_2 kg	Total O_2 kg/RM-day	Total O_2 kg/RM-day P/R = 1.4
82	DE Riv-Lower			
83	Cedar Swamp	220,200	1,295	389
84	DE Riv-Lower	164,078	965	290
85	DE Riv-Lower; Darby Creek	707,043	4,159	1248
86	DE Riv-Lower	68,123	401	120
87	DE Riv-Lower	13,878	82	24
88	DE Riv-Lower			
89	DE Riv-Lower; Mantua Creek	1,339,516	7,880	2364
90	Little Mantua Creek, Main Ditch	230,064	1,353	406
91	DE Riv-Lower; Woodbury Creek	391,813	2,305	691
108	DE Riv-Upper			
109	DE Riv-Upper	4,413	26	8
110	DE Riv-Upper	9,143	54	16
111	DE Riv-Upper; Rancocas Creek	1,624,228	9,554	2866
112	DE Riv-Upper	13,666	80	24
113	DE Riv-Upper			
114	DE Riv-Upper			
115	DE Riv-Upper			
116	DE Riv-Upper			
117	DE Riv-Upper			
118	DE Riv-Upper; Assiscunk and Otter Creeks	169,783	999	300
119	DE Riv-Upper			
120	DE Riv-Upper			
121	DE Riv-Upper			
122	DE Riv-Upper			
123	DE Riv-Upper			
124	DE Riv-Upper			
125	Crafts Creek			
126	DE Riv-Upper	23,010	135	41
127	DE Riv-Upper	6,209	37	11
128	Blacks Creek, Crosswicks Creek	832,716	4,898	1469
129	DE Riv-Upper; Duck Creek	91,597	539	162
130	DE Riv-Upper; Biles Creek			
131	DE Riv-Upper			
132	DE Riv-Upper			
133	DE Riv-Upper			

[1]Rate assumes a linear growing period from March 1 to August 18 (170 days) for both low and high marsh areas. Photosynthesis/Respiration ratio (P/R) is assumed to provide range of possible estimates.

Table 4-26. Average Summertime Fluxes of N and P from various biogeochemical processes in tidal freshwater wetlands.

Process	Nitrogen	Phosphorus
Plant Uptake[1]	- 480	- 80
Sediment-Water Exchange[2]:		
NO_3^- ----> N_2	% 430	NA[3]
$NH_4^+ + NH_3$	% 980	NA
$NO_3^- + NO_2^-$	- 390	NA
o-PO_4^{3-}	NA	% 220
Burial	- 1000	- 160

Fluxes are in **kg/day** and cover the summer time period.
[1]Plant estimates are for aboveground biomass only.
[2](%) - efflux into water from sediment; (-) - uptake into marsh plant or sediment.
[3]NA - Not applicable

Table 4-27. Inorganic N and P Loads to the Tidal Freshwater Zone of the Delaware Estuary.

Source	Nitrogen	Phosphorus
Delaware River @ Trenton	10,400	910
Tributaries (n = 8)	10,900	670
Major WWTP (n = 7)	34,000	630
Minor Dischargers	2,900	ND
Combined Sewer Overflow	170	ND
Atmospheric Deposition	10	ND

Inputs are in **kg/day** for average inputs in 1995.
Data provided by K. Mooney (HydroQual, Inc.). Atmospheric deposition is for annual wet deposition (Scudlark and Church, 1993) only over the area of the tidal freshwater wetlands. ND - No data

5.0 Future Evaluation Regarding Emergent and Submergent Aquatic Vegetation

Tidal marshes, such as those that fringe the Delaware River and Bay, function as important filters for many types of natural and anthropogenic materials. They can improve water quality by removing substances that are harmful to the environment and by producing substances that are beneficial to the ecosystem. Depending on physical, biological and geochemical factors, these wetlands can remove large quantities river-borne pollutants and excess nutrients (see above references), as evidenced by the rudimentary results of this study. Tidal marshes can also enhance organic matter degradation through microbial action within the sediments and overlying waters. Due to the tremendous amount of primary production that also takes place in these systems, tidal marshes are sources of food for organisms living both in, adjacent to, and downstream of the wetlands. Tidal marshes have many other benefits as well, such as providing crucial feeding, nursery, or refuge habitats for both resident and transitory waterfowl and fish.

Tidal marshes in the Delaware Estuary are increasingly imperiled by the impacts of encroaching development. Numerous urban centers have been constructed on filled lands that were formerly tidal wetlands, particularly in the freshwater tidal portion of the estuary that ranges from just above Wilmington, DE to Trenton, NJ. These urban landscapes impinge on the vestigial wetlands directly such as through physical alteration and development, and indirectly via widespread and diverse chemical inputs. The cumulative effects of habitat loss and increased pollution are not well understood in the Delaware Estuary, but are certain to affect the functional value of these wetlands, such as their ability to remove excess nutrients and contaminants. Fortunately, in parts of Delaware Bay (e.g., lower more saline sections), efforts are accelerating to reverse losses of wetland acreage. Salt marshes and tidal freshwater marshes have many similar functions, and the proximate nature of the tidal freshwater marshes to sources of pollution means that they may abate water quality deterioration more directly than those located in the lower Bay. Both natural and restored salt and freshwater marshes are all likely to contribute to the improvement of estuarine water quality, and hence, they must all be included in any Bay-wide assessment of the “filtration” role of wetlands.

Of special interest to the Delaware River Basin Commission is the effect these tidal marshes have on nutrient and oxygen concentrations throughout the estuary. Preliminary evidence presented in the present study strongly indicates that the freshwater tidal marshes within the Trenton-Philadelphia-Wilmington metro region can play a sizeable role in controlling the amount and flux of excess nutrients and possibly dissolved oxygen. There are numerous ways to develop better scientific measures of these functional processes for the Delaware Estuary at large. Below we list a series of recommendations, which would each substantially strengthen and improve our understanding of the functional ecology of these marshes.

1. <u>Improved measures of tidal freshwater EAV biomass and production</u>

In the present study, we collected only living aboveground biomass of emergent aquatic vegetation (EAV) from typical low and high marsh areas. Typical marshes were considered to be a low marsh dominated by *Nuphar* sp. and a high marsh dominated by a diverse assemblage of wild rice (*Zizania aquatica*), arrow arum, smartweed, etc. However, we observed two problems with this approach. First, in certain locations, discrete mono-specific patches of cattails (*Typha*) and common reed (*Phragmites*) were extensive, but were not considered in our estimates of EAV biomass in this preliminary study. Cattail stands contained more than twice the biomass of our mixed species, high marsh community. Literature reports suggest that *Phragmites*, which is more abundant in the lower section of the tidal freshwater estuary, can also contain much greater biomass, perhaps four- to five-fold greater than we measured. We strongly recommend that future EVA studies be expanded to separately quantify biomass, production and nitrogen-phosphorus content of these tall grasses.

Secondly, in the low marsh habitat, EAV biomass and speciation did not follow a clear relationship with elevation, as observed in previous studies. While our sample size was small, we suggest that in the future these measurements be repeated over a broader and more extensive area to determine the robustness of this relationship between biomass and elevation. If it is found to be common phenomenon, then more attention will need to be given to site-specific measurements of typical marsh slopes.

Lastly, to make accurate estimates of emergent plant productivity and its impact on nutrient cycles, estimates need to be made on belowground biomass and nitrogen and phosphorus content. In some cases the belowground biomass can be equal or greater than aboveground biomass and therefore a large reservoir of bound material. Also, both pools of biomass need to be measured at different times during the growing season to obtain better estimates of plant biomass, productivity and turnover rates.

2. Direct measures of nutrient cycling, renewal, and oxygen production by tidal freshwater EAV.

The results of the present study were based on studies from other locations with numerous crucial assumptions. Rather than directly measuring rates of uptake of nitrogen (N) and phosphorus (P) by aboveground EAV that live in the study wetlands, we estimated annual N and P removal simply by looking at the amount bound in aboveground EAV biomass at a single point in time. This might grossly underestimate the role of EAV in N and P removal especially in regards to belowground biomass and plant turnover, and more direct measures are needed. Similarly, for estimating oxygen production by EAV, we compared our crude estimates of EAV production to a standard oxycaloric conversion factor found in the literature; however, such conversion factors can be notoriously variable among higher plants. It was apparent from the literature that emergent plants such as *Nuphar* may not directly impact the water column in regard to oxygen inputs to the water column. This is especially true in the summer time when anaerobic conditions in the sediment may consume excess oxygen than can leak out of the plant through the root, rhizosome system. On the other hand, epiphytic and benthic algae and submerged aquatic vegetation may be important sources of dissolved oxygen to the water column. The impact from epiphytic and benthic algae and SAVs were largely overlooked in this study due to a lack of data. For a better measure of oxygen production within the water column, we recommend direct measurements be made in future research. These studies would entail core and chamber studies to directly measured nutrient and oxygen dynamics in various parts of the marsh (i.e., high, mid, and low sections). In addition, direct measurement of denitrification rates are needed to better determine the magnitude of this loss mechanism in the tidal wetland. Lastly, sediment cores should be taken in various depositional zones of the upper estuary to estimate the actual long-term burial and temporal variations of nitrogen and phosphorus removal and accumulation. All these measures of nitrogen, phosphorus, and oxygen dynamics can be a part of an integrated study of representative wetlands in the tidal river (see below).

3. Expand study beyond EAV to include other primary producers

Numerous other types of primary producers besides EAV are present throughout the estuary and a growing body of evidence suggests that these autotrophs can be much more productive (if not high in biomass) than EAV. These plants include submerged aquatic vegetation (SAV), epiphytic algae, and benthic microalgae and cyanobacteria (i.e., microphytobenthos, MPB). The extent of these plants in the upper Delaware Estuary is largely unknown. Personal observations made in 1997 suggest that there are substantial SAV beds and MPB mats in many tidal freshwater river areas. For example, we observed extensive beds of SAV throughout the Rancocas River in August (1997). Also, in mid-October (1997) after the lower marsh was cleared of *Nuphar* due to dieback and tidal rafting, there were extensive mats of green and possibly blue-green algae on the sediment surface. Therefore, studies should be conducted to determine the extent and diversity of these important aquatic producers. These studies would be labor intensive as remote sensing may be of limited use due to the high turbidity in the water and presence of EAV. In addition, research should be conducted to determine the net exchange of nutrients and oxygen from these plants. Specific studies include light and dark chamber experiments to determine the gross and net flux of nutrients and oxygen. Productivity measurements should also be made to determine how much nitrogen and phosphorus is stored within the plant biomass over the year and how production and respiration affects oxygen concentrations in the water column. This information and nutrient fluxes could be applied to the tidal freshwater portion of the river, and perhaps also be extended throughout the entire estuary.

4. Expand this study of the freshwater tidal wetlands

Our reconnaissance study has suggested that about 25% of what were high marsh wetlands in our study area in 1977 have now become low marsh. This finding needs first to be verified by analysis of current aerial photography or remote sensing data, and second the study should be extended to the rest of the wetlands on the Delaware River. If sediment deposition in some of these wetlands is insufficient to maintain balance with relative sea level rise so that there will continue to be a vegetation change, we need to determine the magnitude of this change and assess its impact on water quality. To accomplish this requires additional careful examination of the available 1977 photography and acquisition of present-day photographic or digital observations. This study would properly be part of the larger consideration of the effect of the rise of relative sea level in the estuary and river and its manifestation in the migration of wetland vegetation zones. Understanding and documenting recent change will be critical to projecting the effect of relative sea level rise on these wetlands and their function.

This reconnaissance mapping of high and low freshwater tidal wetlands should be extended using current photography or digital imagery to all of the Delaware River from circa the Christiana and Salem rivers upstream to Trenton. This is needed to document the present extent and condition of the wetland vegetation prior to any deepening of the river channel by the Corps of Engineers.

As noted in Recommendation 1, there is a need for a comprehensive map of *Phragmites* and *Typha* in the entire estuary. *Phragmites* is, of course, well established in extensive, robust stands in the brackish tidal reaches of the river. In addition, in freshwater tidal marshes there are many locations where *Phragmites* stands appear to exist in patches of stable extent. In contrast, there are many locations in the marshes of the lower estuary where *Phragmites* expansion might best be described as an infestation. Remote sensing can be employed to locate and map stands of *Typha* and *Phragmites* as an aid in identifying present and potential *Phragmites* sites. Multi-spectral imagery of 30-m pixel size and panchromatic imagery of 5-m pixel size can be combined to provide better spatial resolution than can be achieved with 30-m Multi-spectral imagery alone. Even higher spectral and spatial resolution can be attained with airborne hyper spectral scanner imagery which can produce pixel sizes as needed between 1 and 4 m. This imagery also can now be obtained and would be capable of discerning these vegetation stands.

The biomass of many canopies can be quantitatively accessed by satellite or airborne remote sensing data after calibration with some surface optical observations. Biomass harvest sampling at a few locations combined with optical observations of leaf area using a LiCor canopy analyzer provide a means for rapid determination of canopy biomass. Surface observations of the reflectance spectra of these canopies with a portable spectro-radiometer provide the information needed to interpret remotely observed spectra in terms of species type and biomass quantity.

5. <u>Common methodologies for remote sensing of wetlands</u>

There is need for inventory mapping of the high and low marsh tidal wetlands of the entire estuary according to a common methodology, such as that used in this study, to provide a base from which to measure future change. See Item 4 above. The recently completed classification of land cover and land cover change on the Delaware Estuary for the years 1984 and 1993 from Thematic Mapper satellite data by The Center for Remote Sensing of the University of Delaware College of Marine Studies could provide the basis for such an effort. Pilot studies have been conducted within tidal salt marshes in the Delaware Estuary to assess water quality functions (Hardisky et al., 1983). The Marsh Ecology Research Program, Delaware Estuary Program, and The Delaware Estuary Enhancement programs are developing extensive data bases on tidal salt marshes, some of which may contribute to understanding the role of tidal marshes in the estuary.

6. <u>Role of secondary producers on nutrients and dissolved oxygen</u>

Studies need to examine non-producer roles in nutrient removal and oxygen production (e.g. denitrification, food web losses other than to decomposers such as export via fish, shrimp). Tidal [salt] marshes are composed of a complex food web that is fueled by high rates of primary production, but that has been classically described as being "detritus-based." This is not as apparent or well studied for tidal freshwater marshes. This terminology is warranted because the vast majority of EAV biomass is not directly grazed by metazoan consumers, but is instead slowly decomposed after senescence by microbial organisms in the sediments and water column. Many of these decomposers then serve as food or prey for metazoans, perhaps being exported from the system through food chain dynamics. In addition, bacteria can be responsible for considerable N removal through denitrification in the sediments. In contrast to the positive role of many decomposers and metazoan consumers in removing nutrients, however, a significant portion of producer-supplied oxygen might get consumed by these organisms, perhaps abating net oxygen transfer from these wetlands to adjacent open waters. Further study is needed to determine how these other biota modify the EAV- mediated functional processes of nutrient removal and oxygen production. Furthermore, the traditional view of marsh food webs as detritus-based is being questioned since total annual production by non-EAV marsh producers could be greater than that by EAV, and non-EAV production is consumed more readily by metazoans through direct (e.g., non-detrital) microbial pathways.

7. Integrated assessments of marsh systems.

To quantify the net import/export of nutrients and oxygen fluxes in/out of experimental marshes an integrated approach is needed. The most comprehensive way to examine the role of these tidal wetlands in removing nutrients and supplying oxygen is to summarily measure the flux of nutrients (and oxygen) entering and leaving the wetland with each tide and modeling the net exchange. To be robust, this measure would need to be completed *in situ* at replicate sites having similar hydrology, but being characteristic for tidal marshes in general. Furthermore, it will be critical to repeat measurements seasonally since rates of primary and secondary production in the marshes vary greatly through the year. The approach should also be repeated over multiple lunar cycles to determine whether the net flux depends on tidal height. Finally, in an integrated assessment of the Delaware Estuary at large, such flux measurements could vary spatially and must therefore be completed at various points from the mouth of the Bay to the head of tide along both along eastern and western shorelines.

By combining remote sensing data with ground-truthing of the biomass and productivity of EAV and MPB in these experimental marshes, we will be able to statistically examine correlations between marsh water volume, tidal residence time, producer type/species, producer biomass, producer productivity with regard to nutrient removal and oxygen production. Such relationships would enable a rigorous assessment of the role of these wetlands in "filtering" the Bay's water.

As an example, a tidal freshwater marsh with a single channel or entrance to the marsh, such as the well studied Woodbury Creek Marsh, can be used as an experimental system to determine the biogeochemical impact of freshwater tidal wetlands in the urbanized section of the tidal Delaware River. This can be accomplished by monitoring the incoming and outgoing waters from this to system to determine the fluxes from the whole system and its effect on water quality. Specific short-term process measurements within the wetland during this study such as plant uptake rates and oxygen production/consumption, nitrification/denitrification rates, and benthic algal productivity would be needed to assess the results from the tidal inlet study.

Previous studies, while yielding important information, were done over a short period of time. Long-term measurements over a number of years and seasons can be made to assess nutrient, oxygen and contaminant fluxes, emergent plant productivity, benthic edaphic and epiphytic algal production and impacts, and overall material balances. Information from these studies can then be applied to the entire TFW area to determine how these systems filter and modify the flux of material down the Delaware River.

6.0 Summary and Conclusions

We assessed the extent of tidal freshwater wetlands in the upper tidal Delaware River. To accomplish this task we used historical aerial photography, current aerial photography, and ground truth information. We identified a total of 1,412 ha of tidal freshwater wetlands in the lower and upper study areas with slightly more area in the upper estuarine zone (56% of total). Mudflat estimates, while not a major focus of this research, were taken from NWI estimates. Although the upper study area has a higher total area of marsh, on a per river-mile basis the lower study area (10 river-miles) has nearly 62 ha of marsh per river-mile while the upper study area (26 river-miles) has only 30 ha of marsh per river-mile. In both study areas, we found much more high than low marsh habitat (65% and 75% for the lower and upper study areas, respectively). The majority of the marsh areas was located in a small number of tributaries, with very little marsh along the mainstem of the Delaware River. The largest marsh areas were found in Mantua Creek (256 ha), Darby Creek (120 ha), Woodbury Creek (103 ha), Rancocas Creek (466 ha), and Crosswicks Creek (206 ha). However, there does appear to be a significant number of polygons and areas which were dominated by high marsh vegetation which are dominated by low marsh vegetation today.

Our estimates are based on historical data with verification using an aerial overflight and ground-based assessments. In general, every polygon in the sample which we identified as marsh or mud flat in the 1977/78 aerial photography was found still to be marsh or mud flat in 1997. In addition, excepting the created wetland on Duck Island, we did not observe any new marsh areas in 1997 that had not been classified as marsh from the 1977/78 imagery. However, there were some differences between our classification and our assessment of current marsh conditions. It is not always possible to say whether these observed differences are due to the remote sensing techniques (i.e., vertical and oblique aerial photography), poor judgement, or a reflection of change in marsh types over time. Interpretation of the 35-mm and aerial video photography appears to have given consistent results. In this regard, the agreement for the five sample polygons which were identified and separately classified from video and 35-mm slides was 100%.

Twenty-six sample polygons were identified and classified in 1997 either from ground or aerial reconnaissance. In 16 cases the 1997 classification agreed with the photointepretor's classification of the 1977/78 imagery. Of the 10 cases of disagreement, 7 polygons classified as high marsh were found to be low marsh. In one case, a polygon classified as mud flat in the 1977 imagery was found to be low marsh in 1997. An additional sample polygon appeared to be low marsh in 1977/78, currently appears to be high marsh. Finally, one polygon showed disagreement between the 1997 aerial and ground classifications. Most of the disagreement between the photointepretation of the 1977/78 aerial photography and our 1997 verification sampling involved polygons in which the vegetation in 1997 appeared to consist of mixed high and low marsh. In some cases, the identified polygon could be seen from the air or ground to consist of sub-polygons of both high and low marsh. In other cases the vegetation types were intermingled at a fine scale. Of those cases where high marsh vegetation appears to have changed to low marsh, two examples on Mantua Creek appear to be examples of vegetation change from high to low marsh. For these last two, it is reasonable to suppose the change to have been from dominant rice stand to a mixed rice, which constitutes a shift from high to low marsh in our classification scheme.

We re-examined the 1977/78 aerial photography to check our reading of the spectral signatures of sample polygon numbers 6, 43, 44, 106, and 189, which show change between 1977/78 and 1997. In all cases, the original reading was confirmed by this examination. The spectral signature of polygon 6 on the Rancocas was of mixed vegetation, 60% high marsh and 40% low marsh. Polygons 43 and 44 appeared unambiguously to be high marsh with pockets of low marsh. The spectral signature was of tall rather patchy vegetation with mud showing through the canopy. Number 106 was confirmed to be high marsh also with pockets of low marsh. Polygon number 189 was confirmed to have the spectral signature of low marsh, although high marsh vegetation could be seen to dominate the periphery through which we had viewed the polygon in our ground reconnaissance. Consequently, our vantage point on the ground appears to have lead us to an incorrect classification in 1997. These results strengthen our confidence in the consistency of our interpretation of the 1977/78 photography.

We believe that the trend of the changes in polygon numbers 6, 43, 44, and 106 from high marsh to low marsh indicate real changes, not errors in our photointerpretation. Were this not the case, one would expect about as many polygons reclassified from low marsh to high marsh as were reclassified from high marsh to low marsh, and this was not observed. There were 10 polygons for which our air or ground observations in 1997 differed from our 1977 classification. Seven of these were changes from high to low marsh. We interpret this apparent trend from high to low marsh as being consistent with and supporting the findings of Philipp (1995) that "[during the last 100 years,] high fresh and salt water marshes have been transformed to lower marshes. In freshwater marshes this corresponds to a change from high freshwater marsh [vegetation] of *Polygonum*, *Bidens*, and *Acnida* to lower [marsh vegetation of] *Nuphar*, *Peltandra*, and *Zizania* ...". In addition, Lyles et al. (1988) reported that relative sea level has been rising at a rate 0.27 mm/yr for the last 50 years, which would produce a rise of about 60 mm between 1977 and 1997. If these vegetation trends can be substantiated it implies that relative sea water level is rising faster than the rate of marsh accretion in these areas. Further study is needed to substantiate this possibility and to verify these changes.

Numerous quadrants in the upper and lower emergent aquatic vegetation areas (EAVA) were sampled in August of 1997 for this study. Replicate, randomly selected, elevation transects at both Rancocas and Mantua Creeks indicated that considerable variability exists along the shores of these tributary creeks in the elevation profile of the creek banks. The broad-leaved *Nuphar advena* (=*luteum*, =*variegatum*) occurred in uniform stands along the edge of the channel and in ponded and pond-like areas (i.e., low marsh, Type 1 habitat). These areas typically contained at least 95% of the biomass and percent cover in the form of *N. advena*, although both arrow arum (*Peltandra virginica*) and pickerelweed (*Pontederia cordata*) also contributed to the biomass at somewhat less depth of inundation. From low to high marsh (i.e., Type 2), the second prominent EAV type was wild rice (*Zizania aquatica*), which typically formed a narrow to wide band of tall grass just landward from the Type 1 low marsh. Wild rice was widespread at both upstream and downstream locations. Near the low marsh/high marsh boundary, stands of wild rice often intermixed with the succulent, low marsh species. Nonetheless, within the Type 2 area, wild rice composed 66-100% of the total dry EAV biomass, and became nearly monospecific in the bulk of its range. High marsh habitats contained much greater EAV species diversity, but certain species formed patches where they dominated EAV biomass. Cattail *(Typha angustifolia*) and common reed (*Phragmites australis*) also formed nearly monospecific stands having tremendous biomass. These tall plants were very patchy in occurrence, however, often being found only on high marsh levees. We measured the actual EAV biomass within a cattail patch (Type 3) at both the upstream and downstream sites. No measurements of *P. australis* biomass were made. Numerous other species were found during this study, and their spatial distribution in the tidal marsh was nearly always associated with specific flooding conditions for which they are specially adapted.

Based on these flooding characteristics, high marsh vegetation will typically be underwater for up to 4 h per 12-h tidal cycle. The depth of flooding during this time may typically reach 0.3 m. Conversely, the vegetation typically will be out of the water for 2/3 of that tidal cycle. The various low marsh habitat types will be underwater from 3/4 to all of a 12-h tidal cycle up to a maximum depth of 1 or even 2 m. Low marsh vegetated with *N. advena* and *P. virginica* is predominantly within the stream bank to pond-like hydrologic regime, and is drained at normal low tide.

EAV biomass was high in all zones, as might be expected during August at the late peak of the growing season. The low marsh areas dominated by *Nuphar advena* (Type 1) contained 241 g dry weight m^{-2} and 811 g dry weight m^{-2} at Rancocas and Mantua Creeks, respectively (Table 4-5). Although Type 1 EAV biomass at Mantua Creek appeared higher than at Rancocas Creek, these means were not significantly (t-test, $p>0.05$) different from each other because of high variability among replicates at Mantua. The zone of EAV dominated by wild rice (*Zizania aquatica*) contained remarkably consistent biomass at both sites. At Rancocas Creek, the mean biomass in the wild rice zone was 323 g dry weight m^{-2}, and 382 g dry weight m^{-2} was measured at Mantua Creek. These values were not significantly (t-test, $p>0.05$) different between sites. EAV biomass was found to be greatest in high marsh patches of cattails (*Typha angustifolia*) (Type 3).

For the purposes of this study, we sought to establish estimates of EAV biomass in either the low or high marsh because these two zones were most easily identified and quantified by remote sensing (see above). Statistical procedures were therefore needed to determine whether EAV biomass was similar among Types 2, 3 and 4 to permit derivation of a single, characteristic value for high marsh EAV biomass. To estimate the geographical extent of EAV biomass within the designated upstream and downstream river segments of the Delaware Estuary, we sought to calculate a universally applicable average or median EAV biomass for either the low marsh or high marsh, since the remote sensing data were limited to that level of sensitivity.   Since only one vegetation type dominated the low marsh zone at Rancocas and Mantua Creeks, we will simply use the median values measured for the Type 1 habitat in either Rancocas (i.e., represents upstream low marsh EAV biomass) or Mantua (i.e., represents downstream low marsh EAV biomass) Creek. Hence, the median low marsh EAV biomass at the upstream segment (e.g., Rancocas Creek) and downstream segment (e.g., Mantua Creek) was calculated to be 232 and 731 g dry weight m^{-2}, respectively.

To calculate a similar figure for the high marsh, we pooled replicate data for EAV Types 2 and 4 and disregarded data for EAV Type 3 (see above). Therefore, our estimates for the median EAV biomass in the high marsh of the upstream (e.g., Rancocas Creek) and downstream (e.g., Mantua Creek) reaches were 402 and 327 g dry weight m^{-2}, respectively, and it is important to recognize that these figures are conservative estimates since high biomass patches of *Typha* (and *Phragmites*) were not considered.

Overall, our ground surveys of EAV biomass and productivity for each study site produced values that were either consistent with or lower than comparable literature reports. Due to the consistency of the values reported in earlier studies, particularly for the high marsh, we must consider our productivity values to be conservative (i.e., low, underestimated by up to 2- to 3-fold). This could be explained by either or both of the aforementioned sampling problems: 1) we had to assume that no annual EAV production occurred beyond what was measurable as standing biomass in late August (produces an underestimate), and 2) species of EAV known to have much higher biomass and productivity (e.g., cattail and common reed) were not considered herein due to their patchy nature and would underestimate the total biomass.

To determine the effect of EAV on nutrient balances, estimated areas of high marsh, low marsh and mudflat areas, in the tidal freshwater sections of the Delaware Estuary were integrated with: 1) estimates of the median biomass of emergent aquatic vegetation, 2) average biomass elemental concentrations, and 3) the amount and average uptake rate of nitrogen, carbon and phosphorus were determined for the tidal freshwater areas of the upper Delaware Estuary. Aboveground N, C, and P biomass incorporation rates (i.e., kg/river-mile-day) ranged from approximately -0.5 to -131 kg N/RM-day for N, -10 to -3,600 kg C/RM-day for C, and -0.1 to -19 kg P/RM-day for P. These rates are for the time period between 1 July and 15 September. The highest amount of plant biomass, NC and P , and highest rates of production were recorded in Darby (RM 85), Mantua (RM 89), Rancocas (RM 111), and Crosswicks creeks (RM 128). Approximately, similar amounts of N and P in the aboveground biomass were calculated in the upper and lower EAV areas. From our calculations, an estimated 80 metric tons of N (1000 kg = 1 metric ton) and 13 metric tons of P are retained in the living aboveground biomass in the two tidal freshwater wetland areas of the Delaware Estuary in 1997. There is a substantial amount of spatial variation in these results which is most likely due to the amount of high/low marsh area in each zone. Our overall estimate for these elements do not take into account within season turnover of plant biomass by either leaf mortality or herbivory. This can be a significant portion of the annual productivity and produces and underestimate. An additional bias is the lack of belowground biomass storage which was not sampled and could also cause an underestimate.

Estimates for sediment-water exchange processes were made based on literature exchange rates and wetland areas from the current study. Estimates for sediment-water exchange were made for denitrification, ammonium+ammonia, nitrate+nitrite, ortho-phosphate, and dissolved oxygen (i.e., sediment oxygen demand). While there was substantial variations in the literature data, the results indicate that an active exchange of material occurs between the sediments and water column in the Delaware tidal freshwater wetlands. For nitrogen, on average for the entire area and time period, 30 metric tons of nitrate+nitrite-N moved into the sediments while slightly more nitrogen (33 metric tons) was removed from the sediments (and system) via denitrification (i.e., $NO_3^- \div N_{2\,(g)}$). A portion of the nitrate-nitrogen needed for denitrification can be supplied from coupled nitrification-denitrification within the sediments. Also, on average the wetland systems appear to be a large source of dissolved ammonium+ammonia (75 metric tons) to the overlying water, while 77 metric tons of nitrogen was potentially-buried in the wetland areas. On average, approximately 17 metric tons of ortho-phosphate was removed from the wetland sediments to the overlying water while approximately 12 metric tons of phosphorus is permanently buried within the sediments. There was substantial variability in these estimates. Sediment oxygen demand estimates also were variable with approximately 3,000 metric tons of oxygen moving into the sediments from the overlying water.

Another objective of this project was to estimate the effect that emergent aquatic plants within the two study areas have on the oxygen concentration within the water column. Within the scope of this study, we utilized the carbon incorporation rate and production of oxygen via photosynthesis (1 g C per 2.67 g O_2). The amount of oxygen produced per river-mile per day was calculated using a 170-day growing season. Total gross oxygen production ranged from 26 kg O_2/RM-day at river-mile 109 to approximately 9,500 kg O_2/RM-day at river-mile 111. As with the nitrogen, carbon and phosphorus calculations, the higher rates were calculated in sections of the river with highest areas of high and low marsh area (e.g., Rancocas Creek, Darby Creek, and Mantua Creek). It is unclear as to how much of this oxygen actually gets into the water column. Plant respiration could account for a substantial percentage of the oxygen produced, while the flux of oxygen out of the plants through the roots could be consumed by sediment microbial processes. This could be especially important in the warmer summer months when microbial processes are maximal and sediment oxygen demand highest. Therefore, little oxygen may be released from emergent plants into the overlying water.

Our evaluation of information concerning the area of submerged aquatic vegetation (SAV) in the study areas was hampered by the paucity of available data and estimates for nutrient uptake, biomass, or oxygen dynamics. Hence, we were unable to provide estimates for these processes. Literature information, however, does suggest that SAVs can have a substantial impact on water quality and sediment dynamics. As with emergent plants, SAV can oxygenate the sediments affecting nitrification-denitrification dynamics. Additionally, nutrient uptake from both the roots and shoots can have an impact on concentrations within the water column. Future studies should be directed towards determining the extent of SAVs in the tidal freshwater areas, and calculating area-specific uptake rates.

Our results quantified the extent of emergent aquatic vegetation and wetland areas within the tidal freshwater section of the Delaware Estuary. As expected, there is active movement of bioactive elements (e.g., nitrogen, phosphorus, etc) throughout and within the tidal freshwater areas. The comparison between the rates derived from this study, and estimated inputs to the tidal freshwater portion of the Delaware suggests that removal of nitrogen via plant uptake, denitrification, and burial are minor, while on average, approximately 10% of the watershed sources of phosphorus can be buried within the marsh system. Although the current area of tidal freshwater wetlands is greatly reduced compared with their historical extent, the remaining wetlands may have an impact on the chemical form of nitrogen and phosphorus (e.g., oxidized forms versus organic forms) to the more saline portion of the estuary and the eventually the coastal ocean. Compared to salt marshes in the lower estuary, tidal freshwater wetlands are some of the first aquatic habitats to be affected by runoff from the upper Delaware Basin watershed and large urban areas, and hence they could have a greater per-area impact on estuarine water quality. However, to fully evaluate these relationships numerous unknowns need to be addressed. We need improved estimates of wetland areas, understanding the roles of benthic and epiphytic algae, as well as the importance of secondary producers on oxygen and nutrient cycles. The present study relied heavily on published data from other locales, and we recommend site-specific rate data be collected in an integrated manner to enable the complex biogeochemical relationships to be comprehensively modeled for the entire estuary.

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