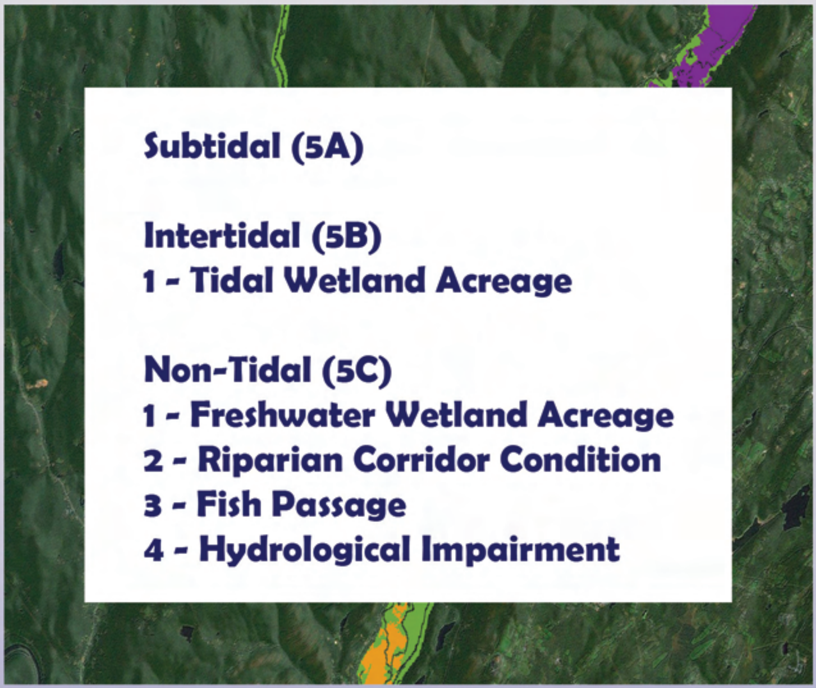
5

Aquatic Habitats



Subtidal (5A)

Intertidal (5B)
1 - Tidal Wetland Acreage

Non-Tidal (5C)
1 - Freshwater Wetland Acreage
2 - Riparian Corridor Condition
3 - Fish Passage
4 - Hydrological Impairment

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5 – Subtidal Aquatic Habitats

Introduction

While surveys of the benthos have occurred in the Delaware Bay and River since the 1950's (Table 5.2) the recent Delaware Estuary Benthic Inventory (DEBI) is the most comprehensive and intensive ever conducted. Due to the extent of the data produced in the DEBI project, it is the focus, though not exclusively, of this indicator.

The DEBI project was lead by The Partnership for the Delaware Estuary, one of twenty-eight National Estuary Programs. In 2005, The Partnership for the Delaware Estuary recognized a fundamental need for a benthic ecosystem assessment that would inventory the physical and biological conditions of the bottom of the open water tidal system of the Delaware River and Bay. This priority need was articulated in early 2005 when the Partnership convened a science and management conference that brought together more than 250 scientists, managers and science-interested people to summarize the current state of science and to identify and prioritize science and management needs for the Estuary. Consensus views from the conference were summarized in the "White Paper on the Status and Needs of Science in the Delaware Estuary" (Kreeger, et al 2006) that called for a better understanding of benthic conditions.

Soon after the white paper, The Partnership and its collaborators around the estuary designed The Delaware Estuary Benthic Inventory (DEBI) program to fill the vital data gap in our understanding of the estuary's ecosystem by characterizing the biological communities on the bottom. By adding a more spatially comprehensive biological layer to existing maps of physical bottom conditions and historical surveys of benthic communities, findings from DEBI are expected to aid scientists and coastal managers interested in trophic relationships, fisheries, pollutant distributions, water quality, and other topics. These results also furnish an important baseline for tracking future ecosystem responses to changing climate and expanded development in the watershed.

A top priority of this project was to use standard methods to examine the spatial distribution and relative abundance of bottom communities living in soft-bottom substrates that span the broad salinity gradient of the Delaware Estuary. Sediment chemistry and water quality were also examined at the same sample stations. A second priority was to explore biological communities living on selected hard-bottom habitats. Although the RARE-funded project was of foundational importance in launching the program and furnishing base layers, follow-up studies are planned to continue DEBI, such as further exploration and mapping of hard bottom communities and mapping of benthic ecosystem services.

By creating a biological layer, to complement existing habitat and bathymetry layers, insight can be gained to the benthic communities that inhabit the bay and river. Benthic invertebrates tend to live a longer life then most planktonic organisms and can therefore suggest the environmental conditions over time. The Delaware Bay and River consist of both hard bottom and soft bottom, each revealing different knowledge. The soft bottom is a dynamic system that can reveal information about anthropogenic inputs, the history of anthropogenic changes caused to hard bottoms in the lower bay and the legacy that it has left is also of relevance. These changes have possibly lead to compositional and structural changes to the biological communities.

PDE

Fig. 5.1. Pictures are from sampling during the 2009 Delaware Estuary Benthic Inventory (DEBI)

As a first step in launching DEBI, the Partnership for the Delaware Estuary (PDE) partnered with US EPA Regions 2 and 3, US EPA of Research and Development (ORD), and other academic and agency partners to create a technical workgroup affiliated with the PDE Science and Technical Advisory Committee. PDE and this workgroup held workshops and summarized existing benthic data from seven prior bay-wide scientific studies. In addition, specimen collections from surveys by William Amos and colleagues in the 1950's were retrieved from storage and digitalized to augment the growing compendium of existing benthic information.

The soft-bottom survey was completed during the summer of 2008, consisting of 230 sampling sites from the mouth of Delaware Bay to the confluence of the Schuylkill and Delaware River, stratified by three salinity zones and sampled using a probabilistic design. EPA Region 3 provided critical in kind support for the 2008 cruises, including ship time and staffing. Bottom grab samples were taken at each station and split for biological taxonomic examination and chemical analyses. EPA Region 3 analyzed samples for a suite of sediment chemistry parameters, and the Delaware River Basin Commission examined splits samples for PCBs. Macroinvertebrate analyses were conducted via a subcontract to Versar Inc.

Exploratory surveys of selected hard bottom habitats were conducted in 2008, 2009 and 2010. Hard bottoms are more difficult to survey than soft bottoms in the Delaware Estuary because of naturally high turbidity and the ineffectiveness of grab samplers used for soft bottoms. Consequently, much less is known about these areas despite the belief that they are biologically active and ecologically important. Epibenthic sleds, oyster dredges, divers, and ROVs were used, where possible, yielding important new information for areas that were surveyed. For example in the lower bay, extensive "sponge gardens" and worm reefs were found in deeper troughs using the dredge, and divers observed greater fish use of these complex habitats compared to adjacent sand soft-bottoms. In the freshwater tidal zone of the estuary, at least two types of SAV and seven species of scarce or rare unionid mussels were discovered in substantial abundance. Two of the mussel species were considered locally extinct by state agencies. These discoveries of sensitive, rare biota were unexpected considering that they were found in the urban corridor which has had historically poor water quality. Although further work is needed to examine their range and abundance, these beds of freshwater mussels and SAV (which coexisted in many areas) could be important for sustaining fish habitat and water quality in the upper estuary.

Taken together, results from the soft- and hard-bottom surveys have yielded important discoveries and provided the most spatially complete biological layer ever for the bottom of the Delaware Estuary. The new biological layer clearly shows that bottom communities of the Delaware Estuary are spatial complex, spanning the many salinity zones and influenced by the presence and absence of sediment chemistry and stressors. From this layer climate change scientists will have a comprehensive baseline to track future changes in biological communities. The Delaware Estuary has over 200 migrant and resident finfish species that use the Estuary for feeding and spawning, and these new data will also provide managers with a better geospatial understanding of how benthic food resources and habitat support fisheries productivity and/or critical habitat for endangered species such as sturgeon. Maps of filter-feeding organisms may lead to a better understanding of pelagic-benthic coupling and ecosystem services that benefit water quality. Certain hard-bottom communities such as intertidal sabellaria reefs and shallow subtital oyster reefs are also increasingly appreciating for helping offset storm surge and coastal flooding.

The work supported by the RARE grant greatly increased our understanding of the estuary's bottom ecology and will have a direct bearing on diverse management priorities. More effort will be needed to build on the DEBI data to increase our understanding of benthic processes, hard-bottoms, and temporal (seasonal or inter-annual) variability that occurs across the Delaware Estuary. To track anthropogenic and climate driven changes, the benthic biota should also be broadly sampled using comparable methods at least every ten years.

5.1 Description of Indicator

Because of their abundance, diversity, sessile nature and recognized responses to environmental conditions, benthic organisms have long been used to assess the "health" of estuarine systems. In this context, the responses of the benthos to disturbance, organic enrichment associated with eutrophication and pollution, including oil and heavy metals, are of particular interest. To obtain benthic faunal data, typically a grab sampler is used to retrieve a bottom sample, and the sample is subsequently sieved to retain animals, which are then preserved. In the laboratory, macrofauna are identified, enumerated and weighed, allowing metrics such as the number of species, diversity indices or other statistical comparisons of stations to be computed. Examinations of patterns in these metrics are then used to infer the state of, or trends in, the benthic community. Alternatively, direct comparison of assemblages between impacted and reference sites may be used to infer habitat degradation and by extension the overall state of the benthic system. The condition of the benthic community is well known to respond to physical (especially salinity and sediment properties such as particle size) and biological (primary productivity, food web structure, especially predators) factors as well as chemical stressors (e.g., organic enrichment, metals, oil and other organics). Typically, estuaries are spatially and temporally variable in these physical, biological, and chemical factors, and benthic species abundance and assemblage composition is accordingly found to be highly variable in time and space as well. In addition, the faunal or assemblage response(s) to a given factor are often not unique, that is, an observed change cannot always be associated with a single causative agent (i.e., chemical), trend, or process, whether natural or anthropogenic. Polluted sites may have assemblages resembling that of naturally disturbed sites and to complicate matters further, stressors may act in combination, and cause and effect may thus be difficult to resolve using simple measures, especially where observed differences are embedded within the overall natural variability of the estuarine environment.

This is the first time an analysis of the subtidal benthic community has been used as a metric in the Technical Report for the Delaware Estuary & Basin report. We review the most recent and most extensive sampling of the bay conducted under the aegis of the Delaware Estuary Benthic Inventory (DEBI) project and present some preliminary findings and conclusions. These results are then placed in the context of past surveys and followed by some consideration of the use of historical surveys for assessing trends across decadal time scales.

5.2 Present Status

In summer 2008, DEBI was conducted to gather soft-bottom benthic data, with extensive benthic grab and water column sampling. 229 sites were allocated throughout the Delaware Bay and River in a design based on random locations within salinity and bottom sediment strata. Sediments were sampled using a 0.04-m^2 modified Young grab, sieved on a 0.5-mm mesh, and processed as described above. A summary of environment parameters measured during this survey is presented in Table 5.1. Benthic species composition, sediment characteristics, and measurements of metal concentrations as potential stressors were analyzed using diversity indices, multivariate ordinations, and dominance curve techniques.

Table 5.1. Summary of benthic Surveys in the Delaware River and Estuary conducted 1951-2008. (< D.L. means below the detection limit)

Parameter	Mean	Minimum	Maximum	Units
Salinity	13.3	0.2	31.8	‰
Temperature	24.8	17.1	27.8	°C
Dissolved Oxygen	6.8	4.3	11.8	Mg/l
pH	7.7	7.0	8.5	-
Turbidity	41.3	3.4	919.2	NT/U
% Sand	58.4	0.8	98.8	%
Total Organic Carbon	1.6	<D.L.	7.8	%
Arsenic	7.35	<D.L.	330	$\mu g \cdot g^{-1}$
Cadmium	0.44	<D.L.	4.6	$\mu g \cdot g^{-1}$
Chromium	23.7	1.1	132	$\mu g \cdot g^{-1}$
Copper	13.5	<D.L.	112	$\mu g \cdot g^{-1}$
Lead	22.6	1.4	256	$\mu g \cdot g^{-1}$

Overall, 233 benthic species were identified in 112 families and 9 phyla. Five stations had 40 or more species and the mean species richness (number of species) was 13. The most diverse groups were: polychaetes (27 families, 79 species), amphipods (15 families, 35 species), bivalves (17 families, 27 species), and gastropods (15 families, 25 species). The mean benthic invertebrate abundance was 8,800 individuals per square meter. The greatest total abundance was 142,000 individuals per square meter at Egg Island Point; this abundance was dominated by the polychaetes, *Sabellaria vulgaris* (See both feature boxes at the end of the section) and *Polydora cornuta*. The most abundant single species at any station was the bivalve, *Gemma gemma* (71,000 individuals per square meter) near Nantuxent Creek. The dominance by polychaetes, bivalves and amphipods was expected for the estuary's mixed sand-silt sediment as well as from previously published studies, although the abundances reported here are considerably larger than some previous reports (as discussed below). Together, the DEBI data represent the most intensive and comprehensive assessment of the Delaware Estuary's benthic fauna ever conducted, and these data are especially valuable in comparison with surveys of Delaware Bay conducted in the 1950's, 1970's, and more regularly since 1990 (Table 5.2).

Table 5.2. Summary of benthic Surveys in the Delaware River and Estuary conducted 1951-2008 in DEBI final report

Metadata	Amos DRIC	Maurer et al.	EMAP	NOAA S&T	MAIA	NCA	DEBI	Comments
Year(s) and Seasonality	1950's, mostly summer	1972-73, summers	1990-1993, summers	1997, September	1997-98, summers	2000-2006, summers	2008, summers	Summertime for peak abundances, most favorable weather
Spatial Domain	Delaware River and Estuary	Delaware Bay	Delaware Bay	Delaware River and Bay and coastal Atlantic	Delaware River (to Trenton) and Bay	Northeast US, Delaware Bay to Maine	Delaware River and Bay	
Number of Stations	Estimated to be about 130	207	25	81	88	138	230	Remarkably, almost stations 900 over all 7 surveys
Sampling Design	Various, piggybacked on hydrographic and zooplankton projects	Lines running along channels, bathymetry	Probabilistic	Probabilistic with strata	Probabilistic	Probabilistic with strata	Probabilistic with salinity and sediment strata	
Sampling Gear	Grabs, dredges, buoy scrapings, plankton tows	0.1 m^2 Petersen grab and 1.0-mm mesh	EMAP grabs and water quality, 0.5-mm mesh sieve	Young modified Van Veen, 0.5-mm mesh sieve	0.04-m^2 Young-modified Van Veen grab sampler, 0.5-mm mesh screen	0.04 m^2 Young-modified Van Veen, 0.5-mm mesh sieve	0.04 m^2 Young-modified Van Veen, 0.5-mm mesh sieve	Note differences in sampling gear and sieve mesh sizes
Additional Data	Hydrographic	Hydrographic and sediment	Hydro-graphic, sediment and stressors	Hydrographic, sediment and stressors	Hydrographic, sediment and stressors	Hydro-graphic, sediment and stressors	Hydrographic, sediment and stressors	Hydrographic: temperature and salinity; sediment: grain size or % sand, % silt-clay; stressors: DO, heavy metals, organic pollutants
Total Number of Species	≈396, but includes plankton, epifauna species	169	268	239	179	203	235 with Taxonomic Serial Numbers (TSN's)	
Mean Abundance	Not applicable, presence/ absence sampling only, abundances not recorded	722 m^{-2}	[to be computed]	Mean densities: 1412.5 m^{-2} to 26985.0 m^{-2}, but Hartwell and Hameedi report mean of 451 m^{-2}(?)	[to be computed]	770 m^{-2} from all stations [to be computed for just Delaware Bay]	Nearly 9000 m^{-2}	Values to be recomputed to ensure valid comparison
Statistical Methodology	n/a, see below	Cluster analysis	EMAP BI	Cluster analyses	Benthic indices	PRIMER MDS ordination and VPI and B-IBI indices	Diversity indices, ordination plots, dominance plots	
Overall Conclusions	1st survey, data exceeded manual analysis, data awaits analysis (2011)	Low abundance implies low productivity, faunal assemblages better related to sediment than salinity	One-fourth of the Delaware Estuary has impacted benthic communities	Diversity and abundance lowest in low salinity dominated by tubificids and oligochaetes; species richness correlated with grain size	One-third of Delaware Estuary received poor score using Paul, et al (1999) benthic index (EMAP-VP)	Ordination suggests salinity and latitude subregions; NCA data with VPI: 34% good, 29% poor, 37% missing	Salinity drives distribution and diversity overall	Distinct estuarine fauna as in, e.g., Remane diagram, but recent studies discount existence of true "estuary species" and interpret distribution and assemblages in light of salinity, sediment and stressors
Key References	Amos (1952, 1954 and 1956) but largely	Maurer et al. (1978), Kinner et al. (1974)	Billheimer et al. (1997), Billheimer et	Vittor (1998), Hartwell et al. (2001) Tech Memo	USEPA 2002. EPA/620/R-02/003	Hale (2011)	[This report is the first look at these data]	
Web URL for Data	Digitized, awaiting analysis	Results published, availability of raw data unknown	http://www.epa.gov/emap/html/data/geographic.html	http://ccma.nos.noaa.gov/about/coast/nsandt/download.aspx	http://www.epa.gov/emap/maia/html/data/estuary/9798/index.html	http://www.epa.gov/emap/nca/index.html	http://www.delawareestuary.org/science_projects_bay bottom.asp	

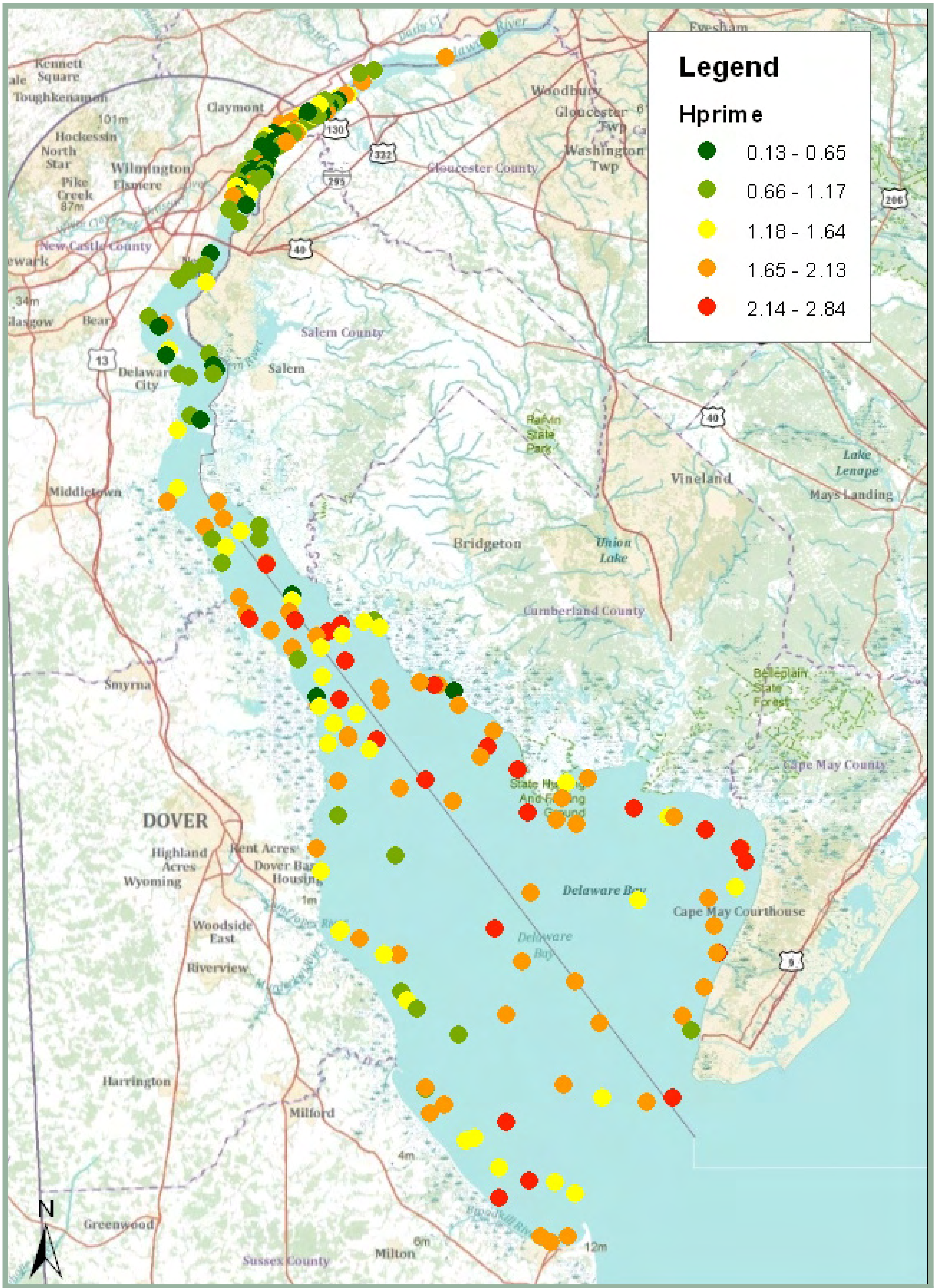


Fig. 5.2. Dots show DEBI sampling locations, and are colored to show benthic diversity in a spatial context, using the Shannon-Wiener diversity index, H'

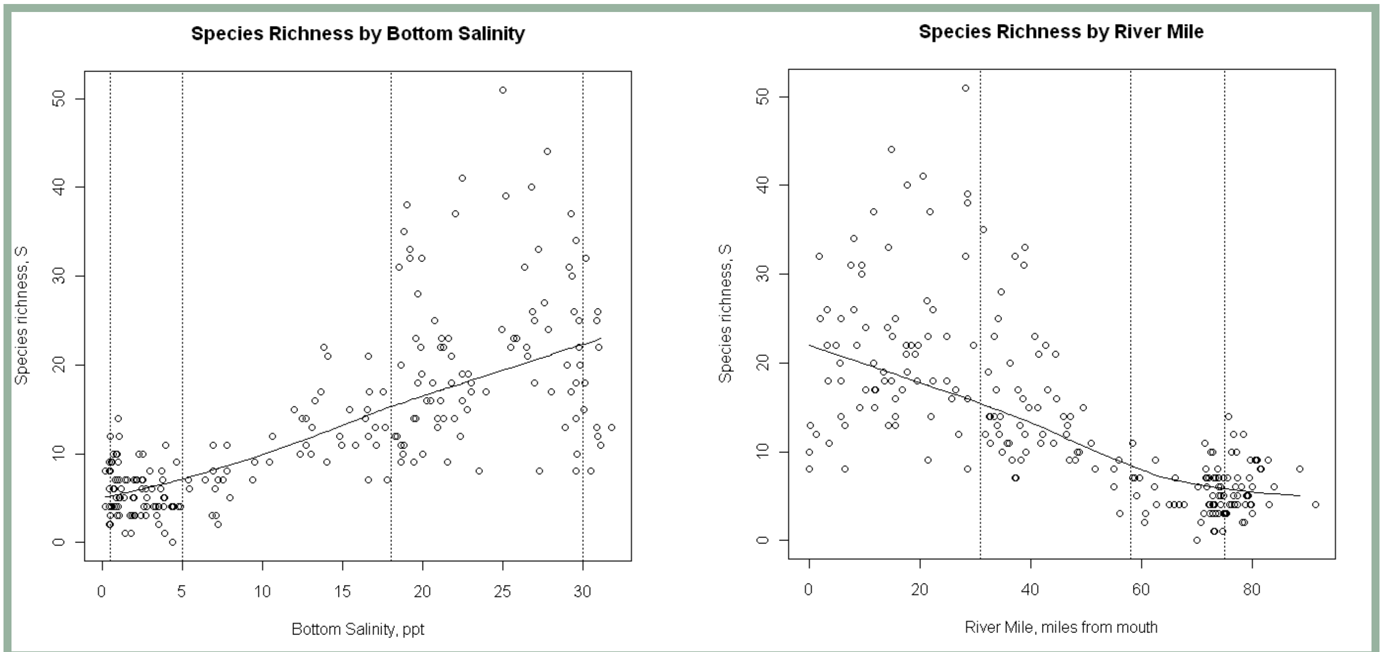


Fig. 5.3. Patterns of benthic species diversity, comparing species richness versus bottom salinity, and comparison of species richness versus river mile.

Figures 5.2 and 5.3 display the estuary-wide patterns of benthic species diversity. Species richness (number of species) versus bottom salinity and river mile, with approximate demarcations of polyhaline, mesohaline, oligohaline, and tidal freshwater zones. Both plots show a characteristic shape of a Remane diagram (Remane and Schlieper 1971) where the pattern is of high diversity at the bay mouth (and at high salinity), decreasing upstream into the mesohaline, reaching a minimum, then higher (and here, more variable) in the oligohaline (near 80 miles from the bay mouth). This is the pattern of benthic diversity commonly seen across estuaries and described in marine ecology textbooks, see Levinton (2001) or Kaiser et al. (2005) and references therein. Figure 5.2 shows benthic diversity in a spatial context using another commonly used metric, the Shannon-Wiener diversity index, H'. The interpretation of this plot is similar to those in Fig. 5.3: the concentration of red and orange dots in the lower bay suggests higher diversity there as compared to the riverine sections of the bay denoted by green and black dots.

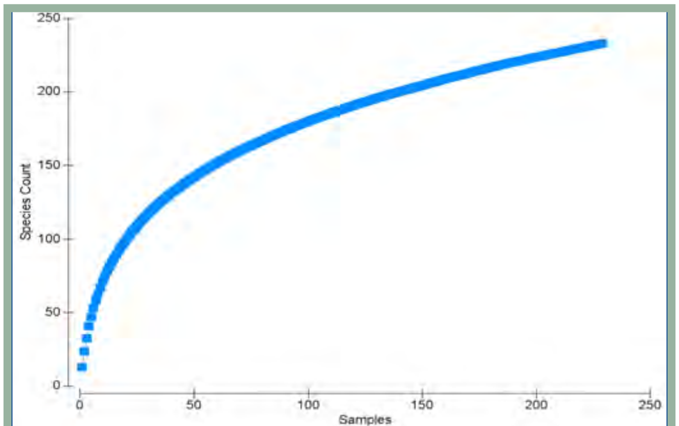


Fig. 5.4. Species accumulation curve, number of species versus number of samples taken during DEBI project.

Figure 5.4 is a species accumulation curve showing the number of species expected versus number of samples taken in the DEBI survey; as more samples are taken, more species are recorded. A leveling off of this curve would indicate that few new species would be recorded by additional sampling, and thus the asymptote represents the total diversity as number of species in the estuary. The shapes of these curves (i.e. initial slope and asymptote) can be compared among studies in order to gauge the effectiveness of sampling and assess the degree to which the full diversity has been sampled. The upward slope at the right of the DEBI curve shown here indicates that even this extensive survey did not capture the full (technically, alpha) diversity of the Delaware Bay soft-bottom benthos. However, the observed diversity of 233 species is generally consistent with other surveys summarized in Table 5.1.

A more detail view of the estuary's benthos is provided using a non-metric multidimensional scaling (MDS) ordination of the full species by assemblage abundance

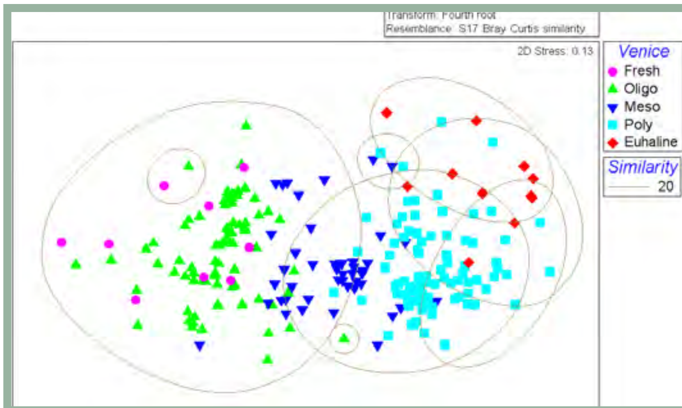


Fig. 5.5. MDS ordination analysis showing species similarities based on salinity zones.

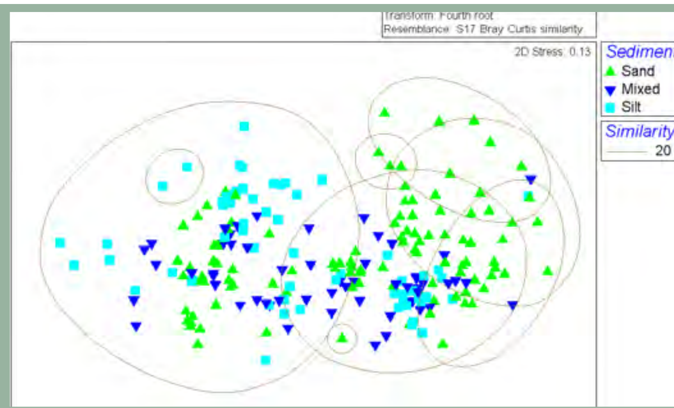


Fig. 5.6. MDS ordination analysis showing species similarities based on sediment type.

matrix. Figures 5.5 and 5.6 show all 299 stations' similarities based on all 233 species using fourth-root transformed abundances and the Bray-Curtis similarity metric, computed using the PRIMER-E package (Clarke and Warwick 2001, Clarke and Gorley 2006). Each symbol represents a station: symbols close together have similar species composition (low dissimilarity), while points far apart differ in species composition (i.e. are dissimilar) in accordance of their separation. The stress value reported here, 0.13, indicates that the two-dimensional plot adequately represents the multivariate (high-dimensional) dissimilarities among stations. The broad ellipses represent groups of stations determined as by a cluster analysis as superimposed on the ordination and are shown here for visual reference. When stations are coded by salinity zone (Fig. 5.5) it is clear that benthic assemblages relate to salinity, with freshwater and oligohaline stations grouped together on the left, mesohaline are concentrated in the middle and polyhaline and euhaline fall together to the right. Figure 5.6 is the same ordination (i.e., the pattern of station points is identical), but the color key represents sediment grain size measured as percent sand. Sandy, silty-sand and silty sites are not separated, but intermixed and not clearly related to species composition, thus sediment composition is not simply associated with broad patterns in species composition. As was found using simple diversity metrics, salinity is the dominant factor correlated with benthic community structure.

Additionally, MDS ordination plots of benthic assemblages can be used to investigate the benthic response to stressors. Figure 5.7 shows four such ordinations (with points identical to those already shown) with the symbol size representing the level of each of four potential stressors: (5.7a) dissolved oxygen near bottom, (5.7 b) total organic carbon, (5.7C) cadmium and (5.7d) chromium. Dissolved oxygen measured near the bottom was in all cases 4.4 mg/l or greater (Table 5.1), and it is not surprising that there is little association of bubble size with stations clusters or broad patterns in the ordination in panel (5.7a). Total organic carbon show larger bubbles associated with stations in the upper and lower bay (5.7b), likely associated with fine sediments (compare

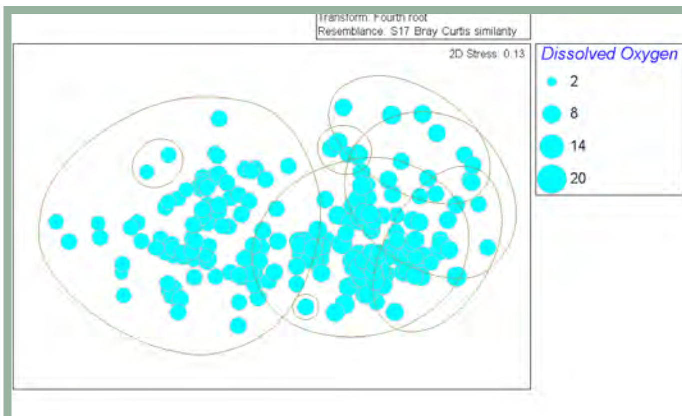


Fig. 5.7a. MDS ordination analysis showing species similarities based on dissolved oxygen.

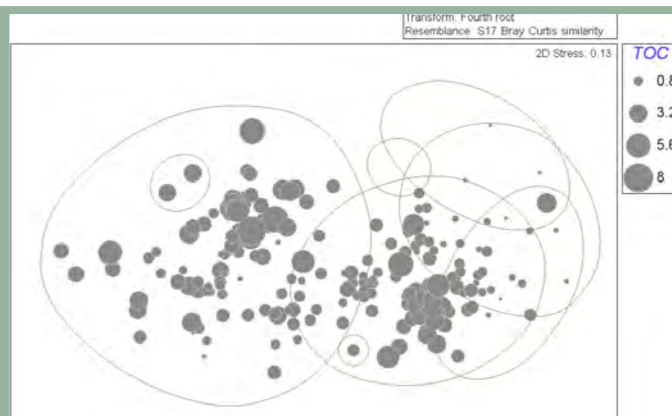


Fig. 5.7b. MDS ordination analysis showing species similarities based on total organic carbon.

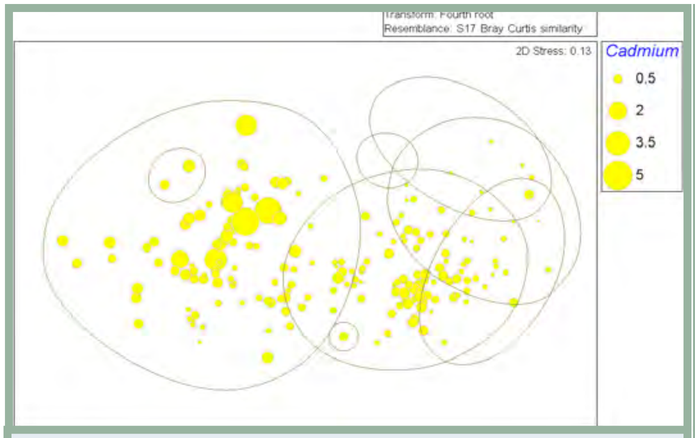


Fig. 5.7c. MDS ordination analysis showing species similarities based on cadmium.

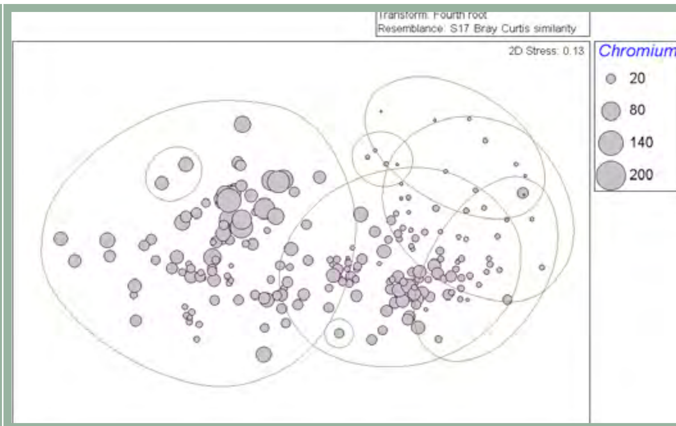


Fig. 5.7d. MDS ordination analysis showing species similarities based on chromium.

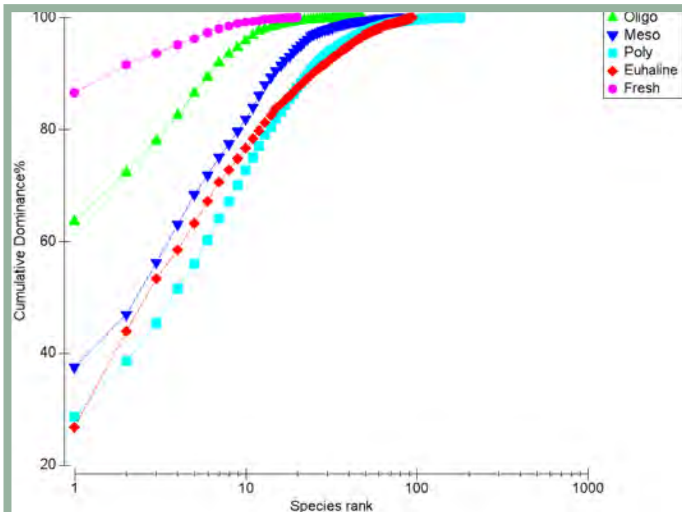


Fig. 5.8a. Dominance curves for DEBI species data, pooled by salinity.

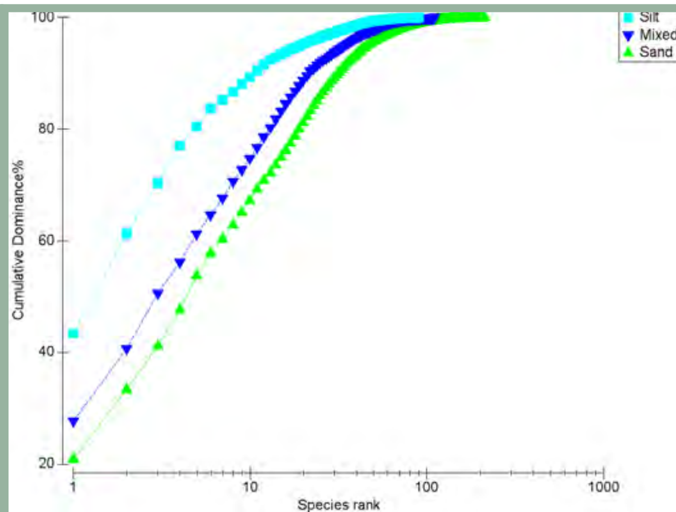


Fig. 5.8b. Dominance curves for DEBI species data, pooled by sediment class.

with Fig. 5.6). A distinct association of high metal concentrations and benthic assemblages and stations is apparent in both panels (5.7 c) and (5.7d) as a knot of large bubbles associated with lower salinity stations (Fig 5.6). This suggests that metal concentrations may be affecting benthic assemblages at these stations and that further analysis is warranted.

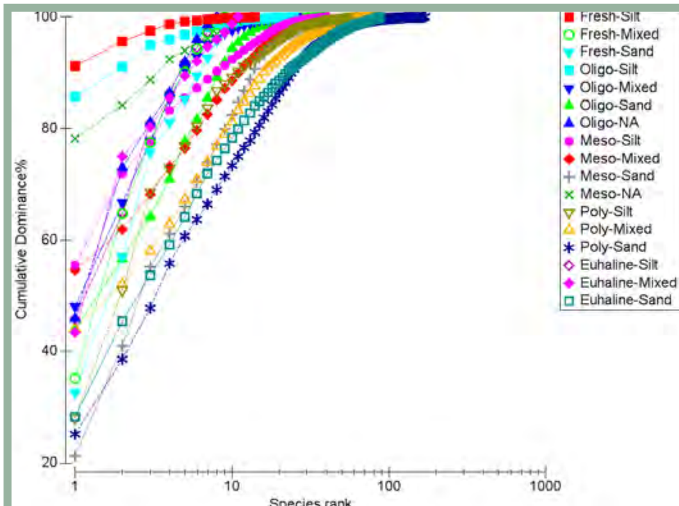


Fig. 5.8c. Dominance curves for DEBI species data pooled by salinity and sediment class.

Dominance curves can likewise be used to investigate patterns in benthic fauna. Potentially disturbed or polluted assemblages have been found to be dominated by few but abundant species (Warwick 1986, Warwick and Clarke 1994, Elliott and Quintino 2007). Figure 5.8 shows these lots for DEBI species data pooled by salinity (5.8a) or sediment class (5.8b) or both jointly (5.8c). The plots show the cumulative percent of individuals for the most abundant species, the second most and so on, by species. A gradual rise to 100% is apparent for these categories, for all sediment classes (5.8b) and mesohaline, polyhaline and euhaline classes, while oligohaline and freshwater curves show higher dominance, higher curve on the left side (5.8a). When jointly classified (5.8c)

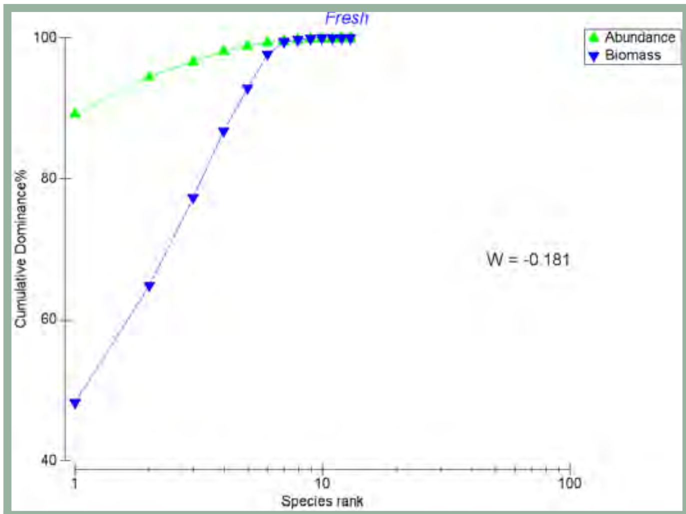


Fig. 5.9a. Abundance-biomass curve for freshwater stations.

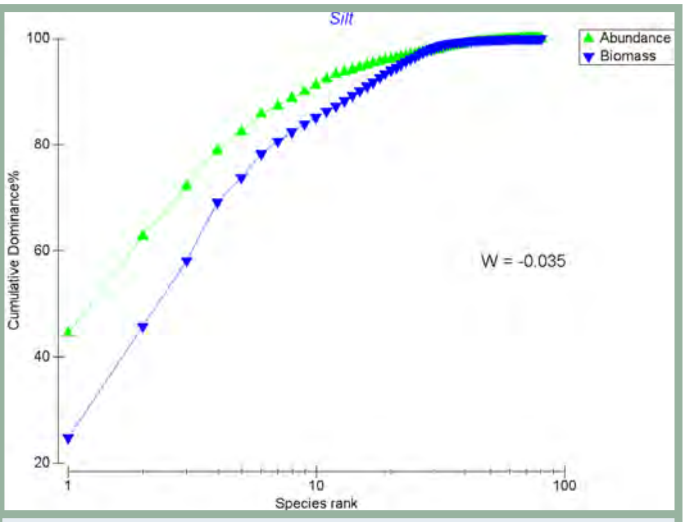


Fig. 5.9b. Abundance-biomass curve for silty sediment stations.

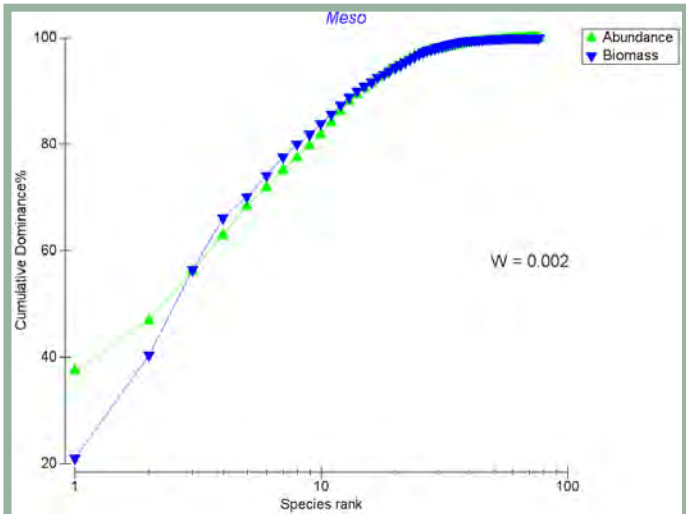


Fig. 5.9c. Abundance-biomass curve for mesohaline stations.

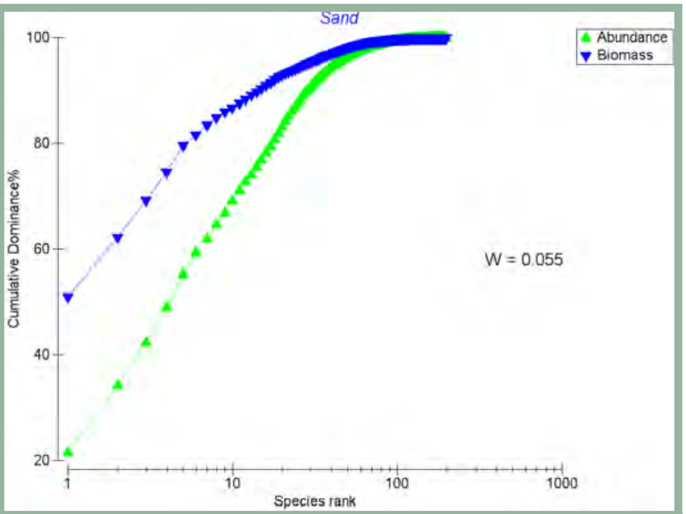


Fig. 5.9d. Abundance-biomass curve for oligohaline and sandy sediment stations.

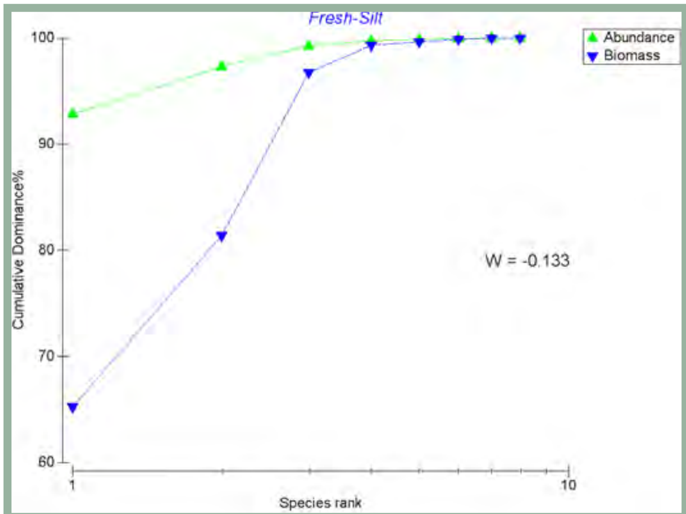


Fig. 5.10a. Abundance-biomass curve for . Oligohaline-silty sediment statinos.

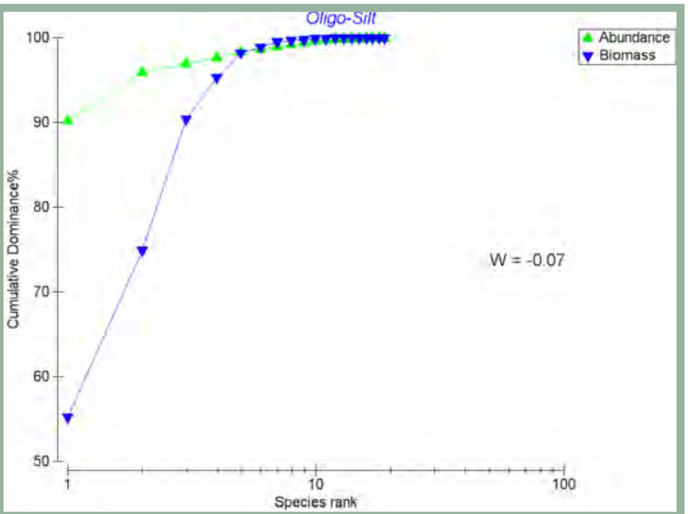


Fig. 5.10b. Abundance-biomass curve for fresh-silty sediments stations.

the oligohaline-silt and fresh-silt stations show high dominance, considerably greater that that of the rest of the salinity-sediment classifications.

Biomass curves can also be used to identify disturbed or polluted conditions: the cumulative percent biomass by species rank is superimposed on the dominance curve in a combined abundance-biomass comparison (ABC) plot. In unpolluted conditions, the biomass curve lies above the abundance curve (Warwick 1986, Warwick and Clarke 1994, Elliott and Quintino 2007), representing an assemblage with many species of moderate abundance and biomass dominated by a few large species, and this interpretation is consistent with that of the classical Pearson and Rosenberg (1978) paradigm (see also Gray and Elliott 2009). In disturbed or polluted conditions, a few but abundant, yet small species dominate (i.e., the large species are eliminated), and the abundance curve lies above that of the biomass. For the DEBI data, fresh and silt ABC curves (Fig. 5.9a and b) are inverted, in comparison to mesohaline and sand (Fig. 5.9c and d). Inversion of the ABC curves is also clearly apparent in the fresh-silt and oligohaline-silt curves (Fig. 5.10a and b), and these stations are located in the C&D Canal to state-line region (and within DRBC's Zone 5) of the estuary. Especially as this area has been characterized as degraded in benthic condition in past studies, these patterns at these stations merit further investigation.

The conclusions from this preliminary analysis are that broad-scale estuarine patterns are as expected for a temperate Atlantic estuary and that the soft-bottom benthic diversity of the Delaware has been sampled to a reasonable though, not exhaustive, extent. Bay-wide, salinity drives the patterns among benthic assemblages to a greater degree than sediment composition, and that high metal concentrations are associated with assemblages at certain stations. Further analysis within salinity and sediment classes reveals assemblages highly dominated by a few, abundant species, which also exhibit inverted abundance-biomass curves, further suggesting disturbed or polluted conditions. In summary, while these overall patterns among the benthic fauna are as expected in terms of abundance, diversity and biomass, stations in the C&D Canal to state line region (DRBC's Zone 5) are distinct in their assemblages, associated with high metal concentrations and have abundance and biomass curves consistent with polluted conditions. This region has been characterized as degraded in past studies on benthic assemblages.

The U.S. EPA recently released the 2011 National Coastal Condition Report IV (U.S. EPA, 2011). The 2006 report divided the analysis not only by region but by estuary as well. Unfortunately, in the 2011 report an assessment was provided only for the Mid-Atlantic Bight and not specifically the Delaware Estuary. The coastal assessment in the Mid-Atlantic Bight of the benthos demonstrates that conditions have remained the same, classified as poor condition, since the last assessment in 2006.

5.3 Past Trends

Starting in the early 1950's, there is an extensive history of scientific benthic study in the Delaware River and Estuary (Table 5.2). Since 1990, surveys have used probabilistic designs for station selection as well as consistent methodologies for sample collection and processing, faunal identification and taxonomy, and data summary and compilation. Specifically, there have been five separate federal programs using the benthos as indicators in Delaware Bay. Conclusions from the early 1990 EMAP survey are reported in Sutton et al. (1996). According to the EMAP benthic index, 93% of the area of the tidal river has benthic communities classified as degraded (68% area) or severely degraded (25 % area). In comparison, only 2% of the bay's area south of the C&D canal was degraded, and no stations were severely degraded. Several benthic indices have been applied to Delaware Bay stations as part of the broader-scale, National Coastal Assessment (NCA) studies beginning in 2000. Using the Virginian Province Benthic Index and 2000-2001 data, 34% of the stations were rated "good," 29% "poor," and 37% "missing," and this mixture of conditions was found throughout the bay and river (US EPA 2006).

In addition to the federal studies, there are "historical" surveys undertaken by Amos in the 1950's and Maurer and colleagues in the 1970's (Table 5.2). In total, sampling has been reported at nearly 900 stations, and the total number of species reported from these studies is consistently 200 or more (cf. Fig. 5.4), with the mean (over stations) total abundances (number of organisms per meter squared) in the expected range of 1000 – 10,000 per square meter, although two surveys reported abundances well below 1000 per square meter. In particular, low abundances were noted by Maurer et al. (1978), wherein they concluded that low abundance reflected low benthic productivity in the Delaware Bay. Low abundance could equally be explained by their use of a 1-mm mesh sieve as compared to the 0.5-mm mesh (a smaller sieve retains more, smaller fauna) used in the present DEBI 2008 sampling as well as other recent federal surveys), although Maurer et al. (1978) discuss this point and explicitly discount this explanation in their report.

The reason(s) for the low mean abundance reported by Hartwell and by Hale are not resolved at present. Future studies by comparing abundance of large species and small (i.e., those not expected to be completely retained by a coarse sieve) selectively, may make it possible to confirm a sieve-bias explanation for at least the Maurer et al. (1978) results.

All or most of the federal data are hosted online although distributed over several federal agency web sites and presented in various data formats. In most cases, data are tabulated as species abundances, and fortunately the consistency of sampling, laboratory analysis, and ready availability of these data will allow synthesis by modern statistical techniques. Any trends in these data over the past 30 years should be resolvable once challenges of data formatting and merging are overcome.

5.4 Future Predictions

Summary plots of diversity, faunal assemblage ordinations, and dominance plots in this section that likely sufficient sampling has been conducted to facilitate development of conclusions and that broad, estuary-scale patterns are as expected based on typical estuarine patterns of diversity. It is important to note that the federal agencies have routinely included stressor variables, such as dissolved oxygen, organic carbon, heavy metals, and organic pollutants in their measurement suite (Table 5.1). These individual surveys have consistently assessed the benthos in light of possible stressors, yet there have been few if any attempts at cross-survey synthesis of these data to assess trends in benthic community structure and condition over time.

5.5 Actions and Needs

The ready availability of extensive data clearly justifies a cross-survey analysis of the past 30 years. Additional effort will be required to determine if differences among data sets are due to a sampling design (spatial allocation of locations) or sampling gear-bias (especially sieve mesh size) or truly represents significant change in estuary conditions. Only limited, broad conclusions can be draw from the simple data summaries and plots presented here. Further analyses using multivariate methods like multi-dimensional scaling and dominance curves may reveal patterns and relationships impossible to discern among multiple possible natural variation and anthropogenic effects. Effective analysis of these benthic data will require additional effort to identify sensitive and tolerant species, reference and control sites (to develop customized and calibrated indices), and the application of more sophisticated multivariate, phylogenetic/taxonomic structural analysis or regression-based species distribution modeling.

5.6 Summary

The benthos of Delaware River and Estuary has been extensively studied and well characterized in surveys conducted over the past 60 years. The most recent, 2008 DEBI survey, represents a firm baseline demonstrating patterns in diversity similar to those found before and typical of temperate estuaries. Overall patterns among the benthic fauna are as expected in terms of abundance, diversity and biomass, but stations in the C&D Canal to state line region are distinct in their assemblages and associated with high metal concentrations. The current DEBI survey data are consistent with other recent studies employing standardized methodology and refute previous conclusions that the bay's fauna is depauperate and unproductive. The availability and congruence of several previous data sets with the current DEBI results clearly justifies a cross survey analysis of all of the data from the past 30 years. Further effort will be required to determine if perceived differences may be due to sampling gear-bias issues, sampling locations differences, or represents real and significant changes in estuary conditions. Effective analysis of these data will require additional effort to identify sensitive and tolerant species, reference and control sites, and the application of more sophisticated multivariate, structural (i.e., phylogenetic/taxonomic) or regression-based species distribution modeling.

Delaware Bay Benthic Mapping Project

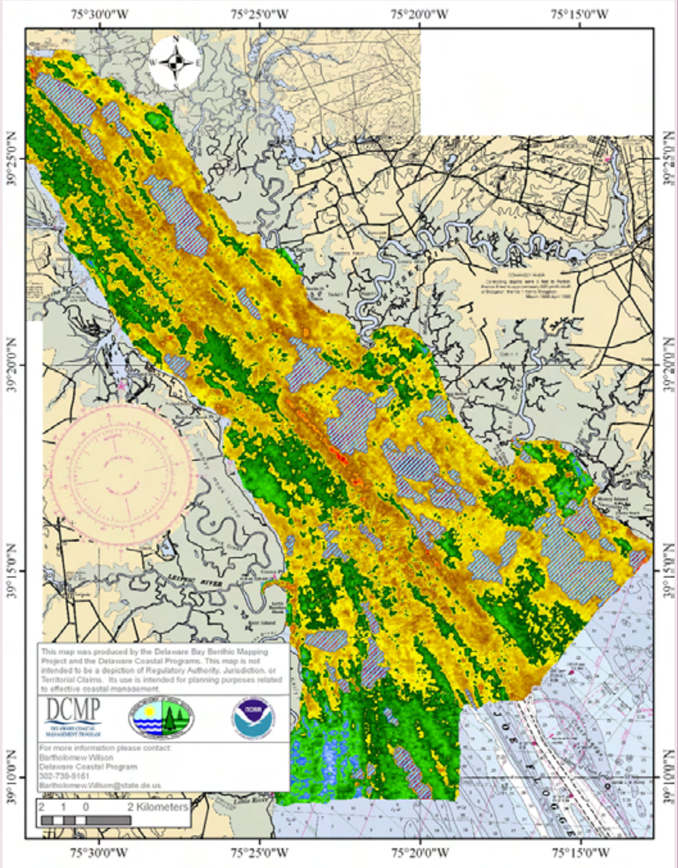
(Author: Bart Wilson)

Through an integrated effort by the Delaware Coastal Programs and the University of Delaware, a benthic and sub-bottom imaging project to identify and map the benthic habitat and sub-bottom sediments of Delaware Bay and River was initiated in 2004. This project would not have been possible without the following partners: University of Delaware Geosciences Department, Delaware Fisheries Section, Delaware Shoreline and Waterway Division, Delaware State University, Partnership for the Delaware Estuary, New Jersey Department of Environmental Protection, and New Jersey Shellfish Bureau.

This project integrates the use of three types of acoustical systems: Roxann Seabed Classification System, CHIRP sub-bottom profiling, and multi-beam bathymetric mapping. Verification of the acoustic data with bottom and sub-bottom sediments is performed through the collection of grab and core samples and underwater video images.

This effort has resulted in many major milestones, which include: mapping over 906 square km, identifying the spatial extent and relative density of the oyster and *Corbicula* beds, identification of borrow sites for beach replenishment, facilitating a greater understanding of the local and regional sediment distribution patterns and pathways, locating key habitats for species (such as: Atlantic Sturgeon, sharks, and *Sabellaria vulgaris*), and starting to understand the relative impact that humans have upon the bay bottom and its living resources. Most importantly integrating the bottom and sub-bottom sediment with species tracking information, in a 3D GIS environment, has provided a new opportunity to assess the habitat relationship between Atlantic Sturgeon and several key regions in the Delaware River.

The program has many accomplishments including an integration of the benthic and sub-bottom data was used to identify sand borrow sites within the Delaware Bay that are located in areas that minimize the impact upon essential fish habitat (especially *Sabellaria vulgaris* habitat). Borrow sites have been located for three coastal communities, and will determine sand resources for 4 additional coastal communities.



DNREC bottom sediment map showing the distribution of sediments and locations of oyster beds over a 620 square kilometers area in the upper Delaware Bay Estuary. In this region, 40 distinct oyster beds were located.

In addition, the project has worked with The Nature Conservancy (TNC) and the Partnership for the Delaware estuary to develop benthic habitat maps for the Delaware Estuary. In September 2011, TNC produced a report entitled; Delaware River Basin Priority Conservation Areas and Recommended Conservation Strategies (http://conserveonline.org/workspaces/nfwfdebasin/documents/all.html). In Appendix V; Benthic Habitats of The Delaware Bay, an attempt was made to create benthic habitat maps using bathymetry, salinity and seafloor substrate. Maps of Ecological Marine Units were created taking into account species data provided by the DEBI project.

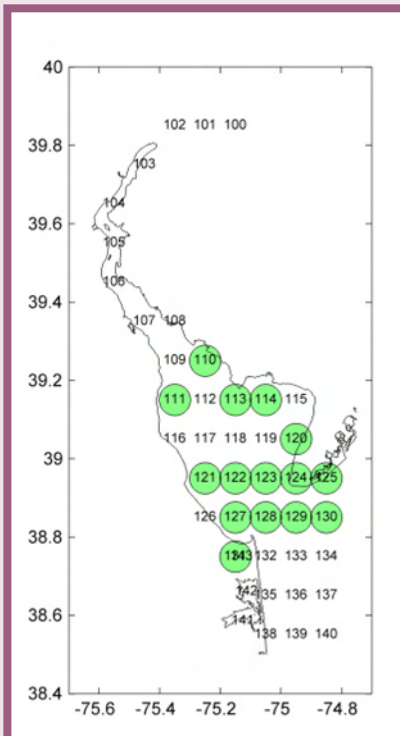
Amos Historical Benthic Collection Analysis

(Author: Douglas Miller)

The Delaware River Invertebrate Collection (DRIC) was the first scientific collection of benthic organisms for the Delaware River and Estuary. William H. Amos' handwritten 5" x 8" data cards along with preserved master specimens from the 1950's are currently housed at the University of Delaware in Lewes. Standing 25 cm (10") high when stacked vertically, these invertebrate cards were scanned for archival purposes in October, 2008 and later digitized.

The Amos DRIC includes over 5,500 records of nearly 400 species from over 130 stations within the Delaware River and Estuary. Information in a locality field in addition to uncovered charts promises to yield much more precise information for sampling locations. These data include collection of benthic organisms by trawl, dredge and Peterson grab, planktonic organisms by net and epifauna as part of the "buoy scrapes" sampling. Chronologically, these data represent mostly the years 1952-54 and 1956, and primarily July and August collections. Many records are included from the Delzoop plankton sampling that occurred several times a year from October 1951 through August 1953.

Amos identified over 400 taxonomic groupings of which about 396 represent species of invertebrates present in the Delaware River and Estuary. This estimate of species number is generally consistent with numbers Amos gave in University of Delaware Marine Laboratory annual reports. Any such "biodiversity" estimate is clearly provisional, depending on updated nomenclature, taxonomic confirmation, and assessment of the influence of sampling effort and gear bias.



Map of Delaware Bay and Amos sector grid, with bubbles showing the number of records of the sandbuilder worm, Sabellaria vulgaris in his pioneering benthic study.

Amos summarize his species distribution data in geographical form using a grid of 40 "sectors" including 37 over the main part of the bay from Philadelphia south, in the bay or just outside, plus Rehoboth Bay, Indian River Bay, and the Lewes & Rehoboth Canal. Samples near Joe Flogger and the Leipsic River have the most records, likely reflecting the intensity of zooplankton sampling in that of the bay. Sectors near Lewes Beach and the Bayside Lab, along the main channel in the lower bay, and at the Shears/Harbor of Refuge have over 200 records each. Most collections are from the main channel and lower Delaware side, and with the exception of the Nantuxent Point area, far fewer are from New Jersey waters.

In addition to representing a time in the history of the Delaware Estuary before major industrialization and development, these data present a uniquely comprehensive picture in terms of the functional group, life habit, and taxonomy of the fauna of the river and estuary. Hopefully now that this historical data set is digitized, scientists around the region will be able to access it and use it in their studies of the benthic ecology of the Delaware River and Estuary.

5 - Intertidal Aquatic Habitats

5 - 1 Tidal Wetland Area

Tidal wetlands are aquatic habitats which lie above the mean low tide line, but below the mean high tide line within an estuary or marine environment. They therefore occupy the intertidal zone between open water and upland areas. Tidal wetlands can be both in fresh water as well as salt water areas.

The traditional definition of a wetland requires that vegetation be present, most typically woody or perennial forms of vascular plants. However, for management purposes, state and federal agencies also consider as wetlands many types of non-vegetated aquatic habitats, such as shallow ponds, mud flats, and some areas dominated by benthic algae (e.g., Cowardin classification system as used by the National Wetland Inventory). For the purposes of this report, the principal focus is on vegetated tidal wetlands, which are a hallmark habitat within the Delaware Estuary.

Tidal wetlands in the Delaware Estuary extend along both shores of the Delaware River and Bay, spanning the broad salinity gradient from the head-of-tide near Trenton, New Jersey, and south to Cape May, New Jersey, and Cape Henlopen, Delaware (Fig. 5.11). The largest portion of tidal wetlands are composed of salt marshes fringing Delaware Bay, which are dominated by smooth cordgrass, *Spartina alterniflora* in the low tidal zone and various (Fig. 5.12) salt-tolerant grasses (e.g., *S. patens* and *Distichlis spicata*) and scrub/shrub vegetation in the "high marsh" zone.

In the upper estuary and in headwater areas of tidal rivers and creeks, nationally rare communities of freshwater tidal vegetation can be dominant wherever salt concentrations are below 0.5 parts per thousand (Fig. 5.13). These freshwater tidal wetlands consist mainly of perennial grasses, sedges and rushes (called emergent marshes), and there are some scrub/shrub and forested tidal wetlands as well.

Typically, freshwater tidal emergent marshes contain greater biodiversity than salt marshes. Species whose presence is diagnostic of this marsh type include wild rice (*Zizania aquatic*), cattail (*Typha sp.*), and low marsh succulents such as spatterdock (*Nuphar luteum*) and arrow arum (*Peltandra virginica*). Like salt marshes, tidal wetlands undergo daily flooding and draining, and are therefore critical components in

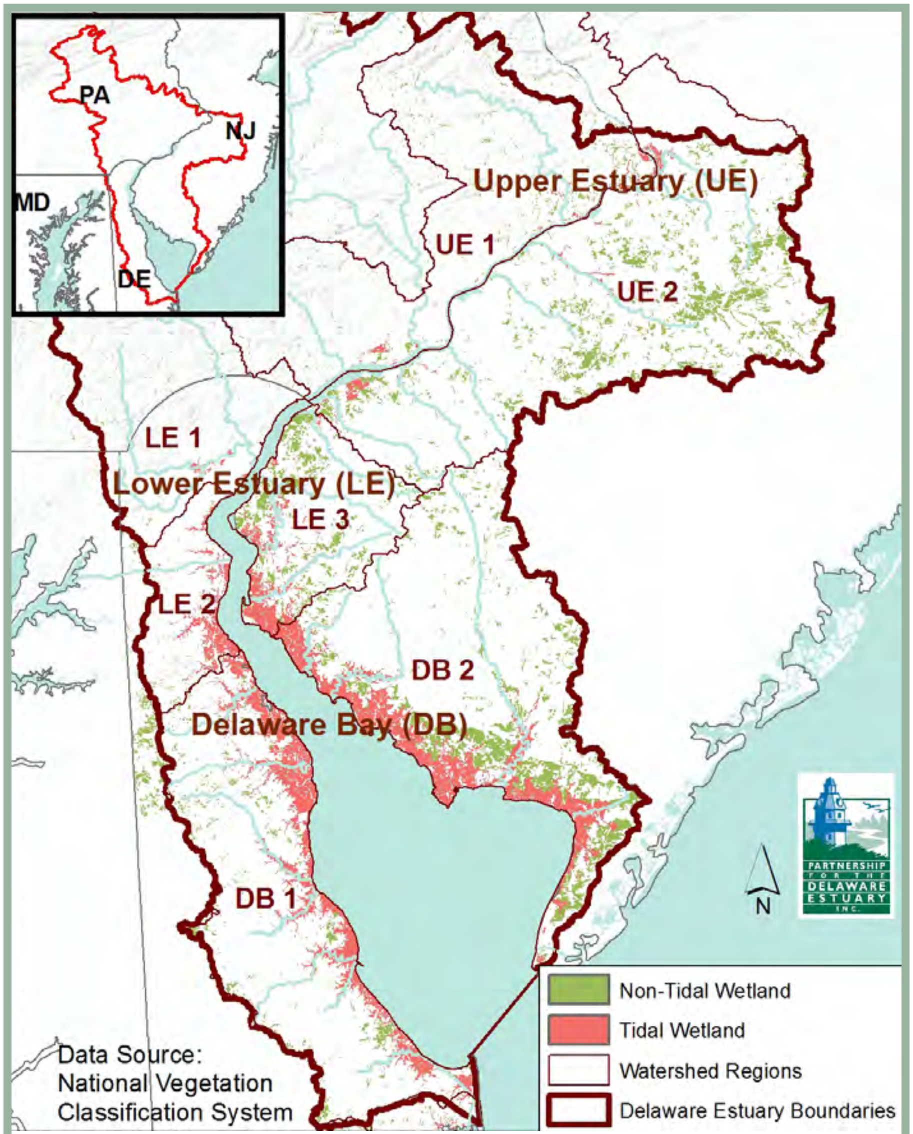


Fig. 5.11. Tidal and non-tidal wetlands in the Delaware Estuary

Key to Watershed Regions

UE - Upper Estuary
LE - Lowever Estuary
DB - Delaware Bay
SV - Schuylkill Valley

David Burke

Fig. 5.12. A characteristic tidal creek salt marsh in Delaware, 2010

Danielle Kreeger

Fig. 5.13. A characteristic freshwater tidal emergent marsh is in Crosswicks Creek, NJ, shown here in July 2011

the sensitive interaction between land and water in the estuary.

Tidal wetlands are among the most productive habitats in the world, and perform a wide variety of vital services. They help protect inland areas from tidal and storm damage; provide water storage to protect against flooding; provide important habitat to a wide variety of wildlife, including waterfowl; serve as a filter to remove contaminants and help sustain water quality; provide spawning and nursery habitat to support commercial fisheries; support active and passive recreation; and provide aesthetic value.

Tidal wetlands are therefore often regarded as the most critical habitat type in the Delaware Estuary for supporting broad ecological health. Assuring that these wetlands remain intact and continue to provide these critical functions is therefore fundamental to the protection and the overall quality of the Delaware Estuary and the Delaware River Basin as a whole.

5 - 1.1 Description of Indicator

The science and management community of the Delaware River Basin has elevated tidal wetland extent and condition as top priorities for monitoring and management, emphasizing that these habitats are one of the leading environmental indicators (Kreeger et al. 2006, PDE 2008). Too little data currently exist to assess the condition of tidal wetlands across the watershed, although efforts are underway via the new Mid-Atlantic Coastal Wetland Assessment (PDE 2011) to fill this data gap for future indicator reporting. Therefore, tidal wetland extent (hectares) is the main environmental indicator that was analyzed for this study.

Despite their importance to the Delaware River Basin, it is difficult to quantify the status and trends in tidal wetland extent due to data gaps and inconsistencies in methods used to track these habitats at different times and in different areas of this large watershed. There are two federal programs, several state programs, and periodic scientific studies that have provided useful data, but to date no data source has yielded a comprehensive, estuary-wide layer at a single pot in time. Furthermore, much of the available data do not differentiate tidal from non-tidal wetlands. The approach here was to inventory the available information on tidal wetland extent and types across the estuary using data that most appropriately reflect wetland areas consistently across each state and the region. The following is a description of the best available data layers for this indicator.

National Wetlands Inventory Data were first gathered for each state from the U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory (NWI). The NWI is a nationwide program which seeks to inventory and assess trends in the nation's wetlands. The US FWS is required to produce a report on the status and trends of the nation's wetlands. The NWI provides detailed, consistent, high-resolution data that enable clear differentiation of wetland types; however, it is of limited value in status and trend analyses for the whole system because of the different dates for which data are collected in different states and areas.

While intended to ensure a consistent and timely picture of wetlands across the country, wetland delineation under the NWI is often highly dependent on funding and input from the states. This leads to a discrepancy in the frequency and (sometimes) methodology of delineation among states. For instance, Fig. 5.14 illustrates the various time periods for the latest NWI data within the estuary, which ranges from the 1970s (in Pennsylvania) through 2009 (in Delaware). The latest NWI data available for New Jersey varies from 2002 in the north to 1999 in the southern coastal areas.

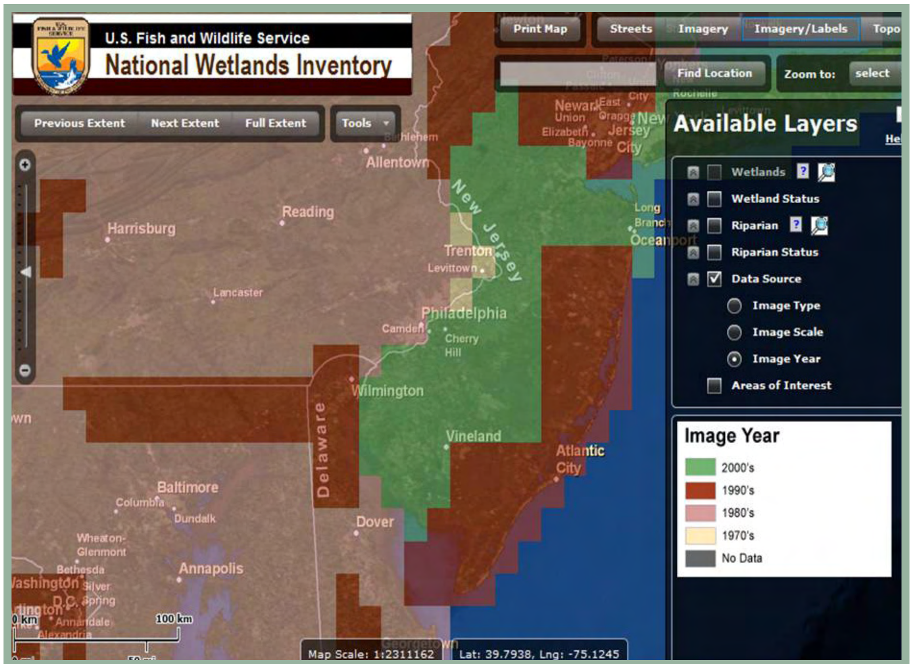


Fig. 5.14. NWI Status in the Delaware Estuary

To determine the current status of intertidal wetlands in the estuary, the latest of each of three state-wide NWI wetlands (Pennsylvania, New Jersey and Delaware) were used. Each of these layers is categorized using the classification scheme developed by Cowardin (Cowardin, 1979). A simplified classification was developed to allow for a synoptic assessment of status and trends of several broad categories of wetlands within the estuary. Table 5.3 lists the classes and the codes used to summarize intertidal wetland types.

Land Cover Data To assess trends in tidal wetland acreage, the National Oceanic and Atmospheric Administration (NOAA) Coastal Services Center (CSC) land cover datasets were used. These data are available for all coastal areas of the U.S., and have been derived from Landsat satellite imagery at a 30m ground resolution. The data are useful for examination of wetlands since there is a relatively high level of detail differentiating wetland types, and since data for the whole estuary are collected periodically at the same time. Categories of wetlands distinguished by the CSC land cover are: Palustrine Forested, Palustrine Scrub/Shrub, Palustrine Emergent, Estuarine Forested, Estuarine Scrub/Shrub, Estuarine Emergent, Unconsolidated Shore, and Palustrine Aquatic Bed. Dates for the CSC land cover data are (nominally) 1996, 2001, and 2006. (Not all states or regions were delineated using satellite from the same epoch, as interpretation requires high-quality, cloud-free imagery; and the use of photography from varying dates during which these conditions were present.)

Although land cover data are useful because they provide more consistent coverage of the watershed at specific times, land cover data sets do not offer the same degree of resolution as NWI, which is derived from high resolution aerial photography and undergoes more comprehensive ground-truthing. More importantly for our indicator analysis, land cover data used do not distinguish between tidal and non-tidal wetlands. There are six wetland categories distinguished in land cover datasets which include tidal wetlands: estuarine emergent, estuarine shrub/scrub, estuarine forest, palustrine emergent, palustrine shrub/scrub, and palustrine forest. Of these six, only one category (estuarine emergent) consists wholly of tidal wetlands (i.e., salt marshes), which represent dominant and ecologically important landscapes within the estuarine system. In general, however, due to the relative abundance of these six categories in our system, the three "estuarine" categories correspond to tidal wetlands, and the three palustrine wetland types represent largely non-tidal wetlands. Assessment of the comparability of the wetland categories of the CSC land cover data with the NWI data for New Jersey and Delaware indicates that the data are comparable with a relatively small percentage difference, especially for estuarine emergent wetlands (Fig. 5.15). Therefore, we mainly used land cover data to assess status and trends in estuarine emergent wetlands (mainly salt and brackish marshes) because of their consistent spatial coverage and ecological importance within the system.

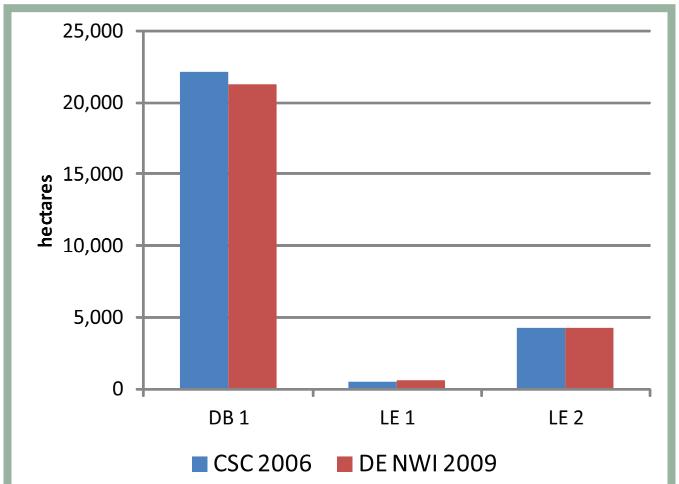


Fig. 5.15. Comparison of measured extent of salt marsh in Delaware watersheds based on CSC and NWI data. Figures agree to within less than 3%. (see map & key on p.133)

5 - 1.2 Present Status

Wetlands types cover a significant portion of the lower Delaware River Basin (Fig. 5.16). From expansive salt marsh complexes in the lower estuary, up to isolated wetlands and ponds in the upland reaches, wetlands are an important part of the ecology and hydrology of the watershed. In all, there are 421,137 acres (170,428 hectares) of wetlands (tidal and non-tidal) in the Delaware Estuary study area (lower half of the basin), representing about 10.8% of the total area. This compares to a national figure of 5.5% total area of wetlands in the contiguous U.S. (US FWS). Of these wetlands in the Delaware Estuary, 39.3% are tidal wetlands and most of those are salt marshes.

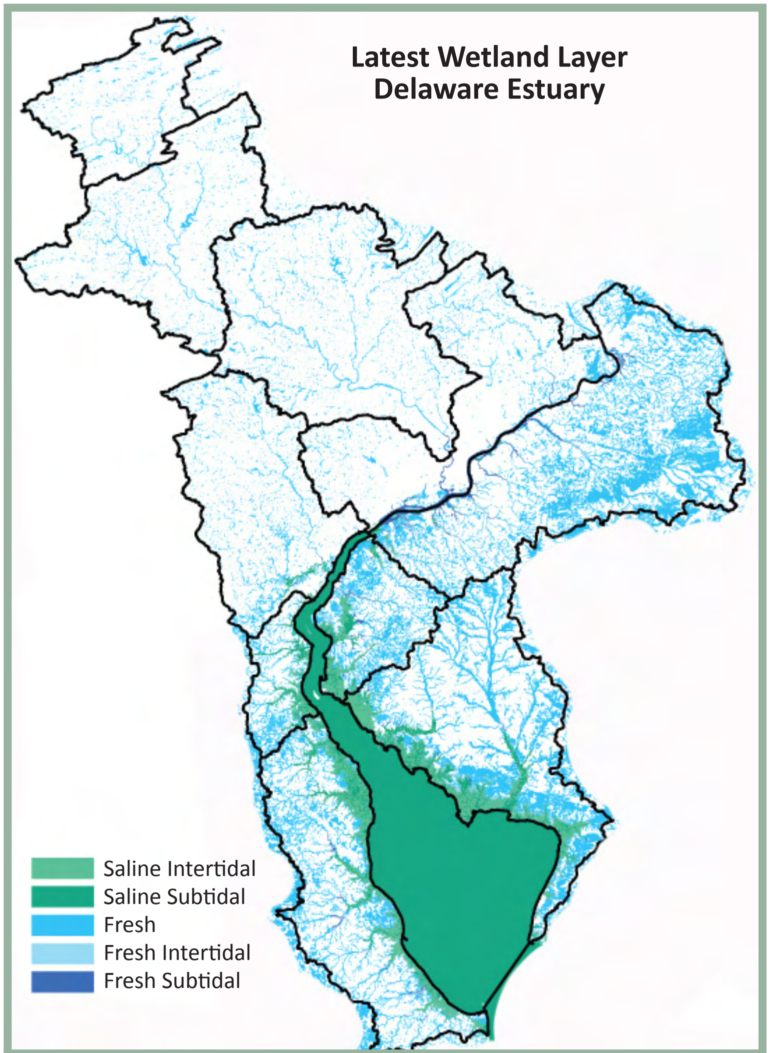


Fig. 5.16. Latest wetland layer for the lower Delaware River Basin based on analysis of both US FWS NWI and NOAA CSC land cover data

Given the disparate dates of the latest NWI data for each of the three states in the Delaware Estuary, total areas of tidal wetlands were considered separately by state. Figures 5.17-20 illustrate the status of wetland acreage based on the latest NWI data for each state. (Note - There is a very small portion of Maryland in the Delaware Estuary, but it is not considered here, particularly since it does not contain tidal wetlands. The New York portion of the Delaware River Basin is not considered here since it also contains no tidal habitat.)

5 - 1.3 Past Trends

It has been estimated that the Delaware Estuary has lost more than half of its wetlands, and more than 95% of our rare freshwater tidal wetlands, since early settlers arrived (PDE 2008). Historical losses occurred primarily because of development and conversion of wetlands for agriculture and other purposes. Despite increased regulatory oversight and "no net loss" policies that have greatly slowed rates of wetland conversion, we continue to lose all types of wetlands within the Delaware River Basin. Indeed, the pace of loss for some types of wetlands might actually be increasing due to a mix of factors (see below). The focus of this analysis was to examine trends in wetland acreage during the past two decades because we do not have data and information to carefully document earlier declines.

To assess trends in the extent of tidal wetlands in the Delaware Estuary, it is important that the data source and classification methodologies be equivalent so that meaningful comparisons can be made. While each state in the estuary has developed programs to map and categorize wetlands, comparing these data across time can be problematic due to differences in source data, interpretation, or methodology. Additionally, since each state has compiled state-wide data layers at different times using different methods, comparison across state boundaries is quite problematic.

Table 5.3. Classification of wetlands in the Delaware Estuary

Category	Code	Description
Saline, emergent vegetation	SAITEM	Typical “salt marsh” characterized by salt tolerant grasses. Predominant intertidal wetland type in the Delaware estuary.
Saline, other vegetation	SAITV	Vegetation other than salt-tolerant grasses, including scrub/shrubs and forest. Typical “high-marsh” habitat.
Saline, non-vegetated	SAIT	Non-vegetated intertidal area, mudflats, pannes, unconsolidated shoreline, beaches. Increases typically accompanies degradation of salt marshes, due to veg. loss, subsidence, and/or Sea level rise.
Fresh, emergent vegetation	FRITEM	Typical freshwater tidal wetlands characterized by emergent vegetation. Generally occur farther up the estuary, or landward of salt marshes in the lower estuary.
Fresh, other vegetation	FRITV	Freshwater tidal wetlands, scrub/shrub and forested wetland.
Fresh, non-vegetated	FRIT	Non-vegetated freshwater tidal wetlands, small portion of wetlands.

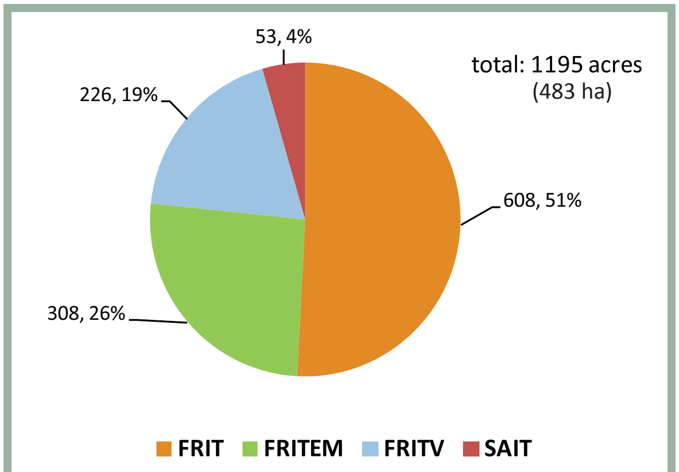


Fig. 5.17. Number of acres (ha) and relative percentage of different tidal wetland types within Pennsylvania based on most recent NWI data (see Table 5.3 for description)

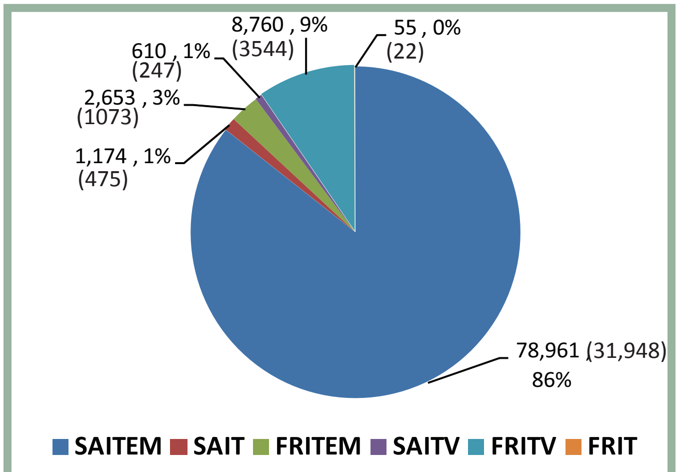


Fig. 5.18. Number of acres (ha) (and relative percentage of different tidal wetland types within New Jersey based on most recent NWI data (see Table 5.3 for description)

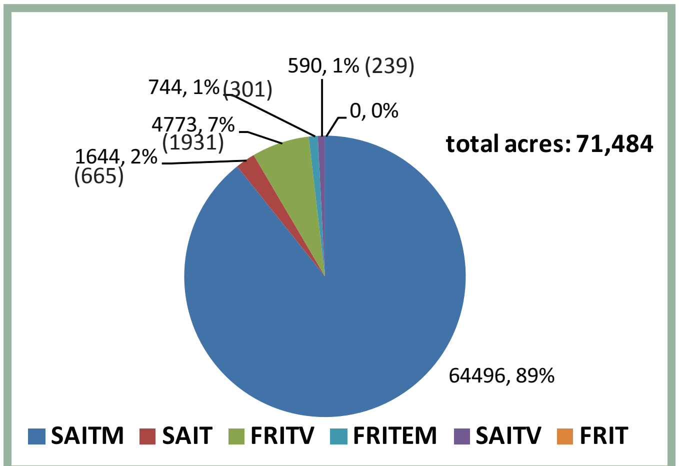


Fig. 5.19. Number of acres and relative percentage of different tidal wetland types within Delaware based on most recent NWI data (see Table 5.3 for description)

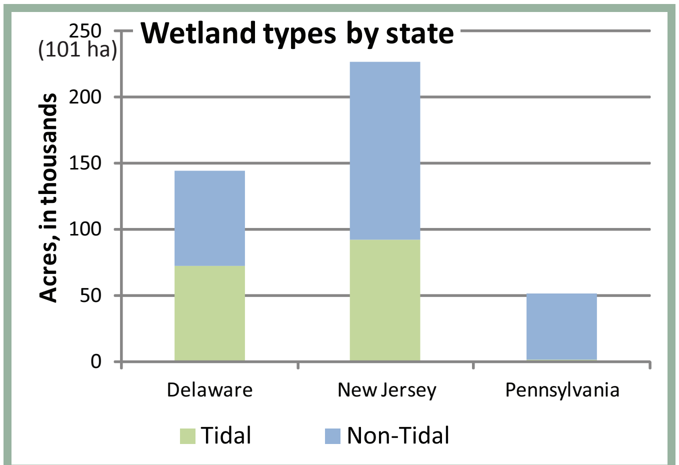


Fig. 5.20. Relative proportion of tidal and non-tidal wetland types within each of the three states in the Delaware Estuary

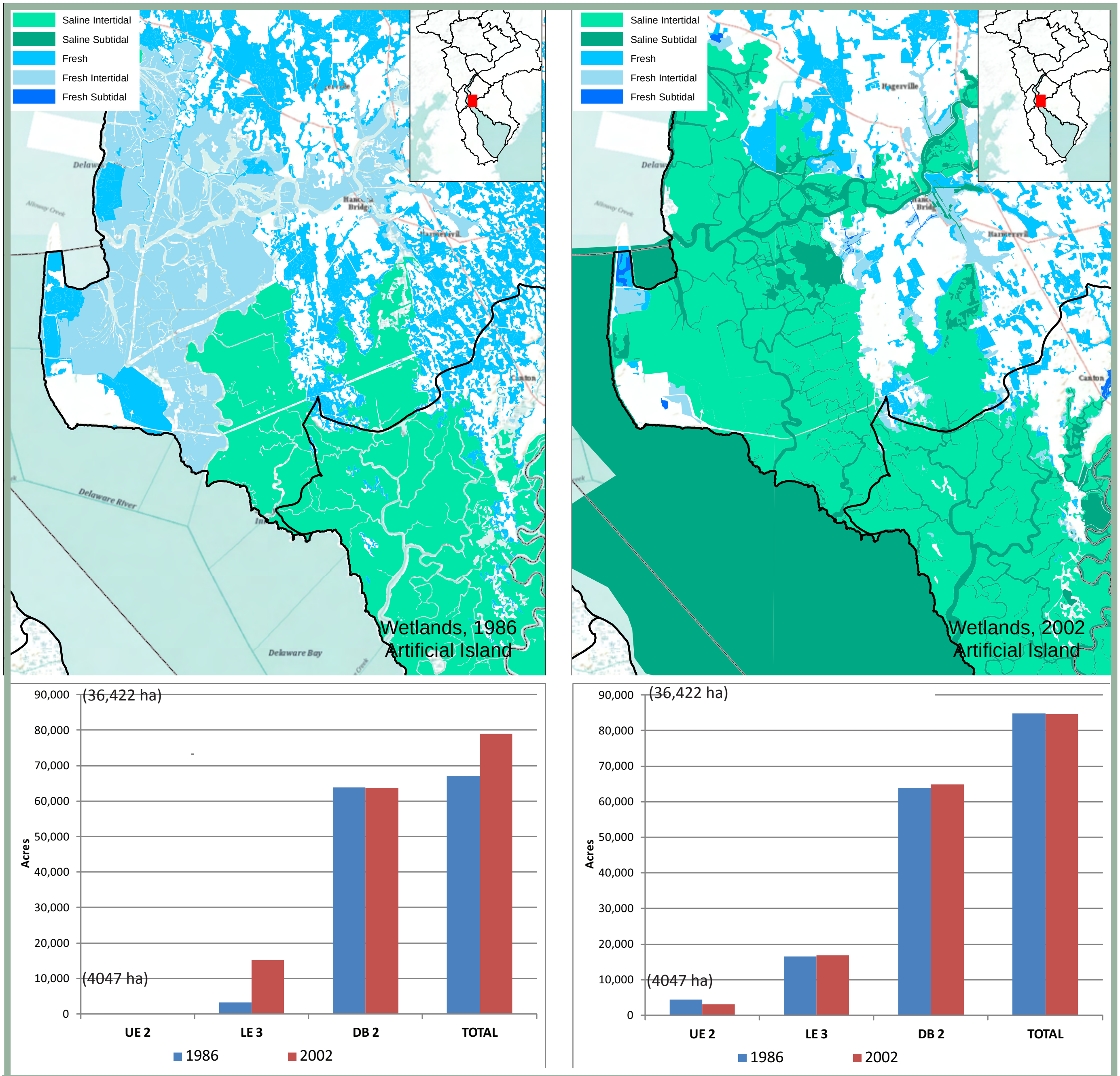


Fig. 5.21. Comparison of wetlands based on NWI classification from 1986 and 2002 for the Artificial Island area, NJ. The increaase in the relative proportion of salt marsh in the lower estuary (LE3) of NJ compare to the total tidal emergent wetlands, might reflect a transition from freshwater tidal marsh to salt marsh due to increasing salinity, or it might have resulted from methodological differences. (see map & key on p.133)

Figure 5.21 illustrates this issue. The two charts and maps depict categories of wetlands as identified in 1986 and in 2002, under the NWI program. This area falls near the typical salt line in the Delaware Bay, near Artificial Island, New Jersey. There is a lack of agreement in the delineation of salt marsh versus freshwater tidal wetlands, a difference which may or may not reflect a real change in wetlands of the estuary. The charts indicate that there appears to be an increase in salt marsh acreage in the Lower Estuary watershed of New Jersey (LE3) and a corresponding loss of freshwater tidal acreage, as can be seen in the maps. The chart showing the total amount of tidal wetlands (both fresh

Table 5.4. Categories of wetlands distinguished in NOAA CSC land cover datasets

Palustrine	Forested
	Scrub/Shrub
	Emergent
	Aquatic Bed
Estuarine	Forested
	Scrub/Shrub
	Emergent
Unconsolidated Shore	
Open Water	

Table 5.5. Change in acres of palustrine wetlands and salt marshes in the Delaware Estuary, 1996 to 2006, based on NOAA CSC C-CAP data. (see map & key on p.133)

Watershed	Palustrine Change	% Change	Salt Marsh Change	% Change
SV 1	36 (15)	463%*	0	--
SV 2	-185 (-75)	-7.7%	0	--
SV 3	-334 (-135)	-2.7%	0	4.9%
UE 1	-288 (-117)	-2.3%	-49 (-20)	-7.6%
UE 2	-514 (208)	-0.5%	-330 (-134)	-8.5%
LE 1	-229 (-93)	-2.3%	-72 (-29)	-5.2%
LE 3	-354 (-143)	-1.2%	-62 (-25)	-0.3%
LE 2	-109 (-44)	-1.3%	-251 (-102)	-2.3%
DB 2	-694 (-281)	-0.6%	-2110 (-845)	-3.4%
DB 1	-582 (-235)	-1.1%	-441 (-178)	-0.8%
TOTAL	-3252 (-1316)	-0.9%	-3316 (1342)	-2.2%

*Due to the small wetland acreage within SV1, this seemingly large percentage increase (from 7 to 43 acres) should be interpreted with caution because it likely falls within the assessment error range.

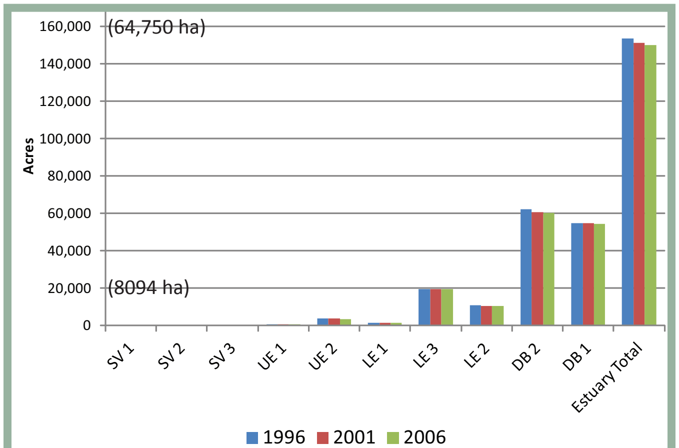


Fig. 5.22. Total acreage of estuarine emergent wetlands (salt marshes) in the watersheds of the Delaware Estuary, 1996 through 2006. (see map & key on p.133)

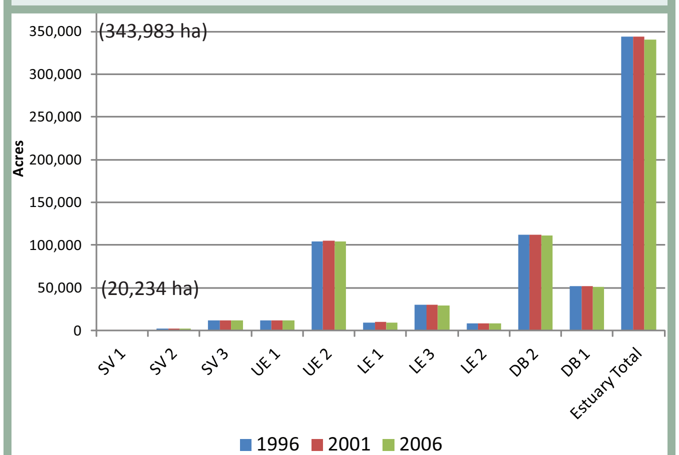


Fig. 5.23. Total acreage of palustrine (vegetated freshwater) wetlands in the watersheds of the Delaware Estuary, 1996 through 2006. (see map & key on p.133)

and salt), indicates that only a very small change in extent occurred when tidal wetlands are considered as a whole. The apparent transition from freshwater tidal marsh to salt marsh might have resulted from increasing salinity in this transition zone due to climate change and sea level rise (see also Chapter 7), but it is not conclusive because of uncertainty in NWI data comparability between the survey years.

To overcome these drawbacks, land cover data from the NOAA Coastal Services Center (CSC) Coastal Change Analysis Program (C-CAP) were compiled for the estuary, as noted above. These data are based on Landsat satellite multi-spectral imagery at a ground resolution of 30 meters. The CSC has derived land cover data for coastal Atlantic states for the years 1996, 2001, and 2006. While focusing on overall land use in the coastal zone, there is a relatively fine level of classification of wetland habitats (see Table 5.4). While the data from the CSC does not differentiate between tidal and non-tidal wetland categories, saline estuarine categories can be analyzed for changes over time. In particular, estuarine emergent wetlands correspond well to tidal brackish and salt marshes (Fig. 5.15).

Across the entire Atlantic seaboard between 1998 and 2004 it is estimated that wetlands have seen considerable losses due to natural and human-influenced causes. Freshwater vegetated wetlands have undergone a loss of 0.5% (from 13,254,960 acres/5,362,957ha in 1998 to 13,188,660 acres/ 5,336,132ha in 2004) (Stedman & Dahl 2008). Over the same period, estuarine emergent wetlands (salt marshes), declined from 1,842,320 acres/745,403ha to 1,822,780 acres/737,497ha, a loss of 19,540 acres/ 7,906ha, or 1.0%. Nationwide for the 6-year period, there was a 0.7% loss of vegetated estuarine wetlands (Dahl,2006).

Compared to these estimates, the rate of tidal wetland loss in the Delaware Estuary was similar or greater over a slightly longer 1-year time period (1996-2006), with a consistent decline in both freshwater wetlands (-0.9%) and salt marsh (-2.2%) (Table 5.5).

The largest losses of salt marsh were in the lower New Jersey bayshore (denoted as Delaware Basin 2, or DB2 in Fig. 5.21), which saw a decrease of 2,110 acres/854ha, or 3.4% (Table 5.5, Fig. 5.22-5.24). Delaware tidal salt marsh wetlands also underwent a significant drop in southern watersheds (LE2 and DB1). Palustrine wetlands (though not necessarily tidal) also saw a consistent decline across the estuary. Fig. 5.25 illustrates the trend for salt marsh (estuarine emergent) and palustrine (vegetated freshwater) wetlands acreage for the years 1996, 2001, and 2006.

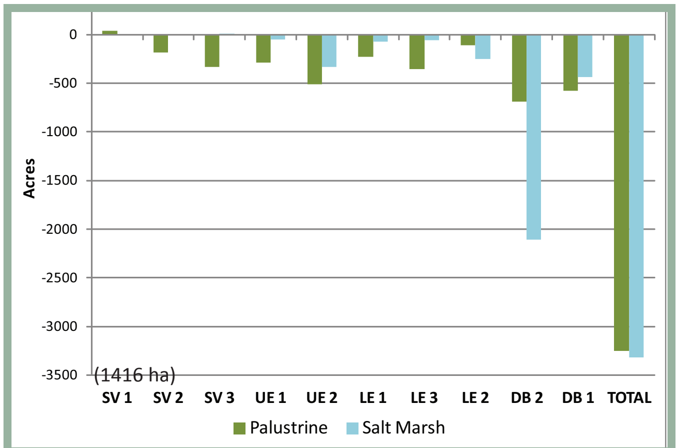


Fig. 5.24. Wetland acreage changes in different watershed regions by type, 1996 to 2006. (see map & key on p.133)

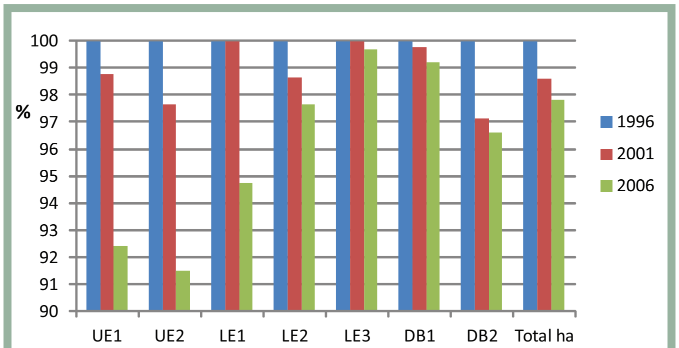


Fig. 5.25. Relative percent loss of wetlands by watershed between 1996 and 2006, with a 1996 baseline at 100%

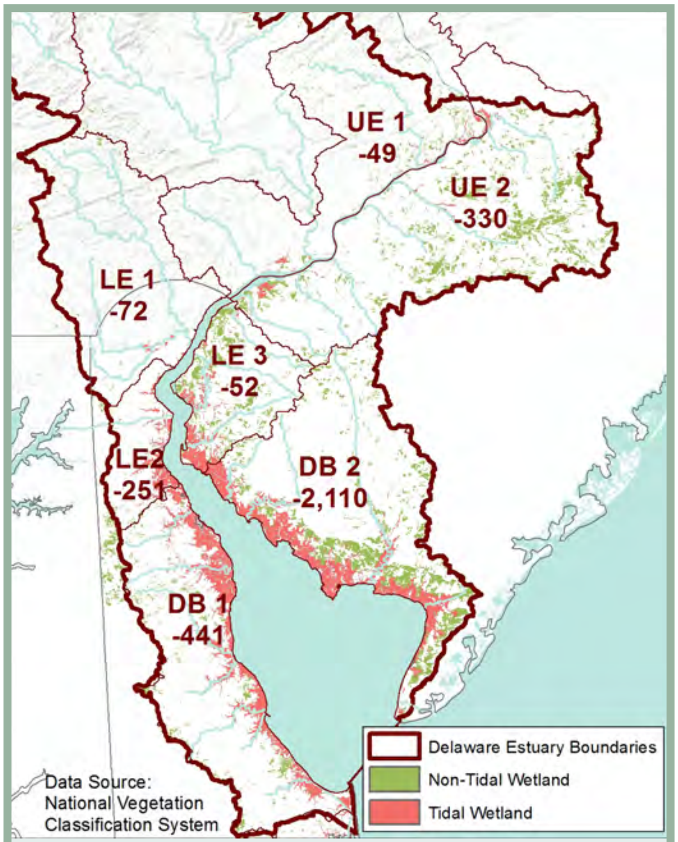


Fig. 5.26. Net change in estuarine emergent wetland acreage in watersheds of the Delaware Estuary, 1996-2006

Taken together, more than 3,200 acres/1295 ha of palustrine wetlands and more than 3,300 acres/1335 ha of salt marsh (estuarine emergent) wetlands were lost in the Delaware Estuary during this 11-year period (Fig. 5.24). The most rapid and sizeable losses occurred in the New Jersey Bayshore area (DB2) where 2,110 acres/854 ha were lost between 1996 and 2006. Percentage of acres lost can been seen in Fig. 5.25, and total acres lost are mapped in Fig. 5.26. These data are supported by on-the-ground observations of rapid, and apparently escalating, erosion and drowning of salt marshes in that area (Fig. 5.27).

Fig. 5.27. High rates of erosion are occurring throughout many areas of the Delaware Estuary as seen here within the Maurice River mouth, New Jersey, 2009

Although losses in the upper estuary are small in absolute terms, they are nevertheless important considering the small amount of tidal freshwater marsh habitat that remains, and their benefits to people, fish and wildlife, and water quality in the urban corridor. In 2009, the Partnership for the Delaware Estuary attempted to assess the health tidal wetland in Pennsylvania by visiting 30 sites at random that were characterized as tidal wetlands in the most recent NWI (1970s-1990s). It was found that many of these sites were no longer wetlands at all; 60 sites were visited until 30 could be found that were still tidal wetlands.This suggests that substantial losses of coastal wetlands continued to occur in recent decades (since NWI data were last collected).

There are many reasons why we continue to lose tidal wetlands in the Delaware Estuary. A recent examination of coastal wetland stressors (EPA 2011) blamed a mix of practices such as mosquito control ditching, continued incremental filling, lack of regulatory oversight, regulatory loopholes for developers, shoreline hardening, hydrological alterations such as dredging, and pollution. Increased rates of sea level rise and the spread of invasive species may also be contributing to the decline of coastal wetlands.

Most tidal wetland losses have converted to tidal open salt water. Nationally, 96.4% of tidal wetland losses were due to conversion to open water, with about 3.5% attributable to human effects in the upland areas (Stedman & Dahl, 2008). Wetland loss to direct human influence is relatively small, but the impacts, particularly on the quality of coastal ecosystems, have undoubtedly been significant. Over 53% of the U.S. population lives in coastal counties, which make up only 17% of the land area of the conterminous U.S. (NOAA Study, 2004). Development pressures and concomitant stresses on estuarine systems in these areas are considerable, and are likely to continue to increase.

As a result of lessons learned from Hurricane Katrina in the Gulf, in recent years attention has turned to assessing sediment dynamics in coastal estuaries and whether channel alterations, dredging, or sediment control projects in watersheds might contribute to coastal wetland losses by possibly starving them of needed sediments. Sea level rise is not a new phenomenon, as evidenced by Figure 5.18 which shows that the shoreline has been retreating with extensive marsh loss since at least the middle of the 19th century. What is not clear is the extent to which an increasing pace of sea level rise will hasten coastal change, possibly pushing tidal wetlands below their maintenance threshold within the tidal prism and relative to sea level.

To maintain themselves, tidal wetlands either need to accrete vertically to keep pace or they need to move horizontally into adjacent landward habitats. Marshes can accrete via the accumulation of organic matter produced *in situ* and/or the passive capture of suspended sediments originating from outside the marsh and brought in by tidal flushing (i.e., mainly rivers). The relative contribution of accumulated organic matter and trapped sediment varies widely from marsh to marsh, but without external sediment supplies most marshes fail to keep pace with sea level rise. Coastal Louisiana was losing a football field of tidal wetland every day for 30 years in part because sediment-laden freshwaters from the Mississippi had been diverted by channels to flow offshore, thereby creating a sediment deficit (Day and Templet 1989, Blum and Roberts 2009). The Delaware Estuary is similar in that it is a naturally muddy, wetland-rich system, and currently more sediments are removed each year through maintenance dredging than enter the system through surface runoff. Although there continues to be high levels of suspended sediments in the water column and the overall sediment budget (inputs and outputs) appears to be in balance (Walsh 2011), these sediment studies also suggest that the budget is currently balanced only because of large inputs of sediments from eroding tidal wetlands.

Another emerging concern is the effect of prolonged, high nutrient concentrations on tidal wetlands. Recent studies indicate that many wetland plants, especially dominant species in salt marshes, are naturally adapted for low nutrient levels and they invest heavily in belowground production of roots and rhizomes as a strategy for scavenging nutrients (Darby and Turner 2008, Turner et al. 2009). This strategy contributes to organic matter in the subsurface and aids in peat accumulation. Nutrient loadings may alter this strategy, resulting in higher ratios of aboveground:belowground production, potentially impairing a marsh's ability to accrete and keep pace with sea level rise. A tell-tale sign of this phenomenon is the presence of taller growth forms of usually short marsh plants, such as *Sparta alterniflora*. across the marsh plain. Paradoxically, a marsh can look its healthiest just before it drowns. Velinsky et al. (2011) reported that many tidal marshes in Barnegat Bay, New Jersey have been significantly degraded by excess nutrients based on careful analysis of diatom chronologies from marsh cores, and areas with highest nutrient loadings are most vulnerable to sea level rise. Increased nutrients can also cause hypertrophic and low-oxygen conditions, affecting the delicate habitats of the marshes and near-shore aquatic beds of the estuary. Since the Delaware Estuary has some of the highest nutrient loadings of any

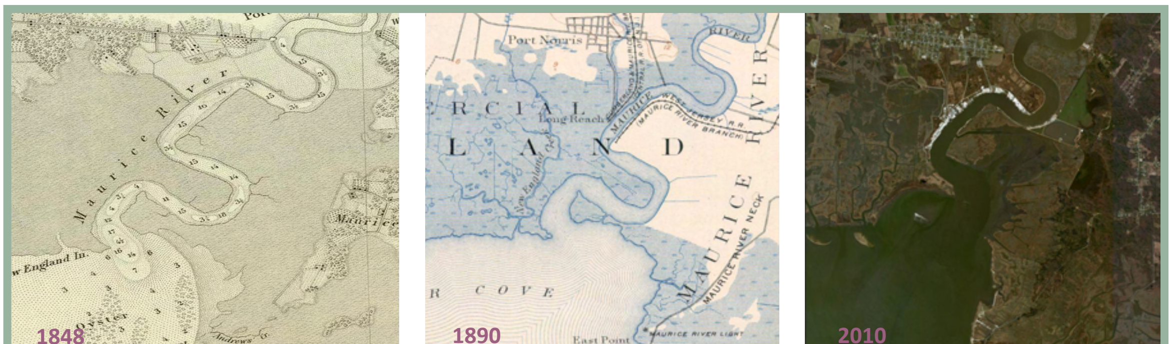


Fig. 5.28. Loss of coastal wetlands in the vicinity of Port Norris and Bivalve, New Jersey, 1848 to present. Although wetland loss has been occurring for a long time, rates of loss may be increasing thereby jeopardizing the safety and economies in coastal towns where these habitats provide flood protection and sustain coastal shellfisheries, fisheries, and ports

coastal area of the United States, it is plausible that these nutrients, especially nitrogen, might be contributing to tidal wetland losses.

Across the country, it has been reported that there has been a net increase in wetlands of approximately 32,000 acres/ 12,947 ha per year between 1998 and 2004 (Stedman and Dahl 2008). Most of these gains, however, are in inland wetland categories, particularly ponds (many on farms). These are not of the same ecological value as natural, vegetated tidal wetlands, and do not provide the same hydrologic and ecosystem services. High quality tidal wetlands, such as those that naturally exist in the Delaware Estuary, are among our nation's most valuable and productive ecosystems.

5 - 1.4 Future Predictions

As discussed above, about half of the pre-settlement acreage of tidal wetlands remain in the watershed, and losses continue to mount every day. Stressors that have contributed to historic and recent losses of tidal wetlands have not gone away in the Delaware Estuary. These include impacts associated with development, including: pollution, shoreline hardening, filling, dredging, ditching, boat wakes, etc. Since the human population is expected to expand by about 80% by 2100 within the Delaware Estuary (PDE 2010), the direct conversion of wetlands for development and the associated environmental pressures by the expanding populace are likely to continue to stress our tidal wetland habitats.

Perhaps even more importantly, increasing rates of sea level rise and associated salinity rise pose mounting threats to tidal wetlands. Although there are limited quantitative data, coastal managers and scientists in Delaware and New Jersey report increasing rates of erosion of seaward marsh edges and rapidly expanding interior open water. Riter and Kearney (2010) reported similar findings from satellite imagery, which suggest that most marshes in the system are showing decreasing amounts of vegetative cover and increasing proportions of open water. Their effort updated the earlier study by Kearney et al. (2002) of both Chesapeake and Delaware Bays, which suggested that more than two-thirds of salt marshes were in a degraded condition.

If the intensity and frequency of storms and associated tidal surges also increase with climate change, this could exacerbate the other threats. Warming trends are expected to boost the incidence of coastal storms, including northeasters and possibly hurricanes. On the other hand, a longer growing season, enriched atmospheric carbon dioxide, and warmer temperatures are likely to enhance primary productivity within wetlands.

In 2010, the Partnership for the Delaware Estuary released a report on the most important changes that are likely to occur as a result of climate change (PDE 2010). Tidal salt marshes were predicted to be highly vulnerable to increasing rates of sea level rise and freshwater tidal wetlands were reportedly highly threatened by salinity rise, among other factors. A panel of wetland experts

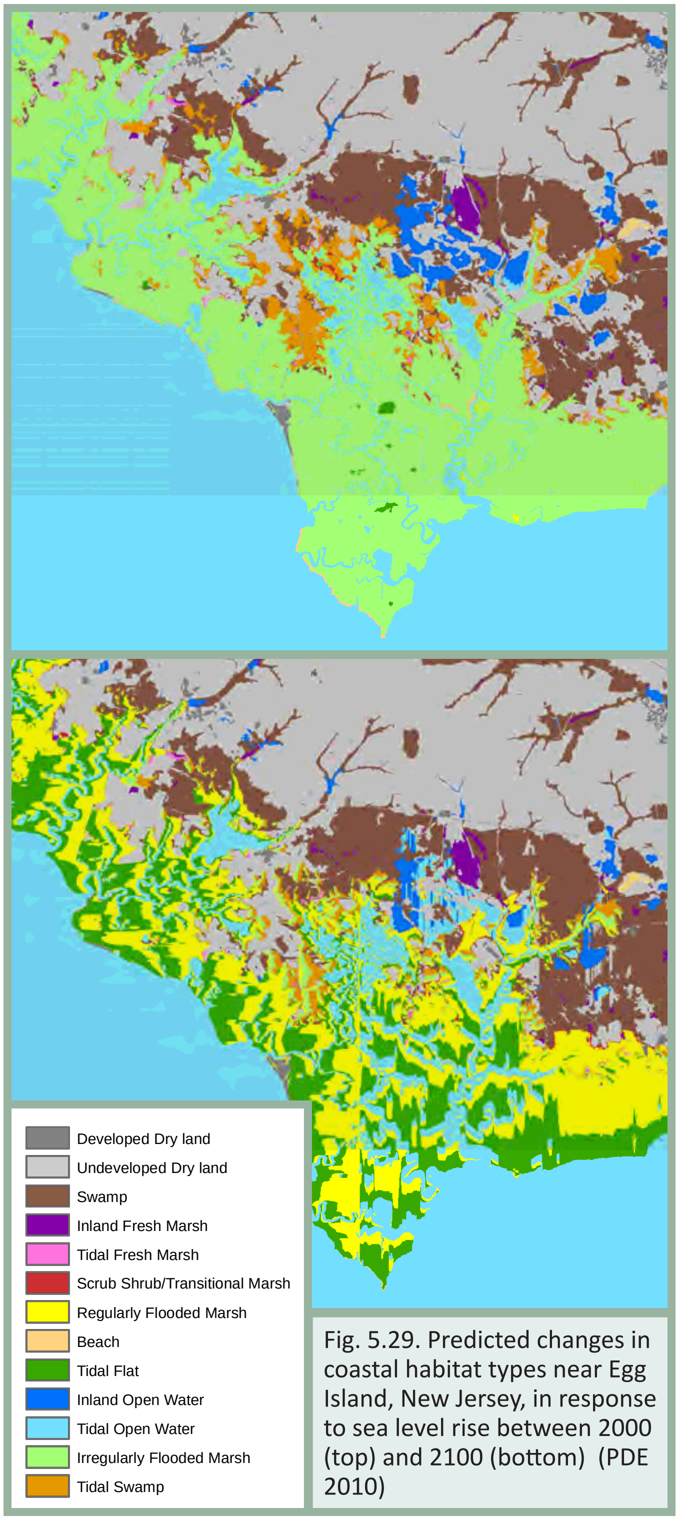


Fig. 5.29. Predicted changes in coastal habitat types near Egg Island, New Jersey, in response to sea level rise between 2000 (top) and 2100 (bottom) (PDE 2010)

predicted that the potential boost to primary production would be dwarfed by the threats posed by sea level and salinity rise (Kreeger et al. 2010).

Moreover, all tidal wetlands face barriers to landward migration within the Delaware Estuary, most significantly in the upper estuary (see PDE 2008, Feature Box). The potential for tidal wetlands to migrate landward is affected by slope, soils, and degree of hardening. Areas with high levels of upland development and shoreline hardening do not allow wetlands to easily migrate landward and thus maintain themselves. In many areas they will need to accrete in place, or face drowning.

With a rise in sea level of one meter by 2100, more than 25% of the system's tidal wetlands are predicted to be lost (PDE 2008). Based on model predictions from the Sea Level Affecting Marsh Model (SLAMM, V.6), this amounts to more than 50,000 acres (20,234 ha) of net loss, resulting from the balance between the landward migration of tidal wetlands into adjacent uplands and non–tidal wetlands (which are expected to >50,000 acres/ 20,234 ha) and a seaward erosion and drowning of tidal wetlands (expected loss of >100,000 acres (40,469 ha). Importantly, since no other habitat types rival tidal wetlands in productivity, the net loss of ecosystem services is expected to be proportionally far more significant than the acreage loss. Based on recent loss trends and revised sea level rise scenarios, we expect total net losses of tidal wetlands by 2100 to exceed 25% (PDE 2010) and perhaps 75% if no action is taken to stem loss. In addition to net losses of acreage, most high marsh in the Delaware Estuary is predicted in this report to convert to low marsh even if it is not eroded.

Sommerfield and Velinsky (2011) reported that accretion rates in tidal marshes are currently greater than rates of sea level rise at sites they studied in the Delaware Estuary. Nevertheless, the Delaware Estuary is experiencing a net loss of these same habitat types. Plausibly, the erosion and loss of some wetlands might be helping to sustain others by subsidizing the sediment supply, but the net balance is still negative per year as determined by decreasing acreage, shoreline retreat, and lower overall vegetative cover.

The current rate of sea level rise in the Delaware Estuary is between 3.5-4.0 millimeters per year, up from about

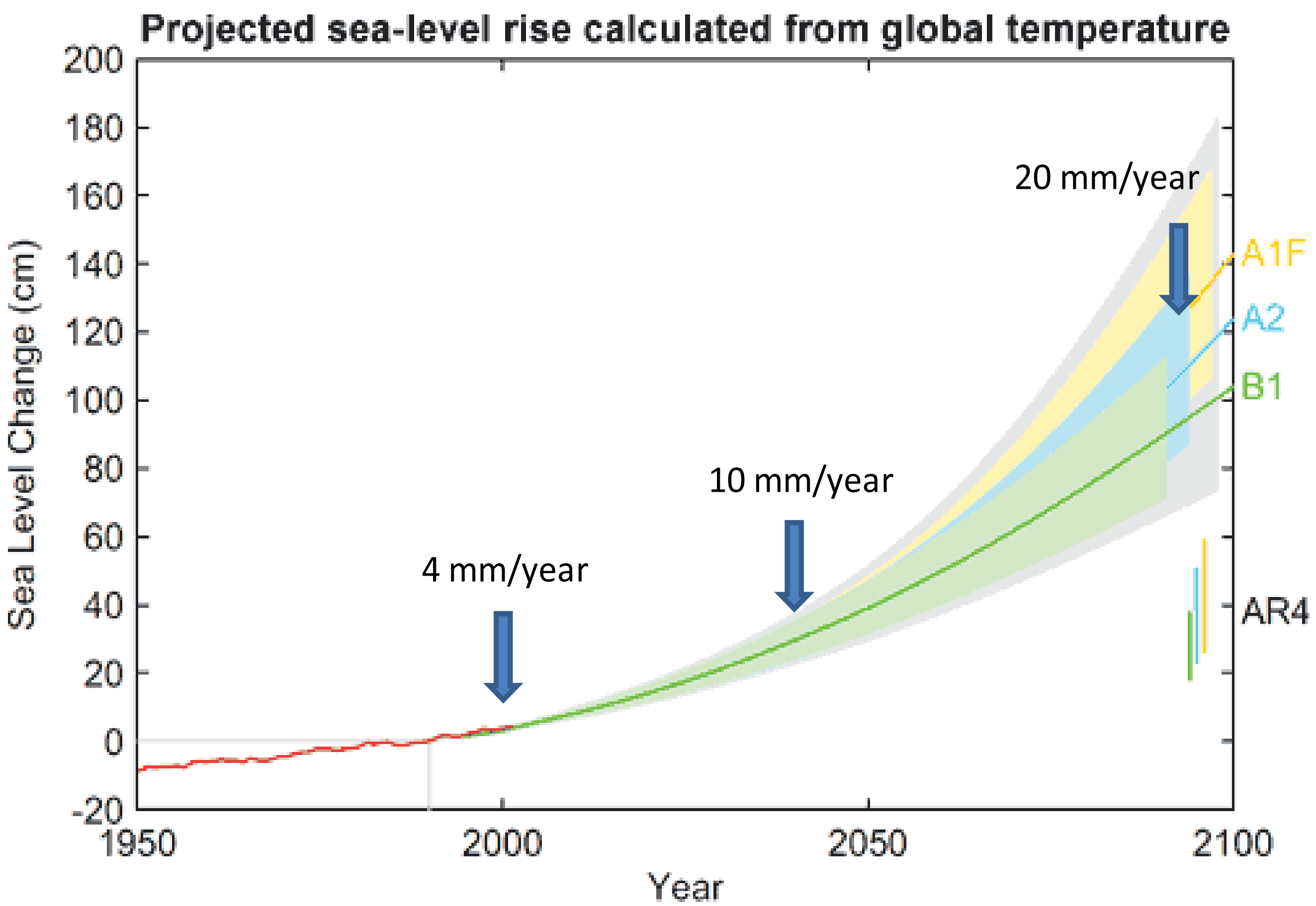


Fig. 5.30. Projected sea level rise calculated from global temperature based on 3 different emissions scenarios (Vermeer & Rahmstorf 2009) with extrapolated rates of sea level rise, assuming a total rise of 1.3m in the Mid-Atlantic region by 2100

1.8 millimeters per year in the early portion of the 20th century (Gill et al. 2011). A one meter rise in sea level by 2100 will require the rate of sea level rise to eventually exceed 10 millimeters per year. The last time that the rate of sea level rise was that high was during the period of post-glacial ice melt up through about 2000 years before present, and during that time period, tidal wetlands were rare along the eastern seaboard, existing only in the most protected areas (Psuty 1986, Psuty and Collins 1996).

In addition, the land is sinking in many areas of the coastal plain due to subsidence from post-glacial rebound. Rates of subsidence appear to be greatest in South Jersey (Sun et al. 1999) where the largest tidal wetland losses have occurred. The interplay between sea level rise and subsidence, compounded by changes in ocean currents (Gulf Stream; see Najjar 2010), will result in greater rates of local "relative" sea level rise than the global forecast models predict. For these reasons, in climate adaptation planning at the Partnership for the Delaware Estuary is expecting 1.3-1.4 meters of relative sea level rise for every 1.0 meter of global sea level rise. To reach 1.3 meters within 90 years, the average annual sea level rise would be 14.4 millimeters if the increase was linear over this period, which it is not. Therefore, the annual rate of sea level rise at the end of the century is likely to be far greater than 14 mm unless significant errors exist in this forecast, or the rise in the rate of sea level slows for other reasons.

Clearly, the rate of relative sea level rise (RSLR) is critically important for determining the fate of tidal wetlands in the Delaware Estuary because of the tipping point that can be breached when the RSLR exceeds the marsh accretion rate. Assuming that this threshold is somewhere between 5-10 millimeters per year for many salt marshes of the Delaware Estuary, and assuming RSLR will reach 1.3 meters by 2100, then a non-linear increase in sea level at the projected rate would likely breach the tipping point within the next 20-25 years for a large proportion of tidal wetlands in the system unless significant actions are taken to aid the vertical accretion of tidal wetlands.

5 - 1.5 Actions and Needs

Sea level rise, salinity rise, development, outdated management paradigms, and pollutants are likely to contribute to the continued degradation and loss of tidal wetlands in the Delaware Estuary unless actions are taken to abate these impacts. Future indicator reporting would also benefit from better monitoring data on tidal wetland extent and condition.

<u>Proactive Adaptive Management</u>

Despite the dynamic nature of the coastline, many regulatory policies continue to treat the landscape as fixed in place. Restoration paradigms set goals based on historic conditions rather than future sustainability. As sea level rises it will be important to update management policies to encourage both the landward migration of tidal wetlands into buffers (Feature Box) and the vertical accretion of tidal wetlands in place (Fig. 5.32). It is still much easier to obtain a permit for a shoreline stabilization project that installs a bulkhead or other hard structurestructure that prevents wetlands from keeping pace with sea level rise and contribute to degradation of tidal wetlands, than it is for a living shoreline (Fig. 5.32). Ditching and filling of tidal wetlands still occur, often without proper monitoring of the effects or understanding of the consequences. To adapt to both climate change and continued watershed development, tidal wetland managers will need to adjust targets, policies and tactics to sustain existing tidal wetland habitat in the future.

In order to address the threats to the intertidal zone in the Delaware Estuary, an approach combining policy and regulatory remedies and actions on the ground is required. The Clean Water Act (1972), Coastal Zone Management Act (1972), and the Coastal Barriers Resources Act (1982), are evidence of the increasing importance of tidal wetlands in the policy and legal arena. Many states and counties have followed the lead of federal agencies and implemented their own regulations covering wetland protection measures such as buffer requirements, impervious cover limitations, and implementation of federal nutrient pollution guidelines. Continued promulgation, refinement, and enforcement of regulations and policies is a critical need, as demonstrated by the various emergency measures that are already underway or being called for in some Delaware and New Jersey areas (e.g. Prime Hook, Delaware; Sea Breeze, New Jersey; Maurice Township, New Jersey) where tidal wetland losses are contributing to the decline of coastal communities. Given accelerating development and population pressures, as well as increases in relative sea level rise, these measures will need to be augmented just to maintain the current integrity of the intertidal zone. In particular, local differences in the extent of regulatory protection provided to wetlands poses a challenge to maintaining consistently high level of wetland quality and function throughout the estuary.

<u>Monitoring Data and Scientific Study</u>

Complete and consistent monitoring data on wetland is a vital need to allow managers to make proper decisions and to enable assessment of wetland status and trends. Such data allows scientists and policy makers to understand the causes of wetland loss and develop approaches to address them. As discussed above, it is still impossible to accurately and consistently report changes in tidal wetland extent because of limited, sustained investment in monitoring. The National Wetlands Inventory is a program designed to address this issue, but differences in the procedures and time frames have made long-term trend analysis problematic. The State of Delaware has developed high-quality datasets, but comparison to New Jersey is not possible. Some areas of Pennsylvania have not been assessed for the NWI since the 1970s. Therefore, basin-wide coordination of NWI assessments is crucial, as is the need to update inventories at least every 5-7 years.

Danielle Kreeger

Fig. 5.31. Scientists from the Academy of Natural Sciences, Partnership for the Delaware Estuary, and Rutgers University installing a surface elevation table in a salt marsh in the Dennis Creek watershed, New Jersey, in March 2011 as part of a new sub-regional monitoring initiative targeting tidal wetlands: the Mid-Atlantic Coastal Wetland Assessment

Since the array of ecosystem services furnished by tidal wetlands are proportional to their condition, better health assessments are also needed. For example, restoration and mitigation targets are based on acreage, and realizing small increases in acreage can be very costly; however, investment in enhancement projects (e.g., living shorelines to stem erosion, beneficial use of dredge material to raise elevation) that boost function and save much larger tracts from being lost might yield greater net value (and acres) in the long run. More scientific studies and restoration pilot projects would contribute to knowledge and strengthen management and restoration practices to sustain greatest tidal wetland acreage.

Investment in consistent tidal marsh monitoring and science is difficult to fund at the scale of the multi-state Delaware Estuary. However, the benefits of tidal wetlands are beginning to be captured and capitalized upon (e.g. flood protection, nutrient and carbon capture, fish production). Tidal wetlands are already regarded as the most valuable natural lands (e.g. NJDEP 2007). Managers should carefully consider how a projected loss of 25-75% of the tidal wetlands in the Delaware Estuary might affect coastal communities (lives and property) and regional economies (fisheries and shellfisheries, property values, nutrient criteria). As markets for ecosystem services develop in the future, there will be increasing demand for essential information on trends in tidal wetland extent and condition. Such information will be vital in the development of strategies to protect and enhance tidal wetlands. Until then, there will continue to be a need to collaborate and leverage funds to fill vital information gaps.

<u>On-the-Ground Action</u>

Efforts at preservation, both through regulatory and physical means, have been having some beneficial impacts across the estuary, but many areas are still undergoing degradation or conversion to open water. New policies and tactics are needed to both facilitate the horizontal, landward migration of tidal marshes and to boost the health and vertical accretion of tidal marshes. Given the rapid pace of change in tidal wetland extent and health, swift action to physically protect or enhance tidal wetlands is warranted to stem losses, even if monitoring and scientific information are still in the development phase. Implementation of Best Management Practices (BMPs) using best possible scientific information has already been shown to help to protect tidal wetlands from landward threats such as nutrient loading, sediment deficits, and contamination, both in agricultural and developed areas. Marsh migration plans are needed and will require conflict resolution and education. Seaward protections and marsh enhancements can be just as difficult to implement due to permitting, logistical, and funding challenges. However, there are efforts underway to explore beneficial use of sediments for enhancement (Bailey-Smith 2011), develop new living shoreline tactics appropriate for the Delaware Estuary (Fig. 5.32) (Kreeger et al. 2009; Whalen et al. 2011), and craft an estuary-wide strategy for living shoreline implementation (e.g. Delaware Estuary Living Shoreline Initiative).

PDE PDE

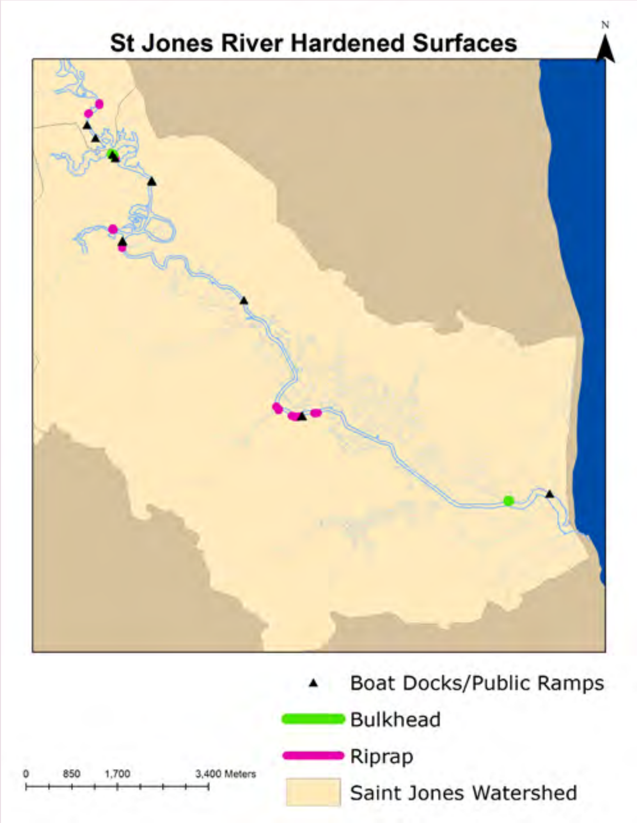
Danielle Kreeger Josh Moody

Fig. 5.32. Installation of a mussel and plant-based living shoreline to help stabilize erosion and improve ecological value of a formerly hardened shoreline at Matt's Landing, New Jersey. This new tactic was developed jointly by the Rutgers Haskin Shellfish Research Laboratory and Partnership for the Delaware Estuary. The photo from Sept 2011 was following Hurricane Irene and Tropical Storm Lee

5 - 1.6 Summary

Tidal wetlands of the Delaware estuary are some of the most productive habitats in the world, and they arguably represent the most ecologically and economically important type of natural habitat in the entire Delaware River Basin. By their very nature, they are transient within the dynamic coastal zone. They absorb tidal energy from the open marine environment, and provide a buffer and sink for contaminants from upland areas. They also provide essential habitat for a wide range of organisms, as well as recreational opportunities for people. As long as the intertidal zone remains in a state of dynamic equilibrium, the benefits that they provide are maintained. However, when the processes which threaten the viability of the intertidal zone come to predominate over the processes which maintain equilibrium, this delicate ecosystem becomes unstable and imperiled. Current trends suggest that tidal wetlands, and hence the ecosystem services and direct financial and aesthetic benefits they provide, are being degraded and lost across all areas of the Delaware Estuary, especially salt marshes around Delaware Bay. Future projections suggest that these losses will increase, perhaps rapidly, likely resulting in a dramatic shift in the character and function of the estuary ecosystem. More study and monitoring, along with proactive management and on-the-ground actions, are urgently needed to minimize ongoing losses since no type of replacement habitat will provide the same net level of ecosystem services as these vital coastal areas.

Hardened Shorelines in the St. Jones (by Kelly Somers)



Shoreline armoring or hardening occurs when non-natural structures are added to a shoreline to offset erosion processes. Examples of hardened structures include bulkheads or rock erosion control such as riprap. Recreational structures (such as docks and piers) also can impact the shoreline's natural habitat. Shoreline hardening alters the structural and functional ecology of the wetlands. Hardened shoreline structures disturb natural shoreline processes (such as sediment exchange) and can lead to increased erosion at the base or downdrift from the structure. Hardened structures also do not allow for natural habitats to migrate inland due to sea level rise and flooding (Castellan et al, 2006). An increasing body of scientific evidence indicates that hardened structures particularly bulkheads, are poor habitats for fish and other biota, in comparison to natural edge habitats which function as biological hot spots.

The St. Jones River Watershed is located southeast of Dover, Delaware. In 2007, with collaboration from Delaware Coastal Programs and NOAA, the Virginia Institute of Marine Sciences completed a shoreline inventory of the St Jones River. Recreational structures and erosion control hardened structures were mapped using a handheld GPS unit to determine their extent and location on the river channel. The project looked at various types of structures including bulkheads, docks, piers, boat ramps, and riprap. The data and report show that of the 43.9 kilometers of shoreline along the St. Jones River, 64 meters were bulkhead (in green) and 499 meters had rip rap (in pink). Hardened recreational structures are found along the main channel; 11 docks and 2 boat ramps (black triangles) (Berman et al, 2008).

As an alternative to shoreline hardening, the Partnership for the Delaware Estuary suggests living shorelines, a more natural erosion control method that also enhances ecological conditions, such as by incorporating native plants, reef-building animals, and structural complexity into shoreline protection projects. These tactics enhance the natural landscape, ultimately providing more habitat for plants and animals.

5 – Non-Tidal Aquatic Habitats

5 – 1 Freshwater Wetland Acreage

Non-tidal wetlands, including forested and shrub swamps, bogs, fens, vernal pools, and riverine wetlands , provide habitat for a diverse array of terrestrial, aquatic, amphibian, and bird species (Davis 1993, Mitsch and Gosselink 2000, Faber-Langendoen et al 2008). Wetlands also serve many hydrologic, biogeochemical, and habitat functions, which are strongly influenced by watershed position (Brinson et al. 1995). Headwater wetlands retain and store precipitation, recharging groundwater resources. They are important sources of water and organic and inorganic materials that support downstream aquatic systems. Riverine and floodplain wetlands can store overbank flows, dissipate energy, provide a local supply of large woody debris, and both supply and retain coarse particulate organic matter. Wetland size, density, and landscape context, including condition of adjacent lands and connectivity among riverine, wetland, and upland habitats, are important indicators of condition.

Large wetlands are critical for maintaining suitable habitat for many of the priority species within the state wildlife conservation plans. For example, the Pennsylvania Comprehensive Wildlife Conservation Strategy (CWCS) emphasizes that conservation of large wetland habitat is especially critical for wildlife conservation (PGC and PFBC 2005). While the CWCS definition of “large wetlands” depends on the wetland type and species of concern, it typically defines large wetlands as between 12 and 100 acres (5 and 40 ha) (or larger).

Separating non-tidal wetlands highlights the value and significance of these systems, which have experienced significant losses in the basin. For example, in the state of Delaware more wetlands were lost between 1992 and 2007 than in the previous 10 years; approximately 99 percent of those losses were to non-tidal/freshwater wetlands (Environmental Law Institute 2010).

5 – 1.1 Description of Indicator

Headwater wetland area and the number of large contiguous headwater wetlands (greater than 100 acres/ 40 ha) were calculated for each subbasin within the Delaware Basin. Together, these serve as potential indicators of the degree to which wetlands are providing critical functions in headwater regions, including recharging groundwater and storing and releasing water and organic and inorganic materials to support downstream aquatic systems.

Fig. 5.33. Riverine and headwater wetlands within the Rancocas Creek watershed, New Jersey.

Non-tidal wetlands were defined by first selecting the woody and emergent wetland land cover classes from the National Land Cover Dataset (NLCD 2001). Open water features such as ponds, lakes, and reservoirs were not included. Non-tidal wetlands were then classified according to the National Vegetation Classification System (NVCS) (Westervelt et al. 2006) and further separated into headwater and riverine wetlands (Fig. 5.33). Riverine wetlands were associated with the floodplains of rivers with drainage areas greater than approximately 40 square miles (10,359 ha). Headwater wetlands exist along the riparian corridors of streams with drainage areas less than approximately 40 square miles (10,359 ha).

Within headwaters, contiguous headwater wetlands were defined as areas with connected wetland landcover (i.e., woody or emergent wetland pixels that are connected on a side or on the diagonal). These contiguous wetlands potentially include multiple wetland types according to various existing classifications, but the overall size is one indicator of potential wetland function. The total area of each contiguous headwater wetland was calculated.

5 – 1.2 Present Status

Figure 5.34 illustrates the total headwater wetland area and the number of contiguous headwater wetlands larger than 100 acres (40 ha) within each subbasin. Despite wetland losses, the Delaware River watershed has several subbasins with abundant headwater wetlands. Noteworthy concentrations are located in the Upper Central and Lehigh Valley subbasins and on the coastal plain within Upper and Lower Estuary and Delaware Bay subbasins.

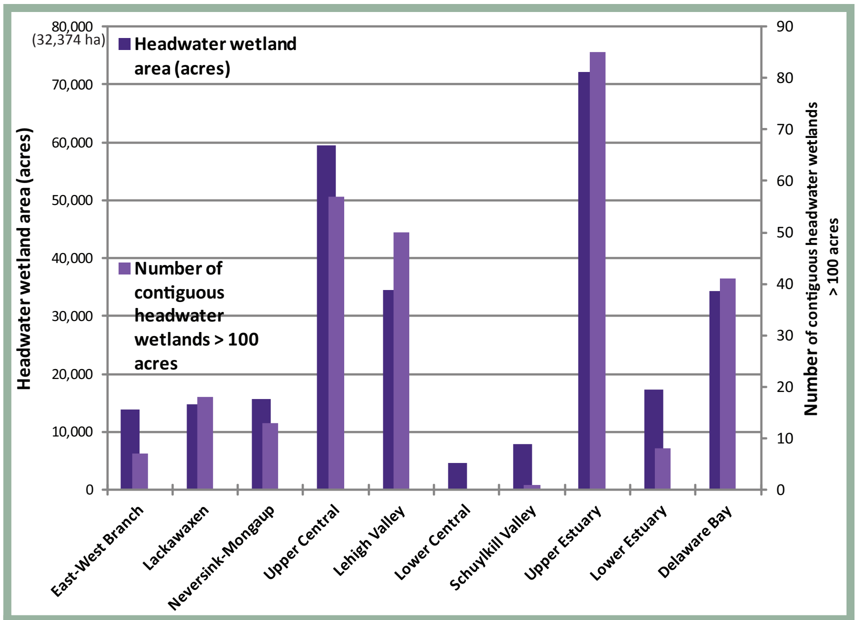


Fig. 5.34. Total headwater wetland area ranges from approximately 4,500 acres (1821 ha) in the Lower Central subbasin to over 72,000 acres (29,137 ha) in the Upper Estuary subbasin. The Upper Estuary subbasin also has 85 headwater wetlands that are larger than 100 acres (40 ha). This is the highest number of any subbasin in the Delaware River watershed.

Both the Upper Central and Lehigh Valley subbasins contain at least 50 headwater wetlands that are larger than 100 acres (40 ha). These subbasins also overlap with the glaciated portions of the Pocono Plateau, which includes the greatest diversity of wetlands in the state of Pennsylvania (Davis 1993). Boreal conifer swamps, oligotrophic kettlehole bogs, cranberry and bog-rosemary peatlands, and acidic broadleaf swamps occur throughout the region. Other unique wetland communities are found along the limestone valley, where mineral-rich groundwater supports calcareous fens, seepage swamps, and limestone wetlands. Cherry Valley National Wildlife Refuge and the Mt. Bethel Fens in Pennsylvania and the Johnsonburg and Sussex Swamps in New Jersey contain examples of these systems. Vernal pools are also scattered throughout the region, with concentrations along the toeslopes of the Kittatinny Ridge.

Although the Upper Estuary subbasin includes Trenton and Camden, NJ, Philadelphia, Pennsylvania, and other urban and suburban areas, this watershed contains over 70,000 acres (28,322 ha) of non-tidal wetlands and 85 wetlands larger than 100 acres. These headwater wetlands are especially abundant on the coastal plain in New Jersey, including along Crosswicks Creek and the North and South Branch Rancocas Creek.

5 – 1.3 Past Trends

Wetlands slow down, capture and cleanse rainwater before releasing it to rivers, oceans, lakes and groundwater. They shelter wildlife and provide breeding and spawning grounds for commercial and recreational fisheries. They store stormwater, releasing it slowly to help prevent floods, and support recreational activities.

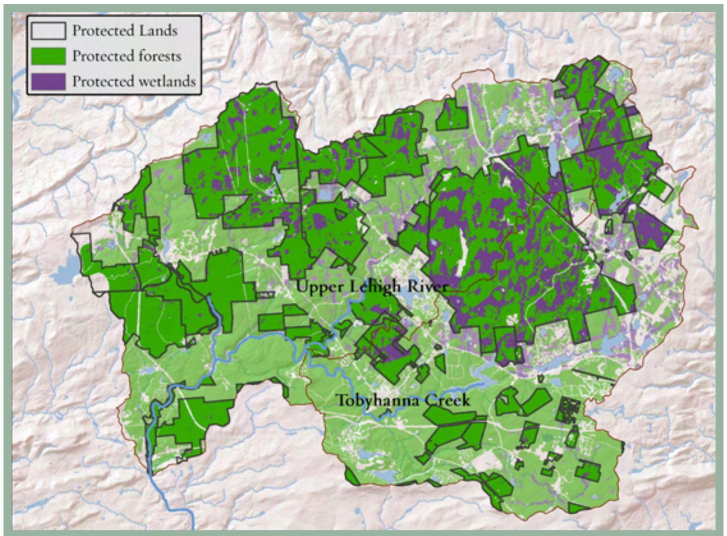


Fig. 5.37. Headwaters within the upper Lehigh Valley subbasin include extensive forests and wetlands within the riparian corridors. Much of this area is also in protected lands.

Yet for much of our history, wetlands have been undervalued. By the mid-1980s half the wetlands in the continental U.S. had disappeared, with losses averaging 500,000 acres (202,343 ha) per year. Regulations to control wetlands loss existed, but were often slow, unpredictable, expensive and frustrating for land owners.

In the summer of 1987, at the request of Lee Thomas, Administrator of the U.S. Environmental Protection Agency, The Conservation Foundation convened the National Wetlands Policy Forum, chaired by Governor Thomas H. Kean of New Jersey, to address major policy concerns about how the nation should protect and manage its valuable wetlands resources.

The goal of the Forum was to develop sound, broadly supported recommendations on how federal, state and local wetlands policy could be improved. In late 1988, the Forum published its final report, a 70-page consensus document that presented approximately 100 recommendations on a variety of issues including promoting private stewardship, improving regulatory programs, establishing government leadership and providing better information. Among the key recommendations was that national policy be guided by a goal of "no overall net loss" of the nation's remaining wetlands and, over the long term, to increase the quantity and quality of the nation's wetlands resources.

This goal has guided national wetlands regulatory and non-regulatory programs and policy ever since.

In the years since the Wetlands Forum, the rate of wetlands loss in the U.S. has slowed dramatically to the point where achieving the goal of "no net loss" may be in sight. This is truly a remarkable accomplishment.

Private land owners have made a major contribution, in recent years enrolling an average of 200,000 acres per year in the national Wetlands Reserve Program, one of the programs recommended by the Forum. Total acreage in the program now exceeds a million acres.

Federal and state agencies stepped up and provided increased leadership in numerous ways and in every Administration since the Forum's recommendations, improving regulatory programs and providing better information. Shortly after the Forum's report, EPA and the Army Corps signed a Memorandum of Understanding to better coordinate regulatory programs, reducing confusion for landowners.

5 – 1.4 Future Predictions

While filling and conversion of wetlands for agricultural and urban development has generally decreased over time, different stressors in the form of new industrial development seeking a location in small headwater watersheds will have to be carefully managed. In addition, it is likely the precipitation patterns of the next 100 years will be more extreme than the past, resulting in changing water budgets at a watershed scale and even greater ecosystem service values attributed to freshwater wetlands in the future.

5 – 1.5 Actions and Needs

Many positive actions are underway and require continued vigilance by Basin management community:

1. Continued attention to quantifying ecosystem service values.

2. Continued attention to harmonizing state and federal regulatory programs.

3. Continued attention to funding conservation initiatives and wetland reserve programs.

4. Continued effort to quantify feedback loops like the USDA Conservation Effects Assessment Program.

5. Passage of the Delaware River Basin Conservation Act of 2011-- championed by Senators Carper and Coons of Delaware, Senator Schumer and Gillibrand of New York, and Senators Menendez and Lautenberg of New Jersey-- which would establish a federal program at the U.S. Fish and Wildlife Service to coordinate voluntary restoration efforts throughout the Delaware River watershed.

5 – 2 Riparian Corridor Condition

Natural riparian corridors are important for stream and river health because they support physical and ecological processes and provide habitat corridors for river-associated birds and mammals. Depending on position within the watershed, riparian corridors play various functions. In headwater areas, hydrology, sediment input, and channel network formation is largely influenced by riparian corridors. Further downstream, riparian corridors often include well-developed floodplains, which may or may not be confined within steep valley walls. Floodplain condition affects channel and bank stability, water quality, sediment storage, and water storage during overbank flows. Riparian condition is one indicator of headwater and floodplain functions throughout a watershed.

5 – 2.1 Description of Indicator

The active river area model and land cover data were used to assess riparian corridor condition throughout the non-tidal portion of the Delaware River basin. The active river area framework is a spatially-explicit approach to identifying the areas within a watershed that accommodate the physical and ecological processes associated with river systems (Smith et al. 2008). The spatial model includes three primary components within the riparian corridor: floodplains, riverine wetlands, and riparian areas that are likely to contribute woody debris, coarse particulate organic matter, sediment, and energy to the riverine system. The area and percent of natural land cover (predominately forest

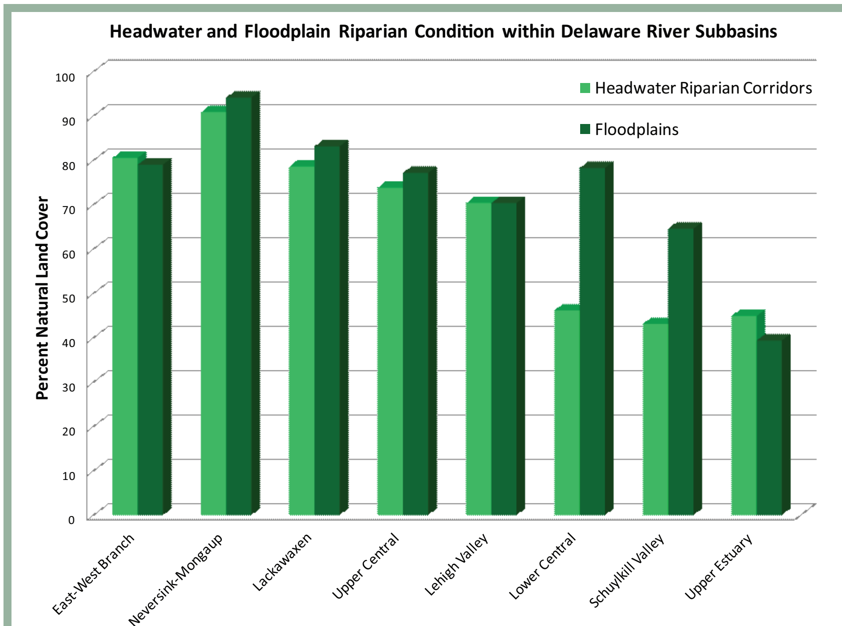


Fig. 5.35. The majority of floodplains and headwater riparian corridors in the Upper and Central Regions of the Delaware Basin contain at least 70% natural cover. Although percent natural cover is lower in the non-tidal portion of the Lower Region, there are still floodplain areas with extensive natural cover, including the portions of the Schuylkill Valley and mainstem Delaware between Allentown, PA and Trenton, NJ (Lower Central subbasin).

and wetland land cover) for headwater riparian corridors (i.e., all streams with drainage areas less than approximately 40 square miles/10,359 ha) was calculated. The area and percent of natural cover within floodplains (i.e., all streams and rivers with drainage areas greater than 40 square miles/10,359 ha) for each major sub-basin was calculated. Comparing riparian condition in headwaters and floodplains is one indicator that reveals how ecological processes may have been altered in various subwatersheds throughout the non-tidal portion of the basin.

5 – 2.2 Present Status

In the Upper and Central Regions of the Delaware Basin, the majority of riparian corridors are at or above 70% natural cover, both in headwaters and in floodplains (Fig. 5.33). The riparian corridors in the Neversink-Mongaup subbasin are in best overall condition compared to any other subbasin; over 90% of the riparian corridors are in natural cover, both within headwaters and within floodplains of larger rivers (Fig. 5.34). Natural riparian corridors in the headwaters, such as those in the Upper Lehigh River and Tobyhanna Creek watersheds, are essential for maintaining water quality and quantity for downstream ecosystems and water users (Fig.5.35). In the Lower Region, riparian corridors are much more developed, although there are still some large areas of natural cover within floodplain riparian corridors in the Schuylkill and Lower Central Subbasins. For example, the floodplain areas along the main-stem between Allentown, PA and Trenton, NJ, are approximately 78% forest and wetland cover. This area includes the Lower Delaware Wild and Scenic River, which is part of the National Wild and Scenic River system managed by the National Parks Service.

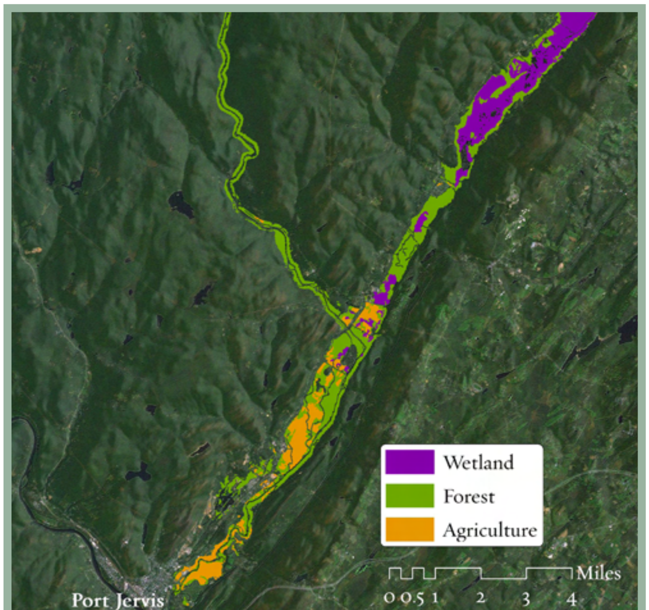


Fig. 5.36. In the Neversink-Mongaup subbasin, approximately 94% of the floodplain area is in forest or wetland land cover.

5 – 2.3 Past Trends

Riparian corridors (floodplains, riverine wetlands and riparian areas) have long been recognized as environmentally sensitive, ecologically diverse, and hydrologically important areas within a watershed. Even though the natural functions of these corridors and the hazards associated with their occupancy are widely known, people have always been attracted to water. Historically, settlements have arisen along waterways because they contain natural features beneficial to human societies (fertile soil, transportation links, water supply, hydropower, and aesthetic beauty). One consequence of human development of riparian corridors is the physical alterations of both stream channels (dams, levee construction, straightening, and dredging) and the floodplain landscape, impacting not only the integrity of the watercourse, but also resulting in significant social and economic consequences. Floods in developed floodplains devastate families, businesses and communities, and cause more damage to life and property than any other natural hazard.

Notwithstanding these problems in many parts of the country, the riparian corridor condition of the Delaware River Basin is relatively good. As noted above, riparian corridors associated with headwater watersheds and floodplains in the Upper Basin enjoy 70% or more natural cover. Similarly, riparian corridor condition associated with the Central Basin Delaware River floodplain has plentiful forest and wetland cover. The national status of the Delaware as the largest free flowing river East of the Mississippi, coupled with high water quality directly attributable to riparian corridor condition have led to inclusion of three-quarters of the non-tidal Delaware River (about 150 miles) in the National Wild and Scenic Rivers System. In contrast, only one quarter of one percent (11, 000 miles) of the 3.5 million miles of rivers in the nation has been included in the System.

5 – 2.4 Future Predictions

In 2004, the four Basin Governors and federal agency Regional Executives signed a forward looking Basin Plan that identified five Key Result Areas, one of which focused on Waterway Corridor Management. Specifically, the Plan specified a Desired Result involving: Waterway corridors that function to minimize flood-induced loss of life, protect property and floodplain ecology, preserve channel stability, provide recreational access, and support healthy aquatic and riparian ecosystems. Work is now underway by many partners to implement the specific goals and objectives enumerated in the plan, including an annual report out of progress at the fall Delaware River Basin Commission meeting.

Another significant milestone in 2011 was realized with the completion of the "Delaware River Basin Priority Conservation Areas and Recommended Conservation Strategies" Report. The report was developed by The Nature Conservancy, Partnership for the Delaware Estuary, and Natural Lands Trust, and funded by the National Fish and Wildlife Foundation. It focuses on Floodplains, Headwaters and Non-Tidal Wetlands and provides a platform for shared conservation and restoration priorities across the basin.

5 – 2.5 Actions and Needs

The Water Resources Plan for the Delaware River Basin ("Basin Plan) Objective 2.3 D called for "Implementing Strategies to protect critical riparian and aquatic habitat" and established milestones for identifying, mapping and prioritizing critical habitats. It also called for development and adoption of protection and restoration strategies.

1. Action: The Final Report for the National Fish and Wildlife Foundation titled "Delaware River Basin Priority Conservation Areas and Recommended Conservation Strategies" was completed in 2011. The report includes detailed maps by Sub-basin showing watershed specific freshwater system priorities. For example, the Upper Delaware River Basin is divided into 22 watersheds and place-specific conservation strategies (Headwater Networks; Floodplain Complexes; Headwater Wetlands; and Riverine Wetlands) are identified and prioritized.

2. Action: The Conservation Plan referenced in Item #1 functions as vehicle for collaborative restoration and protection action.

3. Action: The Conservation Plan also serves as preliminary set of targets for implementation of the Delaware River Basin Conservation Act of 2011, if it is successful in becoming federal law.

4. Need: The Basin conservation community needs to work with its Congressional Delegation to continue to advocate for passage of the Delaware River Basin Conservation Act.

5. Action: The Delaware River Basin Commission Flood Advisory Committee conducted a careful assessment of Floodplain Regulations both in the basin and around the country in 2008 and 2009. In October 2009, they presented a report containing twelve recommendations for more effective floodplain regulations to the Commission. The Committee determined that minimum floodplain regulations, administered by FEMA through the National Flood Insurance Program, do not adequately identify risk or prevent harm. They also found that floodplain regulations are inconsistent from State to State and from community to community. They recommended that floodplain regulations need to be applied more consistently and comprehensively, on a watershed basis that reaches across jurisdictional boundaries.

6. Need: DRBC needs to work with FEMA to advance their Risk Mapping, Assessment and Planning (Risk MAP) strategy to work with local officials to use flood risk data and tools to effectively communicate risk to citizens and better protect their citizens. The DRBC Flood Advisory Committee recommendations could be one component of the FEMA strategy to work with communities at a watershed scale to make the Basin more flood resilient.

5 – 3 Fish Passage

The Delaware River lacks any dams on its main-stem that block passage of fish, a feature which is remarkable for a river of its size. Diadromous fish like American shad, alewife, blueback herring, striped bass, sea lamprey, and American eel can travel over 300 miles (483 km) from the mouth of the river up to its origin (and back out to the ocean) without being blocked by a barrier. Unobstructed stream habitat like this is critical for migratory fish, especially for anadromous fish to be able to access freshwater spawning grounds. Long stretches of connected streams also are important for local movement of resident fish and other aquatic organisms. Some resident species, such as the tessellated darter, also serve as host fish for certain freshwater mussels. Consequently, the ability of fish like this one to move within a stream system is also critical for freshwater mussels, which rely on host fish to disperse their young and colonize new habitats.

Unlike the main-stem, most tributaries of the Delaware River have been dammed over time. Over 1,400 dams within the basin are tracked by various federal and state agencies; additionally, many smaller, unregulated dams that are not captured by these databases exist in the basin. While large dams pose clear barriers to fish passage, small run-of-river dams and even inadequate culverts can impede fish passage. Cumulative effects of barriers can dramatically reduce the amount of accessible habitat for fish within a stream network, although the first few barriers in a stream network have the greatest impact on connected habitat (Cote et al. 2009).

5 – 3.1 Description of Indicator

Using dams in state and Army Corps of Engineers (National Inventory of Dams) databases, as well as a small number of hand-mapped blockages in the Delaware Bay coastal area, we identified the length of each connected stretch of a river network (i.e., portions that have no dams occurring within that stretch) using the Barrier Analysis Tool (BAT, v.1). This tool calculates the total length of a connected stream network by adding the lengths of a river and all connected tributaries between barriers (or between a river origin and the first barrier downstream, or the river mouth and the first barrier upstream). Results of the analysis highlight the longest connected river networks, including those that have no blockages from their headwaters downstream to the Delaware River and out to the Bay.

It is important to note that our analysis included dams that have fish ladders installed on them. These dams were not removed from the analysis primarily because many fishways still pose barriers to fish passage; while they may allow for effective passage of a handful of species similar to those for which they were designed, many fish are still unable to use fish ladders effectively, if at all. Perched, undersized or blocked culverts also can be significant barriers to fish movement; however, this type of barrier was not included in our analysis, due to a lack of a basin-wide culvert dataset.

5 – 3.2 Present Status

The Delaware River is distinguished by being the longest free-flowing river in the Eastern US. Anadromous and catadromous fish species can travel unimpeded through over 500 miles (802 km) of connected rivers and streams, from the mouth of the Delaware River upstream to Hancock, New York and as far upstream on any connected tributary as the first barrier (Fig. 5.38). Many tributaries lack dams in their downstream portions and thus allow migratory fish like river herring to access spawning habitat downstream of any barrier. For example, the Rancocas, Flatbrook, and Neversink River systems all have significant habitat available for migratory fish. A dam removal on the lower Neversink River in 2004 opened up the entire historic habitat available for American shad, while also improving access for American eel and sea lamprey. (In the case of a river like the main-stem Schuylkill River, fish passage structures allow fish like shad to access upstream portions of the river, though our analysis does not recognize this degree of connectivity due to the difficulties in fairly assessing basin-wide where fishways effectively mitigate barriers that dams pose to most fish.)

Despite the fact that the main-stem and connected portions of its many tributaries together provide over 500 miles (805 km) of unblocked aquatic habitat, the Delaware River's tributaries have suffered significant fragmentation from the construction of over 1,400 dams in the 1800s and 1900s. Notwithstanding the fact that they lack a direct connection to the main-stem or bay, some tributary stream networks in the basin still offer significant mileage of connected habitat for resident fish. Some of the largest connected stream networks include the headwaters of the West Branch, the East Branch, the Lehigh River, and the Schuylkill River; a significant section of the middle Schuylkill also lacks tracked dams (Fig. 5.38). The ability to move locally within stream systems like these is important to many species. In particular, potadromous species, such as the white sucker, make instream migrations to complete their life cycles.

It is important to note that while some of the shorter stream systems (e.g, small coastal streams) may not have especially high values in terms of total connected stream length, these streams, which are often highly productive, are 100% connected from their headwaters to the Bay, allowing fish access to their full historic range of stream habitats (e.g., Red Lion Creek or Augustine Creek in Delaware or Oranoaken Creek or Bidwell Creek in New Jersey).

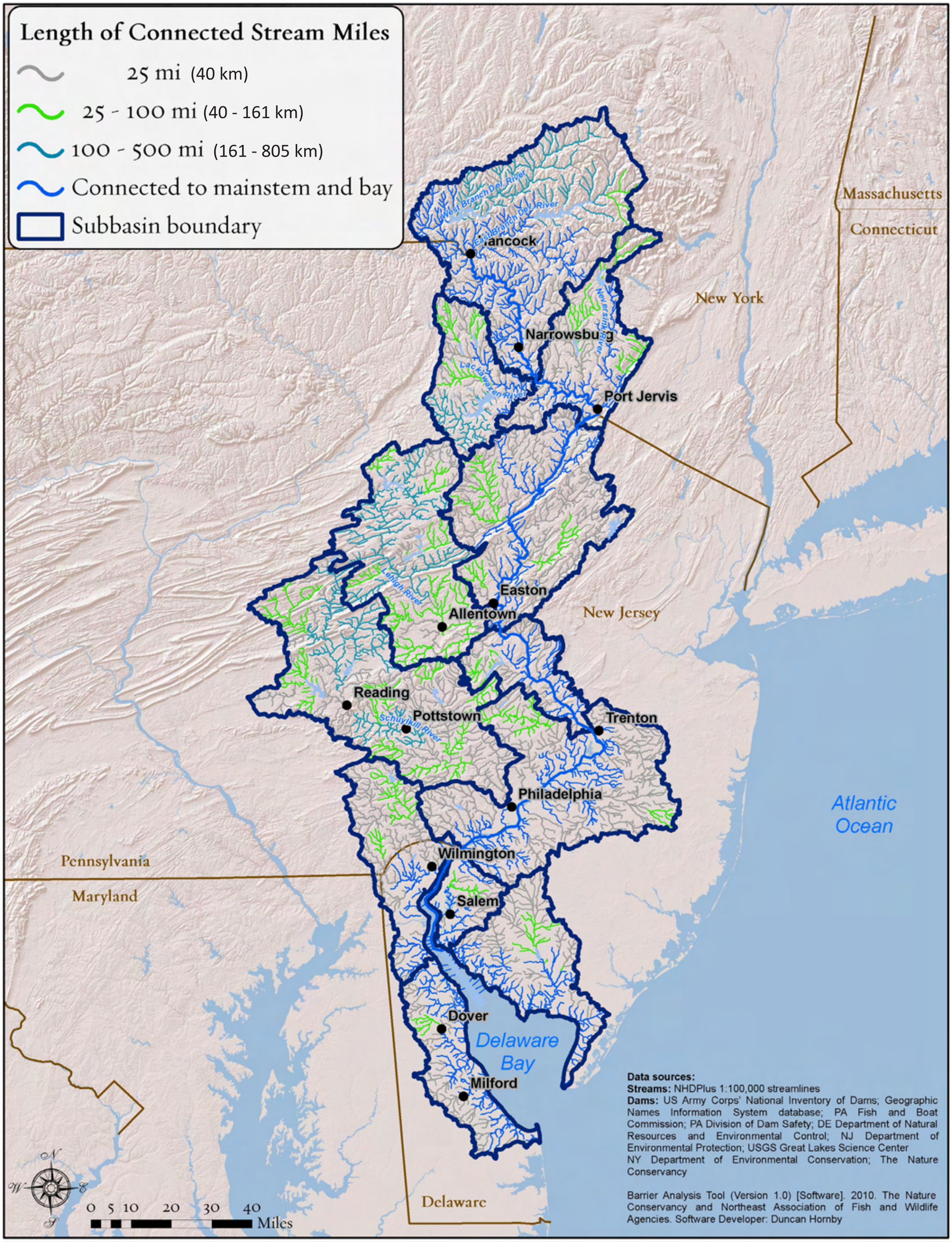


Fig. 5.38. Connected Stream Networks within Delaware River Subbasins.

5 – 3.3 Past Trends

In 1985, the Delaware Basin Fish and Wildlife Management Cooperative identified three priority rivers for fish passage efforts: the Brandywine, Schuylkill, and Lehigh Rivers. How far upstream fish can swim in each of these rivers has changed over time in two of these three rivers as fish passage efforts like dam removal and fishway installation have been implemented (Fig. 5.39).

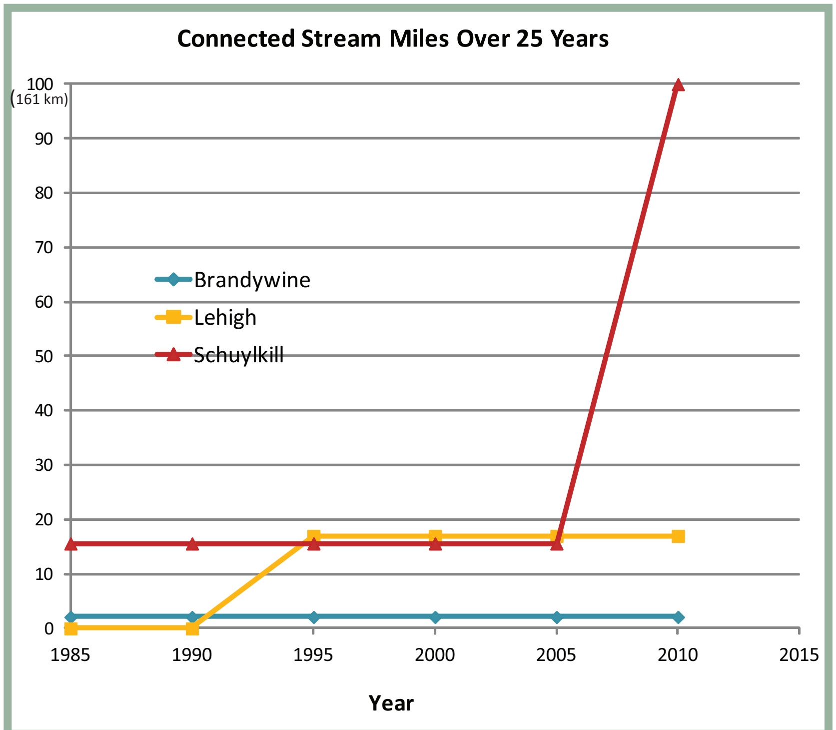


Fig. 5.39. Number of connected stream miles accessible from the mainstem Delaware River or Bay between 1985 and 2010 for each of the three priority fish passage rivers.

On the main-stem Brandywine, fish ladders were installed during the mid-1970's on three of the first four dams, all located within the first four miles of the river. However, after several years of monitoring, the fish ladders were found to be ineffective and were removed. The Brandywine Conservancy has published feasibility studies for addressing fish passage for American Shad in the Delaware (2005) and Pennsylvania (2009) portions of the watershed. The studies included the 11 main-stem Brandywine dams in Delaware (~14 miles/23km of main-stem habitat) and 10 of the 28 current dams in Pennsylvania.

On the main-stem Schuylkill, three fish ladders and four dam removals since 2006 have increased access from river mile 15 up to river mile 100, a dramatic improvement. The effectiveness of the three fish ladders is still largely unknown, with only the Fairmount Dam fish ladder having associated long-term monitoring results published. In addition to the main-stem projects, between 2003 and 2007, five dams have been removed on the Perkiomen Creek main-stem, three on the Wyomissing Creek, and one each on the Tulpehocken and Pickering Creeks.

On the main-stem Lehigh, the first two dams had fish ladders (Easton & Chain) installed in 1994 and later retrofitted in 2000. The third dam, Hamilton St., had a fish ladder installed in 1984. A main-stem dam farther upstream, Palmerton Dam, was removed in 2006. After years of monitoring at both Easton and Chain dams, these fish ladders have been determined to be ineffective in passing their target species, American Shad. As a consequence, the Wildlands Conservancy and the PA Fish & Boast Commission recently requested proposals to evaluate the removal of Easton and Chain dams (July 2011) in the hopes of improving fish passage at these locations. Northampton Dam, the last of the lower four dams, is expected to have a fish passage feasibility study initiated in early 2012. In addition to these main-stem Lehigh projects, between 2000 and 2010, a total of 5 dams have been removed on Saucon Creek, East Branch Saucon Creek, Jordan Creek, Little Lehigh Creek, and Mahoning Creek.

In addition to these three tributary watersheds, there are active fish passage efforts underway in smaller tributaries such as Ridley Creek (DE/PA), Pennypack Creek (PA), Bushkill Creek (PA), Lopatcong Creek (NJ) and the Musconetcong River (NJ).

5 – 3.4 Future Predictions

The importance of river connectivity and associated fish passage is being recognized by many water resource agencies and the public and is evident in the recent number of dam removal projects and feasibility studies recently completed or currently underway. In addition to the direct impact on fish habitat, the relationship between keystone species such as freshwater mussels and their dependence on certain fish species for reproduction and colonization should only add momentum to addressing fish passage. Unless Basin prioritization is revisited, fish passage projects will likely continue to be haphazardly located throughout the Basin with more action occurring in tributaries with active watershed- based organizations and cooperative dam owners rather than in strategic locations.

5 – 3.5 Actions and Needs

Financial resources for addressing fish passage within the Basin are limited, and there is a need for an updated comprehensive evaluation of where best to prioritize fish passage. The prioritization needs to consider the best ecological return for each location addressed as well as the suitability of potential new habitat. An effort ongoing since 2008 by the Northeast Association of Fish and Wildlife Agencies and The Nature Conservancy (TNC), called the Northeast Aquatic Connectivity (NAC) Project, has developed tools and an initial assessment of opportunities for restoration of stream system connectivity across the Northeastern US. With input from the NAC workgroup, TNC calculated 72 ecologically-relevant metrics for almost 14,000 dams across the region and developed tools to allow for tailored assessment of ecological returns of reconnection projects. Tools and final products (expected by 2012) include two assessment scenarios that rank dams for benefits for anadromous fish and for benefits for resident fish, produced using a subset of metrics weighted by the workgroup. While these products and tools will help inform prioritization efforts, site-specific factors still need to be considered in project selection.

In addition to the forthcoming Northeast Aquatic Connectivity Project, Senator Tom Carper (Delaware) recently introduced the Delaware River Basin Conservation Act of 2011, which would establish a federal program at the U.S. Fish and Wildlife Service (US FWS) to coordinate voluntary restoration efforts throughout the Basin and oversee up to $5 million per year of grant funding. It is envisioned that a basin-wide fish passage prioritization project would be an ideal project worthy of funding through the Act and would help guide future distribution of grant monies.

The fish ladders installed in the Lehigh River have also demonstrated that not all fish passage “remedies” are equal, with some being more successful than others. In cases where a dam no longer serves a critical use such as for public water supply, the first remedial option should be removal. In addition, where regulatory opportunities exist with dam owners during permitting actions, regulatory agencies need to adopt and implement a consistent approach as to when and why fish passage needs to be addressed. Many dam owners have argued that if anadromous fish are not present downstream of their dam, then there is no need to address fish passage. For dam locations that do not have anadromous fish downstream, addressing fish passage is still important for resident species.

From the perspective of both anadromous and resident fish, assessing the degree to which road/stream crossing structures also are creating barriers to fish passage will be important, as well. While we currently lack good data, pilot field surveys conducted by The Nature Conservancy and others will provide some insight on the prevalence of problematic culverts within select tributary watersheds in the Basin. Following ecological standards for culvert design and replacement could be helpful to restore connectivity currently hindered by these small structures.

5 – 3.6 Summary

The Basin has experienced a large number of fish passage projects, primarily targeting American Shad, during the past 10 years. Most of the fish passage projects are occurring in Pennsylvania, with both financial and technical support from the state resource agencies. Although three large tributaries were targeted in 1985 for priority consideration, it appears that the only tributary with significant progress may be the Schuylkill River. Recent fish passage efforts do not appear to be a component of a larger restoration plan. A new Basin-wide reassessment of fish passage priorities is needed to ensure that limited resources are being targeted in an efficient and effective manner.

5 – 4 Hydrological Impairment

Natural variations in hydrologic regime—the magnitude, timing, frequency, duration, and rate of change of stream flow—are critical for sustaining healthy river systems (Poff et al. 1997, Richter et al. 1997). Healthy floodplains also are dependent upon natural flows, as they require interaction with rivers whose flow regimes have sufficient variability to encompass the flow levels and events that support important floodplain processes (Opperman et al. 2010). Alterations to the natural flow regime of a river result from a variety of sources, such as flood control, water supply and hydropower dams, as well as water withdrawals and development in the watershed. Paved and other hard surfaces, collectively referred to as impervious cover, often increase the volume of and rate at which precipitation runs off into the stream channel and can increase the flashiness of streams (Leopold 1968). Impairment of a river’s natural hydrologic regime can cause various negative impacts throughout a watershed. Dams that store large amounts of water can significantly change amounts of streamflow downstream of the dam, as well as change seasonal patterns of high and low flows on which many aquatic organisms depend (Poff et al. 1997). In addition, large dams change sedimentation patterns, potentially depriving the river downstream of the dam and causing significant changes in the stream channel and bed. Other impacts include changes in water temperature and nutrient transport, which in turn affect both aquatic and riparian species (Poff and Hart 2002).

5 – 4.1 Description of Indicator

All dams do not have the same effects on downstream rivers, and consequently, using one indicator to predict potential hydrological alteration is difficult across the

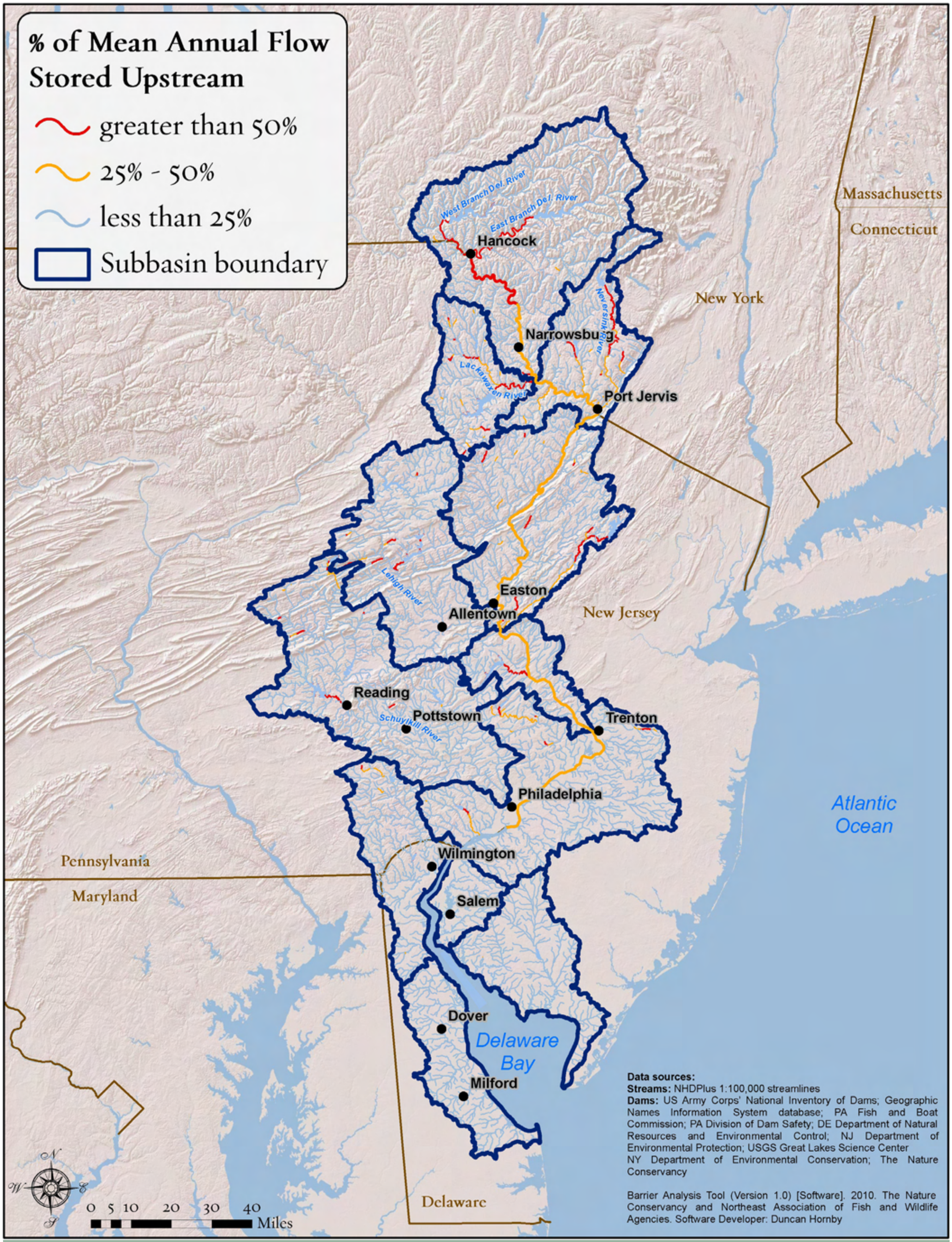


Fig. 5.40. Ratio of upstream dam storage to mean annual flow for river reaches within Delaware River sub-basins.

entire basin. However, one important indicator of potential alteration to the natural hydrologic regime is the ratio of upstream dam storage to mean annual flow downstream (Graf 1999). This ratio is calculated by expressing the cumulative volume of water stored by upstream dams as a percent of the mean annual flow of each downstream river segment. As this proportion increases, so does the likely alteration to natural stream flow. Ratios indicative of a high risk of hydrologic alteration have been demonstrated to be > 50% (Zimmerman and Lester 2006). Using storage values available in state and Army Corps of Engineers (National Inventory of Dams) databases and mean annual flow values associated with NHDplus streamlines, we applied the Barrier Analysis Tool (BAT, v.1) to calculate the percent of mean annual flow that is stored in upstream dams in the Delaware Basin.

This indicator does *not* take into account day to day reservoir operations or specific dam configuration, which can influence the degree of hydrologic alteration in either a positive or negative way. Furthermore, this indicator also does not reflect the effects of other water diversions or withdrawals in the basin, so it is limited to potential impairments to hydrologic regime caused only by dam storage. However, the basin-wide assessment of the risk of hydrologic impairment due to high dam storage is still a useful indicator; across large and small rivers, it can help identify which stream and river reaches may be suffering the hydrologic (and associated ecologic and biologic) impacts of upstream dams and which dams may warrant further investigation to address potential streamflow alteration.

In order to identify places most likely to be suffering hydrologic impairment due to land use change, also examining the percent cover of impervious surface within a watershed can provide a useful complement to the measure of upstream dam storage. The high amounts of impervious cover associated with many highly developed areas are likely to cause hydrologic alteration downstream unless there are adequate stormwater management systems in place. The higher the percent cover of impervious surface across a small watershed, the more likely its streams are to be suffering hydrologic impairment. Because this metric cannot take into account effective stormwater management, it also should be used as a first-cut indicator to identify places that likely would benefit from stormwater management systems if they are not already in place.

5 – 4.2 Present Status

As many dams in the basin are run-of-river dams and have relatively little effect on hydrologic regime, the vast majority of stream miles within the basin are at low risk of hydrologic alteration, as indicated by their ratio of dam storage to mean annual flow value (Fig. 5.40). However, over 300 stream and river miles (483 km) within the basin could be considered at high risk as indicated by ratio values of >50%. Of these 300 miles, over 130 miles (209 km) of high-risk streams and rivers are those which drain less than 38 square miles (9842 ha). High ratios might be expected in these headwater areas where dams occur in small streams that have relatively low mean annual flow values. High risk on larger rivers may be caused by the cumulative storage of many dams upstream or by a major reservoir with significant storage capacity (or a combination of the two). Despite the limitations of the basin-wide analysis of the risk of hydrologic impairment due to high dam storage, this ratio is still a useful indicator of locations where impaired hydrology may be occurring and affecting the health of our streams and rivers. While some significant impacts are occurring in the Delaware Basin, most streams and rivers are at low risk of impairment from dam storage.

Similarly, the vast majority of watersheds within the basin have relatively low (< 10 %) impervious cover (Fig. 5.41). However, streams in or downstream of urbanized areas, particularly those with outdated or insufficient stormwater management in place, are likely to be suffering negative impacts of altered hydrology as well. Most at-risk watersheds are concentrated around the cities of Wilmington, Philadelphia, and Camden, though watersheds along the Lehigh, Schuylkill, and Maurice Rivers also may be experiencing substantial hydrologic impairment due to land use change. Localized land change certainly may also affect hydrology within a watershed, but this basin-wide analysis helps to identify where the greatest impairment is likely to be occurring.

5 – 4.3 Past Trends

Most of the Basin's large reservoirs were completed between 1960-1980 and were not specifically designed to operate with the longitudinal (high and low) and/or the temporal (seasonal) conservation flows that may be needed to maintain native aquatic communities. Recent advances in ecological flow science have resulted in many water resource agencies beginning to factor ecological flow needs into the way that large reservoirs are managed. Some smaller Basin reservoirs currently do not have any conservation release requirements, while most of the larger reservoirs have release requirements based on assimilative capacity needs ("Q7-10" - the consecutive 7-day flow with a 10-year recurrence interval) as opposed to one based on aquatic resource needs. Recent changes adopted by the Decree Parties for the three New York City Basin reservoirs have started to incorporate aquatic resource needs into their reservoir operation plans.

Most of the Basin's existing impervious cover was created prior to modern stormwater management (pre 2000). If any stormwater management did occur prior to 2000, it tended to focus on large storm events (>10 year storm). Modern stormwater management requirements

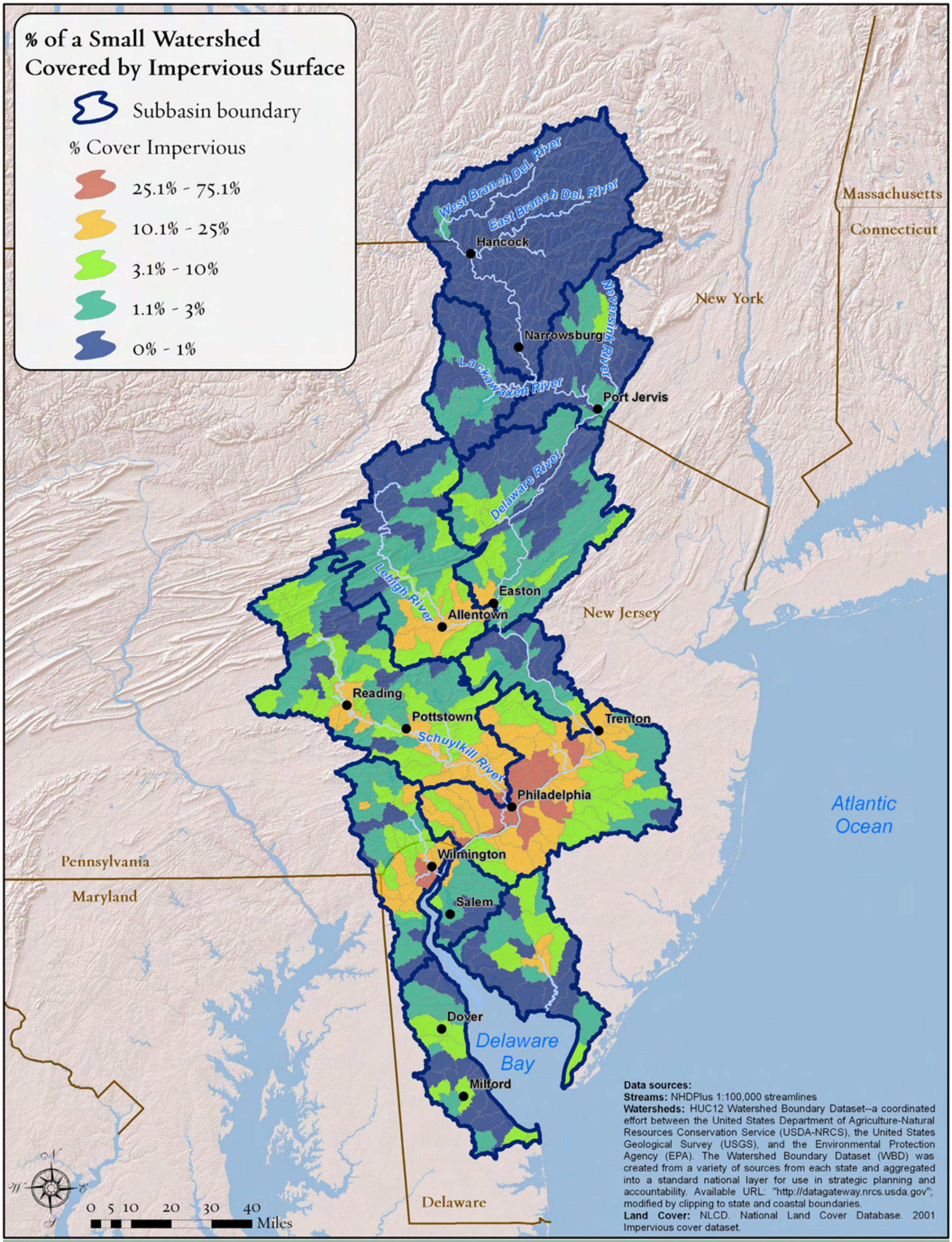


Fig. 5.41. Percent cover by impervious surface across small watersheds in the Delaware River Basin.

have tended to focus on a broader range of rain events (0-100 year storm events), along with minimum infiltration requirements. The modern stormwater management requirements have largely centered on trying to maintain the existing hydrology of a project site from pre to post-development conditions.

5 – 4.4 Future Predictions

As ecological flow science progresses and native aquatic communities' needs are further identified, water resource agencies can start to factor those data into the management of basin reservoirs. New reservoirs will almost certainly be designed and permitted to consider ecological flow needs, while existing reservoirs operations are reviewed during the permit renewal process, which provides opportunities for operational revisions based on the latest science.

Stormwater management will need to focus in two areas – new development and retrofitting existing impervious cover. Almost all new development in the Basin is subject to modern stormwater management requirements. It is anticipated that the level of hydrological impairment due to "new development" will be minimal compared to the existing hydrological impairment caused by existing impervious cover.

5 – 4.5 Actions and Needs

A study of ecological flow needs to protect species and key ecological communities for the range of habitats in the Delaware Basin is necessary in order to provide the scientific basis for any future modifications to reservoir operation plans.

Developing a strategy to deal with existing hydrological impairments due to existing impervious cover is necessary. Options range from mandatory stormwater management retrofits during the redevelopment of a site to voluntary retrofits incentivized by the implementation of stormwater runoff fees.

5 – 4.6 Summary

While most Basin streams are at low risk of hydrological impairment due to dam storage, some significant impacts are occurring in localized areas. The incorporation of ecological flow needs into reservoir management will likely increase in the future as those needs are further identified, which should result in a gradual minimization of impacts in those localized streams.

While most Basin streams are at low risk of hydrological impairment due to existing impervious cover, there are significant impacts in the older urban/suburban areas of the Basin. Implementing stormwater management on existing impervious cover is expensive and may take several decades to address.

Example effects of dam storage and operations on hydrologic impairment: Neversink River

The basin-wide indicator of dam storage ratios does not take into account actual dam operations. For example, this analysis indicates a high level of alteration downstream of the Neversink Reservoir. Indeed, the biologic effects of hydrologic alteration have been documented in the Neversink River, where macroinvertebrate surveys indicated that species composition in the river downstream of the reservoir showed signs of degradation similar to stretches impaired by acidity in other parts of the watershed (Ernst et al. 2008). Altered temperatures and low flow in river stretches immediately downstream of the reservoir appeared to favor Chironomidae taxa over Ephemeroptera, Plecoptera, and Trichoptera taxa, similar to how pH and aluminum in the East Branch of the Neversink River appeared to influence macroinvertebrate composition there. This change in the biotic community of the river downstream of the reservoir likely was caused by adverse effects from dam storage (Ernst et al. 2008). However, more recently, a detailed study of the effects of changes in the management of the Neversink Reservoir just within the past few years illustrates that recent management changes have improved the degree of alteration to the Neversink River's natural hydrologic regime (Moberg et al. 2010). The figure below shows how the natural range of variability in flow on the Neversink has changed with the implementation of the Flexible Flow Management Plan. Whether the biotic communities of the Neversink River downstream of the reservoir have shown any positive response to the return of a more natural hydrologic regime has not yet been studied.

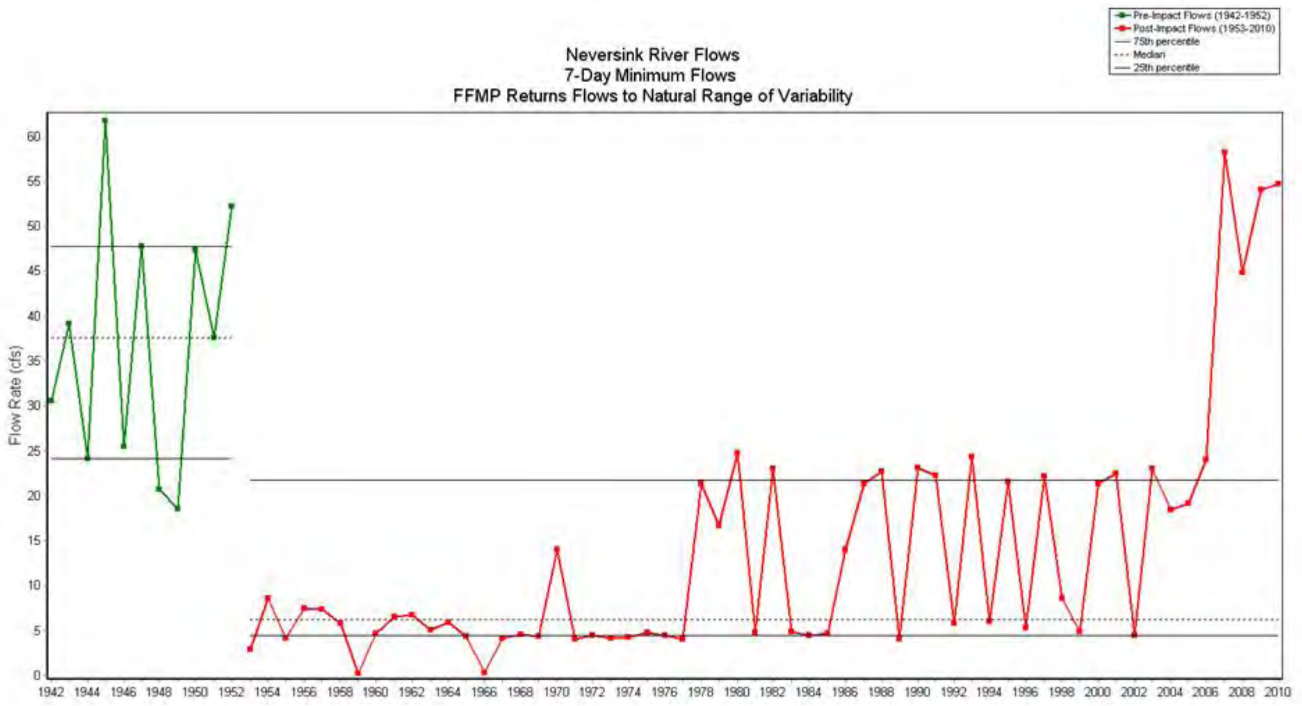


Illustration of the effects of dam operations on the Neversink River, compared to pre-dam conditions. Patterns during implementation of the Flexible Flow Management Plan indicate minimum flows within the range of natural variability.

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