Water Resources Research Center
Annual Technical Report
FY 2009

Introduction

The major water science issue in Maryland is the health of the Chesapeake Bay. It is one of the largest economic assets in the State. Research, education, and information transfer projects of the Maryland Water Resources Research Center nearly all have a focus on the Bay and related water quality.

Research Program Introduction

With 104B funding, after peer review, the Maryland Water Resources Research Center supported two research projects and awarded two graduate student summer fellowship. All have a water quality focus and Chesapeake Bay implications • Nearshore Sediment Inputs due to Shore Erosion in the Maryland Chesapeake Bay, Lawrence Sanford, University of Maryland Center for Environmental Science

• Low-cost Anaerobic Digesters for Dairy Manure Treatment and Renewable Energy Production, Stephanie Lansing, Department of Environmental Science & Technology, University of Maryland

• Role of Invertebrate Bioturbation in Phosphorus Retention of Agricultural Ditch Soils--Summer Fellowship, Alan W. Leslie, Department of Entomology, University of Maryland

• Correlating Nutrient, Pesticide, and Bacteria levels in the Choptank River--Summer Fellowship, Gabriela T. Nino de Guzman, Department of Civil and Environmental Engineering, University of Maryland

Also, a Maryland 104G project continued during 2009:

• "Integration of Stormwater Management Ponds into Urban Communities: Long-term Water Quality Protection, Wildlife, and Environmental Awareness." Joel Snodgrass, Towson University

Another project, funded in a previous year, was completed in 2009:

• "Responses of Species-Rich Low-Salinity Tidal Marshes to Sea Level Rise: a Mesocosm Study." Andrew Baldwin, University of Maryland

Responses of Species-Rich Low-Salinity Tidal Marshes to Sea Level Rise: a Mesocosm Study

Basic Information

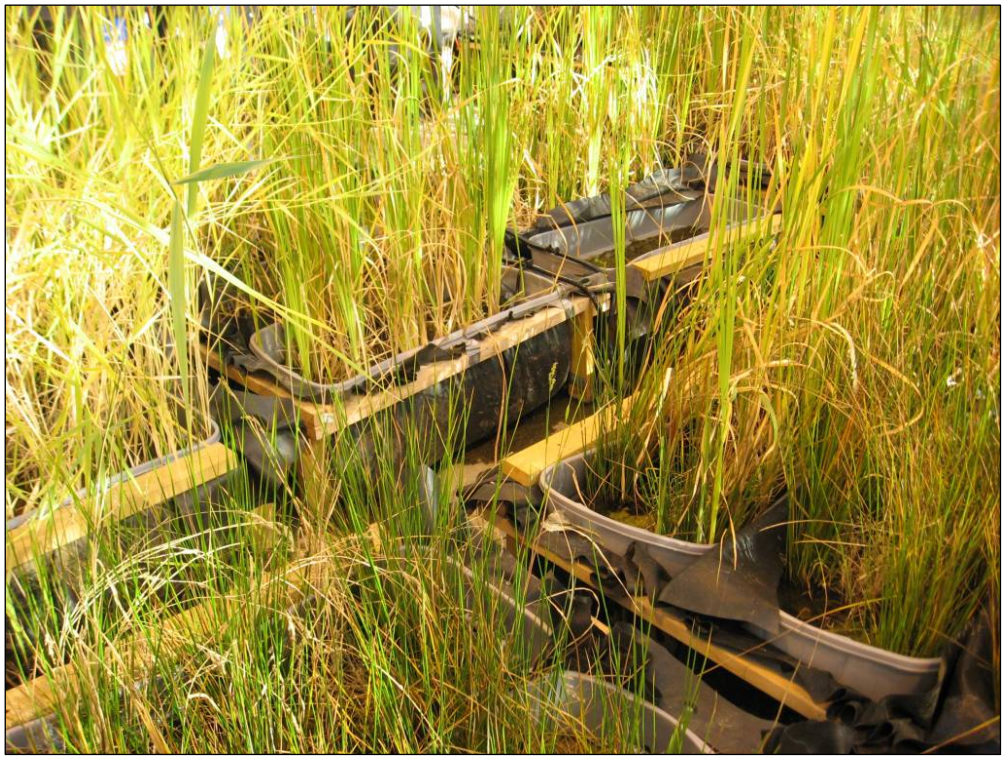
Title:	Responses of Species-Rich Low-Salinity Tidal Marshes to Sea Level Rise: a Mesocosm Study
Project Number:	2007MD143B
Start Date:	3/1/2007
End Date:	3/1/2009
Funding Source:	104B
Congressional District:	5th & 8th Congressional District of Maryland
Research Category:	Biological Sciences
Focus Category:	Wetlands, Ecology, Climatological Processes
Descriptors:	
Principal Investigators:	Andrew Baldwin, Peter James Sharpe

Publications

There are no publications.

Final Report FY 2009

Responses of Species-Rich Tidal Freshwater Wetlands to Sea Level Rise: a Mesocosm Study



Project duration: March 1, 2007-March 1, 2010

Prepared for:
Maryland Water Resources Research Center

Principal Investigator:
Andrew H. Baldwin
baldwin@umd.edu
301-405-7855

Co-PI:
Peter J. Sharpe

Prepared by:
Peter J. Sharpe and Andrew H. Baldwin
Department of Environmental Science and Technology
1423 Animal Science Building
University of Maryland
College Park, MD 20742

Submitted May 26, 2010

Summary of Project Outcomes

This project supported the Ph.D. research of one student, Peter Sharpe, who has graduated and is currently in a post-doctoral position at IRTA in Spain. His dissertation is available through the University of Maryland libraries (Sharpe, P.J. 2009. Patterns of wetland plant species richness across estuarine river gradients. Ph.D. Dissertation, Department of Environmental Science and Technology, University of Maryland, College Park, MD. PP. 1-217). A manuscript resulting from this project is nearing submission (Sharpe, PJ and AH Baldwin. Marsh plant community responses under accelerated sea-level rise conditions, *Aquatic Botany*). The research was featured in an invited talk at the 2009 MEES Colloquium by Peter Sharpe, the only graduate student selected for this honor. Several undergraduate students also participated in data collection and sample analysis during this project.

Introduction

Statement of Critical Regional or State Water Quality Problem

Sea level rise is threatening coastal wetlands worldwide. Increases in sea level may cause shoreward movement of salt-tolerant species such as *Spartina alterniflora* (Donnelly and Bertness 2001) or conversion of coastal wetlands to open water (Baumann et al. 1984). In the Chesapeake Bay, where the relative rate of sea level rise since 1900 has been 2.5-3.6 mm/year (Lyles et al. 1988; Stevenson and Kearney 1996), extensive marshes such as those at Blackwater National Wildlife Refuge on Maryland's eastern shore have been lost (Stevenson et al. 1985; Kearney et al. 1988). Much of the research on effects of sea level rise on coastal wetlands has focused on brackish and salt marshes, where increases in relative water level due the combined effects of land subsidence and eustatic (background) sea level rise have been implicated as a dominant factor in loss of these wetlands (Stevenson et al. 1985, 1986; Morris et al. 2002). However, little is known about the effects of sea level rise on low-salinity tidal wetlands, which include the species rich, high-productivity tidal freshwater and intermediate or oligohaline marshes (Tiner and Burke 1995). In addition to increases in water level, the salt-sensitive vegetation of low-salinity wetlands also is likely to exhibit stress or mortality due to saltwater intrusion from sea level rise (McKee and Mendelssohn 1989; Baldwin and Mendelssohn 1998). Therefore, sea level rise arguably poses a greater risk to low-salinity wetlands than to salt and brackish marshes.

The Chesapeake Bay contains one of the greatest concentrations of tidal low-salinity marshes in the United States, covering about 16,000 hectares in Maryland alone (Tiner and Burke 1995; Mitsch and Gosselink 2000). Extensive low-salinity tidal marshes are associated with many of the rivers flowing into the Bay, including the Patuxent, Choptank, Wicomico, and Pocomoke Rivers in Maryland and the James, York, and Rappahannock Rivers in Virginia (Tiner and Burke 1995). These wetlands are of tremendous importance to the Chesapeake Bay ecosystem. Due to their low salinity, the plant communities of tidal freshwater marshes are considerably more diverse than those of salt and brackish marshes. Additionally, tides and river flooding supply abundant nutrients, generating primary productivity as high as any ecosystem on earth, including agroecosystems (Tiner 1993; Mitsch and Gosselink 2000). The combination of high plant diversity and productivity and low salinity stress supports diverse and abundant fish and wildlife populations. For example, almost 300 bird species have been reported in tidal freshwater marshes, and the majority of commercially important fish species rely on tidal low-salinity wetland for some phase of their lifecycle (Odum et al. 1984; Odum 1988). These include the rockfish or striped bass, *Morone*

saxatilis, a multimillion dollar fishery industry in Maryland. Reportedly 90% of east coast rockfish are spawned in the tidal fresh and oligohaline portions of tributaries of the Chesapeake Bay, where their larvae congregate in and along the edges of low-salinity marshes (Berggren and Lieberman 1977; Odum et al. 1984). In addition to supporting plants, fish, and wildlife, tidal low-salinity wetlands are used heavily for hunting, fishing, and nature observation by humans, and act to protect shoreline properties from coastal erosion and storm surges (Mitsch and Gosselink 2000).

Clearly, the loss of tidal low-salinity marshes, or their conversion to brackish or salt marshes, in the Chesapeake Bay due to sea level rise would have dramatic socioeconomic and ecological consequences. While sea level rise itself cannot be readily controlled, measures can be taken to stabilize or restore coastal wetlands. These include addition of sediment to increase elevation, a technique that has been used in coastal Louisiana to mitigate wetland loss due to sea level rise (Ford et al. 1999), and which is being considered for restoration of wetlands at Blackwater National Wildlife Refuge.

While the broad responses of vegetation to increases in salinity and soil waterlogging are understood, the potential for vegetation dieback or changes in species composition in tidal low-salinity marshes of the Chesapeake Bay and other Atlantic Coast estuaries in response to changes in salinity and waterlogging acting together has not been studied. Because of their position in the estuary, these marshes may experience increases in salinity, but not waterlogging if sedimentation patterns continue to provide adequate accretion to keep pace with increases in water level (Kearney et al. 1988). Alternatively, salinity and water level both may increase. Currently little quantitative information exists upon which to base predictions of changes in species diversity or composition in tidal low-salinity marshes, or even whether vegetation will die back under different projected sea level rise scenarios (IPCC, 2001). Because of the ecological and socioeconomic significance of tidal low-salinity marshes of the Bay and elsewhere, quantitative information and predictive models are invaluable tools for understanding how coastal wetlands will respond to increases in sea level and in designing mitigative measures or wetland restoration projects in the face of sea level rise.

Preliminary Research

During 2006 we studied patterns of plant diversity and composition across low-salinity tidal marshes in the upper estuaries of the Patuxent and Nanticoke Rivers in Maryland. Vegetation cover was described in 1000 m^2 plots located across an approximately 50-km gradient at roughly 5-km intervals, extending across tidal freshwater and oligohaline marshes into the brackish marsh zone in both estuaries. Sampling used the module method for non-destructive sampling, which combines large-scale sampling (1000 m^2) with nested plots of 100 m^2,

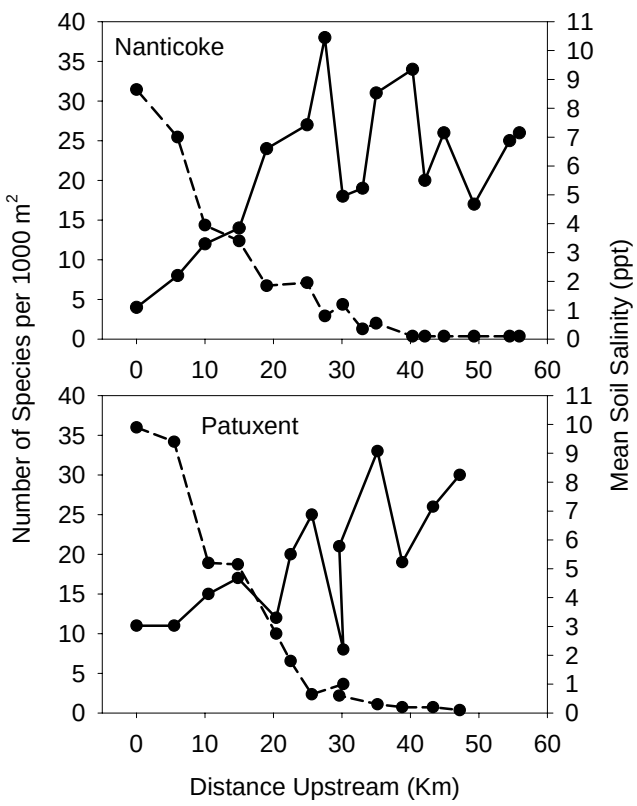


Figure 1. Plant species richness in 1000-m^2 plots (solid lines, left axis) and porewater salinity (dashed lines, right axis) in tidal marshes in the upper Nanticoke and Patuxent Rivers, Maryland (May/June 2006).

10 m^2, 1 m^2, 0.1 m^2, and 0.01 m^2 (Peet et al. 1998). This methodology is a powerful but rapid method that provides composition data across a relatively large area of marsh and allows comparison of species richness at different spatial scales.

Preliminary results show a species richness peak occurring in areas 15-25 km downstream from the uppermost tidal freshwater marshes. In these reaches salinity periodically increases of 2-5 ppt (mddnr.chesapeakebay.net) during periods of low river discharge, which typically occur in late summer during drought years; our springtime 2006 measurements also detected salinity (Fig. 1). This observed peak in plant species richness occurs within the fresh-brackish transition (oligohaline zone) of both rivers and challenges the popular belief that plant species richness is uniformly and inversely related to salinity in tidal marsh ecosystems (Anderson et al. 1968; Tiner 1995; Odum 1988). We hypothesize that the principal abiotic mechanisms controlling the observed plant species richness peak is periodic salinity stress, which reduces the competitive advantages afforded many freshwater plant species and allows less competitive brackish marsh plants to survive in this transition zone.

These preliminary results document the considerably higher plant diversity in low-salinity tidal marshes and that increases in salinity associated with sea level rise will reduce the diversity of these wetlands. Furthermore, if marshes are unable to migrate landward, as is expected in many regions due to coastal steepening, the low-salinity marshes may succumb to the so-called "coastal squeeze" between saline marshes and uplands (Taylor et al. 2004).

While these preliminary findings demonstrate correlation between salinity and plant diversity in coastal wetlands, stronger cause-and-effect relationships can be examined using manipulative experiments than is possible in observational studies. Questions not addressed by this preliminary research are: 1) how do increases in salinity concentration alter species richness and composition in low-salinity coastal marshes?; and 2) does soil waterlogging, also predicted to increase due to sea level rise, reduce or interact with changes in salinity? These questions are the subject of our research.

Nature, Scope, and Objectives of the Project

Our overall goal for the research proposed here is to understand how changes in salinity and water level will influence diversity and ecosystem function of these tidal low-salinity marshes. Specifically, our objectives were to:

1) Create experimental wetland mesocosms containing species from tidal oligohaline and freshwater marshes
2) Subject mesocosms to a factorial arrangement of salinity and inundation treatments
3) Relate changes in plant communities and indices of ecosystem function to potential changes in water level and salinity predicted under various sea level rise scenarios

Through these objectives we tested the following hypotheses, developed based on literature discussed previously and later in the Related Research section:

H1: Increases in salinity will tend to reduce plant diversity (species richness and diversity index) and indices of ecosystem function (biomass, nutrient pools, and soil respiration), but maximum diversity will occur at low salinity rather than in fresh water.

H2: Increases in salinity will result in a shift toward salt-tolerant species.

H3: Increases in soil waterlogging will reduce plant diversity and growth of all species.

H4. Salinity and waterlogging will interact in a synergistic manner to reduce diversity and ecosystem function.

Methods

To examine the potential future responses of low-salinity marsh vegetation to sea-level rise, we developed a greenhouse experiment subjecting marsh mesocosms (the experimental unit) to a range of salinity and soil flooding conditions. The experiment tested the effects of various salinity and flooding regimes on species richness, species composition, and indices of ecosystem function (i.e. above and below ground biomass). Specifically, we subjected synthetic plant communities to three levels of soil flooding and five levels of salinity (0, 1.5, 3, 6, and 12 parts per thousand or ppt) in a 3 x 5 factorial treatment arrangement. For reference, the salinity of ocean water is about 35 ppt, and the salinity classification of coastal marshes is <0.5 ppt for freshwater, 0.5-5 ppt for oligohaline or intermediate marshes, 5-18 for mesohaline or brackish marshes, and >18 for polyhaline or salt marshes (Cowardin et al. 1979).

Mesocosm Configuration

Because of possible gradients in light, temperature, or humidity across greenhouse benches, as well as greenhouse space limitations, experimental units were arranged in a split-plot randomized complete block design (Figure 2). Each block represented a replicate for salinity (i.e. two replicates for salinity) this represented the whole-plot effect, with the sub-plot factor being flooding frequency and having three replicates per trough (Figure 2).

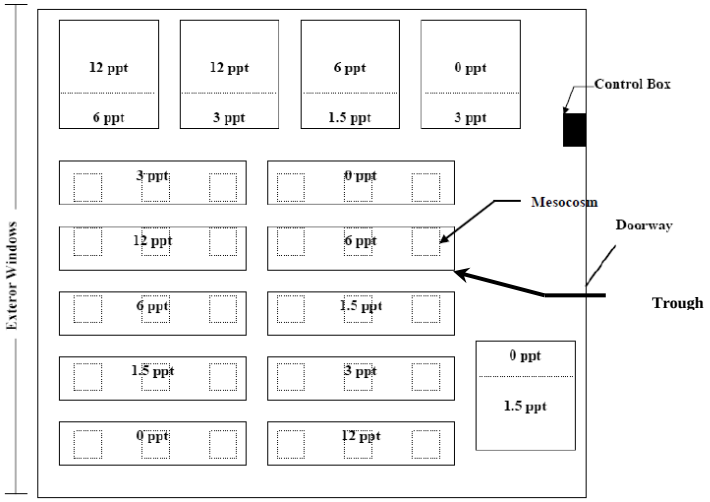


Figure 2. Plan view of experimental treatments and layout for the greenhouse mesocosm study (total experimental units = 30, salinity replicates = 2, and flooding frequency replicates = 10).

The mesocosms consisted of a container design that allowed control of water level and supply of salinity and nutrients. The mesocosm itself was a 56 x 44 x 44 cm (h x l x w; 151.4 L), Rubbermaid® Square Brute container Atlanta, GA with 16, 1.3-cm diameter perforations along the bottom to allow for exchange of water within the watering trough. Each mesocosm also had mesh screens installed at the bottom of each mesocosm over the drainage holes to prevent soil loss. The screens were made from plastic and had a 4 mm^2 mesh size. The watering troughs were made from pressure treated lumber and were (61 x 196 x 56 cm, 666 L). The troughs were designed to house three mesocosms each and were fed by a dedicated reservoir randomly assigned to that particular trough (Figure 2). The reservoirs were also constructed from pressure treated lumber and were (56 x 117 x 117 cm, 767 L) and were randomly located within the greenhouse. To prevent leaking, the troughs and reservoirs were lined with 45-mil thick black Firestone Pond liners Nashville, TN (Figures 3 and 4). Submersible pumps (Little Giant 115 Volt, Franklin Electric, Blufton, IN) were placed in the reservoirs and troughs to move water into and out of the system. The pumps were attached to a circuit board and timing mechanism set to a six hour interval rate. The circuit controller activated the pumps and allowed the reservoirs to fill over a period of 6 hours, at the end of the 6 hour cycle the system activated a second set of pumps and drained the system over a another 6 hour period. This 6 hour pumping cycle was established to simulate the natural tidal cycles of marshes within the Chesapeake Bay. Target salinity levels were achieved through the addition of Instant Ocean Sea Salt to our targeted treatment level and verified through the use of a handheld YSI-30 SCT meter.

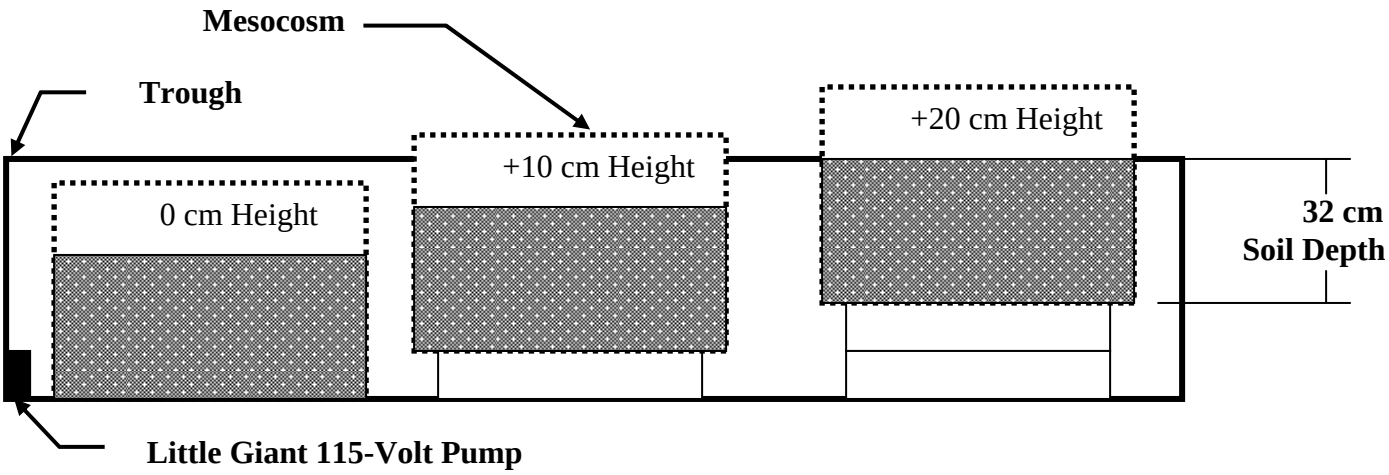


Figure 3. Profile drawing showing a conceptual layout of the marsh mesocosms within a trough.

Flooding frequencies were altered by elevating the mesocosms on concrete blocks; each mesocosm was randomly assigned a height of + 0 cm, +10 cm, or +20 cm above the trough bottom. These heights corresponded to a flood frequency (percent of hours in a 24 tidal cycle) that the soil surface was inundated with water 23%, (+20 cm), 44% (+10 cm), and 62% (+0 cm). Flooding frequencies were verified using an automatic water level (WL-15 Global Water, Inc Gold River, CA) recording device placed inside a representative trough and measured over a period of 24 hours. For reference, flooding durations measured from 29 marsh plots along the Nanticoke and Patuxent Rivers averaged 35% in 2006.

Experimental Plant Community – Mesocosm and Seedbank Studies

Figure 4. The lined trough and mesocosms in July 2007.

The goal of this experiment was to create a diverse assemblage of plant species representative of conditions across one of our previously surveyed river gradients. This goal was accomplished by inoculating the mesocosms with homogenized soils containing seeds collected along the Patuxent River marsh gradient and supplementing the seed bank with some dominant planted perennials identified previously (Chapter 2), and representative of the entire fresh-brackish salinity gradient. The rationale for including some species of brackish marsh plants was to provide a source of vegetative material that would allow plant communities to potentially shift from salt-intolerant to salt-tolerant communities if environmental conditions became appropriate, as occurs in coastal wetlands experiencing high rates of relative sea-level rise that do not convert directly to open water (Boesch et al. 1994; Perry and Hershner 1999). Previous research has used sections of marsh soil and vegetation collected intact from wetlands rather than synthetic plant communities proposed here (Baldwin and Mendelssohn 1998; Baldwin et al. 2001). However, we decided to use synthetic plant communities because we wished to assemble a diverse suite of propagules and vegetative material from a range of coastal wetland types to better understand how the diversity and composition of wetland vegetation would respond to different combinations of salinity and flooding treatments. Synthetic plant communities also have the added benefit of reducing variation between experimental units, allowing reduced numbers of replicates, and therefore greater numbers of treatment factor levels, than would be possible with more variable soil-vegetation sections.

Marsh surface soils were collected from four marsh locations (two freshwater sites, one transitional site, and one brackish site) along the Patuxent River on March 19-21, 2007. Marsh soils were collected by hand using 5 x 4.75 cm (h x d) corers. A total volume of 38 L (of the top five centimeters of topsoil) was collected from each of the four sites. An additional freshwater marsh site was needed due to concern that a sufficient number of freshwater annual plants would not germinate from a single site. As commercially grown wetland annuals are difficult to obtain, the additional fresh marsh site was included to ensure adequate representation of each salinity class in our mesocosms. The collected marsh topsoil was stored in 19 L buckets and placed in refrigerated conditions until April 17, 2007 when the soils were homogenized.

Marsh surface soil samples from each location were homogenized in a cement mixer and five (284 cm^3) samples from the homogenized soil from each site were extracted, and spread in a uniform 1-cm thick layer on top of a 2-cm thick layer of Sunshine LC1 potting soil mix within 4 x 14 x 20.3 cm (H x W x L) aluminum pans. Next the collected topsoil across all four marsh locations was homogenized by placing one bucket of topsoil from each marsh type into a cleaned and rinsed cement mixer. The cement mixer was run for seven minutes and the resulting mixture was placed back into the four empty buckets. This process was repeated for the remaining four topsoil sample buckets. Next, two buckets from each of the mixed sets were chosen haphazardly (four buckets total) and mixed again for five minutes and poured back into the empty buckets. This

process was repeated for the remaining four buckets. This process of mixing and re-mixing of the collected topsoil samples was utilized to achieve a homogeneous soil mixture.

Five 284-cm^3 volumes of soil were then extracted from the homogenous mix and placed in the aluminum pans as part of the seed bank variability component of this study. This process allowed us to characterize the seed banks of the individual collection sites, as well as the homogenized seedbank that was used in all the mesocosms.

The seedbank trays were randomly placed on a misting bench in the University of Maryland Research Greenhouse Complex and emerging seedlings counted by species. Soil seed banks contain seeds of several dominant annual species in low-salinity marshes, including *Polygonum* spp., *Impatiens capensis, Bidens* spp., and *Pilea pumila* (Baldwin and DeRico 1999; Peterson and Baldwin 2004). Application of a homogeneous soil sample is an effective way to introduce these species, many of which cannot be purchased from nurseries and for which seed collection would be necessary throughout the year. We anticipated that between the planted perennials and plants recruited from the seed bank would approach stem densities similar to those of natural marshes (e.g., 250 stems/m^2 in July and 150 stems/m^2 in August; Darke and Megonigal 2003).

Upon completion of topsoil homogenization and seedbank study set-up, mesocosm containers were filled with 30 cm of SUNGRO Professional Blend potting soil and inoculated with a 2-cm thick layer of collected marsh topsoil. The resulting mesocosms were put on a freshwater drip-line irrigation system, placed outside 4 April 2007 and then moved into the greenhouse (5 May 2007) and allowed to acclimate to greenhouse conditions until 11 July 2007 when the mesocosms were placed into our tidal system. Perennial wetland plants (two inch plugs) purchased from Environmental Concern, Inc. (St. Michaels, MD) were randomly planted at each of 16 positions (2 of each) within each marsh mesocosm on May 31, 2007. The perennial plants were selected based on availability and relative indicator value from a previous study (Chapter 2). The plant species were: *Acorus calamus, Distichlis spicata, Leersia oryzoides, Spartina alterniflora, Typha angustifolia, Spartina patens, Phragmites australis, and Spartina cynosuroides*. *P. australis* and *S. cynosuroides* were grown in the greenhouse from rhizomes harvested along the Patuxent River as these two species were not commercially available. All of the aforementioned perennial species were from Maryland ecotypes and two of each species were randomly placed into each mesocosm with the exception of *S. cynosuroides*. The *S. cynosuroides* rhizomes did not successfully generate enough viable plants for more than one of that particular species to be planted per mesocosm.

Mesocosm Operation

After the May 31, 2007 perennial planting event the mesocosms were maintained on a freshwater drip line system, the planted perennials were censused and dead planted perennials were removed and replaced prior to salinity treatment initialization on July 27, 2007. Salinity was altered by creating solutions of reconstituted sea water using Instant Ocean® sea water

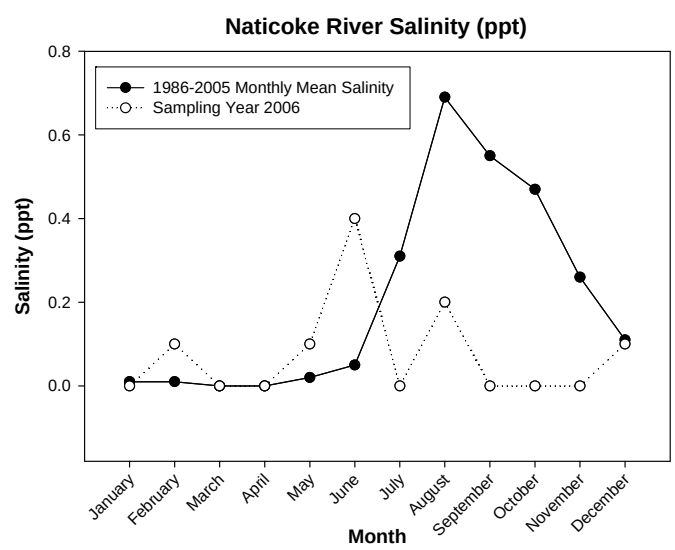


Figure 5. Nanticoke River Salinities Measured at Maryland DNR surface water quality station ET6.1 – Sharpetown, Maryland (near plot N35W) showing the mean monthly salinities measured from 1986-2005 and the mean monthly salinities from the 2006 sampling effort.

mix. After salinity treatments began for all reservoirs (except for the two fresh water troughs), final reservoir salinities were gradually ramped up over a period of twelve days. The initial salinity treatment brought reservoir salinity concentrations up to 0.75 ppt initially; followed by increases every other day, to the final levels of 1.5, 3.0, 6.0, 9.0 and 12.0 ppt. For those treatments whose target salinities were less than 12.0 ppt, no further salts were added to the system once the target salinity level was reached, except where necessary to maintain the treatment salinity level. The salinity levels were raised gradually to avoid shocking the plant communities with high salt concentrations. Historic salinity data from the Nanticoke and Patuxent River (Figures 5 and 6) also show that salt concentrations tend to spike in late July and August, so this procedure was employed to mimic field conditions on these two river systems (Maryland DNR "Eyes On The Bay Program, 2007). Apart from simulating natural salinity increases, this procedure also prevented inhibition of early season germination due to salinity (Odum et al. 1984 and Baldwin et al. 1996)

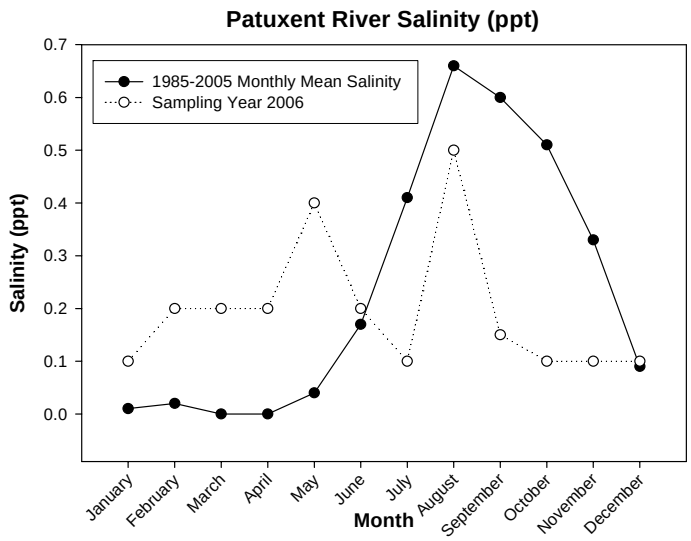


Figure 6. Patuxent River Salinities Measured at Maryland DNR surface water quality station TF1.5 at Nottingham, MD (near plot X30E) showing the mean monthly salinities measured from 1985-2005 and the mean monthly salinities from the 2006 sampling effort.

The mesocosms were operated from the middle of the growing season (July 2007) to the end of July 2008. The salinities in all tanks above 0 ppt were reduced by 50% from 9 October 2007 until 1 May 2008 to simulate the seasonal retreat of the salt front from the fall through early summer. Due to evapo-transpiration losses the water within each mesocosm system was replaced, on average, once per week. Flooding regimes in the mesocosms were maintained 10 cm below the soil surface for 2 weeks so that plants could acclimate, after which water levels were adjusted to their appropriate experimental treatment condition (0cm, +10 cm, and +20 cm). This occurred concurrently with the salinity exposure.

Vegetation and Environmental Measurements

Vegetation in mesocosms was censused non-destructively by using species presence/absence determinations and by estimating percent cover of each plant type using cover class from the North Carolina Vegetative Survey protocol (Peet et al. 1998). This census was performed at the beginning of the salinity/flooding treatments in June 2007, September 2007, and July 2008. The purpose of the initial monitoring was to describe variation in the initial structure of plant communities between mesocosms and track potential treatment effects within and between the mesocosms. Experimental treatment water was also periodically analyzed for salinity, pH, and temperature using YSI portable

meters. Treatment water samples were also analyzed periodically for nitrate-nitrogen levels using a portable spectrophotometer (Hach 2000). Study mesocosm soils were also collected dried at room temperature, ground, and analyzed for water soluble-P (USDA 2000), Mehlich-3 extractable aluminum (Al), potassium (K), iron (Fe), calcium (Ca), and phosphorus (P). The purpose of the water and soil chemistry data collection was to identify any potential covariates that might affect the hypothesized outcomes.

At the conclusion of the experiment in July 2008, the aboveground biomass was harvested, separated by species, dried to a constant mass at 70°C, and weighed. The below ground biomass was harvested by using a high pressure water hose and sieve system to separate the roots from the soil matrix. Plant roots were dried to a constant mass at 70°C, and weighed.

Data Analysis

Species richness was calculated using July/September 2007 and July 2008 species count data. Shannon-Wiener diversity values were calculated using the July 2008 data. Above and below ground biomass values were analyzed separately as dependent variables using a two-way analysis of variance (ANOVA) using SAS version 9.1. Additionally, average plant species richness from 2007 and 2008 were analyzed in an ANOVA analysis against salinity and flood frequency independent variables. In instances where no significant block effects were found in the initial ANOVA analysis, the blocking factor was removed and the analysis was rerun to improve statistical power.

The environmental variables such as trough water pH, nitrate-nitrogen, and temperature were analyzed using repeated measures ANOVA analysis (proc mixed procedure) in SAS version 9.1. All soil chemistry data was analyzed using the split-plot ANOVA analysis in SAS described previously.

Non-metric multidimensional scaling (NMS) was also employed as a multivariate analysis tool for determining the relative strength of relationships between vegetation, salinity, and flooding frequency variables. The NMS analysis used a Sorenson (Bray-Curtis) distance measure with a 0.0000001 stability criterion and a maximum of 500 iterations (McCune and Grace 2002). In the NMS analysis plots were identified as Group 1-5 based on the salinity treatment for that set of mesocosms (Group 1 = 0 ppt, Group 2 = 1.5 ppt, Group 3 = 3.00 ppt, Group 4 = 6 ppt, and Group 5 = 12 ppt). NMS analysis was completed using PC-ORD Version 5.0 (MjM Software Design, Gleneden Beach, OR).

Results

Seedbank Observational Study

The results of the seedbank community study showed some significant variation in plant species richness and dominant plants between collection sites. The upper most fresh marsh community (Fresh 2) differed significantly from the brackish marsh seedbank ($p = 0.01$, Tukey adjusted) and there were no significant differences between the fresh and oligohaline seedbanks (Figure 7). As was expected the mixed seedbank, which was an amalgamation of seeds from all four sites, displayed the highest average richness, and was significantly higher then the brackish ($p < 0.01$) and lower fresh marsh site (Fresh 1) ($p < 0.01$). *Eleocharis parvula* and *Pluchea purpurascens* were the most frequently observed plant species from the brackish seedbank ($\bar{x} = 662 \pm 83$ and $\bar{x} = 42 \pm 4$ seeds/sample respectively) and the mixed community seedbank ($\bar{x} = 43 \pm 36$ and $\bar{x} = 12 \pm 1.5$ seeds/sample respectively). A total of 36 species were observed across all seedbank communities (Table 1); average frequencies for most seedbank species ranged from 1 to 20 individuals.

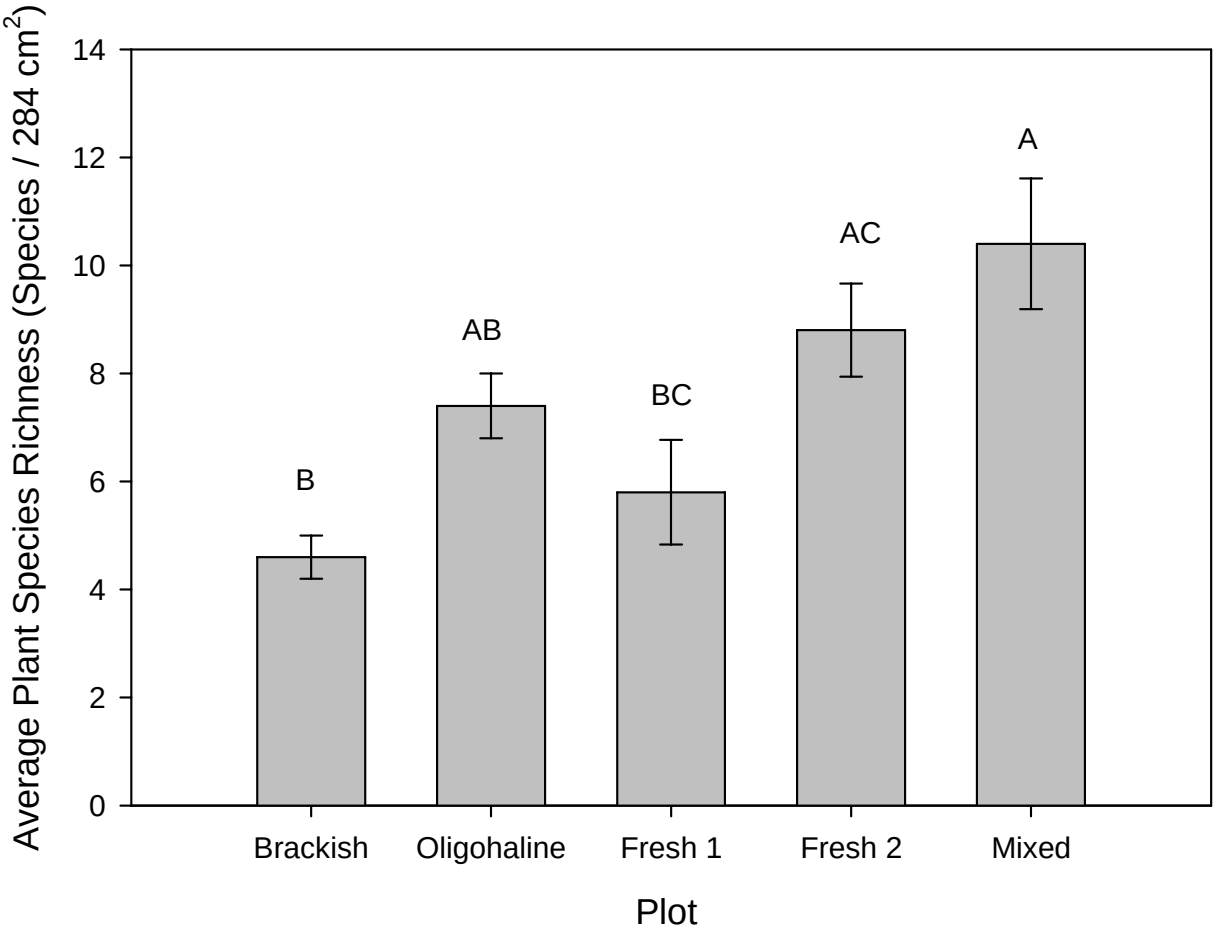


Figure 7. Average plant species richness (± standard error) from seedbanks collected from four locations along the Patuxent River, MD (n = 5). Means that do not share any letters are significantly different (Tukey's HSD, $p < 0.05$).

Table 1. Plant species observed within collected tidal marsh seedbanks along the Patuxent River (values are means, SE is the standard error).

	Fresh 2		**Fresh 1**		**Oligohaline**		**Brackish**		**Mixed**	
Species	Mean Frequency	SE	Mean Frequency	SE	Mean Frequency	SE	Mean Frequency	SE	Mean Frequency	SE
Alnus rugosa (du Roi) Spreng.			0.20	0.20						
Amaranthus cannabinus (L.) Sauer	0.60	0.24	0.20	0.20	0.20	0.20			0.60	0.24
Aster puniceus L.	1.60	0.68	0.80	0.37					0.20	0.20
Aster simplex Willd.	1.00	1.00								
Atriplex sp.							0.20	0.20		
Boehmeria cylindrica (L.) Sw.	0.80	0.49	0.60	0.24					0.80	0.37
Cardamine pensylvanica Muhl. Ex Willd.					4.60	2.60	0.60	0.40		
Cinna sp.									0.20	0.20
Cuscuta gronovii Willd. Ex Schult.	0.20	0.20	0.20	0.20						
Cyperus erythrorhizos Muhl.	4.20	0.58	0.20	0.20					2.00	0.32
Cyperus odoratus L.							0.20	0.20		
Cyperus spp.	0.20	0.20	0.20	0.20						
Echinocloa sp.	0.20	0.20					0.20	0.20	0.20	0.20
Eleocharis parvula (Roem. & Schult.) Link ex Bluff, Nees & Schauer	0.40	0.40	8.60	8.60			662.80	82.58	43.60	36.28
Hibiscus moscheutos L.					0.40	0.24				
Iva frutescens L.							3.00	1.95	0.40	0.24
Juncus effusus L.	0.20	0.20								
Kosteletzkya virginica (L.) Presl			0.20	0.20						
Leersia oryzoides (L.) Sw.	2.40	0.60			5.20	0.80			1.40	0.24
Lobelia cardinalis L.									0.20	0.20
Lythrum salicaria L.					0.80	0.49				
Mentha arvensis L.									0.20	0.20

Species	**Fresh 2** Mean Frequency	SE	**Fresh 1** Mean Frequency	SE	**Oligohaline** Mean Frequency	SE	**Brackish** Mean Frequency	SE	**Mixed** Mean Frequency	SE
Pilea pumila (L.) A. Gray	7.00	1.38	6.20	1.36	0.40	0.24			3.40	1.30
Pluchea purpurascens (Sw.) DC.	0.20	0.20			0.80	0.37	42.20	4.44	12.00	1.58
Polygonum arifolium L.	0.20	0.20			0.20	0.20			0.40	0.24
Polygonum punctatum Elliot					1.40	0.40				
Polygonum sagittatum L.			0.20	0.20						
Sagittaria latifolia Willd.	0.20	0.20								
Schoenplectus fluviatillis (Torr.) M.T. Strong									0.20	0.20
Schoenplectus robustus (Pursh) M.T. Strong							1.00	1.00		
Schoenplectus tabernamontani (C.C. Gmel.) Palla	1.00	1.00	1.40	1.40	2.00	2.00				
Spartina cynosuroides (L.) Roth					0.60	0.40	0.20	0.20	0.80	0.80
Spartina patens (Aiton) Muhl.							4.80	4.55		
Teucrium sp.					0.40	0.40				
Typha angustifolia L.			0.20	0.20						
Typha spp.	0.80	0.37	1.40	0.60	1.40	2.59			2.40	0.93

Mesocosm Study

Data were originally analyzed as a block design, but the block effect was not significant, therefore it was removed from the model. The results of the overall split plot ANOVA supported our initial hypothesis regarding the impact of salinity on plant species richness, specifically that salinity would create significant differences in low versus high salinity treatment mesocosms (Table 2). This is also supported by the clear trend observed in the July 2008 mesocosm richness data that show a clear downward trend in richness between the low-salinity oligohaline mesocosms (1.5 ppt) and the most saline treatment mesocosms (12 ppt) (Figure 8). Flooding frequency and the interaction between flooding frequency and salinity effects were also not significant, which was contrary to our original hypothesis that flooding has a strong influence on tidal marsh plant diversity.

Table 2. Overall Type III Test of Fixed Effects using plant species richness (July 2008) as the response variable and salinity, flooding frequency, and salinity*flooding frequency as independent variables. Richness values are from species counts per mesocosm.

Effect	Num DF	Den DF	F Value	Pr > F
Salinity	4	15	6.01	0.0043
Inun (Flooding Frequency)	2	15	1.79	0.2016
Salinity*Inun	8	15	0.54	0.8057

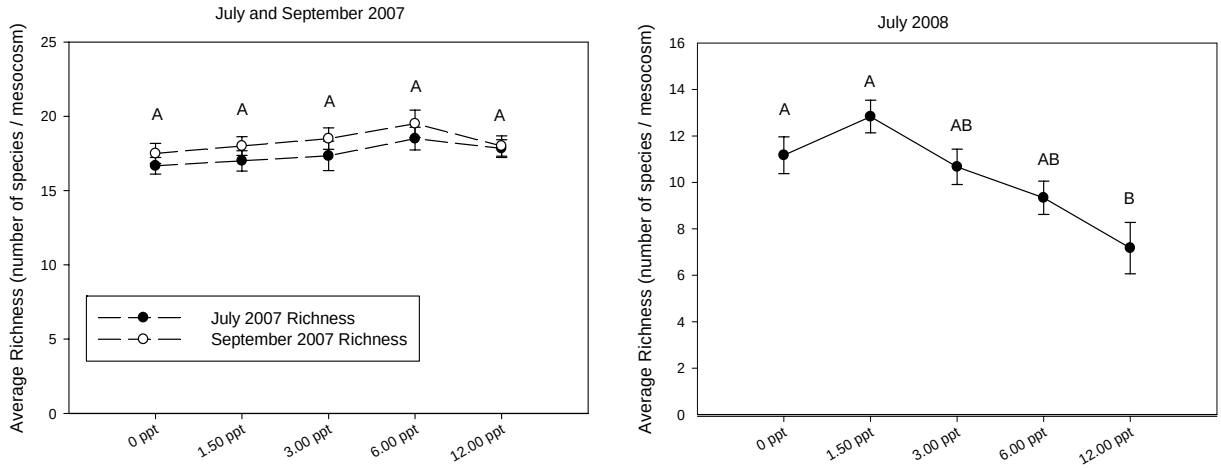


Figure 8. Mesocosm plant species richness during the initial portion of the experiment (2007) and following one entire year of salinity treatments (July 2008). Bars depict mean plant species richness based on salinity treatment group with standard error bars and significant differences depicted. Means with different letters are significantly different with each date (Tukey's HSD, $p < 0.05$). Richness values are from species counts per mesocosm.

These results of the July 2008 richness data differ from the preliminary findings of this study in 2007 which found no significant differences in plant species richness between salinity treatments at either the initial (June 2007) or late growing season (September 2007) plant surveys. Additionally, mean plant species richness within all of the mesocosms showed a marked decline between 2007 and 2008 (Figure 11). This was likely due to little or no influx of seeds from 2007 to 2008 and no cold stratification within the greenhouse environment between growing seasons. However, exposure of seedlings to elevated salinity levels early in the growing season of 2008 produced trends in the low-salinity oligohaline (1.50 ppt) mesocosms similar to those observed along the Nanticoke River in 2006 (see Figure 1 and Chapters 1 and 2). The observed trend in the July 2008 data in Figure 11 was also the same as the Nanticoke River data in that the low-salinity (1.5 ppt) mesocosm community had a average richness values comparable (not significantly different) to the fresh marsh community. As in Chapter 2 this difference was not significant at the 0.05 level. Additionally, average Shannon-Wiener indices of plant species diversity across all salinity treatments yielded no significant differences (Figure 9).

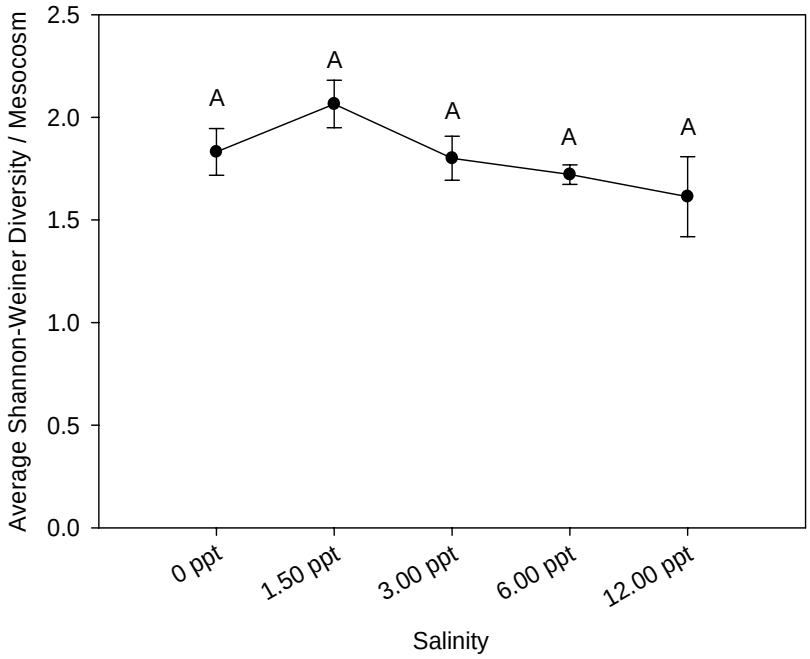


Figure 9. Average Shannon-Wiener Diversity (± SE) based on the July 2008 biomass data. Means with different letters are significantly different (Tukey's HSD, $p < 0.05$)

No significant differences with regards to plant species richness were observed between flooding frequency treatments (Figure 10).

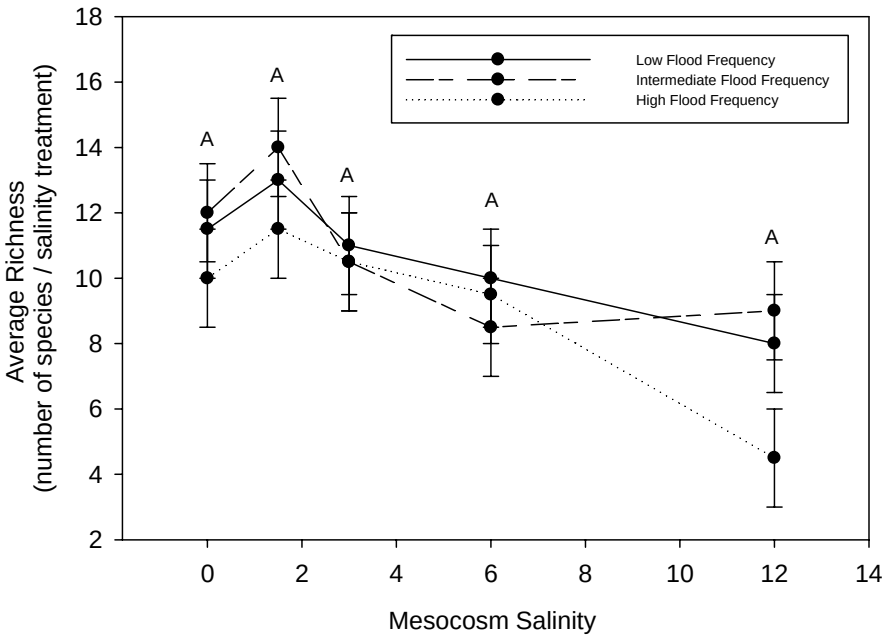


Figure 10. Average mesocosm plant species richness values based on the July 2008 biomass data and separated out by flood frequency to show potential interactions and trends. Means with different letters are significantly different (Tukey's HSD, $p < 0.05$)

Several species of plants did not regerminate and grow between the 2007 and 2008 sampling years, some of these species included *Apios americana*, *Bidens laevis*, *Cyperus esculentus*, and *Zizania aquatica* (Table 3). Additionally, species such as *Amaranthus cannabinus*, which was a dominant plant throughout many of the marsh mesocosms in 2007 based on aerial cover ($\bar{x}$ = 45 - 35% from 29 mesocosms), was present for final sampling in July 2008, but had a much lower presence and cover value ($\bar{x}$ = 15% from 5 mesocosms). Species such as *Iva Frutescens*, *Rumex* sp., and *Samolus parviflorus* were not observed in 2007 but volunteered in 2008. Of the plant species observed in the mesocosms in September 2007 80% (31 species) of them grew from the seed bank of the mesocosms, with the remaining 20% (8 species) being species which we planted randomly within each mesocosm. The July 2008 plant species list shows a 75% recruitment of plant species from the seedbank (24 species), there was also a slight drop in the total number of species between September 2007 (39 species) and July 2008 (32 species), as well as a minor drop in total cover following treatments (Table 3).

Table 3. Mesocosm mean plant species cover and standard errors for June and September 2007, and July 2008. Mean species cover was averaged across all 30 mesocosms.

Species	Mean Cover June 2007	Mean Cover September 2007	Mean Cover July 2008
Acorus calamus L.	1.76 +/- 0.21	3.65 +/- 0.79	1.63 +/- 0.43
Amaranthus cannabinus L.	44.82 +/- 4.32	35.36 +/- 3.63	16.88 +/- 3.71
Apios americana Medic.	1.5 +/- 0.00		
Aster puniceus L.	0.5 +/- 0.00	3 +/- 0.46	9.9 +/- 1.30
Atriplex sp.			
Bidens laevis L.	17.5 +/- 0.00	29.17 +/- 5.35	
Bidens sp.		0.50 +/- 0.00	
Bidens coronata (L.) Britt.	24.17 +/- 2.11		
Boehmeria cylindrica (L.) Sw.	0.50 +/- 0.00	3 +/- 0.55	8.83 +/- 1.48
Cinna sp.		0.5 +/- 0.00	1.17 +/- 0.11
Cuscuta gronovii Willd.	2.93 +/- 1.48	3.21 +/- 0.48	
Cyperus sp.		1.83 +/- 0.28	
Cyperus strigosus L.		1.5 +/- 0.00	13.11 +/- 2.59
Cyperus esculentus L.	0.5 +/- 0.00	1.83 +/- 0.28	
Cyperus filicinus Vahl		1.5 +/- 0.00	
Decodon verticillatus (L.) Ell.		0.5 +/- 0.00	
Distichlis spicata (L.) Greene	7.5 +/- 0.00	2.27 +/- 0.69	4.07 +/- 1.36
Echinochloa muricata (Pursh) Nash			9.43 +/- 2.80
Echinochloa walteri (Pursh) Nash	17.39 +/- 2.40	28.77 +/- 3.75	
Eleocharis parvula (R.&S.) Link			1.8 +/- 0.44
Galium tinctorium L.	0.5 +/- 0.00	0.75 +/- 0.09	2 +/- 0.39
Galium palustre L.	0.5 +/- 0.00	0.5 +/- 0.00	
Hibiscus moscheutos L.		7.5 +/- 0.00	
Hibiscus sp.	2 +/- 0.39	5.5 +/- 0.52	
Impatiens capensis Meerb.	7.5 +/- 0.00		
Iva frutescens L.			1.5 +/- 0.00
Juncus effusus L.			0.5 +/- 0.00
Juncus sp.		0.5 +/- 0.00	
Kosteletzkya virginica (L.) Presl		1.5 +/- 0.00	13.5 +/- 1.15
Leersia oryzoides (L.) Sw.	16.03 +/- 0.70	28.2 +/- 3.15	26.99 +/- 4.50
Lythrum salicaria L.	0.5 +/- 0.00	3.5 +/- 0.00	8.5 +/- 1.21
Mikania scandens (L.) Willd.	0.74 +/- 0.14	19.07 +/- 2.06	40.85 +/- 4.55
Murdannia keisak (Hasskarl) Hand.-Mazz	0.5 +/- 0.00	2.5 +/- 0.51	2.15 +/- 0.47
Nasturtium offiicinale R. Br.	7.3 +/- 1.03		
Peltandra virginica (L.) Schott & Endl.	1.86 +/- 0.82	1.33 +/- 0.21	0.23 +/- 0.04
Phragmites australis (Gav.) Trin.	3.1 +/- 0.67	9.35 +/- 1.43	27.47 +/- 3.10
Pilea pumila (L.) Gray	28.05 +/- 2.65	16.95 +/- 2.24	0.55 +/- 0.12
Pluchea purpurascens (Sw.) DC.	33.5 +/- 2.72	8.73 +/- 1.16	14.69 +/- 3,14
Poaceae sp.			7.5 +/- 0.00
Polygonum arifolium L.	41.39 +/- 3.70	19.75 +/- 2.80	0.5 +/- 0.00

Species	Mean Cover June 2007	Mean Cover September 2007	Mean Cover July 2008
Polygonum punctatum Ell.	16.34 +/- 2.00	18.83 +/- 2.25	
Polygonum sagittatum L.	5.23 +/- 0.73	4.5 +/- 0.37	
Polygonum sp.	7.5 +/- 0.00		
Rorippa islandica (Oeder) Borbas	0.5 +/- 0.00	0.5 +/- 0.00	
Rumex sp.			8.17 +/- 1.48
Samolus parviflorus Raf.			8.42 +/- 1.53
Schoenplectus sp.	0.5 +/- 0.00		3.13 +/- 0.59
Senecio sp.	0.5 +/- 0.00		
Sonchus sp.	17.5 +/- 0.00		
Spartina alterniflora Loisel.	7.5 +/- 0.00	2.23 +/- 0.65	0.06 +/- 0.05
Spartina cynosuroides (L.) Roth	1.3 +/- 0.26	2.5 +/- 0.38	5.05 +/- 0.90
Spartina patens (Ait.) Muhl.	7.5 +/- 0.00	6.3 +/- 1.04	19.75 +/- 4.00
Typha angustifolia L.	0.5 +/- 0.00	0.68 +/- 0.07	1.34 +/- 0.37
Zizania aquatica L.	12.5 +/- 1.29	7.5 +/- 0.00	
Unidentified Dicot		0.5 +/- 0.00	0.1 +/- 0.00
Unidentified Dicot 2			0.1 +/- 0.00
Total Species Count	**37**	**39**	**32**
Total Cover	**339.91**	**285.77**	**259.84**

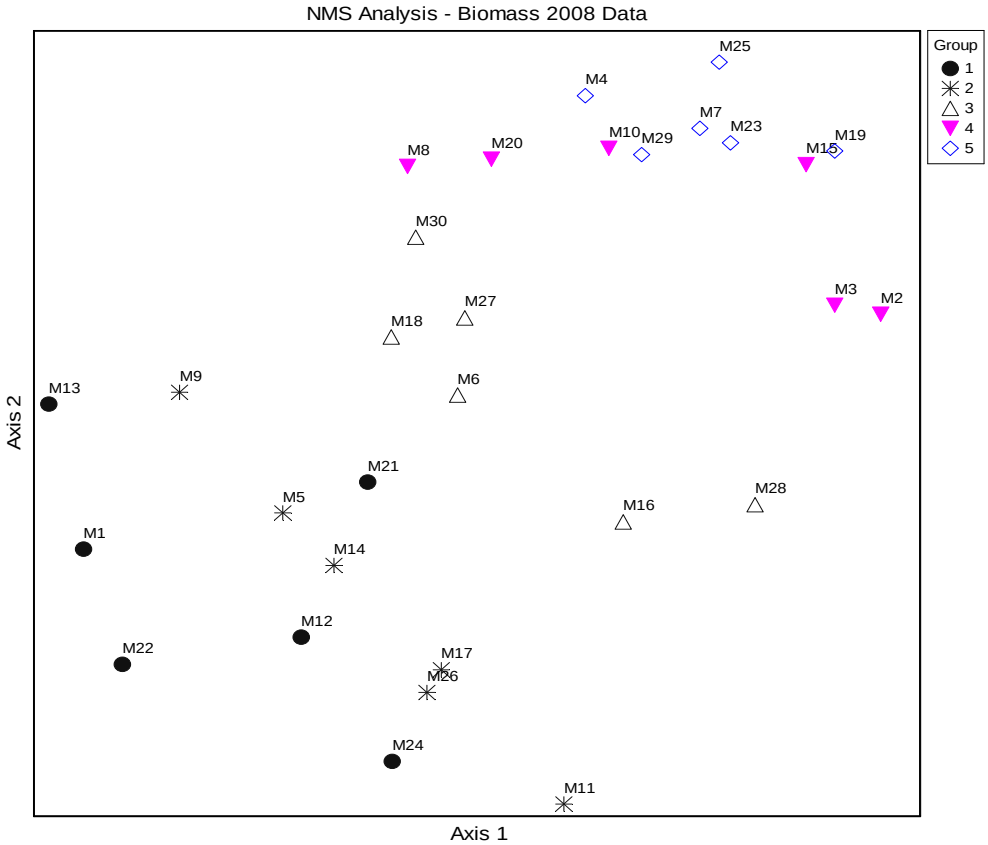


Figure 11. NMS two dimensional graph showing the mesocosm biomass data, individual points are mesocosms from the final harvest in July 2008. The groups are arranged by salinity treatment with Group 1 = 0 ppt, Group 2 = 1.5 ppt, Group 3 = 3 ppt, Group 4 = 6 ppt, and Group 5 = 12 ppt. Points are individual mesocosms.

Not only were there observed changes in individual plant species occurrence and abundance between 2007 and 2008, but there was also a strong shift in the plant species communities themselves in response to the salinity treatments. Figure 11 shows an NMS graph of the mesocosm species biomass from July 2008 relative to salinity and flooding. Clear patterns in the plant communities shown by distinct clustering of mesocosms arranged by salinity treatment can be readily observed. These results suggest that over the course of the 2007-2008 year the plant communities began to shift in response to the treatments, with fresh water marsh species doming minatin low-salinity ranges and salt tolerant species dominating in the high salinity mesocosms. This outcome supports the hypothesis of plant community shifts in response to the salinity treatments. Differences in the above ground biomass of the ten most abundant plant species based on biomass and frequency of occurrence within study mesocosms also varied as a function of salinity. Fresh marsh plant species such as *Mikania scandens*, *Cyperus* sp.1, and *Leersia oryzoides* displayed higher biomass in the low-salinity ranges of the experiment (0-1.5 ppt) and a general decline in biomass as salinity increased. *Phragmites australis* and *Spartina cynosuroides*, two species common in oligohaline-brackish marshes along the Patuxent and Nanticoke Rivers, showed no pattern of biomass differences across the salinity range (0-12 ppt) (Figure 15). *Spartina patens* had higher average biomass in mesocosms exposed to salinity treatments ranging from 6-12 ppt in 2008. Fresh marsh plant species such as *Mikania scandens*, *Cyperus* sp.1, and *Leersia oryzoides*

displayed higher biomass in the low-salinity ranges of the experiment (0-1.5 ppt) and a general decline in biomass as salinity increased.

Phragmites australis and *Spartina cynosuroides*, two species common in oligohaline-brackish marshes along the Patuxent and Nanticoke Rivers, showed no pattern of biomass differences across the salinity range (0-12 ppt) (Figure 12). *Pluchea purpurascens and Kosteletzkya virginica*, two species also found in oligohaline-mesohaline marshes showed distinct peaks at 3 and 6 ppt respectively. As the graphs in Figure 12 only show the average plant biomass/salinity treatment, it's possible that at extreme fresh water and salt water conditions the combination of salinity and flooding frequency at one end, versus competition and flooding at the other imparted restrictions on these species distributions and caused their peak biomass to occur near the middle of the salinity range.

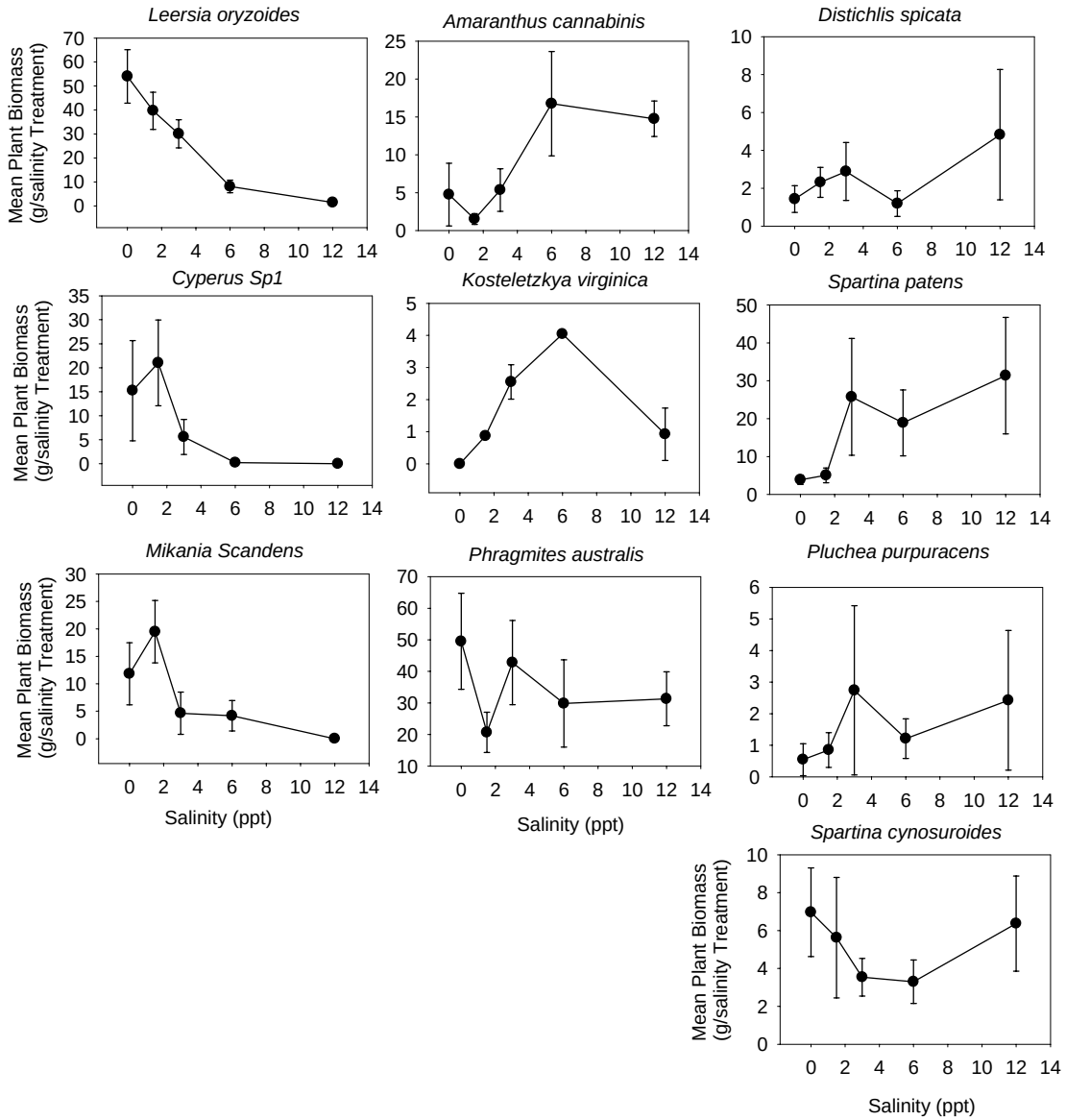


Figure 12. Graphs showing the above ground biomass (g/salinity treatment) of the ten most abundant plant species from the July 2008 biomass data from all 30 mesocosms. Individual points represent mean species biomass per salinity treatment ± SE.

Mesocosm Chemistry

No significant differences in the water soluble-P or in water nitrate-nitrogen levels were observed between treatment groups. Significant differences in average Mehlich-3 extractable magnesium, potassium, and calcium levels were observed with the high salinity mesocosms (12 ppt) having higher magnesium and potassium concentrations in the soil compared to the 0 ppt and 1.5 ppt mesocosms. Mean calcium levels were significantly higher in the purely fresh water (0 ppt) mesocosms compared to the higher salinity level treatments which was likely due to calcium precipitating out in the high salinity mesocosms as $CaSO_4$. These elemental differences were not unexpected as the Instant Ocean mix contains these micronutrients and was added to the water supply of all the salt treated tanks. There were no significant differences in average mesocosm porewater pH which ranged from 6.08 to 6.97. A significant overall difference ($p = 0.0002$, $F_{9,306}=3.69$) was observed between trough water temperatures (Figure 13).

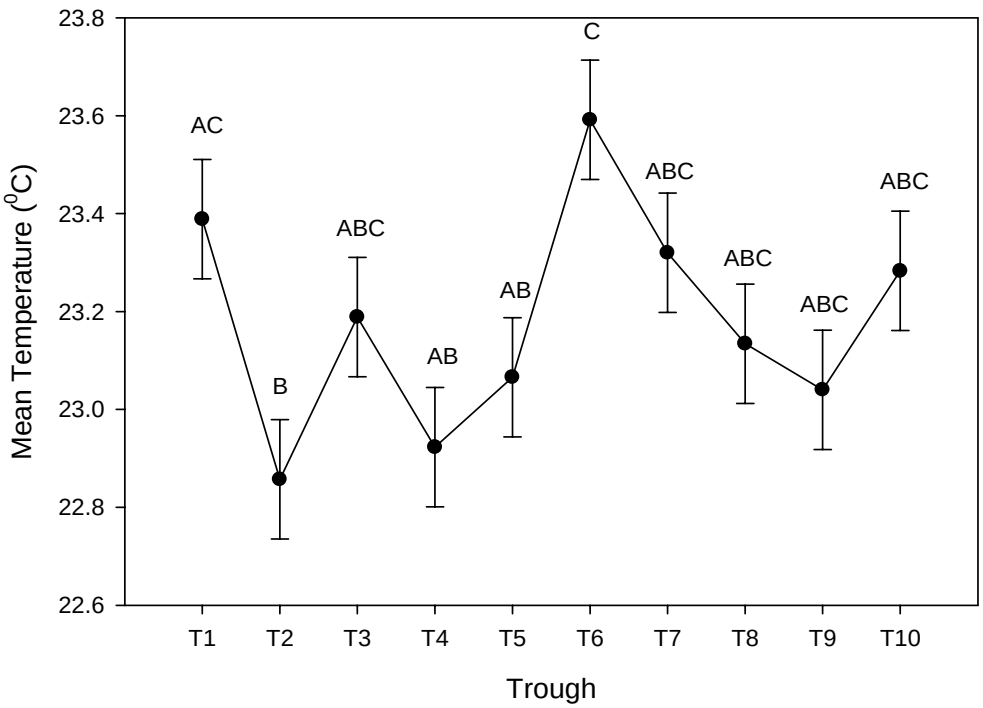


Figure 13. Average trough water temperatures measured at 35 different times over the course of the experiment (2007-2008). Means which share a letter are not significant at the 0.05 level.

Though these data show significant differences between some of the experimental trough water temperatures, it's unlikely that these differences are significant at a biological level as the difference between the highest mean temperatures (Trough 6 – 23.59 ^{0}C) and the lowest mean temperatures (Trough 2 – 22.85^{0}C) was less than 1^{0}C during the growing season.

Biomass

Our initial hypothesis was that above and below ground plant biomass would be significantly higher in the marsh mesocosms subjected to lower salinity and flood frequency disturbances. The results of the ANOVA analysis found no significant differences in mean above ground biomass across salinity and flood frequency treatment levels for the study mesocosms at the 0.05 level (Figure 14). These results coupled with the NMS output (Figure 11) and individual species biomass graphs (Figure 12) suggest that as some species are eliminated with increasing salinity they are replaced by salt tolerant species (assuming seed or propagule material is available). This replacement of species helps offset the loss of biomass in the marsh mesocosms. Mean above ground biomass among salinity treatments separated out by flooding frequency shown in Figure 15 also show no significant differences between salinity treatments.

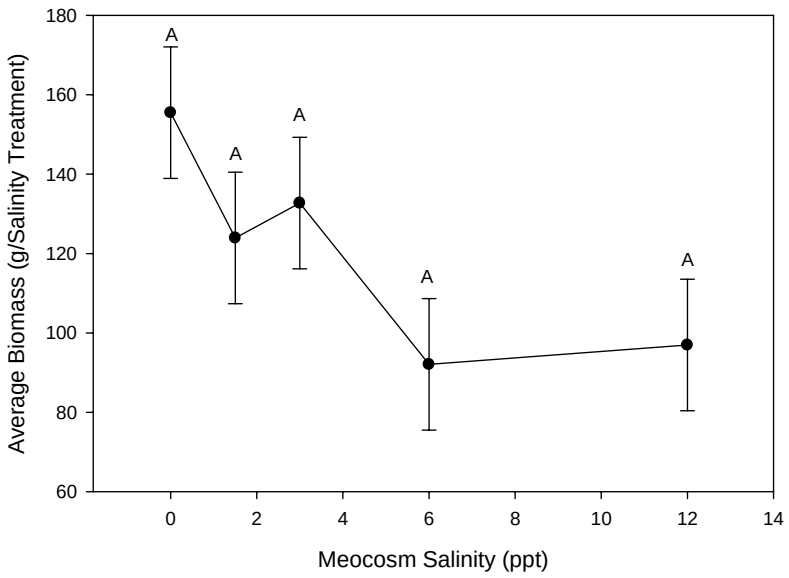


Figure 14. Mean above ground biomass versus salinity for the five salinity groups. Different letters designate significant differences values are salinity group means (n=6) ± SE.

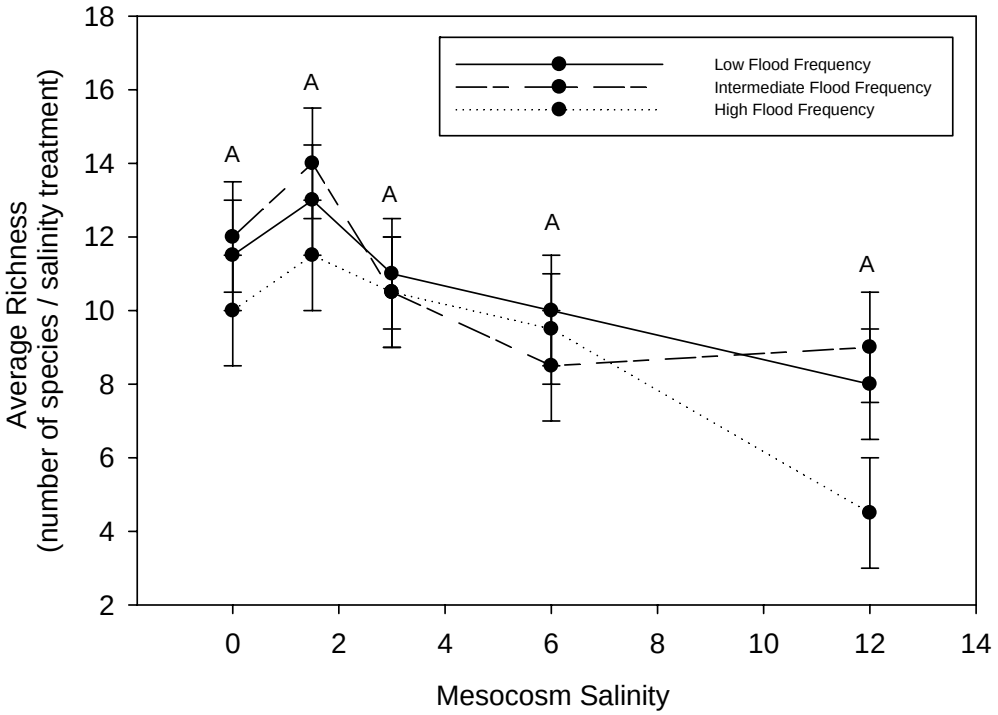


Figure 15. Mean above ground biomass versus salinity group for the three flooding treatments using July 2008 mesocosm biomass data.

Mean below ground biomass (July 2008) versus salinity treatment for the three flooding treatments displayed no significant statistical differences between these means that is consistent with the above ground biomass data in Figure 14 and suggests that the more saline tolerant species were able to minimize the impacts of increased salinity and flood frequency on the marsh mesocosms.

Discussion

Considerable research has been conducted on how salt and brackish marshes will respond to sea-level rise (Mitsch and Gosselink 2000; Morris et al. 2002; Turner et al. 2004). Much of this research has focused on the ability of salt marshes to accrete vertically at sufficient rates to keep pace with sea-level rise and the role of macrophytes in marsh stability or loss (Kearney et al. 1994; Roman et al. 1997; Day et al. 1999), or on the responses of marsh vegetation to increases in salinity and water level or soil waterlogging (Mendelssohn et al. 1981; Pezeshki et al. 1993; Broome et al. 1995; Naidoo et al. 1997; Gough and Grace 1998). These and other studies have demonstrated the importance of mineral sediment and organic matter deposition, which are critical to maintaining elevation (Reed 1995), and tolerance of marsh vegetation to increases in salinity and water logging (Kozlowski 1997). In general, growth and survival of salt and brackish marsh vegetation is reduced by increases in soil waterlogging, such as those that may occur due to sea-level rise (e.g., Webb et al. 1995; Mendelssohn and Batzer 2006). Loss of salt and brackish marshes in areas such as the Mississippi River delta plain (Louisiana) and the Chesapeake Bay is believed to primarily be the

result of an inability of marsh elevation to keep up with relative sea level, which increases soil waterlogging and anoxia, stressing or killing salt marsh plants (Stevenson et al. 1985; Boesch et al. 1994).

In contrast to salt and brackish marshes, responses of tidal low-salinity marshes to sea-level rise have received little attention, with the exception of those in the Louisiana delta plain. Research in Louisiana has shown that increases in salinity, as well as soil waterlogging, due to high rates of relative sea-level rise result in vegetation dieback and wetland loss (McKee and Mendelssohn 1989; Boesch et al. 1994; Flynn et al. 1995; Webb and Mendelssohn 1996). These findings suggest that low-salinity marshes in other estuaries are similarly sensitive to increases in both relative water level and salinity. In the Chesapeake Bay, Kearney et al. (1988) found that marsh losses in the Nanticoke River estuary since the 1920s had occurred primarily in the lower portions of the estuary; tidal freshwater marshes remained relatively stable, probably because they occur in the sediment-trapping portion of the estuary. However, it is likely that as sea level rates continue to accelerate, the salt wedge and the zone of major sediment deposition will move farther upstream (Meade 1972; Officer 1981), resulting in vegetation dieback or conversion to salt-tolerant species.

The overall goal of this research was to understand how changes in salinity and water level influenced diversity and ecosystem function of tidal marsh communities grown in a controlled greenhouse environment. Our preliminary research hypothesis was that marsh mesocosms subjected to increased salinities and flood frequencies would display diminished plant species richness, diversity, and productivity with an associated shift to fresh marsh plants at low salinities and brackish marsh plants at the high end of the spectrum. Additionally, we were curious as to whether or not average plant species richness would be highest in mesocosms subjected to low oligohaline (0.75-1.50 ppt) salinity conditions similar to the pattern observed in the Nanticoke River (Chapter 2).

Salinity and Tidal Marsh Plant Species Richness

The preliminary species richness and plant community data collected in June and September 2007 showed no significant differences based on the main effects of salinity and flood frequency. Additionally no significant shifts in the plant communities from the initial mixtures were observed between June and September 2007. The results in 2007 were contrary to our research hypothesis, however, this was likely due to salinity and flooding treatments not being initiated until July of 2007 which allowed the plants to establish themselves and grow undisturbed for three months prior to treatment. Changes to the salinity and flooding regimes within the mesocosms are likely to have less of an effect on vegetation that has already become established and thus more resistant to environmental perturbation.

Plant species community data from the second year (2008) following seedling exposure to salinity and flooding treatments yielded results more consistent with our research hypothesis. However while average plant species richness was highest in the low-salinity oligohaline marsh mesocosms (0.75 – 1.50 ppt), it was not significantly different than purely fresh marsh mesocosms. This finding supports the results from Chapter 2 regarding the similarity in pattern between the low salinity mesocosms in the experiment and Our observed findings from the Nanticoke River in 2006. It would appear from Our observations and this experiment that plant species richness/diveristy along some estuarine systems can be more accurately described by a sigmiodal response to salinty rather than a simple linear relationship.

Ecological modeling determined that salinity and inundation frequency were more important overall than the MDE (Chapter 3). Therefore, we hypothesized that the observed pattern in plant species richness was the result of periodic salt water intrusions into low-salinity marshes, which

suppressed the more competitively dominant fresh marsh plants, and allowed the salt tolerant species to survive and grow promoting high plant species richness/diversity. The results of this experiment which removes the influence of the MDE by mixing all short and large range species together, support this hypothesis and suggest that low-salinity oligohaline marshes may have plant species richness and diversity values equal to or sometimes even higher than purely tidal fresh water marshes. These findings lend further support to the theory of a more complex pattern of plant species richness along estuarine gradients which is contrary to the general trend of decreasing richness with increasing salinity noted widely elsewhere (Anderson et al. 1968, Odum 1988, Mitsch and Gosselink 2000, Greenberg et al. 2006).

Elevated Flooding and Salinity Effects on Tidal Low-Salinity Marshes

This research suggests that tidal marsh plant communities continuously exposed to salinities as high as 12 ppt with a concurrent increase in flooding frequency midway through the growing season are somewhat resilient to perturbation, provided the plant community is well established prior to disturbance. However, continued exposure to elevated salinity and flooding frequencies (particularly early in the growing season) caused a shift in the plant community types from more fresh-marsh plants to more brackish-marsh plants. Based on direct observation and statistical analysis of the harvested biomass it appears that the plant communities were able to convert to more mesohaline systems without a significant diminishment in biomass, provided that a source of seed/propagules of salt/flood tolerant species were available.

Implications

These findings suggest that low-salinity tidal marshes subjected to increases in flooding and salinity can maintain vegetation albeit with reduced plant biomass (at least initially), provided that they have a diverse enough assemblage of salt and flood-tolerant species in the seedbank or as available rhizome material. One plant species that seemed particularly adapt at surviving and growing under our range of salinity and flooding treatments was *Phragmites australis*. In general this plant did not show a significant diminishment in biomass across the salinity range, except under extreme flooding and salinity treatments. Given that *Phragmites australis* is a C3 plant, can propagate from seed or rhizome material, and can tolerate high flooding and salinity conditions it is already well adapted for marsh growth under elevated atmospheric carbon dioxide, salinity, and flooding conditions. Additionally, despite many efforts to remove or limit this plant species from tidal marshes within Chesapeake Bay, it still remains prevalent throughout much of the Bay ecosystem. We suggest that natural resource managers and agencies interested in restoring and protecting tidal marsh ecosystems without using invasive plants such as *Phragmites australis* focus on selecting species with similar physiological traits, as current climate model trends in Chesapeake Bay suggest an increase in salt intrusions into estuarine river systems and continual increases in relative sea-level rise (Hayhoe et al. 2007, Pyke et al. 2008).

As tidal marshes face increasing threat from anthropogenic forces, sea-level rise, and invasive plant species, understanding the principal mechanisms affecting species richness has become increasingly important. Resource managers intent on maintaining tidal marsh plant species diversity with the goal of providing ecosystem services such as high habitat diversity for wildlife should focus their efforts on low-salinity oligohaline marshes as well as on tidal freshwater systems. Invasive species such as *Phragmites australis*, though viewed by many in the natural resource community as undesirable, may be able to offer insights regarding plant selection and management of restored tidal marsh ecosystems. Our hope is that this research can be utilized to predict tidal

marsh community changes over time and develop additional controlled experiments examining plant community responses to altered physical and biotic conditions, such as those caused by global climate changes.

Literature Cited

Anderson, R. R., R. G. Brown, and R. D. Rappleye. 1968. Water quality and plant distribution along the upper Patuxent River, Maryland. *Chesapeake Science* 9: 145-156.

Baldwin, A. H., K. L. McKee, and I. A. Mendelssohn. 1996. The influence of vegetation, salinity, and inundation on seed banks of oligohaline coastal marshes. *American Journal of Botany* 83: 470-479.

Baldwin, A. H., and I. A. Mendelssohn. 1998. Effects of salinity and water level on coastal marshes: An experimental test of disturbance as a catalyst for vegetation change. *Aquatic Botany* 1250: 1-14.

Baldwin, A. H., and E. F. DeRico. 1999. The seed bank of a restored tidal freshwater marsh in Washington, DC. *Urban Ecosystems* 3: 5-20.

Baldwin, A. H., M. S. Egnotovich, and E. Clarke. 2001. Hydrologic change and vegetation of tidal freshwater marshes: Field, greenhouse, and seed-bank experiments. *Wetlands* 21: 519-531.

Baldwin, A. H., and F. N. Pendleton. 2003. Interactive effects of animal disturbance and elevation on vegetation of a tidal freshwater marsh. *Estuaries* 26: 905-915.

Baldwin, A. H. 2004. Restoring complex vegetation in urban settings: the case of tidal freshwater marshes. *Urban Ecosystems* 7: 137.

Baumann, R. H., J. W. Day, Jr., and C. A. Miller. 1984. Mississippi deltaic wetland survival: Sedimentation versus coastal submergence. *Science* 224: 1095.

Berggren, T. J., and J. T. Lieberman. 1977. Relative contributions of Hudson, Chesapeake, and Roanoke striped bass, *Morone saxatilis*, to the Atlantic coast fishery. *Fisheries Bulletin* 76: 335-345.

Boesch, D. F., M. N. Josselyn, A. J. Mehta, J. T. Morris, W. K. Nuttle, C. A. Simenstad, and D. J. P. Swift. 1994. Scientific assessment of coastal wetland loss, restoration and management in Louisiana. *Journal of Coastal Research* SI20: 1-103.

Broome, S. W., I. A. Mendelssohn, and K. L. McKee. 1995. Relative growth of *Spartina patens* (Ait.) Muhl. and *Scirpus olneyi* Gray occurring in a mixed stand as affected by salinity and flooding depth. *Wetlands* 15: 20-30.

Cowardin L.M., Carter V., Golet F.C. & LaRoe E.T. Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31. 1979. Washington, D.C., U.S. Fish and Wildlife Service. Ref Type: Report

Crain, C. M., B. R. Silliman, S. L. Bertness, and M. D. Bertness. 2004. Physical and biotic drivers of plant distribution across estuarine salinity gradients. *Ecology* 85: 2539-2549.

Darke, A. K., and J. P. Megonigal. 2003. Control of sediment deposition rates in two mid-Atlantic Coast tidal freshwater wetlands. *Estuarine Coastal and Shelf Science* 57: 255-268.

Day, J. W., J. Rybczyk, F. Scarton, A. Rismondo, D. Are, and G. Cecconi. 1999. Soil accretionary dynamics, sea-level rise and the survival of wetlands in Venice Lagoon: A field and modeling approach. *Estuarine Coastal and Shelf Science* 49: 607-628.

Donnelly, J. P., and M. D. Bertness. 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. *Proceeding of the National Academy of Sciences* 98: 14218-14223.

Faulkner, S. P., W. H. Patrick, Jr., and R. P. Gambrell. 1989. Field techniques for measuring wetland soil parameters. *Soil Science Society of America Journal* 53: 883-890.

Flynn, K. M., K. L. McKee, and I. A. Mendelssohn. 1995. Recovery of freshwater marsh vegetation after a saltwater intrusion event. *Oecologia* 103: 63-72.

Ford, M. A., D. R. Cahoon, and J. C. Lynch. 1999. Restoring marsh elevation in a rapidly-subsiding salt marsh by thin-layer deposition of dredge material. *Ecological Engineering* 12: 189-205.

Gough, L., and J. B. Grace. 1998. Effects of flooding, salinity and herbivory on coastal plant communities, Louisiana, United States. *Oecologia* 117: 527-535.

Greenberg, R., J. Maldonado, and M. V. McDonald. 2006. Tidal marshes: a global perspective on the evolution and conservation of their terrestrial vertebrates. Bioscience 56:675–685.

Hayhoe, K., C. P. Wake, T. G. Huntington, L. F. Luo, M. D. Schwartz, J. Sheffield, E. Wood, B. Anderson, J. Bradbury, A. DeGaetano, T. J. Troy, and D. Wolfe. 2007. Past and future changes in climate and hydrological indicators in the U.S. Northeast, *Clim. Dyn.*, 28: 381–407.

Hotes, S., E. B. Adema, A. P. Grootjans, T. Inoue, and P. Poschlod. 2005. Reed die-back related to increased sulfide concentration in a coastal mire in eastern Hokkaido, Japan. *Wetlands Ecology and Management* 13: 83-91.

IPCC. 2007. Climate Change 2007 Synthesis Report: Contribution of working groups I, II, and III to the fourth assessment report of the Intergovernmental Panel on Climate Change. Pachauri, R.K. and Reisinger, A. (Eds.), IPCC, Geneva, Switzerland. pp 104.

Kearney, M. S., R. E. Grace, and J. C. Stevenson. 1988. Marsh loss in Nanticoke Estuary, Chesapeake Bay. *Geographical Review* 78: 206-220.

Kearney, M. S., J. C. Stevenson, and L. G. Ward. 1994. Spatial and temporal changes in marsh vertical accretion rates at Monie Bay: implications for sea-level rise. *Journal of Coastal Research* 10: 1010-1020.

Kozlowski, T. T. 1997. Responses of woody plants to flooding and salinity. *Tree Physiology* Monograph No. 1: 1-29.

Lyles S.D., Hickman L.E. & Debaugh H.A. Sea level variations for the United States 1855-1986. 1988. Rockville, Maryland, National Oceanic and Atmospheric Administration, National Ocean Service. Ref Type: Report

McCune, B. and J. B. Grace. 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, OR, USA.

McKee, K. L., and I. A. Mendelssohn. 1989. Response of a freshwater marsh plant community to increased salinity and increased water level. *Aquatic Botany* 34: 301-316.

Meade, R. H. 1972. Transport and deposition of sediments in estuaries. *Geological Society of America Memoir* 133: 91-120.

Mendelssohn, I. A., K. L. McKee, and W. H. Patrick. 1981. Oxygen deficiency in *Spartina alterniflora* roots: metabolic adaptation to anoxia. *Science* 214: 439-441.

Mendelssohn, I. A., and D. P. Batzer. 2006. Abiotic constraints for wetland plants and animals. Pages 82-114 *in* D. P. Batzer, and R. R. Sharitz editors. Ecology of freshwater and estuarine wetlands. University of California Press, Berkeley and Los Angeles, California.

Mitsch W. J., and J. G. Gosselink. 2000. Wetlands, Third edition., Third edition. John Wiley and Sons, New York.

Morris, J. T., P. V. Sundareshwar, C. T. Nietch, B. Kjerfve, and D. R. Cahoon. 2002. Responses of coastal wetlands to rising sea level. *Ecology* 83: 2869-2877.

Naidoo, G., K. L. McKee, and I. A. Mendelssohn. 1992. Anatomical and metabolic responses to waterlogging and salinity in *Spartina alterniflora* and *S. patens* (Poaceae). *American Journal of Botany* 79: 765-770.

Neff, K. P., and A. H. Baldwin. 2004. Seed dispersal into wetlands: techniques and results for a restored tidal freshwater marsh. *Wetlands* 25: 392-404.

Odum W.E., Smith III T.J., Hoover J.K. & McIvor C.C. The ecology of tidal freshwater marshes of the United States east coast: A community profile. FWS/OBS-83/17, 1-177. 1984. Washington, D.C., U.S. Fish and Wildlife Service. Ref Type: Report

Odum, W. E. 1988. Comparative ecology of tidal freshwater and salt marshes. *Annual Review of Ecology and Systematics* 19: 147-176.

Officer, C. B. 1981. Physical dynamics of estuarine suspended sediments. *Marine Geology* 40: 1-14.

Peet, R. K. 1974. The measurement of species diversity. *Annual Review of Ecology and Systematics* 5: 285-307.

Peet, R. K., T. R. Wentworth, and P. S. White. 1998. A flexible, multipurpose method for recording vegetation composition and structure. *Castanea* 63: 262-274.

Perry, J. E., and C. H. Hershner. 1999. Temporal changes in the vegetation pattern in a tidal freshwater marsh. *Wetlands* 19: 90-99.

Peterson, J. E., and A. H. Baldwin. 2004. Seedling emergence from seed banks of tidal freshwater wetlands: Response to inundation and sedimentation. *Aquatic Botany* 78: 243-254.

Peterson, J. E., and A. H. Baldwin. 2004. Variation in wetland seed banks across a tidal freshwater landscape. *American Journal of Botany* 91: 1251-1259.

Pezeshki, S. R., and R. D. DeLaune. 1993. Effects of soil hypoxia and salinity on gas exchange and growth of *Spartina patens*. *Marine Ecology-Progress Series* 96: 75-81.

Pyke, C. R., R. G. Najjar, M. B. Adams, D. Breitburg, M. Kemp, C. Hershner, R. Howarth, M. Mulholland, M. Paolisso, D. Secor, K. Sellner, D. Wardrop, and R. Wood. 2008. Climate Change and the Chesapeake Bay: State-of-the-Science Review and Recommendations. A Report from the Chesapeake Bay Program Science and Technical Advisory Committee (STAC), Annapolis, MD. 59 pp.

Reed, D. J. 1995. The Response of Coastal Marshes to Sea-Level Rise - Survival Or Submergence. *Earth Surface Processes and Landforms* 20: 39-48.

Roman, C. T., J. A. Peck, J. R. Allen, J. W. King, and P. G. Appleby. 1997. Accretion of a New England (USA) salt marsh in response to inlet migration, storms, and sea-level rise. *Estuarine Coastal and Shelf Science* 45: 717-727.

Sharpe, P.J. and A.H. Baldwin. 2009. Patterns of wetland plant species richness across estuarine gradients of chesapeake bay. *Wetlands* 29(1): 225-235.

Stevenson, J. C., M. S. Kearney, and E. C. Pendleton. 1985. Sedimentation and erosion in a Chesapeake Bay brackish marsh system. *Marine Geology* 67: 213-235.

Stevenson, J. C., L. G. Ward, and M. S. Kearney. 1986. Vertical accretion rates in marshes with varying rates of sea-level rise. Pages 241-259 *in* D. A. Wolfe editor. Estuarine variability. Academic Press, New York, New York.

Stevenson, J. C., and M. S. Kearney. 1996. Shoreline dynamics on the windward and leeward shores of a large temperate estuary. Pages 233-259 *in* K.F.Nordstrom, and C.T.Roman editors. Estuarine Shores: Evolution, Environments and Human Alterations. John Wiley and Sons, Ltd..

Taylor, J. A., A. P. Murdock, and N. I. Pontee. 2004. A macroscale analysis of coastal steepening around the coast of England and Wales. *Geographical Journal* 170: 179-188.

Tiner R. W. 1993. Field guide to coastal wetland plants of the southeastern United States. The University of Massachusetts Press, Amherst, Massachusetts.

Tiner R.W. & Burke D.G. Wetlands of Maryland. 1-193. 1995. U.S. Fish and Wildlife Service, Ecological Services, Region 5, Hadley, Massachusetts and Maryland Department of Natural Resources, Annapolis, Maryland.

Turner, R. E., E. M. Swenson, C. S. Milan, J. M. Lee, and T. A. Oswald. 2004. Below-ground biomass in healthy and impaired salt marshes. *Ecological Research* 19: 29-35.

USDA-CSREES. 2000. Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters. Southern Cooperative Series. Bulletin No. 396. G.M Pierzynski editor.

Valery, L., V. Bouchard, and J. Lefeuvre. 2004. Impact of the invasive native species *Elymus athericus* on carbon pools in a salt marsh. *Wetlands* 24: 268-276.

Webb, E. C., I. A. Mendelssohn, and B. J. Wilsey. 1995. Causes for vegetation dieback in a Louisiana salt marsh: a bioassay approach. *Aquatic Botany* 51: 281-289.

Webb, E. C., and I. A. Mendelssohn. 1996. Factors Affecting Vegetation Dieback of an Oligohaline Marsh in Coastal Louisiana: Field Manipulation of Salinity and Submergence. *American Journal of Botany* 83: 1429-1434.

APPENDIX A
EXPERIMENT PHOTOGRAPHS

Photograph showing a 19 liter bucket filled with collected marsh topsoil

Marsh mesocosms in May 2007 prior to experimental treatments

Randomly planting supplemental plant species not recruited from the seedbank May 31, 2007

Marsh mesocosms in July 2007 just before the start of treatments

Timing box and relay switchboard for controlling the flow of water into and out of the mesocosm troughs

View of the marsh mesocosms in July 2007.

View of the marsh mesocosms at high tide in May 2008. Water level treatment effect is visible.

Peter Sharpe giving a presentation on the mesocosm experiment to group of graduate students from the University of Hamburg, Germany (August 2008).

Integration of Stormwater Management Ponds into Urban Communities: Long-term Water Quality Protection, Wildlife, and Environmental Awareness

Basic Information

Title:	Integration of Stormwater Management Ponds into Urban Communities: Long-term Water Quality Protection, Wildlife, and Environmental Awareness
Project Number:	2007MD160G
Start Date:	2/1/2008
End Date:	1/31/2010
Funding Source:	104G
Congressional District:	3
Research Category:	Water Quality
Focus Category:	Water Quality, Hydrogeochemistry, Ecology
Descriptors:	
Principal Investigators:	Joel Wade Snodgrass, Ryan E. Casey, Edward R Landa, Steven Lev

Publications

1. Brand, A. B., and J. W. Snodgrass. 2010. Value of Artificial Habitats for Amphibian Reproduction in Altered Landscapes. Conservation Biology 295-301.
2. Brand, A. B., J. W. Snodgrass, M. T. Gallagher, R. E. Casey, and R. Van Meter. 2010. Lethal and sublethal effects of embryonic and larval exposure of Hyla versicolor to stormwater pond sediments. Archives of Environmental Contamination and Toxicology 58:325–331.
3. Gallagher, M. T., J. W. Snodgrass, D. R. Ownby, A. B. Brand, R. E. Casey, S. Lev. In review. Watershed-scale analysis of pollutant distributions in stormwater management ponds. Urban Ecosystems.
4. Rodgers, D., S. Lev, J. W. Snodgrass, D. Ownby, L. Prince, and R. Casey. In review. Development of an enriched stable isotope tracer technique to estimate the soil Zn pool available to Lumbricus terrestris (L.) across a salinization gradient. Environmental Toxicology and Chemistry.

Report of Activities Under Subaward Agreement Z592801: Integration of Stormwater Management Ponds into Urban Communities

PIs: Joel W. Snodgrass, Steve M. Lev, Ryan E. Casey and Ed R. Landa

Stormwater management ponds are common features of more recent development and are required by most state and local governments as part of more comprehensive stormwater management practices. While stormwater ponds are human created habitats, they may superficially resemble natural wetlands and attract wildlife. Moreover, while short-term (individual storm event) studies indicate storm water ponds are affective at removing pollutants, the effectiveness of ponds over longer time scales (years) and the interaction of these ponds with human populations have received little or no attention. Our work seeks to evaluate pollutant movement between ponds and streams through groundwater transport, the role of ponds as wildlife habitat for amphibians, and social perception and understanding of ponds. Below we outline our progress under four specific goals.

Goal 2: Determine to what degree metals (primarily Cr, Cu, Ni, Pb, and Zn) and Cl^- are transported via ground water from ponds where they accumulate to natural surface waters

To quantify contaminant loading to stormwater ponds and flux to surface waters in the Red Run watershed via groundwater transport, we set up a dense monitoring network at two ponds. We installed 50 drive point piezometers within the ponds and on the floodplain between the ponds and a second-order tributary to Red Run. We place water level and conductivity loggers at inputs to the ponds and up and down stream of the floodplain input within the second-order tributary. We measured stream discharge and collected water samples from the wells and stream on a regular basis.

At the study site, discharge in the second order tributary downstream of input from the stormwater ponds is generally 2.5 times higher and chloride levels are 5 times higher than an upstream reference site. Surface water measurements immediately downstream of storm water derived input record elevated conductivities year round in the stream, peaking at approximately 2.5 mS/cm. A chloride enriched groundwater plume moving down gradient from the retention ponds has also been identified. Groundwater conductivities remained elevated throughout the year, peaking at > 20 mS/cm in ground water immediately under the ponds in late winter. Under the floodplain, ground water conductivties also remain elevated year round with a high of 5.85 mS/cm occurring during the winter months. These finding clearly indicate that road salts entering retention ponds are being transferred to ground waters where they are stored and, ultimately discharged to streams.

Goal 2: Determine if there are interactions between the types of pollutants that accumulate in stormwater ponds that might facilitate or otherwise influence ground water transport of pollutants.

To assess the effects of road salt contamination of soils on bioavailability of Zn we conducted a series of experiments with a common earth worm, *Lumbricus terrestris*. In the first experiment *L. terrestris* was exposed to OECD artificial soil amended with Zn and NaCl or $CaCl_2$. After salt application OECD soil exhibited the intended treatment effect, with Na^+ and Ca^{2+} accounting for 74 and 96% of soil cation exchange sites, respectively. Deionized water phase Zn also varied between treatments, averaging 3.4 times higher in the Ca^{2+} treatment. Despite this difference in available Zn, earthworms did not accumulate Zn or other trace metals in either treatment over the course of a 22-day exposure. We observed complete mortality in the Na^+ treatments at day 22 (8 worms), and consequently considered that a relationship between the biologically relevant ions Na^+ and K^+ may have caused stress.

In our second experiment we chose to further explore the importance of Na^+:K^+ in earthworms by treating a field derived soil with a suite of five concentrations, which allowed us to achieve Na^+:K^+ ratios in the soil ranging from 3.5 to 190; values both greater and less than those observed in local

stormwater pond soils. Increasing amounts of Na^+ in the soil led to marked changes in soil cation composition, with all major cations except Na^+ showing decreases over time. Earthworm biology was also affected, with average percent weight losses of 5.7, 12, 17, 17 and 43 for the five treatments. While Na^+:K^+ ratio did seem to be significantly higher in salt treated soil than the control, we did not observe a dose-dependent effect. Our results suggest that the road salts may be affecting soil communities by limiting the availability of major cations.

Goal 3: Determine the range of pollutants and hydrological conditions exhibited by typical ponds and the degree to which they degrade habitat for developing embryonic and larval amphibians.

To address the potential for pollutant exposure for wildlife, we randomly selected 68 stormwater ponds in the Red Run watershed of Baltimore County, Maryland. We sampled sediment and water in the 68 ponds to estimated the proportion of ponds in a third-order watershed that exceed toxicity guidelines for trace metals and polycyclic aromatic hydrocarbons in sediments and chloride in surface waters. Ninety-six percent of ponds exceeded consensus-based threshold effect concentrations for at least one trace metal. Nine percent of ponds exceeded chronic toxicity levels of chloride on all sampling dates, and 21% exceeded acute toxicity concentrations on at least one sampling date.

We also surveyed hydrology and Wood Frog (*Rana sylvatica*) and American toad (*Bufo americanus*) use of the 68 randomly selected ponds. Wood Frog use of ponds was associated with both hydrology and concentrations of Cl^- in pond waters. Wood Frogs only bred in ponds with relatively long hydroperiods (drying only in mid to late summer) and Cl^- levels less than ~250 mg/L. A set of laboratory bioassays involving exposure of embryos and larval Wood Frogs to sediments from six ponds confirmed that road salt contamination was at least partially responsible for limiting Wood Frog use of ponds. Pond treatments with chloride concentrations in water above approximately 260 mg/L saw reduced or no larval survival. In contrast, the occurrence of American toads was best predicted by hydroperiod, but was not related to pollutant levels in water or sediment. In bioassay using sediment from the same ponds used with wood frogs, toad survival was not affected.

Overall, our results suggest that the use of ponds by wood frogs is limited by road salt contamination, but less sensitive species such as American toads are not affected to the same extent. Future studies should address the potential interaction among road salts and other contaminants commonly found in stormwater management ponds.

Goal 4: Examine breeding habitat choice in natural and recently urbanized landscapes to determine if amphibians select or avoid stormwater ponds as breeding sites.

To investigate the potential impacts of stormwater ponds on amphibian populations we intensively surveyed three second-order watersheds of the larger Red Run watershed and three second-order watersheds that were predominately forested (Brand and Snodgrass, 2010). In suburban watersheds, most (89%) of the wetlands that had breeding activity were either stormwater ponds or were otherwise artificial. This pattern was also evident in the forested watersheds, where amphibians were primarily found breeding in wetlands created by past human activity. Late-stage larvae were found only in anthropogenic wetlands in all study areas because the remaining natural wetlands did not hold water long enough for larvae to complete development. Our results suggest that in urban and suburban landscapes with naturally low densities of wetlands, wetlands created by current or historic land uses may be as important to amphibian conservation as natural wetlands or pools.

Number and degree of students supported:

Six undergraduate students
Four Master's students

Papers

Brand, A. B., and J. W. Snodgrass. 2010. Value of Artificial Habitats for Amphibian Reproduction in Altered Landscapes. *Conservation Biology* 295-301.

Brand, A. B., J. W. Snodgrass, M. T. Gallagher, R. E. Casey, and R. Van Meter. 2010. Lethal and sublethal effects of embryonic and larval exposure of *Hyla versicolor* to stormwater pond sediments. *Archives of Environmental Contamination and Toxicology* 58:325–331.

Gallagher, M. T., J. W. Snodgrass, D. R. Ownby, A. B. Brand, R. E. Casey, S. Lev. *In review*. Watershed-scale analysis of pollutant distributions in stormwater management ponds. *Urban Ecosystems*.

Rodgers, D., S. Lev, J. W. Snodgrass, D. Ownby, L. Prince, and R. Casey. *In review*. Development of an enriched stable isotope tracer technique to estimate the soil Zn pool available to Lumbricus terrestris (L.) across a salinization gradient. *Environmental Toxicology and Chemistry*.

Any notable awards and achievements

Brand and Snodgrass (2010) received coverage on the front page of Conservation Maven, an online hub for the conservation community (http://www.conservationmaven.com/frontpage/2009/8/17/do-created-wetland).

Nearshore sediment inputs due to shore erosion in the Maryland Chesapeake Bay

Basic Information

Title:	Nearshore sediment inputs due to shore erosion in the Maryland Chesapeake Bay
Project Number:	2009MD192B
Start Date:	3/1/2009
End Date:	2/28/2010
Funding Source:	104B
Congressional District:	Maryland First
Research Category:	Climate and Hydrologic Processes
Focus Category:	Sediments, Water Quality, Geomorphological Processes
Descriptors:	None
Principal Investigators:	Lawrence Paul Sanford, Lawrence Paul Sanford

Publication

1. Bell, L., and L. Sanford, “Dynamic Equilibrium Beach Profiles: Forces of Erosion and Offshore Sediment Transport” ; ‘Response of Shallow Photic Systems to Multiple Stressors’ Session, Atlantic Estuarine Research Society, Spring 2010

Nearshore sediment inputs due to shore erosion in the Maryland Chesapeake Bay

MWRCC Project #

Dr. Lawrence Sanford
University of Maryland Center for Environmental Science
Horn Point Laboratory, Cambridge, Maryland 21613

Final Report

Published Abstracts

Bell, L., and L. Sanford, "*Dynamic Equilibrium Beach Profiles: Forces of Erosion and Offshore Sediment Transport"* ; 'Response of Shallow Photic Systems to Multiple Stressors' Session, Atlantic Estuarine Research Society, Spring 2010

Student Support

One graduate student (Lynda Bell; MS student enrolled in the Marine, Estuarine, and Environmental Science Program) was supported by this grant from 2009-2010 to work on in depth analysis of the 2008 Maryland Chesapeake Bay shoreline erosion data acquired by the Maryland Geological Survey (MGS) and the University of Maryland Center for Environmental Science (UMCES) under NOAA Grant 14-08-1218 CZM 237. This work and analysis are the foundation of Ms. Bell's Master's Thesis.

Statement of Critical Regional or State Water Problem

Estuaries and coastal embayments of the Mid-Atlantic region have been significantly impacted by erosion, loss of submerged aquatic vegetation (SAV), loss of marsh, increasing shoreline hardening, nutrient enrichment, declines of key species from overharvesting and disease, and hypoxia and sustained algal blooms (e.g. Hagy et al. 2004, Gilbert and Magnien 2004, Kemp et al. 2005). In addition, climate change and climate variability interact with these stressors to alter temperature, freshwater flow, sea level, and ultimately population dynamics of both benthic and pelagic species (Kimmel and Roman, 2004, Kimmel et al. 2006).

Sediment input is greatest in nearshore waters due to shore erosion in the northern (Maryland) reaches of the Chesapeake Bay. The effects of erosion on nearshore water quality and habitat are complex, as are the effects of different types of shore protection structures. In general, erosion leads to increased nutrient loads, all of which degrade nearshore water quality. Furthermore, erosion is a source of sediment deposition into navigable channels that then require dredging (Marcus and Kearney 1991; Hobbs et al. 1992). However, natural eroding shorelines also can provide ecosystem services such as beach habitat and a source of sediment for SAV beds and marshes, which in turn improve water quality and help protect shorelines from further erosion.

Thus, management of erosion is a significant challenge, particularly in estuarine systems (STAC, 2003).

For example, shoreline hardening has been suggested to benefit SAV in the Chesapeake Bay by improving water quality via reduced nutrient and suspended particle input. Yet, not only has this benefit not yet been proven, but new findings (Koch et al. 2007) suggest that shoreline hardening may actually be detrimental to SAV in some cases. Studies (Stevenson et al 2002, Wicks 2005) have shown that the type of shoreline being eroded needs to be considered when determining the benefit of hardening shorelines. As marsh shoreline characterized by fine, organic particles is eroded, shoreline retreat leaves behind highly compacted peat in the sub-tidal. This sediment is unsuitable for SAV colonization leading to unvegetated adjacent areas. In this case, shoreline hardening may be beneficial to SAV. In contrast, if a sandy deposit is being eroded it leads to the deposition of sand in the nearshore and allows SAV to follow the landward migration as the shoreline regresses. In this case, shoreline hardening would be disastrous to the local SAV community (E. Koch, personnel communication). *Zostera marina* (eelgrass) requires substrates with more than 65% sand for its survival (Koch et al. 2007). By cutting natural sand supplies via shoreline hardening, this vital source of sand is removed and over time sediments become too fine to support eelgrass growth.

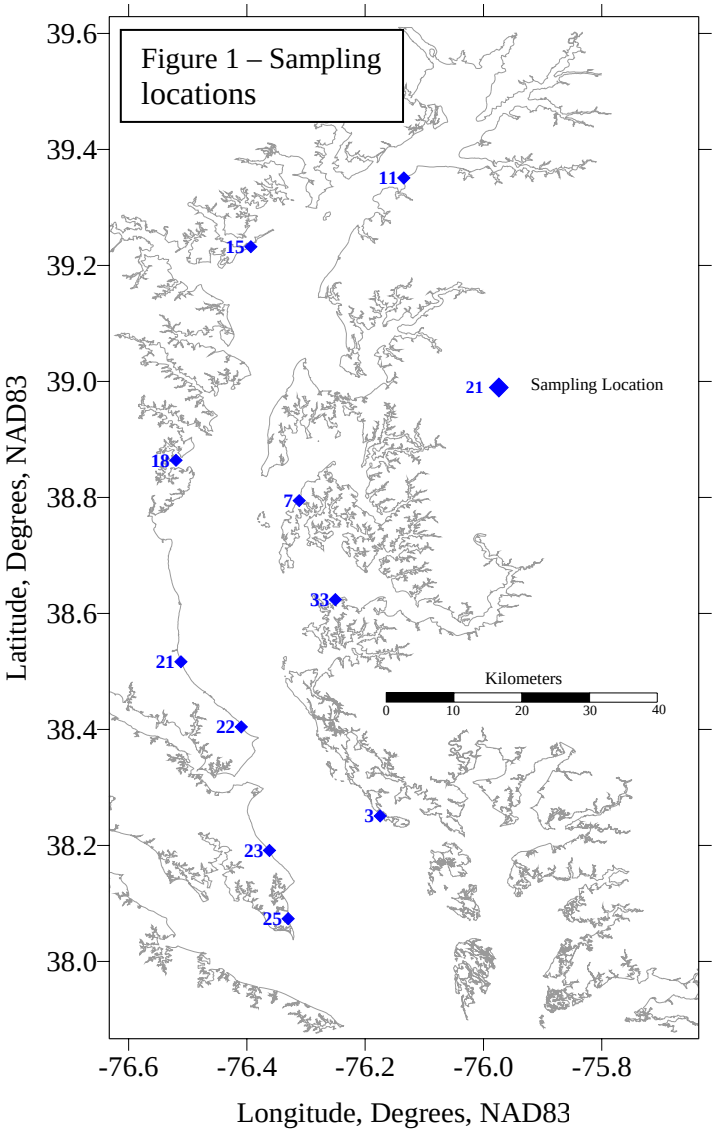
As the human population increases, so too does the economic pressure to protect eroding shorelines. Currently 69% of Maryland's shoreline is eroding (Hennessee et al., 2003), and erosion rates in the southern reaches of Chesapeake Bay are slightly higher than northern areas (0.4m/yr vs. 0.3m/yr; Hardaway and Anderson, 1980). Recent surveys indicate that overall 12% of the state's shoreline is hardened, and in developed urban and suburban areas such as Baltimore and Anne Arundel Counties the hardening exceeds 40%. Even in more rural areas such as Talbot County hardening now exceeds 30% (VIMS, Comprehensive Coastal Inventory Program: http://ccrm.vims.edu/gisdatabases). Managers charged with advising landowners on shoreline protection measures and issuing permits obviously must take into account the economic, aesthetic, and cultural value of the eroding land, but they also should consider the ecological impact of various protection measures. Indeed, recent legislation in Maryland specifically promotes installation of so-called "living shorelines" where appropriate; these are a hybrid protection measure in which offshore breakwaters contain emplaced sandy sediments that are planted with marsh plants above Mean Low Water (MLW) and SAV below MLW.

Several important types of information are required to evaluate and manage shore erosion appropriately. Among the most important is an understanding of historical rates of erosion and how they relate to environmental factors such as wave attack, tidal height/flooding, bank composition and height, nearshore sediment composition, the nearshore depth profile, and rates of sea level rise. Given a reasonable understanding of these factors, it should be possible to estimate the response of unprotected shorelines to changes in sea level and weather, due to climate change. Several studies have been carried out by different agencies working in the Chesapeake Bay region to document past and present rates of shore erosion (USACE 1990; Halka 2005; STAC 2006). However, for the most part estimates of future erosion rates simply extrapolate the most recent measured erosion rates forward in time, due to lack of an adequate understanding of the processes involved. Another key factor for managers is a reliable estimate of the rates and composition of sediment input into nearshore waters associated with shore erosion, for reasons explained above. Clearly, bank height, bank composition, and shore erosion rate are important. It

has also been recognized in recent years that downcutting of the nearshore subaqueous bottom can be a significant component of the overall sediment inputs (STAC 2006). Nearshore bottom sediments may account for a third to a half of the total sediment input due to shore erosion. There are very few detailed data sets available on nearshore bathymetry and bottom sediment composition in Chesapeake Bay, however, such that recent estimates of sediment input due to shore erosion (e.g., Hennessee et al. 2003; Dr. Carl Cerco, PI for USEPA Chesapeake Bay Water Quality Model development, personal communication) have been based on application of an assumed split between bank and nearshore contributions applied uniformly to all locations. The assumed split is based on the characteristics of the few sites where data are available, but even in these cases data on the bottom sediment composition is generally not available.

Project Objectives

The principle objective of this project was to begin an in-depth analysis of a data set on offshore bathymetry and bottom sediment characteristics collected during the summer of 2008. Shore normal offshore transects were measured at ten sites in the Maryland Chesapeake Bay (Figure 1) by the Maryland Geological Survey (MGS) and the University of Maryland Center for Environmental Science (UMCES), supported by NOAA Grant 14-08-1218 CZM 237. The surveyed area included bay shorelines from Kent County, north of the Chesapeake Bay Bridge (very near the Estuarine Turbidity Maximum), as well as shorelines on the eastern shore (from Tilghmans Island to Hoopers Island), and the western shore of Maryland's Chesapeake Bay (St. Mary's and Calvert County). Data collected at each site included bathymetric surveys along shore-normal transects, differential leveling at the shoreline, and collection of sediment cores and grab samples along a surveyed offshore profile. The parent project was completed, a data report was prepared and delivered to the Maryland Chesapeake and Coastal Program, and the information was added to Maryland Shorelines Online (http://shorelines.dnr.state.md.us/).



Task I: Combine shore erosion rate estimates, bank height and composition data, bottom sediment composition data, and depth profile information to obtain better estimates of the inputs of fine and coarse sediments that accompany shore erosion at each of the sites.

Task II: Determine how well the offshore depth profiles fit the classic Bruun equilibrium profile (described in Methods below) by completing the following:

a. Explore potential patterns in the Bruun profile fits, and in deviations from those fits, that help explain the data.

b. Explore how well Bruun's law for the shoreline retreat accompanying a given sea level rise or a modified version of it, fits the historical data at each of these sites.

Task III: Determine feasibility of using these techniques for estimating future estimates of sediment inputs from shoreline erosion and see if these results can increase the predictive capabilities at other sites around Chesapeake Bay.

These explorations may provide improved techniques for extrapolating shore erosion rates into the future and estimating the amounts and impacts of associated sediment inputs, and therefore changes in water quality and nearshore habitat quality. At the very least, we will gain a better understanding and estimation of sediment inputs at the 10 sites investigated. Note that these sites were purposely chosen to be as free of the influence of shoreline protection measures as possible in the modern Bay, and to avoid convergences or divergences in longshore littoral transport that might invalidate the essentially 2-D approach proposed.

Background – the 2008 data set

In order to explore and develop various models of dynamic beach profiles using the 2008 data set, a classic beach profile equation was tested, known as the Bruun equation or the 2/3 rule. Bruun inferred an equilibrium profile on sandy shores that would result in a profile shape that could be explained by the following equation:

$$h_{eq} = A'y^{2/3}$$

where h_{eq} is the equilibrium depth, y is the distance offshore, and A' is a constant that is related to grain size. The parameter A' in the Bruun equation has been shown to be related to sediment grain size as a consequence of the ability of the incoming wave energy to erode and move particles of different sizes (Dean et al., 2002). Bruun's Model states, "The profile at any location is presumed to represent a dynamic equilibrium", [Bruun 1954].

In order to define the parameter A' in this equation, grain size analysis was used to test the relationship between the equilibrium constant (A') and grain size. Initial calculations of A' were made using equilibrium depth, and the distance offshore (Figure 2). Further examination of the relationship between A" and grain size was made using the core data itself. Grain size analysis was completed on samples taken of the top of each core and analyzed for percent sand, silt, and clay. (Figure 3). By looking at variation in beach types/heights as well as amount of sand vs. fines at each site, we were able to examine variations in the profile

Figure 2: Calculated A' values for the 2008 Bruun Profile Survey
(Fine Sands: A' = 0.063 to 0.15; Finer than Sand: A' < 0.05)

Site Number (Figure 2)	Site Name	Calculated A'	R^2
22	Calvert Cliffs State Park	0.04319 *	0.8138
18	Cheston Point, Rhode River	0.06972	0.9316
23	Elms Beach	0.06964	0.9105
7	Long Point	0.04519 *	0.8356
11	Meeks Point	0.15098	0.9116
15	Pleasure Island	0.03618 *	0.8663
3	Richland Point	0.01754 *	0.6781
21	Scientist's Cliffs	0.07917	0.9320
25	Scotland Beach	0.05440	0.9124
33	Todds Point	0.03618 *	0.8825

Figure 3: Grain size analysis Results

	RESULTS						
Sample ID	%H20	Bulk Densit	%GRAVEL	%SAND	%SILT	%CLAY	SHEPCLASS
PleasureIsl	5.285996	2.49331	0.533049	99.36034	0.10661	0	Sand
PleasureIsl	9.055118	2.353454	23.63549	76.17216	0.192354	0	Sand
PleasureIsl	19.79167	2.02922	0	93.73149	6.268509	0	Sand
PleasureIsl	15.37255	2.151205	0	50.21946	34.88724	14.89329	Silty-Sand
PleasureIsl	17.27534	2.096928	0	97.64005	2.359952	0	Sand
PleasureIsl	19.71891	2.031116	0	98.01258	1.987421	0	Sand
MeeksPtCc	6.595092	2.442889	7.661376	92.2328	0.10582	0	Sand
MeeksPtCc	7.838115	2.396865	53.87624	45.34139	0.782361	0	Sand
MeeksPtCc	16.17043	2.128107	68.60805	10.34161	8.113994	12.93634	Sand-Silt-Clay
MeeksPtCc	19.71644	2.031181	0	99.7416	0.258398	0	Sand
MeeksPtCc	15.41067	2.15009	0	90.81935	4.667889	4.512763	Sand
RichlandPt	19.89529	2.026525	0	13.90836	51.62624	34.46541	Clayey-Silt
RichlandPt	18.89265	2.052902	0	98.47248	1.527522	0	Sand
RichlandPt	21.81091	1.97797	0	96.46707	3.532934	0	Sand
ElmsBeach	7.358612	2.414412	59.02479	40.64741	0.327802	0	Sand
ElmsBeach	18.95032	2.051366	0	98.70558	1.294416	0	Sand
ElmsBeach	17.36057	2.094561	0	98.92231	1.077694	0	Sand
ElmsBeach	14.29079	2.183333	17.30465	79.78488	2.910471	0	Sand
ScientistCli	3.738686	2.555657	45.92655	53.86239	0.21106	0	Sand
ScientistCli	16.61143	2.115552	0	99.08055	0.919448	0	Sand
ScientistCli	20.87399	2.001424	0	93.53933	6.460674	0	Sand
ScientistCli	9.77354	2.328558	0	99.39394	0.606061	0	Sand
ToddsPtCo	12.38162	2.242441	0	96.32337	3.676633	0	Sand
ToddsPtCo	19.69528	2.031733	0	97.85503	2.14497	0	Sand
ToddsPtCo	18.10409	2.074135	0	97.80248	2.197518	0	Sand
ToddsPtCo	16.78832	2.110557	7.573907	90.74029	1.685805	0	Sand
CalvertCliff	8.619975	2.368794	5.673599	92.67631	1.65009	0	Sand
CalvertCliff	22.99438	1.949118	0	79.57065	12.63417	7.795179	Sand
CalvertCliff	17.43869	2.092395	0	99.40901	0.590987	0	Sand
CalvertCliff	17.14633	2.100521	0	99.75828	0.241721	0	Sand
ScotlandBe	6.243603	2.456226	51.83688	47.95219	0.210933	0	Sand
ScotlandBe	9.955202	2.322346	1.028393	98.14442	0.827185	0	Sand
ScotlandBe	17.86848	2.080564	15.85804	82.79425	1.347709	0	Sand
ScotlandBe	24.79339	1.906837	1.427372	88.11923	10.4534	0	Sand
ScotlandBe	27.72727	1.841684	0.552588	15.43562	34.90514	49.10665	Silty-Clay
RhodeRive	21.46451	1.986577	0.258023	57.79713	18.0616	23.88324	Clayey-Sand
RhodeRive	8.897196	2.358998	0	99.57219	0.427807	0	Sand
RhodeRive	19.06028	2.048444	0	98.48973	1.510275	0	Sand
RhodeRive	16.43411	2.120582	0	97.63743	2.362573	0	Sand
LongPtNCc	3.75	2.55519	0.430769	99.30256	0.266667	0	Sand
LongPtNCc	17.78063	2.082971	0	98.60477	1.395231	0	Sand
LongPtNCc	9.765458	2.328835	0	99.18839	0.811608	0	Sand
LongPtNCc	12.21926	2.247616	0	99.19335	0.806649	0	Sand

Methods and Preliminary Results

Methods and procedures will be illustrated in this section using one of the sites as an example. Todds Pt., on the southern shore of the lower Choptank River, is indicated as site 33 in Figure 1. Todds Pt. was the site of a cooperative study by MGS and UMCES from October 2002 through November 2003 (Halka and Sanford 2008). Long term shore erosion rates average about 1.8 m y^{-1}, but were as high as 2.5 m y^{-1} during the study period. Bank heights at the study site are approximately 1 m, and bank composition is a poorly sorted mix of about 30% sand, 45% silt, and 25% clay. The bottom sediments offshore were not sampled during the original Todds Pt. study, though they were sampled during the summer of 2008 when the site was revisited. They were identified visually in the earlier study as a sticky, partially consolidated estuarine sediment of the Kent Island Formation, similar to the soils at the base of the bank, overlain by a thin lens of sand that occasionally disappeared entirely to expose the underlying cohesive matrix.

Figure 4 – Eroding Todds Pt. bank

Comparison of 1847 and 1942 shorelines and bathymetries allowed a direct estimate of the contribution of nearshore bottom sediment erosion, as well as bank erosion. At this site, the nearshore bottom sediments contribute 52% of sediment inputs and bank erosion 48% of sediment input. Using the composition of the bank sediments to represent both the nearshore and bank contributions, Halka and Sanford (2008) estimated that the average sediment input per day over the 283 m unprotected reach was 5.21 metric tons, with 1.25 metric tons of sand, 2.61 metric tons of silt, and 1.35 metric tons as clays. The fate of these sediments was not possible to determine with certainty, however. Separate measurements of suspended solids and physical forcing indicated that erosion occurred mostly during large, wave-forced events when the tide level was at low to mid-bank. Very high concentrations of suspended silts and clays appeared to dissipate rapidly after the end of an event, most probably through offshore transport. The fate of the sands was not clear; they may have been transported partly offshore or downdrift, and/or they may have remained behind to maintain the thin surface veneer of sand in the nearshore. It is notable that observations offshore a hardened portion of the Todds Pt. site did not reveal evidence of the nearshore sand veneer, presumably because the shoreline source was cut off. It is also notable that the bottom at the base of the hardening structure appeared to deepen at the same rate as on the unprotected portion of the site, so that hardening eliminated the bank source, but did not completely eliminate the nearshore bottom sediment source.

Figure 5 shows some of the data available from the summer 2008 observations. The offshore bathymetry and the Bruun profile fits in the upper 2 panels are virtually identical to their counterparts obtained in the earlier study, which supports our (implicit) assumption of an inwardly translating, constant geometry depth profile accompanying shore erosion.

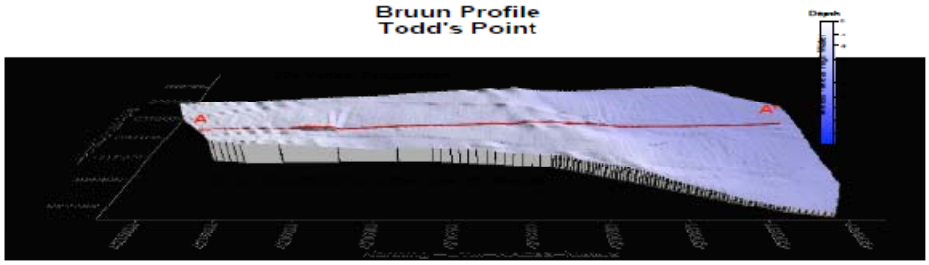


Figure 1. Three dimensional image of the local bathymetry collected at the sample site. The analyzed transect is highlighted in red with the most shoreward point labelled as A and the most offshore point labelled as A'. The bathymetry has been oriented for the best view of the transect.

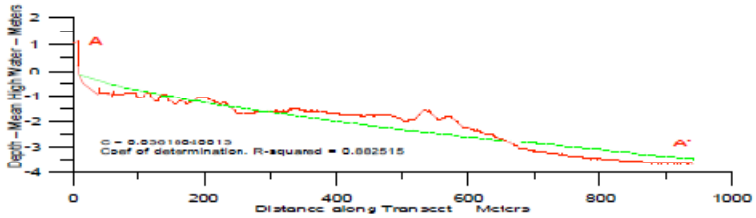


Figure 2. Bathymetric cross-section showing collected bathymetry in red and the fitted Bruun profile curve in green. The calculated coefficient and the coefficient of determination are presented for the fitted line.

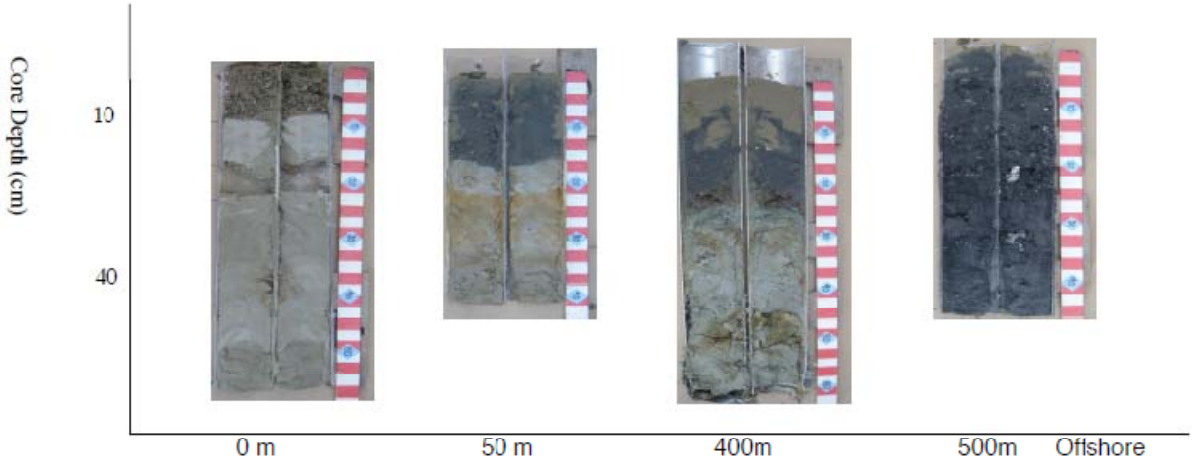


Figure 5 - Visual representation of data obtained during summer 2008 observations at Todds Pt. Upper panel shows a three dimensional image of local bathymetry with the location of the depth profile transect (middle panel) shown in red. The depth profile in the middle panel has been fitted to a Bruun 2/3 power law profile, with C=0.036 and r^2 = 0.88. Lower panel shows photos of core samples taken along the transect from 0m (beach) to 500m offshore. The scale interval for the ruler in each core photo is 2 cm with 10 cm marks annotated in blue.

One of the most important aspects of the summer 2008 data set is that sediment cores and grab

samples were taken along the offshore depth profiles (e.g., Todds Pt. Cores in Figure 5). At Todds Pt., each core had a layer of sand in the top 10-20 cm, with the coarsest being the gravelly tan beach sands at 0m offshore, finer tan sands interbedded with black anoxic sands at 50m and 400m, and a thin cap of fine sand at 500m. Toward the bottom of each core out to 400m, a substantial layer of fine grained muds (silts and clays) was present. Samples were obtained for all layers of each core and are currently stored in refrigeration at MGS. Only the samples from the upper sediment layer of each core were used for further grain size analysis. These samples were analyzed using standard MGS procedures for grain size analysis. Photographs and Munsell color classification were also used to examine offshore geology and translation of sediments from the top of the core. These results were then used to extrapolate estimates of the extent of the mobile sand unit.

In order to estimate the extent of the mobile sand unit and the shape of the offshore profile, Bruun's Rule was defined with the following parameters:

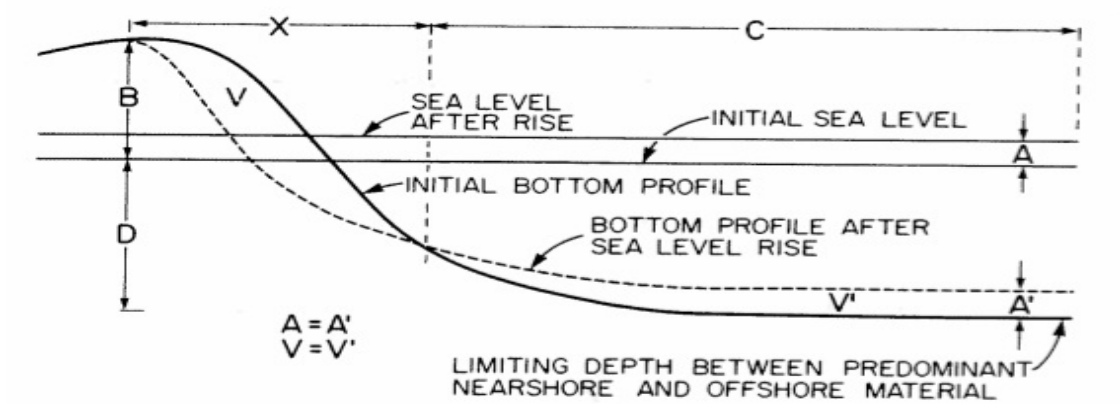


Fig.4. The Bruun Rule relating sealevel rise and shore erosion (after Bruun, 1962).

$$X = AC/(B+D)$$

where X - shore retreat B - shore elevation A - sea level rise

*C - distance to limiting depth from shore
*D - The limiting depth between predominant near shore and offshore material
(*principle unknown parameters)

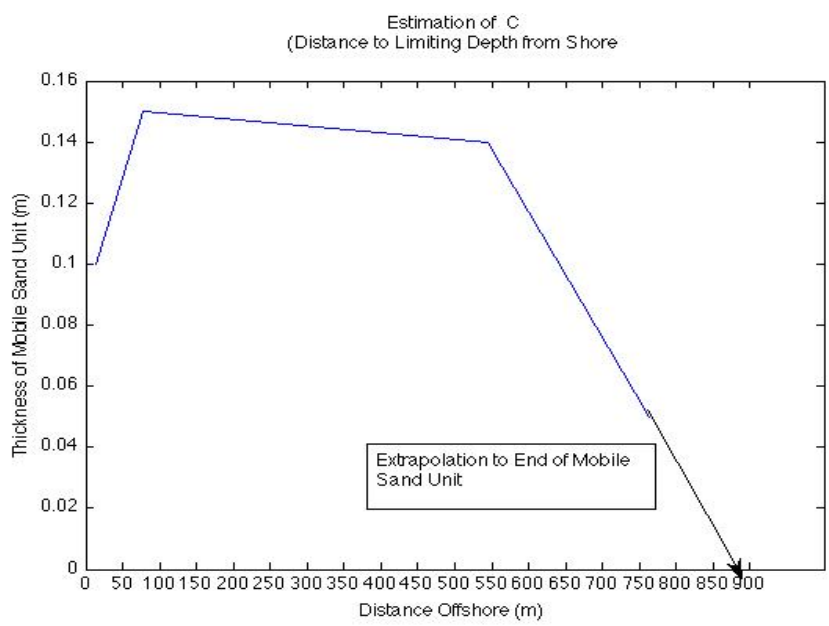
For Todds Pt., initial inputs of sea level rise over the last century, as well as the 1990s were used from the NOAA tides data base and Stevenson et al, 2002. In the Chesapeake Bay, where there is postglacial fore-bulge collapse, sea level rise has increased from 0.5 mm per year, from 1000 to 1850AD, to more than 3.2 mm per year during the 20th century (Stevenson et al, 2002). Current sea level rise in the Chesapeake Bay ranges from 3.1 mm per year in Baltimore Harbor to 7.0 mm per year at the Chesapeake Bay bridge tunnel in Virginia. The decadal rate in the 1990s was very high, more than 1.3 cm per year. By using these inputs of sealevel rise (A), measured shoreline elevations (B), measured limiting depth between predominant near shore and offshore material from the 2008 bathymetry data (D), and an extrapolated distance to limiting depth from shore (C) (Figure 5), estimates of shoreline erosion were made.

A - sea level rise *(0.0035 m/yr over last century; 0.013 m/yr 1990s)**
B - shore elevation *(0.9 m at Todd's Point)*
C - distance to limiting depth from shore *(extrapolated from C plot)*
D - The limiting depth between predominant near shore and offshore material *(from 2008 bathymetry data)*

Estimated X from 2008 Bruun Profile data: ~0.73 – 2.72 m/yr
Estimated X from Historic MGS Shoreline Data: ~1.80 m/yr
Estimated X from 2002/2003 GPS Survey: ~2.48 m/yr

* NOAA tides database; Stevenson et al, 2002

Figure 5: Estimate of C at Todds Point (distance to limiting depth from shore)



This technique has allowed us to use a series of sediment core and bathymetry data taken along shore-perpendicular profiles to examine offshore geology *(depositional history)* and translation of sediments from beach to beginning of channel *(estimates of the mobile sand unit)*. We have also used a variation in beach types and heights as well as amount of sand vs. fines at each site to examine variations in the offshore profile. This allows us to estimate sediment input to the near shore environment as well as the landward translation of the shore (shore erosion) in response to sea level rise.

The Todds Pt. example was presented by Ms. Bell at the 2010 Spring meeting of the Atlantic Estuarine Research Society.

Discussion and Future Work

Analysis is continuing along the lines of the Todds Pt example shown above. The technique of estimating the point of 0 thickness of the mobile sand unit to estimate C and D as applied at Todds Pt. does not work for all sites, such that a bathymetric variation approach is being explored as well.

Future applications of this work will be to attempt to use the Bruun profile model as a "baseline" in analyzing offshore profiles in the Chesapeake. This can allow for further exploration of relationships between grain size, beach types, composition and strength of eroding sediments, as well as additionally model the effects on the profile of longshore drift as well as effects of engineered coastlines. Finally, these estimates of sediment input to the nearshore environment will help to more accurately assess turbidity of the water column and provision of sand to SAV beds.

This work will form the basis of the MS thesis of Ms Lynda Bell, who expects to defend her thesis at the end of 2010.

References

Bruun, P., 1954, Coastal erosion and development of beach profiles: US Army Corps of Engineers, Waterways Experiment Station, Technical Memorandum 44.

Dean, D. R., D. D. Kriebel, et al. (2002). Cross-Shore Sediment Transport Processes. Coastal Engineering Manual Outline, Part III, Coastal Sediment Processes, Chapter III-3, Engineer Manual 1110-2-1100. D. T. Walton. Washington, D.C., U.S. Army Corps. of Engineers: 85.

Glibert PM, Magnien RE (2004) Harmful algal blooms in the Chesapeake Bay, USA: Common species, relationships to nutrient loading, management approaches, successes, and challenges. In:

Hagy JD, Boynton WR, Keefe CW, Wood KV (2004) Hypoxia in Chesapeake Bay, 1950–2001: Long-term change in relation to nutrient loading and river flow. Estuaries 27: 634-658

Halka, J. P. (2005). SEDIMENT IN THE CHESAPEAKE BAY AND MANAGEMENT ISSUES: TIDAL EROSION PROCESSES, Chesapeake Bay Program Nutrient Subcommittee Sediment Workgroup Tidal Sediment Task Force: 17.

Halka, J. P. and L. P. Sanford (2008). Contributions of shore erosion and resuspension to nearshore turbidity in the Choptank River, Draft report to the USEPA Chesapeake Bay Program.

Hardaway CS, Anderson GL (1980) Shoreline erosion in Virginia. Virginia Institute of Marine Science, Gloucester Point, VA

Hennessee, L., M. J. Valentino, et al. (2003). Updating shore erosion rates in Maryland. Baltimore, MD, Maryland Geological Survey: 26.

Kemp WM, et al (2005) Eutrophication in Chesapeake Bay: Historical trends and ecological interactions. Marine Ecology Progress Series 303: 1-29

Kimmel DG, Roman MR (2004) Long-term trends in mesozooplankton abundance in Chesapeake Bay, USA: influence of freshwater input. Marine Ecology Progress Series 267: 71-83

Kimmel DG, Miller WD, Roman MR (2006) Regional scale climate forcing of mesozooplankton dynamics in Chesapeake Bay. Estuaries and Coasts 29: 375-387

Koch EW, Wazniak C, Karrh L, Wells D (2007) Sediment as a limiting factor to seagrass distribution in the Maryland Coastal Bays, USA, in preparation

STAC (2003) Chesapeake Futures: Choices for the 21st Century. Chapter 9. Once and Future Bay

STAC (2006). Assessing Cumulative Impacts of Shoreline Modification Workshop Report: Chesapeake Bay STAC Proactive Workshop, CBP Scientific and Technical Advisory Committee. Publication 07-003: 11.

Stevenson JC, Kearney MS, Koch EW (2002) Impacts of sea level rise on tidal wetlands and shallow-water habitats: A case study from Chesapeake Bay. In: McGinn NA (ed) Fisheries in a Changing Climate. American Fisheries Society, Bethesda, MD, pp 23-36

USACE (1990). Chesapeake Bay Shoreline Erosion Study. Baltimore, Md, Department of the Army, U.S.Army Corps of Engineers: 111.

Wicks EC (2005) The effect of sea level rise on seagrasses: Is sediment adjacent to retreating marshes suitable for seagrass growth? M.S. degree thesis, University of Maryland, College Park

Low-cost Anaerobic Digesters for Dairy Manure Treatment and Renewable Energy Production

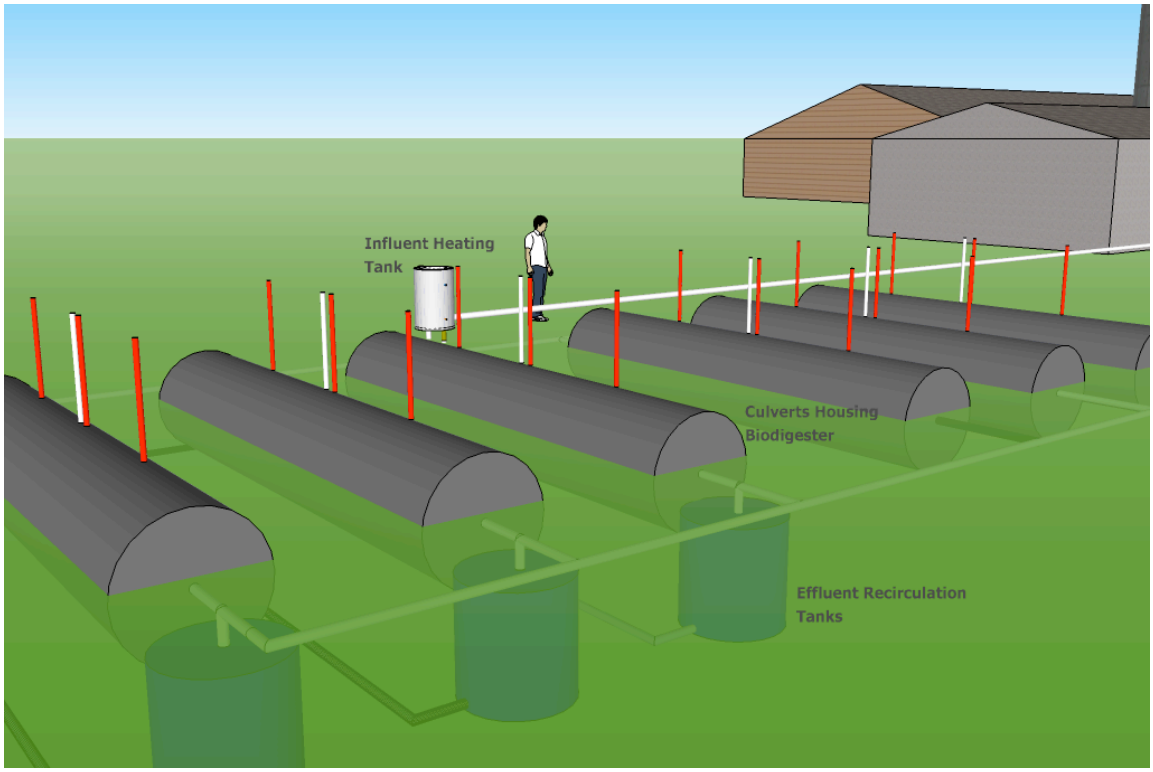
Basic Information

Title:	Low-cost Anaerobic Digesters for Dairy Manure Treatment and Renewable Energy Production
Project Number:	2009MD193B
Start Date:	5/1/2009
End Date:	5/31/2012
Funding Source:	104B
Congressional District:	5
Research Category:	Water Quality
Focus Category:	Wastewater, Treatment, Agriculture
Descriptors:	None
Principal Investigators:	Stephanie Lansing

Publication

1. None

Report as of FY2009 for "Low-cost Anaerobic Digesters for Dairy Manure Treatment and Renewable Energy Production"



Reporting period: March 1, 2009-February 28, 2010
Project duration: March 1, 2009-February 28, 2011

Prepared for:
Maryland Water Resources Research Center

Principal Investigator:
Stephanie Lansing
slansing@umd.edu
301-405-1197

Prepared by:
Dr. Stephanie Lansing
Andrew Moss (M.S. student)
Katharine Strass (M.S. student)
Department of Environmental Science and Technology
1445 Animal Science Building
University of Maryland
College Park, MD 20742

Submitted May 26, 2010

Background Information

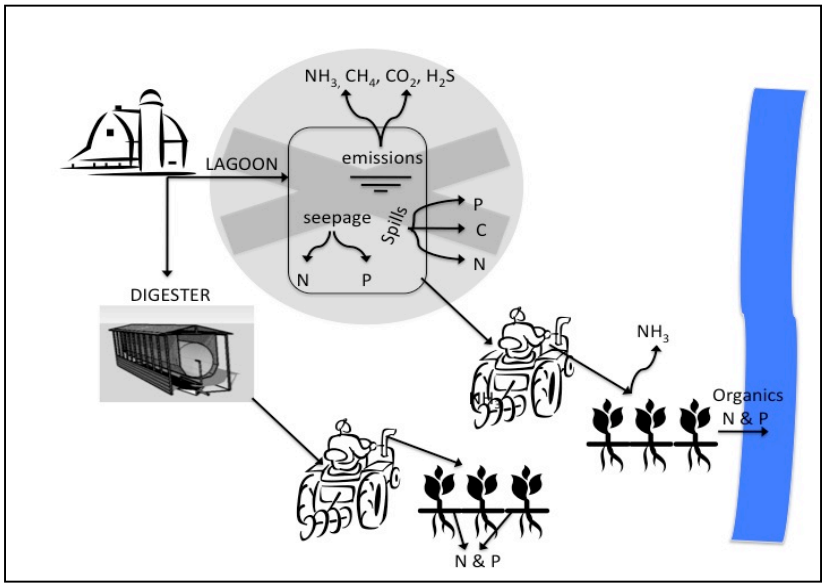
GOAL

The long-term goal is to provide low-cost treatment options for small to medium-scale farmers that produce renewable energy and reduce environmental degradation and greenhouse gas emissions. To achieve this goal, two types of modified low-cost digesters will be designed, implemented, studied, and modeled over a three-year period to provide long-term research on a system that has not been fully explored. A research protocol will be developed for studying low-cost digestion systems, and an operation and maintenance plan will be developed that is appropriate for farmers wanting to adopt this accessible technology at their farms.

LITERATURE REVIEW

Improper treatment of dairy manure results in contamination of waterways, noxious odors, and the release of methane, which is a greenhouse gas with 21 times the global warming potential of carbon dioxide (USEPA, 2004). Agricultural runoff and wastewater from animal operations are currently the dominant source of nonpoint-source pollution to surface waters in the United States, contributing to pathogen proliferation, siltation, oxygen depletion, and elevated nutrient concentrations in streams and reservoirs (USEPA, 2002). The majority of generated waste from animal containment facilities is held in waste lagoons. After initial settling, the wastewater is land-applied onto nearby fields. There are numerous potential public health and environmental concerns associated with these lagoon-sprayfield waste management systems (Hill, 2003). Lagoon failures and flooding have caused waste spills, groundwater has been contaminated due to seepage, and surface waters have been impacted from sprayfield runoff (Mallin et al., 1997; Parker et al., 1999). In addition, there are negative social perceptions associated with lagoons due to odor emanation and negative landscape appeal (Schiffman, 1998).

An anaerobic digestion system can be used in place of a lagoon system and transform animal waste into an environmental and economic benefit. A digester provides an optimal environment for microorganisms that produce methane by using the wastewater as a nutrient source. The digestion process results in a number of benefits: the captured methane becomes a source of renewable energy, a liquid fertilizer is created, and wastewater pollution, greenhouse gas emissions, and noxious odors are sharply reduced (USEPA, 2004). A digester sharply reduces the organic matter



and phosphorus content of the wastewater, thus reducing surface pollution from land application and drastically reducing its potential odor and water pollution problems (Figure 1). In addition, a digester can add value to manure, which is often viewed as a waste and not a resource.

Figure 1: Digesters can reduce water and odor pollution from dairy facilities by eliminating lagoon management systems, increasing nitrogen (N) and phosphorus (P) soil infiltration, and reducing surface runoff of organic pollutants, N, and P and emissions of methane (CH_4), carbon dioxide (CO_2), hydrogen sulfide (H_2S), and ammonia (NH_3) both from the lagoon and the fields after application.

In the United States, previous digester research has focused exclusively on industrialized systems that are capital and management-intensive, and with an average cost of $1.0 million, are inaccessible to medium and small-scale farmers (USEPA, 2006). Of the 114 existing digesters in the United States, 88 are located on dairy farms. Due to capital requirements, the U.S. EPA recommends digester installation for herds with more than 500 cows, which puts this beneficial technology out of the hands of the overwhelming majority of Maryland stakeholders (USEPA, 2006). Ninety-four percent of Maryland dairies (774 farms) have less than 200 dairy cows, and only 8 farms in Maryland have more than 500 dairy cows. This trend is observed throughout the Chesapeake Bay watershed, with 94.5 % of the dairy farms having less than 200 cows and only 1.2 % having 500 cows or more (USDA, 2004). Dairy operations with fewer than 200 cows have been found to be significant contributors of water pollution and could greatly benefit from the environmental and economic benefits of anaerobic digestion (MacDonald et al., 2007).

Low-cost anaerobic digestion is a proven technology in developing countries, with over 10 million low-cost digesters in India, China, and Latin America (Abraham et al., 2007; Lansing et al., 2008b). The transfer of this technology to temperature zones in the United States has not been explored previously. This project seeks to fulfill this research gap by investigating modified low-cost digesters in Maryland for technology transfer to dairy farms throughout the Chesapeake Bay watershed. The results of this research will support the development of digesters that can be used by the vast majority of dairy farmers in Maryland to treat their manure, obtain renewable energy and a higher quality fertilizer, and increase the economic viability of their farm by offsetting electricity and heating costs. Lowering the cost of anaerobic digestion systems in the Chesapeake Bay watershed could lead to high adoption rates of this technology, resulting in less organic pollutants and nutrients reaching the Chesapeake Bay from the over 15,000 small to medium-scale dairy farms in the watershed and greater energy independence and economic incentives for waste treatment for the majority of farms in the region.

In low-cost, plug flow digesters, 90 % of the organic material is removed, which is 40 % higher than the industrial mixed digesters favored in the United States (Lansing et al., 2008b). Releasing organic matter into aquatic environments can lead to deoxygenation of waters and large-scale fish and invertebrate kills. Additionally, solids associated with organic wastes increase turbidity, reduce light penetration for photosynthetic organisms,

and settle on the substratum, altering the benthic invertebrate communities. Low-cost plug-flow digesters have higher organic matter removal efficiencies than completely stirred industrial digesters due to longer solids retention time, which promotes further degradation of the organic material and results in improved water quality in surrounding waterways (Hobson, 1990). Additionally, over 97 % of the pathogens are removed from the wastewater during digestion in low-cost, mesophilic systems (Khang and Tuan, 2002; Lansing et al., under review).

Total nitrogen and phosphorus concentrations are reduced by 35 to 70 % in plug-flow digesters, and the remaining nutrients are transformed from an organic to dissolved state during digestion, which is a more useful form for plant uptake. Due to higher total phosphorus retention rates in low-cost digesters, the fertilizer value of the effluent is higher than raw manure and effluent from completely mixed digestion systems, which do not retain phosphorus. The average N:P ratio of plug-flow digester effluent is 7:1, which is higher than undigested manure (4:1) and closer to corn nutrient uptake needs (7.5:1) (Edwards and Daniel, 1992). Previous studies have shown that digester effluent enhances crop production in relation to raw manure (Massé et al., 2007). The liquid effluent from plug-flow digesters has also been used successfully as an aquaculture feed and treated to tertiary standards in wetland cells (Lansing et al., 2008b).

The benefits of anaerobic digestion also include reductions in greenhouse gas emissions and nitrogen (N) losses. In anaerobic digestion systems, methane that is naturally produced from stored manure is captured and combusted. Additionally, the digested effluent has a decreased C:N ratio, resulting in a lower viscosity, which increases N mineralization and plant uptake shortly after application, and decreases N leaching, N loss through denitrification, and ammonia emissions due to increased infiltration into the soil surface (Chaussod et al., 1986; Rubaek et al., 1996; Massé et al., 2007). The environmental benefits of low-cost anaerobic digestion in terms of water quality, renewable energy production, and greenhouse gas reductions make this technology timely to address the needs of farmers, Chesapeake Bay watershed stewards, and the state of Maryland's carbon reduction and renewable energy targets.

Low-Cost Digesters

Digesters provide an environment conducive to the growth of the bacteria involved in the transformation of organic matter to carbon dioxide and methane. Two temperature ranges are optimum for anaerobic digestion: mesophilic (25 – 40 ºC) and thermophilic (50 – 65 ºC). The digesters in this proposed study will be mesophilic (28 ºC). Low-cost digesters are often plug-flow, the contents are not mixed, and continuous management or internal heating is not required when located in a tropical climate (Chará et al., 1999). The solids tend to settle out, resulting in better degradation of solids in these systems compared to industrialized completely mixed reactors, which use a portion of the produced biogas to heat and mix the digester (Hobson, 1990; Berglund and Börjesson, 2006). The majority of digester studies have been conducted on lab-scale completely mixed reactors.

Previously, little information existed on low-cost digesters. The PI conducted three investigations on low-cost digesters in Costa Rica from 2005 – 2008 (Figure 2). The studies investigated variability in biogas quality and water pollutant reductions in seven digesters located at small farms in Costa Rica (Lansing et al., 2008a), the production of electricity using combined biogas from low-cost swine and dairy digesters (Lansing et al., 2008b), and

methane production, wastewater pollutant reductions, and fertilizer utility in 12 field-scale, low-cost digesters co-digesting swine manure and used cooking grease (Lansing et al., 2010).

The digester variability study revealed that the digesters reduce organic matter by 84.1 %, produce biogas with 66.3 % methane, and increase NH_4-N by 78.3 % (Lansing et al., 2008a). In the electricity production study, the dairy and swine digesters produced 33.5 m^3/day of biogas, with 64.8 % methane and reduced organic matter by 89.0 %, meeting 82 % of the farm's 2-hour peak electricity demand (25.8 kW/day). The $21,000 capital cost of the project will be recovered in 10 years through electricity savings and wastewater fines reductions (Lansing et al., 2008b).

The co-digestion experiments revealed that combining swine manure with 2.5 % used cooking grease increased the methane production by 111%, and no additional benefits were seen by increasing the grease concentration beyond 2.5 % due to the lower quality of biogas produced. No adverse effects were observed from co-digesting with 2.5 % grease in terms of organic matter removal (94.7 %), pathogen reduction (98 %), and grease removal (98 %). There was less phosphorus reduction with co-digestion (33.5 %), resulting in lower N:P ratios in the grease treatment groups compared to the control (0 % grease) (Lansing et al., 2010).

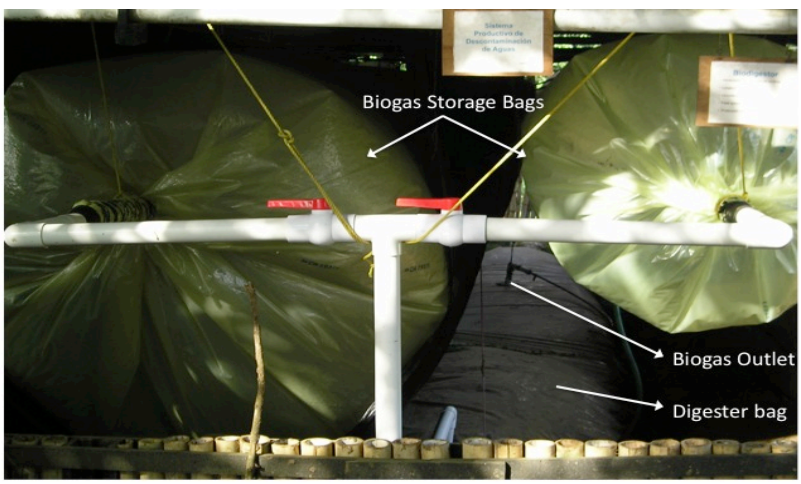


Figure 2: Pictured is a low-cost digester that treats dairy manure and produces electricity in Costa Rica. The biogas from the digester bag flows through the biogas outlet to two biogas storage bags located above the digester.

In addition to these investigations, similar wastewater reductions and biogas production have been found in other low-cost digestion studies in Costa Rica (Botero and Hernández, 2005), Colombia (Pedraza et al., 2001), Vietnam (Khang and Tuan, 2002), and Mexico (Esquivel et al., 2002). The baseline data and experiences from these digester studies were used in designing both the digesters and experimental protocol used in this study. These previous investigations demonstrated that small-scale digesters are ideal systems for rural farmers to treat livestock wastewater and obtain renewable energy from their waste products when located in a tropical climate. The current study takes low-cost digesters to the next level by utilizing the basic design principles of low-cost digesters and adapting them to temperate climates.

Digester Design

The digesters used in these previous studies were made of polyethylene and were constructed using the Taiwanese biodigester design developed by Raul Botero and T.R. Preston (Figure 3) (Botero and Preston, 1987). The material to be used in the proposed project is a thicker, more durable, and insulating geomembrane material and has been successfully used in the highlands of Mexico (Eaton, personal communication). There have been previous digesters built in Costa Rica and Bolivia at high altitudes that were enclosed in a greenhouse structure similar to the structure in the design of the current project (Marti Herrero, 2007; Gonzalez, personal communication).

The preheating of the digester influent is a cost-saving method that has been studied at the field-scale level on sophisticated digestion systems by Avatar Energy, Inc. (Roberts, personal communication). This system included both preheating of the digestion materials and continual heating of the digester environment. When continual heating is used, the digester materials become much more expensive, thus putting anaerobic digestion technology beyond the financial ability of the majority of dairy farms. Utilizing only pre-heating digestion apparatus and keeping the simpler geo-membrane PVC digester design of low-cost digesters has not been explored.

Specifically, in the current research, it will be determined whether preheating alone can maintain the mesophilic temperatures (25 – 40 ºC) needed for optimal methane production or if recirculation of the heated digester effluent is needed to maintain an optimal temperature range. Neither of the proposed designs has been tested on the field-scale level. Recirculation of the effluent has been previously used in an internally heated, plug-flow digester in Minnesota (Martin, 2005). The recycling of the effluent to the front of the digester was used for used to improve treatment capability in this study, but the digester cost was $550,000, in large part due to the materials needed for internal heating of the digestion environment.

A report has been released by the Minnesota Project that details current options for small-scale producers (Goodrich, 2007). The capital cost for a 100-cow farm based on five current technology options was $105,000 - $184,000, with the lowest cost model being an untested model that was proposed by the report as a possible future lower-cost design. All of the digesters in this report were internally heated, which drastically increases the cost of materials and places these digesters out of the financial means of the average Maryland farmer. The proposed design strives to lower the overall capital costs to $15,000 - $30,000 while maintaining the benefits of digestion technology.

OUTCOMES AND BENEFITS

This investigation will quantify methane production, as well as the removal and transformation of organic material, solids, and nutrients in low-cost digesters modified for a temperate climate. Nine field-scale low-cost digesters, using three separate operating designs, will be constructed at the USDA Beltsville Agricultural Research Center (BARC) dairy farm and monitored for one year in order to determine the optimal design for low-cost digesters in a temperate climate and statistically analyze variability between digester designs over time.

The study results will be utilized in an emergy analysis to further understanding of nutrient dynamics and facilitate predictions of treatment capacity, renewable energy

production, and capital cost recovery time for future digestion installations on different sized dairy farms. Ultimately, the investigation results will facilitate comparisons between low-cost and higher-cost digestion systems and access economic viability, treatment capacity, and renewable energy production potential of low-cost anaerobic digestion systems for medium to small-scale dairy farmers in the temperate United States.

In addition to publication in peer-review journals, an extension bulletin will be created at the end of the one year study to address the lack of information on low-cost digestion options that are available to small and medium-scale dairy farmers. The results from the emergy and financial comparisons between different types of digesters will also be included in the extension bulletin in order to allow farmers to identify digestion possibilities given the numbers of dairy cows, manure management style, and capital investment capabilities.

The U.S. EPA states that the main barriers to the widespread adoption of anaerobic digestion technology are the cost and sophistication of the operating equipment (USEPA, 2006). This research will advance the field of digestion technology by creating a system that produces renewable energy, while maintaining the simplistic design advantages of low-cost digesters. The research reaches across disciplinary boundaries by integrating traditional farming specialties, i.e. animal production and crop growth, with ecological engineering. The new knowledge created from this research will improve scientific understanding of digesters and provide medium to small-scale farmers with a digestion system that produces energy to meet farm needs while providing fertilizer for their crops, and reducing nutrient translocation, pathogens, environmental degradation, and greenhouse gas emissions.

OBJECTIVES

The specific objectives of the project are to: (1) test the ability of low-cost digesters to produce energy and treat manure throughout the year in a temperate climate; (2) determine the optimal design for low-cost digestion systems in the temperate U.S.; (3) develop a research, operational, and maintenance protocol for low-cost digestion systems; (4) develop a emergy analysis using the system results in order to enhance understanding of the digester process and its energy use or "energy memory," known as "emergy," and compare results between small-scale and industrial-scale digesters.

Project Update

Several changes were made to the original project.

1. The project site was moved from the Clarksville Maryland Research and Education Center (CMREC) dairy farm to USDA Beltsville Agricultural Research Center (BARC) dairy farm. The movement of the project site will allow for direct comparisons between an industrial mixed digester, which is currently operating at BARC, and the experimental low-cost digesters that are being built and study as part of this project. The movement of the project site did cause a delay in construction, but also allowed

more time for the PI and her graduate students to optimize the design and add additional testing components to the overall system.

2. While the movement of the project site did allow for a comparative study between two different types of digesters and allowed for new collaborative research efforts between the PI from UMD and USDA BARC scientists on digestion technology, it meant that the existing treatment wetland at Clarksville could not be utilized. There is room to add a treatment wetland to the end of the experimental digesters in the future, but since wetland construction was not part of the grant budget, a polishing treatment wetland will not be included in this phase of the project. It has been proposed that the results from this experimental digester study, which is treating 10% of the waste produced at BARC, be used to construct a full-scale digester system, with a treatment wetland, at the Clarksville site in the future.
3. The design of the digesters has been completed. There are nine digesters being constructed with a geo-membrane material, laid in insulative foam beds surrounded by radiant barriers, and enclosed within 42-inch corrugated drainage culverts to protect, insulate, and help maintain the desired shape of the biodigesters. After the manure is heated in the kettle, it is released into a digester and the hot water from the kettle jacket is circulated through radiant piping located underneath each digester bag in order to provide additional heating throughout the length of the digester (Figure 3). The culverts will be partially buried for insulation and protected from the elements by a windshield structure.
4. The digesters were placed in the ground in May 2010. During June 2010, the digesters will be tested, and in July 2010, they will start to receive wastewater and the sample collection period will begin.
5. The dynamic model of the original proposal has been replaced by an emergy and financial analyses. With the movement of the project site to BARC, and the side-by-side comparison of the two digester types, the financial component and energy needs/processing of the two designs were seen as the more appropriate analyses.
6. In following with the research objectives, an additional focus has been developed in which the digestion of post-solid-separation manure and un-separated manure will be compared in terms of energy returns, treatment efficacy, and overall cost savings. Screening – or the separation of the liquid and solid components of the manure – represents additional mechanization and cost in the anaerobic digestion process (Wilkie, 2005) and may potentially be avoided in low-cost systems. In addition, by studying nutrient dynamics in both digesters that treat the entire wastewater stream and those that treat only the effluent from the solid separation, we will be able to evaluate the efficiency of solid separation/composting for digestion technology both in terms of energy production and nutrient removal.

Project Gallery

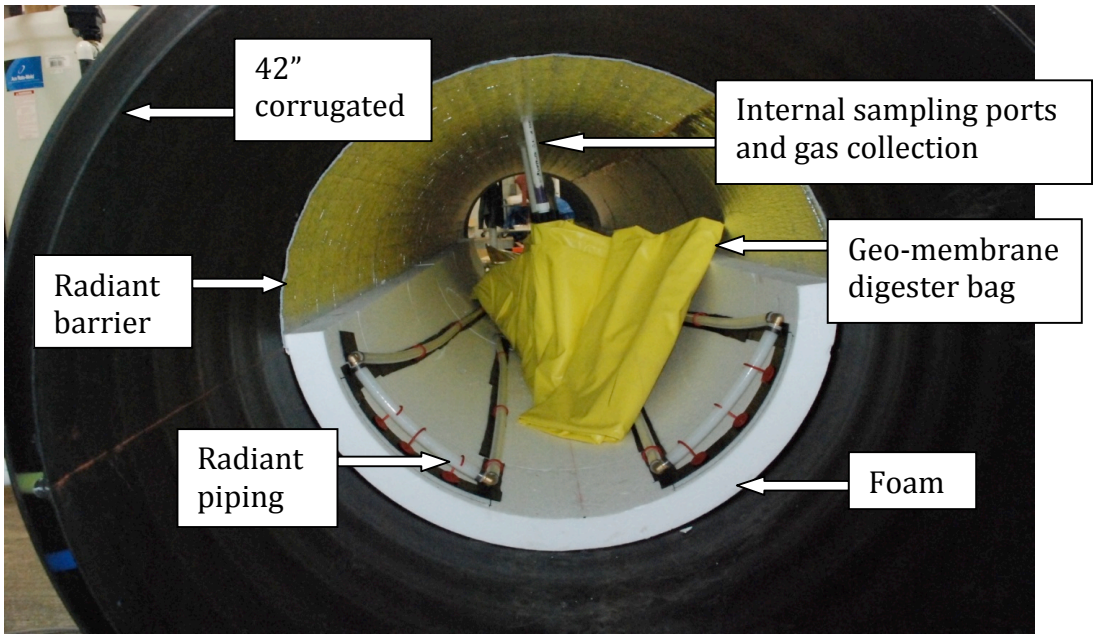


Figure 3: Modified plug-flow digester highlighting various components

Figure 4: Undergraduate assistant, Scott Allen, installs sampling ports into a partially constructed digester.

REFERENCES

Abraham, E.R., S. Ramachandran, V. Ramalingam, 2007. Biogas: Can it be an important source of energy? Environmental Science and Pollution Research 14 (1), 67-71.

APHA, 1998. Standard methods for the examination of water and wastewater. 20th ed., Clesceri, L.S., Greenberg, A.E., Eaton, A.D. (Eds.). American Public Health Association, Washington, D.C.

Berglund, M., P. Börjesson, 2006. Assessment of energy performance in the life-cycle of biogas production. Biomass and Bioenergy 30, 254-266.

Botero, R., T.R. Preston, 1987. Low cost biodigesters for the production of fuel and fertilizer from animal excreta: Manual for installation, operation, and utilization. Centro para la Investigatión en Sistemas Sostenibles de Producción Agropecuaria, Cali, Colombia.

Botero, R., C. Hernández, 2005. Productive management of waste in tropical cattle farms (Manejo productivo de excretas en sistemas ganaderos tropicales), In: Revista INFOHOLSTEIN. Asociación de Criadores de Ganado Holstein de Costa Rica, San Jose.

Chará, J., G. Pedraza, N. Conde, 1999. The productive water decontamination system: A tool for protecting water resources in the tropics. Livestock Research for Rural Development 11 (1).

Chaussod, R., G. Catrouz, C. Juste, 1986. Effects of anaerobic digestion of organic wastes on carbon and nitrogen mineralization rates: laboratory and field experiments, in: Kofoed, A.D., Williams, J.H., L'Hermite, P. (Eds.), Efficient Land Use of Sludge and Manure. Elsevier Applied Science Publishers, London, pp. 56–60.

Eaton, A., personal communication (2007 – 2008). Director of International Renewable Resources Institution – Mexico, Tlacotalpan, Mexico.

Edwards, D.R., T.C. Daniel, 1992. Environmental impacts of on-farm poultry waste disposal. A review. Bioresource Technology 41, 9–33.

Esquivel, R.R., M.D. Méndez y Cazarín, T.R. Preston, and G. Pedraza, 2002. Important aspects for introducing biodigesters in small-scale dairy operations. (Aspectos importantes al introducir biodigestores en explotaciones lecheras a pequeña escala). Livestock Stock Research for Rural Development, 14 (3): www.cipav.org.co\lrrd\lrrd14\3\viey143.htm.

Derico, E.F., 2001. Nutrient dynamics and plant community structure of a constructed wetland system for treating dairy manure milkhouse waste. M.S. thesis. University of Maryland.

Goodrich, P.R., 2007. Anaerobic digester systems for mid-sized dairy farms. The Minnesota Project 2005. Report submitted to AgSTAR Program, US Environmental Protection Agency. http://www.mnproject.org/pdf/agstar%20report%20full%20update.pdf

Gonzalez, P., personal communication, 2007-2008. Owner: Biosinergia Alternativa S.A. Agro-Environmental Consulting. Guápiles de Pococí, Costa Rica.

Hill, V.R., 2003. Prospects for pathogen reduction in livestock wastewater: a review. Critical Reviews in Environmental Science and Technology, 30 (2): 187-235.

Hobson, P.N., 1990. The treatment of agricultural wastes. In: A. Wheatley (Ed), Anaerobic Digestion: A Waste Treatment Technology. Critical Reports on Applied Chem. 31, 93-138.

Jones, R.M., J.F. MacGregor, K.L. Murphy, E.R. Hall, 1992. Towards a useful dynamic model of the anaerobic wastewater treatment process: a practical illustration of process identification. Water Science and Technology, 25 (7): 61-71.

Khang, D.N., L.M. Tuan, 2002. Transfer of low-cost plastic biodigester technology to producers in Southern Vietnam. (Transferencia de la tecnología del biodigestor plástico de bajo costo a los productores del sur de Vietnam). In: Julián David Chará, (Ed), Biodigestores plásticos de flujo continuo: Investigatión y transferencia en países tropicales. Foundación CIPAV. Cali, Columbia. pp. 47-54.

Lansing, S., R.B. Botero, J.F. Martin, 2008a. Wastewater treatment and biogas production in small-scale agricultural digesters. Bioresource Technology 99, 5881-5890.

Lansing, S., J. Víquez, H. Martínez, R. Botero, J. Martin, 2008b. Optimizing electricity generation and waste transformations in a low-cost, plug-flow anaerobic digestion system. Ecological Engineering. In Press.

Lansing, S., J.F. Martin, R.B. Botero, T. Nogueira da Silva, E. Dias da Silva, Under Review. Wastewater transformations and fertilizer value when co-digesting swine manure and used cooking grease in low-cost digesters. Water Research.

MacDonald, J.M., E.J., O'Donoghue, W.D., McBride, R.F., Nehring, C.L., Sandretto, R., Mosheim, 2007. Profits, costs, and the changing structure of dairy farming. Economic Research Report Number 47. United States Department of Agriculture, September 2007.

Mallin, M.A., J.M. Burkholder, G.C. Shank, H.B. Glasgow, Jr., B.W. Touchette, J. Springer, 1997. Comparative effects of poultry and swine manure waste lagoons spills on the quality of receiving waters. Journal of Environmental Quality, 26: 1622-1631.

Marti Herrero, J., 2007. Democratizing an appropriate technology: Low-cost family digesters in Bolivia. (Democratizando una tecnología apropiada: biodigestores familiares de bajo costo en Bolivia). Energía y Desarrollo 30: May 2007.

Martin, Jr., J.H., 2005. An evaluation of a mesophilic, modified plug flow anaerobic digester for dairy cattle manure. Report submitted to AgSTAR Program, US Environmental Protection Agency. http://www.focusonenergy.com/data/common/ dmsFiles/W_RW_REFR_GordondaleFeasReport.pdf.

Massé, D.I., F. Croteau, L. Masse, 2007. The fate of crop nutrients during digestion of swine manure in psychrophilic anaerobic sequencing batch reactors. Bioresource Technology 98, 2819-2823.

Parker, D.B., D.E. Eisenhauer, D.D. Schulte, D.L. Martin, 1999. Modeling seepage from an unlined cattle feedlot storage pond. Transacitons of the ASAE, 42 (5): 1437-1445.

Pedraza, G., J. Charrá, N. Conde, S. Giraldo, L. Giraldo, (2001) Evaluación de los biodigestores en geomembrana (PVC) y plástico de invernadero en clima medio para el tratamiento de aquas residuals de origin porcino. Livestock Stock Research for Rural Development 14 (1), www.cipav.org.co\lrrd\lrrd14\1\Pedr.14.1.htm.

Roberts, Guy, personal communication, 2008. Avatar Energy – A Renewable Energy Company, South Burlington, VT.

Rubaek, G.H., K. Henriksen, J. Petersen, B. Rasmussen, S.G. Sommer, 1996. Effects of application technique and anaerobic digestion on gaseous nitrogen loss from animal slurry applied to ryegrass (*Lolium perenne*). J. Agric. Sci. 126, 481–492.

Schiffman, S., 1998. Livestock odors: implications for human health and well-being. Journal of Animal Science, 76 (5): 1343-1355.

Siegrist, H., D. Renggli, W. Gujer, 1993. Mathematical modeling of anaerobic mesophilic sewage sludge treatment. Water Science and Technology, 27 (2): 25-36.

USDA, 2004. 2002 Census of Agriculture: United States Summary and State Data, Vol. 1: Geographic Area Series, Part 51. AC-02-A-51.

USEPA, 2002. National water quality inventory, 2000 report. Office of Water, Washington D.C. EPA 832-F-02-025

USEPA, 2004. AgSTAR Handbook: A manual for developing biogas systems at commercial farms in the United States, 2nd ed. Roos, K.F., Martin, Jr., J.H., Moser, M.A. (Eds.). EPA 430-B-97-015.

USEPA, 2006. Market opportunities for biogas recovery systems: A guide to identifying candidates for on-farm and centralized systems. AgSTAR Program: USEPA, USDA, and DOE Energy and Pollution Prevention. EPA-430-8-06-004.

Wilkie, 2005. Anaerobic Digestion of Dairy Manure: Design and Process Considerations. Dairy Manure Management: Treatment, Handling, and Community Relations. NRAES-176, p.301-312. Natural Resource, Agriculture, and Engineering Service, Cornell University, Ithaca, NY, 2005.

Role of Invertebrate Bioturbation in Phosphorus Retention of Agricultural Ditch Soils--Summer Fellowship

Basic Information

Title:	Role of Invertebrate Bioturbation in Phosphorus Retention of Agricultural Ditch Soils--Summer Fellowship
Project Number:	2009MD199B
Start Date:	3/1/2009
End Date:	2/28/2010
Funding Source:	104B
Congressional District:	5th
Research Category:	Water Quality
Focus Category:	Agriculture, Hydrogeochemistry, None
Descriptors:	None
Principal Investigators:	Allen Davis, William Lamp

Publications

There are no publications.

Role of Invertebrate Bioturbation in Phosphorus Retention of Agricultural Ditch Soils

Alan W. Leslie, Department of Entomology, University of Maryland

Summary of Research during MWRRC 2009 Summer Graduate Fellowship

Agricultural drainage ditches on the Eastern Shore of Maryland primarily drain water from fields with poorly drained soils to allow crops to be grown, and secondarily have been engineered to manage the loads of nutrients such as nitrogen and phosphorus being conveyed to local watersheds in agricultural runoff. By controlling rates at which water leaves drainage ditches, nitrates can effectively be removed by promoting anaerobic conditions in the soils, and particle-bound phosphate can settle out and be prevented from being carried downstream. However, loads of dissolved phosphorus are not effectively managed by controlled drainage. The purpose of this study was to determine how the activity of aquatic invertebrates burrowing in the soils within ditches might be altering the chemistry of ditch soils, and how this might influence the amount of dissolved phosphorus released into the water column.

The hypothesis being tested was that small aquatic animals burrowing in the upper soil layers add oxygen to those soil layers by pumping oxygen-rich water into burrows dug in soils otherwise depleted of oxygen. This may prevent phosphorus from being released into the water by maintaining an oxygenated environment where phosphorus preferably adsorbs to coatings on soil particles. To test this hypothesis, mesocosms were constructed using ditch soils inundated with water both in the presence and absence of aquatic oligochaete worms. The glass mesocosms were constructed to be of very thin profile, so that burrowing activity could be monitored, while allowing enough room for insertion of probes. Soil and oligochaete worms were collected from the research farm at the University of Maryland, Eastern Shore (Princess Anne, MD). Oligochaetes were identified as belonging to the family Lumbriculidae, and the soil was silt loam with approximately 5.5% organic matter. Mesocosms were kept at 23ºC, which approximated the average temperature within the ditch from mid May to mid June.

After a week of bioturbation by the aquatic worms, oxidation-reduction potential (redox) measurements were made on each of the mesocosms using platinum electrodes and a Calomel reference. If the worm burrows do in fact increase the circulation of oxygenated water within the soil, then mesocosms with worms should have a higher redox potential than the controls. However, the results of the experiment showed that soil mesocosms with aquatic worms actually had a slightly lower redox potential than the controls. Although this difference was not statistically significant (i.e. $P = 0.08$), there is a trend towards lower redox potential in soils being bioturbated by aquatic worms.

The results of this study suggest that the burrowing activity of aquatic worms may actually accelerate the release of dissolved phosphorus within the water column, which would promote the loss of nutrient pollution to local watersheds, and the Chesapeake Bay. Through observation of the worms burrowing in the mesocosms, it was clear that through undulating motions, water was being translocated within the upper soil layer; however the effect was

definitely not the aeration of soils adjacent to the animals. Further observation of the worms revealed the deposition of castings, or feces at the soil surface, near the entrance of their burrows, nearly covering the entire soil surface at the completion of the experiment. The effect of the deposition of feces may be to mix soil materials, and to incorporate organic matter deeper within the soil profile. This mixing of organic matter, coupled with excretion by the individual worms may have led to increased bacterial respiration within the soil, and the observed decreased redox potential. In conclusion, the original hypothesis of increased soil aeration through bioturbation by aquatic worms is not supported, and a new hypothesis, that aquatic worms promote anaerobic conditions within drainage ditch soils is suggested.

Correlating Nutrient, Pesticide, and Bacteria levels in the Choptank River--Summer Fellowship

Basic Information

Title:	Correlating Nutrient, Pesticide, and Bacteria levels in the Choptank River--Summer Fellowship
Project Number:	2009MD200B
Start Date:	3/1/2009
End Date:	2/28/2010
Funding Source:	104B
Congressional District:	5th
Research Category:	Water Quality
Focus Category:	Agriculture, Hydrogeochemistry, None
Descriptors:	None
Principal Investigators:	Allen Davis, Alba Torrents

Publications

There are no publications.

“Influence of poultry runoff on a small stream: Indigenous bacteria, nutrients, pesticides, metals, and antibiotics”: A Research Progress Report Submitted to the Maryland Water Resources Research Center

Gabriela Niño de Guzmán, Department of Civil and Environmental Engineering, University of Maryland, College Park

The research conducted this summer focused on determining if and/or how runoff from a medium-sized poultry facility was affecting the local aquatic microbial population. This was done by collecting water column and sediment samples upstream and downstream of the facility, and by paying particular attention to the water-sediment interface layer that is very easily disturbed. Nutrients, pesticides, metals, antibiotics, and bacteria were all studied as factors of disturbance.

The original objectives of my work were to 1. investigate poultry facilities in four sub-watersheds in the Choptank River watershed, 2. gather bacterial, pesticide, and nutrient data, 3. create a water model from these data describing pollutant dispersion and dissipation from the small tributaries to the larger river body, and 4. compare pollutant behavior in poor- and well-drained soil areas. A closer look at one of the selected well-drained watersheds yielded this site: one poultry facility in the immediate area of interest with surrounding land ditched into a nearby stream. Fairly straight waterways, close proximity to a river body, no major obstacles or interfering facilities (that might confuse data), and accessible sampling points added to this site’s appeal. I therefore decided to re-focus my work here as it is better suited for discerning direct impact of a poultry facility on a stream and surrounding environment.

Six sampling sites along this stream have been selected (Figure 1), and the original control site has been kept. In order to get a more detailed idea of poultry facility

Figure 1: Aerial view of sampling locations; control site (G7) not pictured

and surrounding agriculture impact on the local aquatic microbial population, I decided to expand the original list of investigated pollutants and add sediment sampling. Water column *and* sediment samples are analyzed for antibiotics and bacteria (*E. coli*, *Enterococcus sp.*, *Clostridium sp.*); water column samples are additionally screened for nutrients (nitrogen, phosphorus), metals (arsenic, copper, iron), and pesticides (atrazine, metolochlor). All weather conditions are observed and logged, water quality parameters (pH, temperature, DO, conductivity) are recorded for all sites, and pictures are taken for visual record. Stream depth is also measured at every site for later correlation to flow.

Starting at the end of May, preliminary samples were taken from sites 1 through 4 to determine suitability of sampling locations, establish sampling procedures, and investigate the constituent concentrations to be expected. These samples were analyzed for arsenic, pesticides, antibiotics, and bacteria. To more fully describe the data collected at site 4 (site closest to the Choptank River and furthest downstream), sites 5 and 6 were added. These two sites do not see any input from the poultry facility, but may see some impact from surrounding agricultural fields. These sites (upstream from site 4) describe a different tributary and are located in the thick of the buffer zones surrounding the adjacent streams. Nutrients, copper, and iron analyses were added shortly after the preliminary sampling to investigate other sources of pollution (natural, anthropogenic). Analysis methods have either been developed or modified from previous environmental research to detect the constituents of interest. In particular, the methodology for the detection of antibiotics in sediment and water column samples continues to evolve, as more compounds of interest are added to my “seek” list.

Sampling has proceeded every two weeks to present, with the exception of the month of August, with only one sampling event. In mid-August, I presented a poster at the national American Chemical Society Convention in Washington, D.C, where I was awarded one of the Graduate Student Travel Grants for my presented research. At the suggestion of several scientists, whom I met at the conference, I am conducting more preliminary bacterial experiments for other indicator organisms that suggest poultry contamination, as well as analyzing for pesticide degradation products.

Using GIS has also been an essential element in my current work, not only as a mapping tool but also to analyze LIDAR (Light Detection and Ranging) data to determine direction of overland flow and land use.

With the progress made this summer, I am wrapping up the sampling events for the season due to the lack of agricultural activity and the ephemeral state of some of the sampling sites. After October, sampling will resume in April, so as to catch the spring flush. Preliminary modeling and more in-depth data analysis are beginning, as are further antibiotic and bacterial investigations.

I cannot thank enough the Maryland Water Resources Research Center for their support of my project. None of this would have been possible without it, especially the developments and modifications I have been able to pursue. Thank you.

Award No. G09AP00142 Comparison and Integration of Lock Control Policies

Basic Information

Title:	Award No. G09AP00142 Comparison and Integration of Lock Control Policies
Project Number:	2009MD205S
Start Date:	7/22/2009
End Date:	1/21/2010
Funding Source:	Supplemental
Congressional District:	
Research Category:	Not Applicable
Focus Category:	None, None, None
Descriptors:	
Principal Investigators:	Allen Davis, Paul Schonfeld

Publication

1. Wang, S.L., Yang, N. and Schonfeld, P. “Simulation-based Network Maintenance Planning and Scheduling,” Transp. Res. Record 2100, July 2009, pp. 94-102.

Lock Control Policies

-

An Application of the Detailed Single Lock Simulation Model (DLSIM)

Prepared for the Institute for Water Resources
U.S. Army Corps of Engineers

by

Shiaaulir Wang
Paul Schonfeld

University of Maryland, College Park

May 2010

Table of Contents

Table of Contents 2
List of Tables 4
List of Figures 5
Introduction 6
Features of DLSIM 7
  Four-stages / three-conditions lockage operation 7
  Lockages 8
    Multi-Chamber Lockage 8
    Vessels Priorities 9
    Multi-Vessel Lockage 9
    Open Pass / Navigable Pass 10
  Control Policies 10
Lockage Policies 11
  Static Control Policies 12
    FIFO 12
    N-Up M-Down 12
    One-way 14
    Longest Queue 15
    SPF (Shortest Processing Time First) 16
    FSPF (Fairer Shortest Processing Time First) 18
  Dynamic Control Policies 20
    Definition 20
    Thresholds 20
    Example 20
Performance Measurements 21
  Total Number of Lockages 21
  Average Overall Lockage Time 22
  Average Waiting Time in Queue 23
  Average Number of Tows in System 24
  Average Queue Length 26
  Lock Utilization Rate 27
  Overall Simulation Time 27
Model Test 27
  Study Lock 27
  Input Data 28
    Lock Information 28
    Vessel Information 28
    Operation Information 32
    Processing Time Distributions 32
  Performance Curves for Different Control Policies 33
    Vessel Demand 33
    Change of Demand Level 34
    Designed Scenarios of Static Control Policies 37

Performance Curves for Different Control Policies..... 38
Comparison of Different Control Policies 43

List of Tables

Table 1 Four-Stages / Three-Conditions Locakge Component 7
Table 2 Chamber Characteristics 11
Table 3 Static Control Policies 11
Table 4 Dynamic Control Policies 11
Table 5 Definition of Policy Group for Dynamic Control Policies 12
Table 6 2007 Lock Information at Marmet 28
Table 7 2007 Vessel Information at Marmet 28
Table 8 2007 Vessel Configuration at Marmet 30
Table 9 Cut Information of Shipment List 31
Table 10 2007 Operation Information at Marmet 32
Table 11 Tested Demand Level 37
Table 12 Designed Scenarios for Static Control Policies 37

List of Figures

Figure 1 Lock Domain Configuration..... 7
Figure 2 Approach Area Interference 8
Figure 3 Gate Area Interference 9
Figure 4 Operational Logic of N-up M-down..... 14
Figure 5 Operational Logic of One-Way 15
Figure 6 Operational Logic of Longest Queue 16
Figure 7 Operation Logic for SPF Control 17
Figure 8 Operation Logic of FSPF Control 19
Figure 9 Logic for Switching Control Policies 20
Figure 10 Dynamic Policy Evaluation..... 21
Figure 11 Total Number of Lockages upon Simulation Termination..... 22
Figure 12 Four-Component Lockage Process..... 22
Figure 13 Time Frame of Vessels' Lockage Time in Lock System 23
Figure 14 Time Frame of Vessels' Waiting Time in Lock System 24
Figure 15 System Status for A Single Lock..... 25
Figure 16 Time Path of Average Number of Tows in System 26
Figure 17 Time Path of Average Queue Length..... 26
Figure 18 A Realization of Time-Average Variables 27
Figure 19 Vessels' Arrival Pattern..... 29
Figure 20 Vessels' Configurations..... 30
Figure 21 Cut Information for 2007 Shipment List 31
Figure 22 A Shipment List Input for Simulation Model..... 33
Figure 23 Logics of Changing Vessel Demand Level for Simulation Test 35
Figure 24 Change of Shipment List Input for Simulation Model 36
Figure 25 Performance Curves for FIFO 39
Figure 26 Performance Curves for N-Up M-Down 39
Figure 27 Average Queue Length for Group of N-Up M-Down..... 40
Figure 28 Performance Curves for Longest Queue 41
Figure 29 Performance Curves for SPF 42
Figure 30 Performance Curves for FSPF..... 42
Figure 31 Average Queue Length for Group of FSPF..... 43
Figure 32 Comparison of Average Time in Queue..... 44
Figure 33 Comparison of Average Queue Length (Upstream)..... 44
Figure 34 Comparison of Average Queue Length (Downstream)..... 45

Introduction

A Detailed Lock Simulation Model (DLSIM) has been developed at the University of Maryland (UMD) for analyzing any single lock in detail without involving other network models at any level. By considering the detailed lockage process which includes four stages (approach, entry, chambering, and exit) and three conditions (fly, exchange, and turnback), rather than by considering one simple processing time for overall lockage service, DLSIM is able to perform more sophisticated analysis while simulating each lockage stage in detail. With detailed analysis of lockage processes in various stages, the interferences among vessel movements at the approach and gate areas are first modeled in DLSIM. In addition, this detailed simulation model,also models multi-cut, multi-chamber and multi-vessel operations.

Beyond the above lockage details, DLSIM also models several operational control policies. Since a waterway lock is considered as a queuing system, the queue dispatching rule is quite important for operations. Among various lock control policies that have been proposed, the following six have already been modeled in the Detailed Single Lock Simulation Model (DLSIM)

a. FIFO
b. N-Up M-Down
c. One-Way
d. Longest queue (direction)
e. Shortest Processing Time First (SPF)
f. Fairer Shortest Processing Time First (FSPF)

Some of the above policies, such as FIFO, Longest Queue, One Way and N-Up M-Down, are already employed in real-world operations. Some of the modeled control policies, such as SPF and FSPF, are proposed for improving the current lockage operation. The performance under various traffic conditions can be evaluated with the developed DLSIM. Therefore, in order to help lock operators increase the system efficiency, or provide further information to decision makers, this study aims to (1) develop methods for comparatively evaluating and integrating various control policies that may be considered in the future with a full network simulation model and (2) to evaluate and integrate the lock control policies.

To our knowledge, no performance analysis has yet been conducted with DLSIM for any control policies. The effects of any operational changes depend on traffic volumes, queue lengths, lock characteristics and various vessel characteristics. An early “off line” study of system performance under different control policies, combinations of policies or transitions among policies as conditions change should enable us to identify which policies are preferable in various situations, and how the parameters of those policies should change with system conditions. If the future network simulation model is run with one predetermined policy for any given lock and year, then guidelines generated by this proposed study will help model users in pre-selecting the most effective policies. Such predetermined guidelines can also greatly reduce computation time in an optimization process relying on detailed simulation.

Features of DLSIM

Four-stages / three-conditions lockage operation

Vessels move through a lock in four stages: approach, entry, chambering and exit. Approach and exit can also be conditioned with fly, exchange and turnback based on the travel directions of the previous or following vessels. Table 1 shows the definitions of the four-stages and three-conditions in the lockage process.

Table 1 Four-Stages / Three-Conditions Lockage Components

Stage	Condition	
Approach	Fly	If there is no vessel in queue
	Exchange	If the current vessel travels with different direction from the previous vessel
	Turnback	If the current vessel travels with the same direction as the previous vessel
Entry	-	-
Chambering	-	-
Exit	Fly	If there is no vessel in queue
	Exchange	If the current vessel travels with different direction from the previous vessel
	Turnback	If the current vessel travels with the same direction as the previous vessel

Figure 1 logically shows a conceptual configuration of a general lock domain. A lock domain is defined between two approach points. Vessels start and end their lockages from and at the approach points. For turnback conditions, two extra stages, pre-approach and post-exit, are represented to turnback approach and turnback exit. For those lockage stages, four standards and two extras, different time increments should be applied. If multiple cuts are required for vessels, then multiple entries, multiple chamberings and multiple exits are performed.

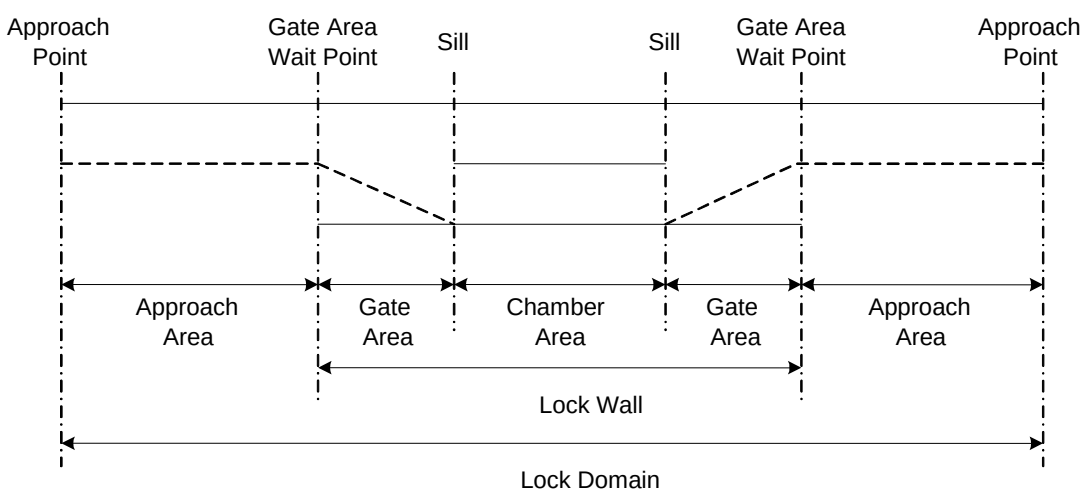


Figure 1 Lock Domain Configuration

Lockages

Multi-Chamber Lockage

1. Interference

Some physical interference between vessels is observed at multi-chamber locks (Lisney, 2005). Such lock interference actually compels a vessel to wait while another vessel, using the other chamber, finishes an action, even though the chamber intended for the waiting vessel is ready for service. Recreational craft and light boats cannot cause and are not affected by interference.

Interferences can be classified into two categories: approach area and gate area interferences. Each of them may incur extra waiting time for certain vessels. Approach area interference considers lockage at the two-chamber locks as passing through a series of "single-server" approach area, "two-server" chamber area, and another "single-server" approach area (as shown in Figure 2). After a vessel served at a lock completes its chambering, and while it exits and occupies the approaching area in the conflicting direction, another approaching vessel cannot start its lockage even if the other chamber is idle. That is, if the subject vessel and the next travel in the same direction, the approach area for the next vessel is clear and no interference occurs.

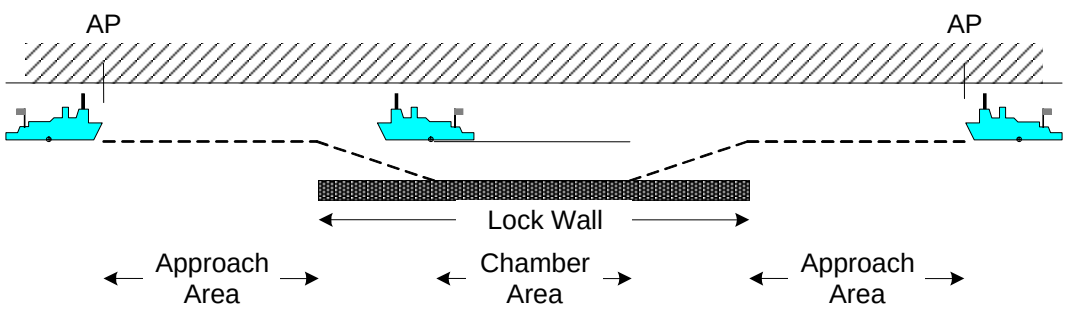


Figure 2 Approach Area Interference

Gate area interference occurs when vessels are in the entry process and arriving at the gates, or while they are assembled or disassembled into cuts (as shown in Figure 3). If the breaking cuts are waiting outside the gate area or arriving vessels are entering the gate, a vessel finishing in another chamber cannot start exiting unless the remaining space in gate area is large enough for both vessels to pass through.

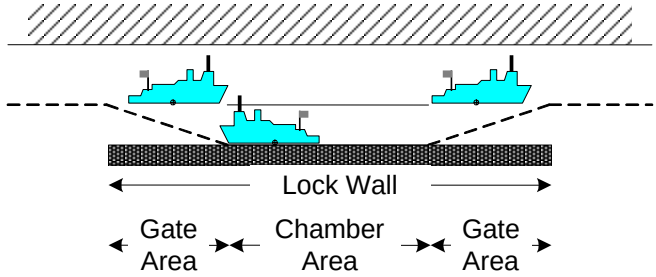


Figure 3 Gate Area Interference

2. Preference and Exclusion

Preference is mainly related to the chamber assignment. Although the main and auxiliary chambers are both available when a vessel is coming, the lock operators may assign the specific chamber to certain kinds of vessels (e.g. recreation crafts are often assigned to the auxiliary chamber).

Sometimes the width of the vessel would exceed that of one of the chambers, and reconfiguration is required for the vessel so that it can fit into the chamber. However, the multi-cut operation after the reconfiguration may involve the chamber exclusion logic due to the limitation of cuts. The chamber exclusion usually applies in the auxiliary chamber.

Another exclusion rule determines whether a particular vessel is allowed to use particular types of chambers (e.g. recreation craft are not allowed touse the main chambers).

Vessels Priorities

There are 6 types of vessels designed in the DLM: tows, lightboats, recreational craft, passenger vessels, government vessels, and other vessels. Generally, n terms of priority, those 6 types of vessels can be categorized into 3 groups: (1) high priority vessels including passenger and government vessels (H vessels), (2) tows, lightboats, and other vessels, and (3) recreational craft. The H vessels have the highest priority and are sent to the front of the queue upon their arrivals. Usually, recreation craft are expected to wait for up to three commercial lockages. Then a lockage is dedicated to recreation craft. When there is a recreation lockage, all the recreation craft in the queue should, if possible, be locked together regardless of how many commercial lockages other recreation craft have already waited. Tows, lightboats, and other vessels are treated similarly and processed with whatever control policy is in effect. Small vessels are highly likely to be included in multi-vessel lockages.

Multi-Vessel Lockage

Multi-vessel lockages are those lockages in which two or more commercial vessels are served in a single lockage cycle. Only commercial vessels are considered in defining multi-vessel lockages; the definition does not apply to recreation craft, government and commercial passenger vessels. A typical multi-vessel lockage serves two small tows which fit at the same time in a large chamber. In addition, a tow moving a

hazardous commodity cannot participate in multi-vessel lockages. Generally, a significant distance, say 100-200 feet, is needed between the vessels while they are in the chamber. If the queue is very long, say 25 vessels in each direction, the algorithm should not search through the entire queue looking for small vessels to make up multi-vessel lockages. There should also be a feature which allows the user to disallow multi-vessel lockages for each chamber. Based on empirical data, the model must have a provision which allows the user to specify the time added to the entry and exit portions of the lockage for each additional vessel.

Open Pass / Navigable Pass

Open pass and navigable pass logic will be implemented in the DLM. Some locks, such as L&D 52 and L&D 53 on the lower Ohio River have movable wicket dams; others have relatively low fixed crest dams. These dam types afford vessels the opportunity to move past a lock site without actually locking through the lock chambers. They pass by the lock by navigating over the dam, hence the term navigable pass. Whether a vessel can pass a lock using navigable pass or must lock through the chambers depends upon water levels. The lock may be in navigable pass for weeks on end, or it may alternate between locking and navigable pass several times in one week. Historic LPMS/OMNI data can provide statistics which describe which times of the year navigable pass is likely to occur, and how long is it likely to last. The model must have the ability to place a lock in navigable pass mode. When a lock is in navigable pass mode, processing time distributions will be needed for upbound vessels and for downbound vessels. If vessels are in queue, they will be removed from queue based on either a FIFO or N-Up M-Down policy, or other control policies.

Control Policies

In addition to baseline control policy of the FIFO, there are other lockage policies available in practice, as well as in designed DLSIM, including:

1. N-Up M-Down;
2. One-Way;
3. Longest queue (direction);

Two additional control policies are modeled. SPF (Shortest Processing Time First) has already been discussed in at least one simulation text book (Law and Kelton, 2000). It provides a way of re-sequencing the queue. Re-sequencing may affect the perceived fairness of a control policy and that fairness should be taken into account. Some studies (Ting and Schonfeld, Wang and Schonfeld) have been conducted to comparatively evaluate the performance of FCFS and SPF at single waterway lock or in a waterway network. A fairer SPF (FSPF) has also been proposed to consider a fairness constraint (Ting and Schonfeld, Wang and Schonfeld).

Lockage Policies

As chamber specific characteristics, two types of control policies are considered in DLSIM (as shown in Table 2). One is static control policies which are fixed over time. The other group includes dynamic control policies based on the congestion level (e.g., queue lengths or waiting time) at the lock as the simulation progresses.

Table 2 Chamber Characteristics

tblChamber : Table

ChamberID	LockID	ChamberDesc	Main	Length	Width	CutLimit	ControlPolicyID	DynamicControl
38	54	Marmet	☑	360	56	5	1	☑
84	54	Marmet	☐	360	56	1	6	☐
0	0		☐	0	0	0	0	☐

Table 3 lists the defined static control polices. Policies with different parameters, though with the same policy name, are labeled with different control policy ID. For instance, three N-up M-down policies are listed with their own UpCount and DownCount parameters.

Table 3 Static Control Policies

LockPolicyID	LockPolicy	Direction	UpCount	DownCount	LockPolicyDescription	FairnessValue
1	FIFO	0	0	0	FIFO	0
2	N-Up M-Down	0	3	3	N-Up M-Down	0
3	N-Up M-Down	0	6	6	N-Up M-Down	0
4	N-Up M-Down	0	12	12	N-Up M-Down	0
5	One Way	1	1	0	One Way	8
6	One Way	2	2	0	One Way	8
7	Longest Queue	0	0	0	Longest Queue	0
8	SPF	0	0	0	Shortest Processing Time First	0
9	FSPF	0	0	0	Fairer Shortest Processing Time First	5
10	FSPF	0	0	0	Fairer Shortest Processing Time First	7
NULL	NULL	NULL	NULL	NULL	NULL	NULL

Table 4 and Table 5 define the dynamic control policy which include a group of static control policies. The thresholds triggering policy switch are user defined. For example, during low congestion, the policy will probably be FIFO. Then, as congestion and queue lengts rise to say 6 tows in one direction, the policy will switch to 3-up 3-down. If congestion rises further, the policy may change to 6-up 6-down. Then, when congestion decreases, the control policy may switch back to FIFO. The queue lengths that trigger such changes in control policy will be data inputs.

Table 4 Dynamic Control Policies

	LockPolicyDyna...	LockPolicyGrou...	LockPolicyID	ThresholdType	QueueTH	TimeTH
▶	1	1	1	Q	0	0
	2	1	2	Q	4	0
	3	1	3	Q	7	0
	4	1	4	Q	13	0
	5	2	1	Q	0	0
	6	2	2	Q	5	0
	7	2	3	Q	8	0
	8	3	1	T	0	0
	9	3	8	T	0	100
	10	4	1	T	0	0
	11	4	8	T	0	100
	12	4	9	Q	7	0
*	NULL	NULL	NULL	NULL	NULL	NULL

Table 5 Definition of Policy Group for Dynamic Control Policies

	LockPolicyGrou...	LockPolicyGroupDesc
▶	1	FIFO / 3-level N-up M-down
	2	FIFO / 2-level N-up M-down
	3	FIFO / SPF
	4	FIFO / SPF / FSPF
*	NULL	NULL

Six control policies, with their static or dynamic features, are modeled in DLSIM. Those control policies are only applied to tows / other vessels, not recreational or high priority vessels. Each control policy has its own assumptions and specific operation rules. More details are provided below.

Static Control Policies

FIFO

First in, first out (FIFO) is a most common service policy in inland waterway network. FIFO is viewed as the fairest control policy which locks vessels based on their arrival order, without any service preference. Whenever an available chamber looks for next serving vessel, the earliest arriving one is always chosen. Chamber turnback is necessary if the chosen vessel travels in the same direction as the previous served vessel.

N-Up M-Down

The N-Up M-Down notation represents the serving sequence of the waiting queues. All iterations should be completed by serving N and M vessels from upbound and downbound directions, respectively. If the starting direction is downbound, the system will try to look for M vessels, one by one, from downbound queue with earliest arrival time that satisfies cut limits as the next vessel. It should noted that N-up M-down

is not multi-vessel lockage. N or M vessels are served individually, multiple chamber turnback times.

A few assumptions are made about the N-Up M-Down control policy:

(1) Once we decide to use the N-Up M-Down control policy, the starting direction (i.e. up or down) is initialized by the first arriving vessel when the chamber is idle.
(2) The N-Up M-Down policy only applies to tows and other vessels. Passenger / government vessels and certain specific recreational crafts still have the priority to use the available chamber.
(3) For 3-Up 3-down, it is ok to lock vessels with 3-Up 1-down, 3-Up 2-down, 1-Up 3-Down, or 2-Up 3 down since one direction might not have that long queue compared with other direction. The policy automatically ends when the queue in any direction has dissipated.

The basic logic of N-up M-down is shown in Figure 4.

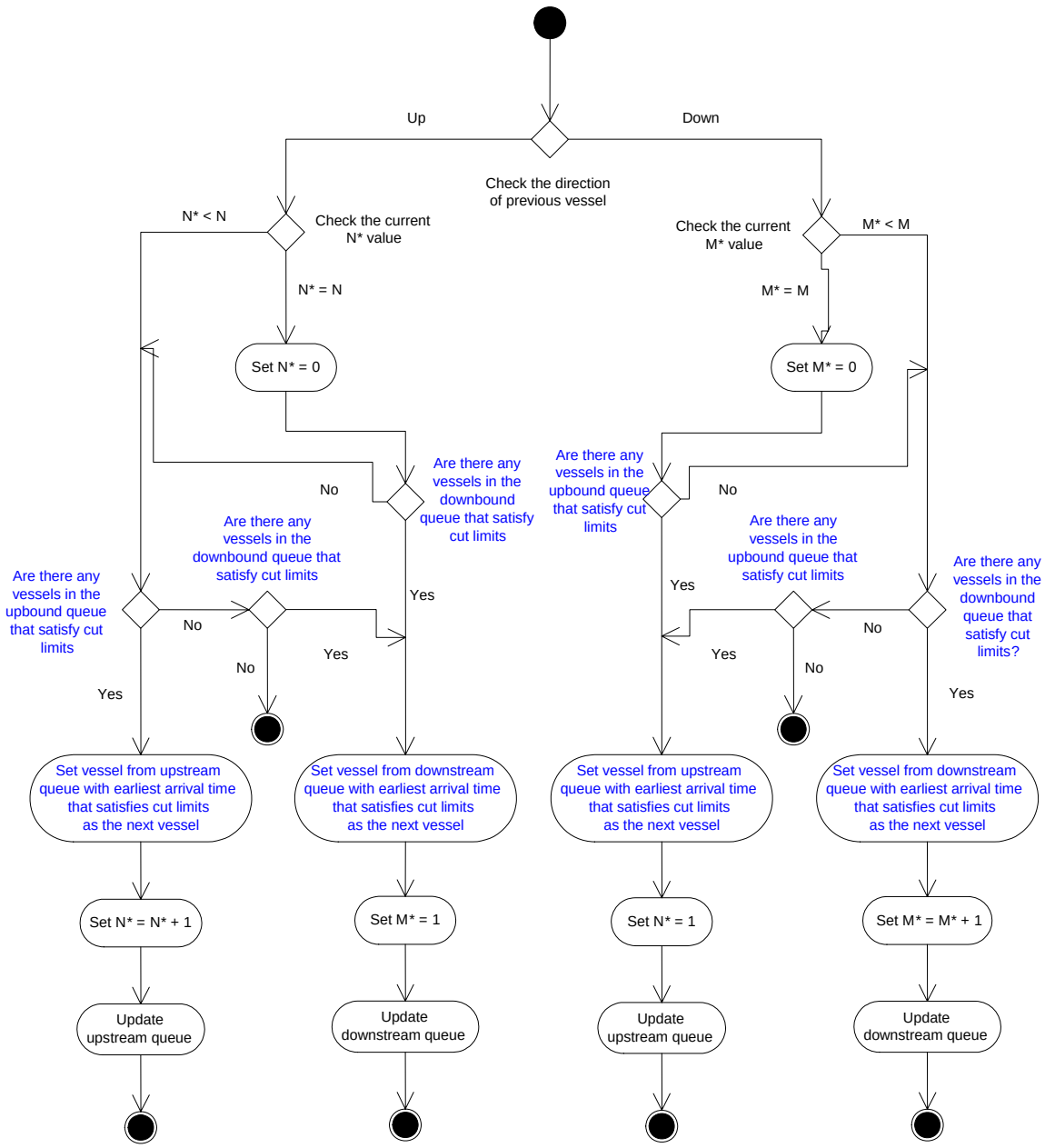


Figure 4 Operational Logic of N-up M-down

One-way

In the one-way policy upbound and downbound traffic are each served by different chambers. Each chamber is uniquely assigned to one direction. Thus, a single chamber lock cannot have a one-way policy.

The control policy for one-way operation is similar to 'one direction FIFO'. The overall logic is also similar to FIFO while dealing with traffic from one direction. This preferred direction is pre-determined among chambers. There could be a fixed one-way policy or preferred one-way policy. That is, if no vessel is waiting in the preferred

direction or the queue is very unbalanced directionally, the available chamber can temporarily switch to serve the opposite direction, until the next vessel from the preferred direction arrives. Figure 5 shows the logic for the one-way policy.

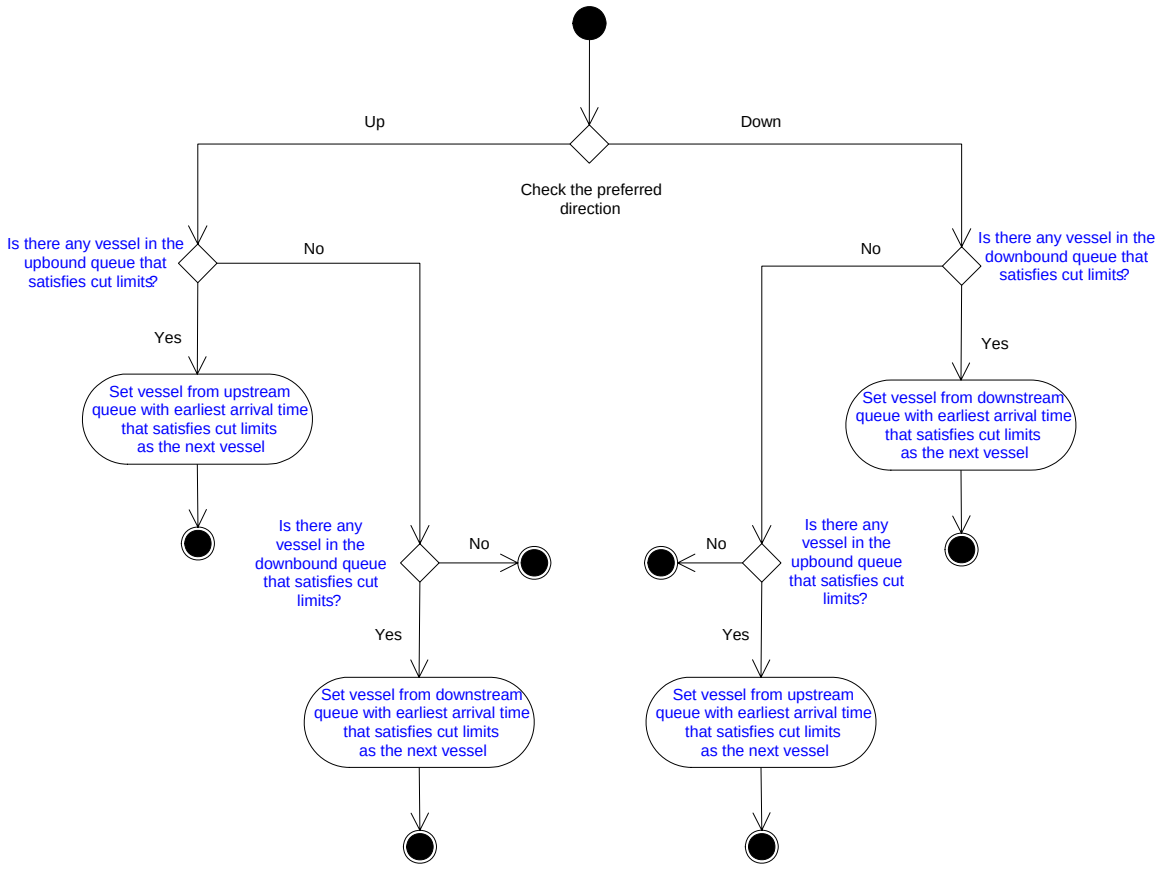


Figure 5 Operational Logic of One-Way

Longest Queue

With the longest queue policy, the next vessel is selected from the direction with the longest waiting queue. Similarly, the overall logic is the same as the FIFO, but only based on the longest queue.

At beginning, the lockage operator should check the lengths of all queues. Once the longest queue direction has been determined, the earliest vessel waiting in this queue that satisfies the cut limits constraint is assigned to the available chamber. If none of the vessels in the longest queue satisfy the cut limit restriction, the available chamber will look for the next vessel from the opposite direction. If the queue lengths in both directions are identical, the chamber will look for the earliest vessel satisfying the cut limit constraint. The basic logic of longest queue policy is shown in the Figure 6.

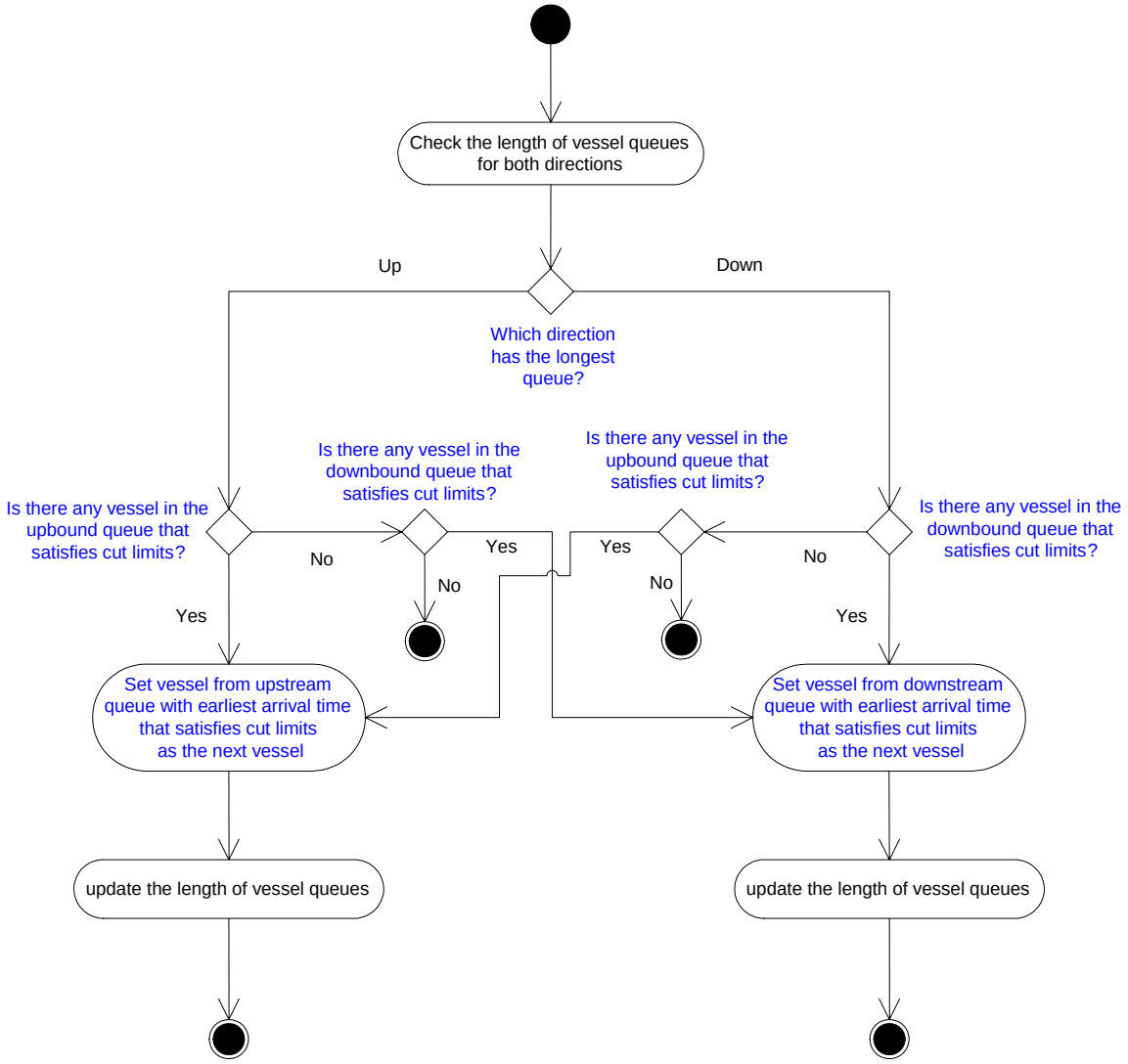


Figure 6 Operational Logic of Longest Queue

SPF (Shortest Processing Time First)

Based on the definition, Shortest Processing Time First operation might select the tow based on the "average service time tow". The tow with the lowest service time (usually the smallest tow) has the first priority to be processed. However, with different tow sizes, measuring the service or delay times per barge should be better than per tow due to the size variations. Thus, the SPF in current model is designed to assign the tow with minimum service time per barge (i.e. usually the largest tow) rather than the minimum service time per tow (i.e. the smallest tow). The SPF operation logic is shown in Figure 7. The SPF factor is calculated based on the average service time per barge.

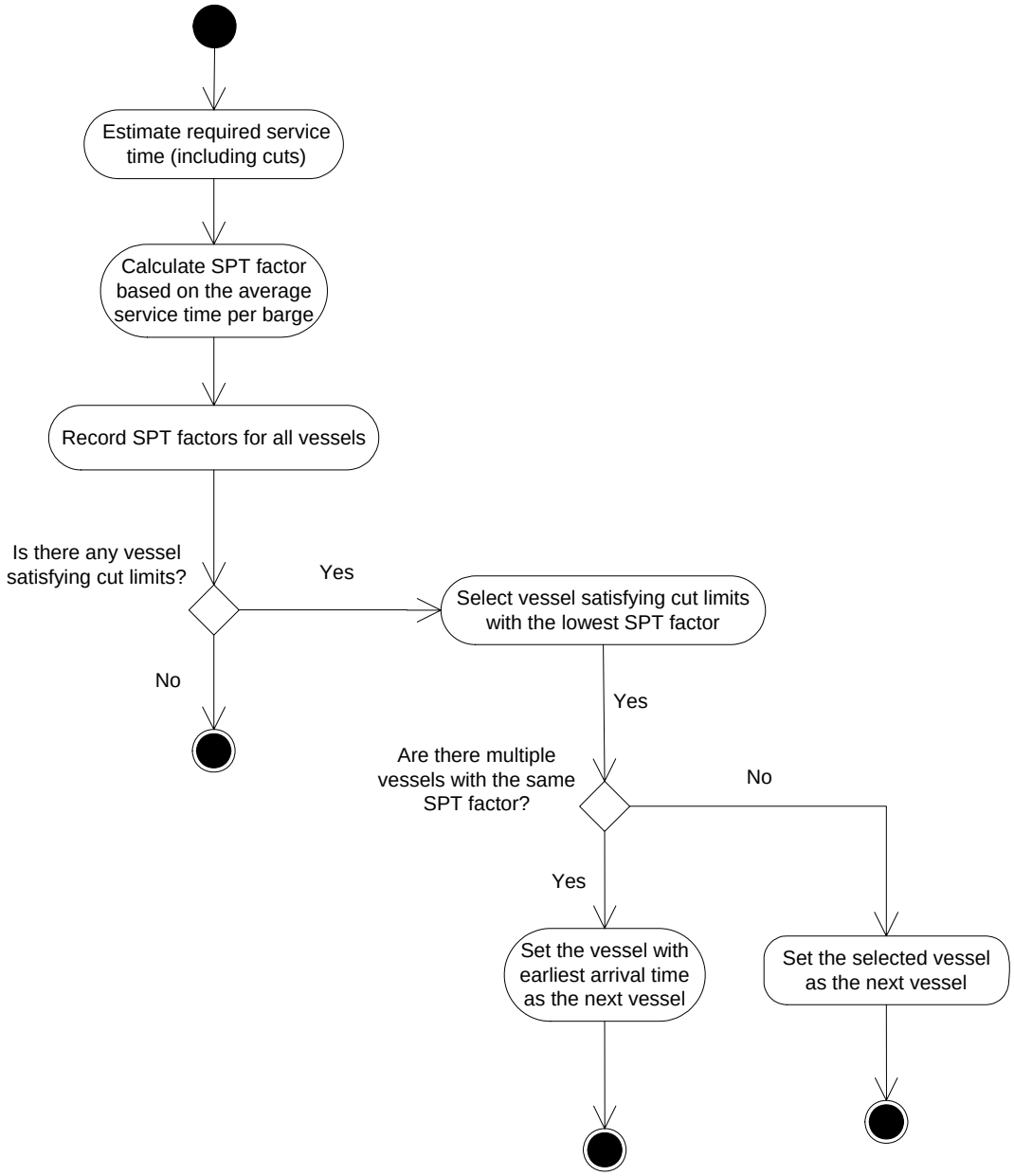


Figure 7 Operation Logic for SPF Control

The SPF assumptions are as follows:

1. Shortest processing time records the 'shortest service time per barge' and assigns the available chamber to this tow; however, all exclusive rules should be considered.
2. For the same number of cuts, a tow with more barges has higher priority than one with fewer barges.
3. For different numbers of cuts, we need to estimate the total incremental chamber turnback time for tows with more cuts; furthermore, we can calculate the average service time per barge for each tow and determine which tow should be served next.

4. If there are multiple vessels with the same SP time, then we select the next vessel based on the FIFO operation rule.

In the current DLM, the processing time is determined by the number of cuts. Therefore, the tow with the fewest cuts has the highest priority with the SPF policy.

FSPF (Fairer Shortest Processing Time First)

From the system point of view, SPF can reduce total system delays through the pre-specified dispatching priority. However, large tows with more cuts may experience greater delays with SPF than FIFO. In order to balance the system efficiency and fairness among individual tows, FSPF is proposed as an intermediate policy between SPF and FIFO. FIFO is a fairer SPF control policy that gives priority to tows which have waited for a certain number of lockages (F*) based on the SPF rules. F* is the fairness value pre-defined in the input table. Different fairness values will influence the system performance. In general, if F*decreases, FSPF will be quite similar to FIFO. However, if F*increases, FSPF will be close to SPF.

Furthermore, the average number of tows in the queue is a significant indicator for evaluating the system. FSPF with lower fairness value gives larger tows more chances to leave the waiting queues while they keep being passed by smaller tows. Also, in the contrast, the system will be more efficient with lower delays and shorter queues based on the higher F*. All other assumptions of FSPF are similar to those of SPF. The FSPF operation logic is shown in Figure 8.

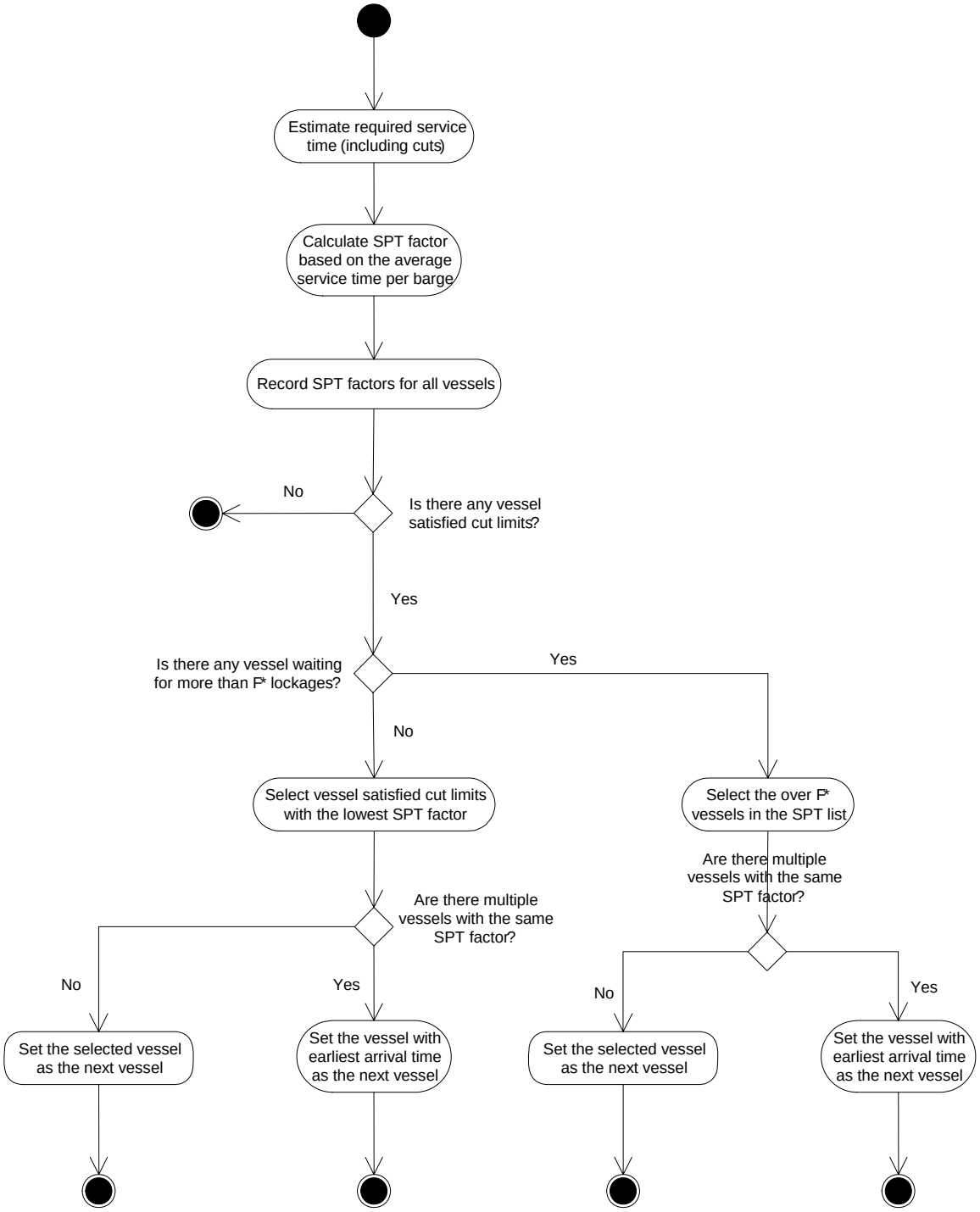


Figure 8 Operation Logic of FSPF Control

Dynamic Control Policies

Definition

In control problems dealing with queuing systems, the "switch"-form dynamic control policies usually be considered as a more effective strategy. A dynamic control policy can automatically switch between regular static queue disciplines when one or several levels of service (LOS) reach pre-determined thresholds. The selection of a proper level of service and determination of the value of thresholds are also optimization problems which need further studies. Thus the LOS and the thresholds should be input items in current phase.

The current DLM supports dynamic control policies which can switch policies in real-time according to the current total queue length or the current time period.

Thresholds

As shown in the Figure 9, a dynamic control policy consists of transitions among several static control policies (Policy *i*). The control kernel selects the corresponding static policy based on the evaluation of the current LOS (level of service) and pre-determined thresholds (T_i).

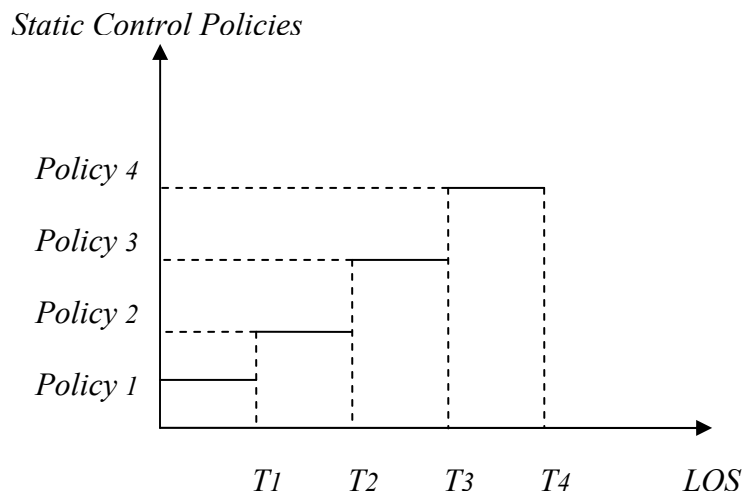


Figure 9 Logic for Switching Control Policies

Example

Figure 10 demonstrates an example of dynamic N-up M-down control policy, where

(1) T_1 is the threshold value between FIFO and "3-up 3-down," T_2 is the threshold value between "3-up 3-down" and "6-up 6-down," and T_3 is the threshold value between "6-up 6-down" and "12-up 12-down."

(2) Among FIFO, "3-up 3-down," "6-up 6-down," and "12-up 12-down" can change to others through a 'reevaluation' procedure which measures the current LOS.

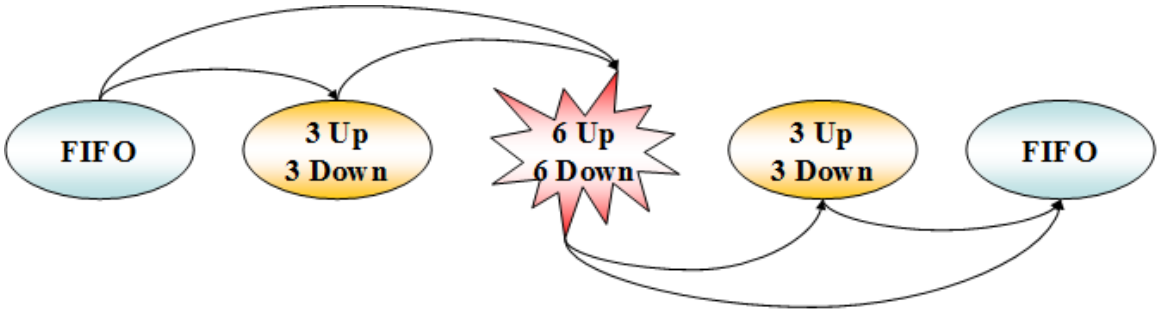


Figure 10 Dynamic Policy Evaluation

Performance Measurements

Various types of performance measurements are used to evaluate lock operations. Usually the most relevant ones for lock operator as well as customers (e.g. tows or vessels) are the total number of completed lockages, lock utilization rate, the average time spent in lockage, average time spent in queue and the average number of vessels waiting in queue. In DLSIM, some measurements, such as number of completed lockages (*TOL*), average lockage times (*ALT*) and average waiting times in queue (*AQT*), are directly calculated by accumulating the values at the times whenever there are events of vessels arriving at or leaving locks. However, some measurements are relevant time-average data, such as the average number of tows in the system *Q(t)*, the average number of tows in the queue *q(t)* and the lock utilization rate *U(t)*, which should be updated over the simulation time by updating the areas under the functions needed for those continuous-time statistics. The overall simulation time could also be a performance measurement. It simply implies the relation between the traffic level and the required time for running the simulation.

Total Number of Lockages

The number of completed lockages is updated when a vessel is leaving a lock. If the simulation is terminated after a fixed running period (e.g., 10,000 hours), it is possible at the end of simulation to have more arrival vessels then departing vessels due to the vessel queue in system. However, if the simulation is terminated after locking the last arrival vessel, the number of completed lockage should be the number of arrivals with no vessels left in queue. Figure 11 shows how the number of lockages is counted along the time axis. A and L indicate the arrival and departure time, respectively, for each single vessel. If the simulation is terminated at T_2, all the vessels will be locked and number of lockages is counted as number of arrived vessels. If the simulation is terminated at T_1, the number of lockages should be less than the number of arrivals.

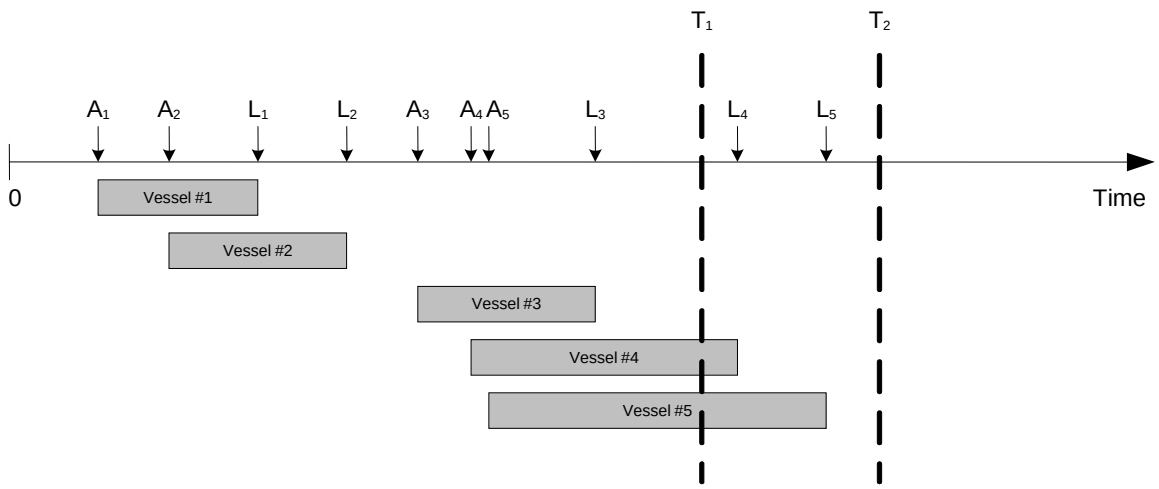
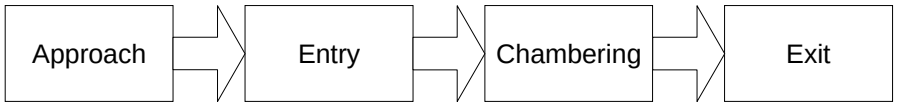


Figure 11 Total Number of Lockages upon Simulation Termination

Average Overall Lockage Time

Average overall lockage time is updated at the end of each lockage. It is calculated by summing each vessel's lockage time and averaging with the total number of lockages counted at the end of simulation.

As shown in Figure 12, DLM features a detailed lockage process of four stages and three conditions. After approaching, vessels physically go through steps of entering, chambering, and exiting to end the lockage. Therefore, the entire time frame for overall lockage process can be divided into four separate processes in series which provide those locking stages of approach, entry, chambering, and exit. The time points for separating those lockage processes are recorded in LPMS data as SOL (start of lockage), BOS (bow over sill), EOE (end of entry), SOE (start of exit) and EOL (end of lockage). Most of the time, those four processes are consecutive steps and their time components could be calculated based on the difference between those time points.



SOL (Start of Lockage)
BOS (Bow Over Sill)
EOE (End of Entry)
SOE (Start of Exit)
EOL (End of Lockage)

Approach Time = BOS - SOL
Entry Time = EOE - BOS
Chambering Time = SOE - EOE
Exit Time = EOL - SOE

Figure 12 Four-Component Lockage Process

Figure 13 shows an example of time frames for each arriving vessel in a lock system if the FIFO control policy is in effect. The overall lockage time for each vessel is calculated by its SOL and EOL regardless of how much time vessels spend for each lockage component. If the chamber is available when a vessel arrives, the vessel's SOL is its arrival time. If the chamber is not available when that vessel arrives, the vessel's SOL is previous vessel's EOL.

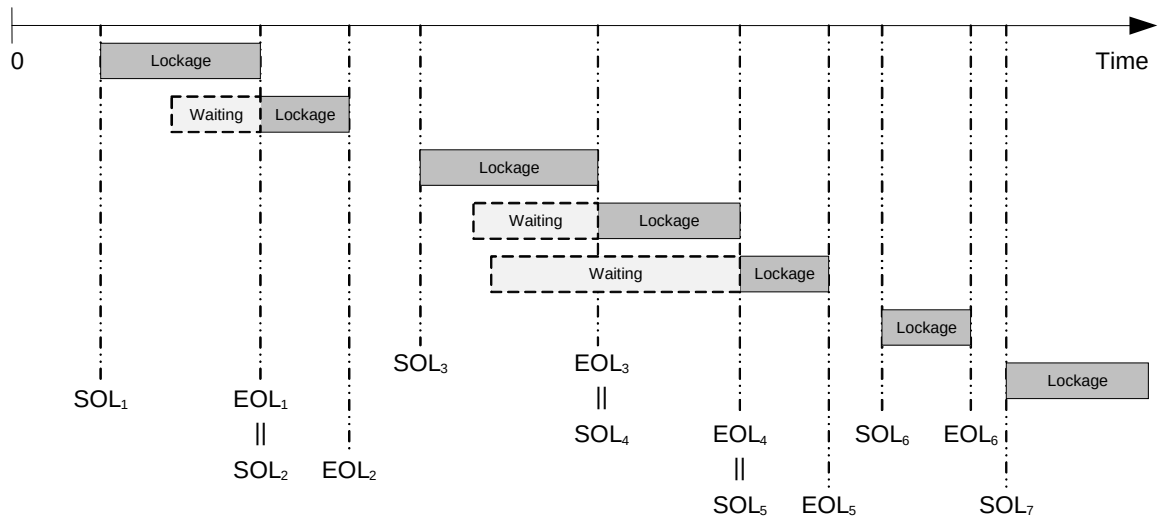


Figure 13 Time Frame of Vessels' Lockage Time in Lock System

The average overall lockage time (ALT) is then calculated from sum of total lockage time divided by number of vessels.

$$ALT = \sum_{n} (EOL_i - SOL_i)/n$$

Average Waiting Time in Queue

Waiting time in queue for each vessel is calculated at the start of lockage time (SOL). The arrival time (A) for each vessel is recorded when that vessel reaches the approach point. If the chamber is available at that time, the vessel starts the lockage immediately upon its arrival. There is no queue and waiting time is zero. However, if the chamber is not available at that time, vessels are staying in the queue and wait to be called for lockage. The time a vessel spends in queue is then the difference between the arrival and the start of lockage times.

Figure 14 shows an example of time frames of vessels' stay in a lock system. If the FIFO control policy is in effect, upon arrival, vessels #1, #3, #6 and #7 receive lockage service immediately due to chamber's availability. That is, their arrival time is also their SOL. However, vessels #2, #4 and #5 start their lockages upon vessel #1, #3 and #5's EOL, respectively, and thus experience waiting time in the queue.

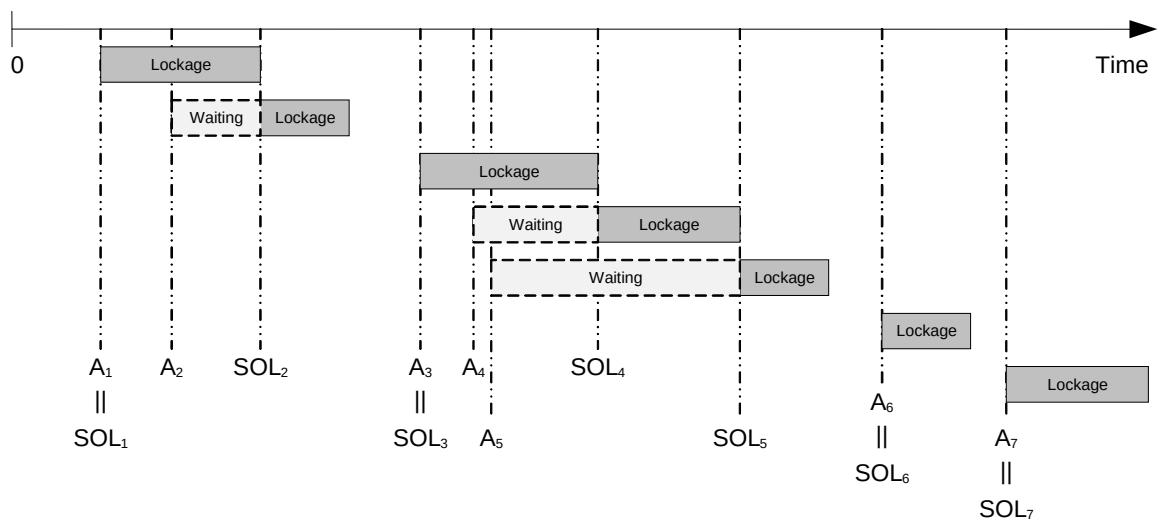


Figure 14 Time Frame of Vessels' Waiting Time in Lock System

The average waiting time (AQT) in the queue is then computed as $\sum(SOL\text{-}A)$ divided by the total number of vessels.

$$AQT = \sum_{n} (SOL_i - A_i)/n$$

It should be noted that the average waiting time in queue is a system-wide variable. It is based on the total number of vessels in the system, rather than the number of vessels which are really in queue. Even though there are vessels receiving lockage service immediately upon their arrivals, their zero waiting times are recorded and accumulated for calculating the average waiting time in the queue.

In addition, in DLSIM, there are other possible waiting among various components of lockage process due to the requirement of chamber turnback and operationally considered gate area interference. However, those extra times are not really spent waiting in the queue, but included in the total lockage time. Thus, average waiting time in queue is calculated by summing each vessel's actual waiting time in queue and averaging with the total number of lockages counted at the end of simulation.

Average Number of Tows in System

Number of tows in system is updated at the time points when there is an arriving vessel or a departing vessel. It is defined as the summation of tows in service and tows in queue. Since it is a time-relevant variable, system status (e.g. number of tows in system) alone the time frame (e.g. each time segment) is considered. An example of system status is explained in Figure 15. The number of tows in the system changes over time, e.g., 0 tow in system, 1 tow in system, 2 tows in system, 3 tows in system, etc., when there is an arrival or a departure. Each status (e.g. number of tows in system) lasts a specific period (e.g., time t) until there is new arrival (e.g., system status changes from 2 tows in system

to 3 tows in system) or a departure (e.g., status changes from 2 tow in system to 1 tow in system).

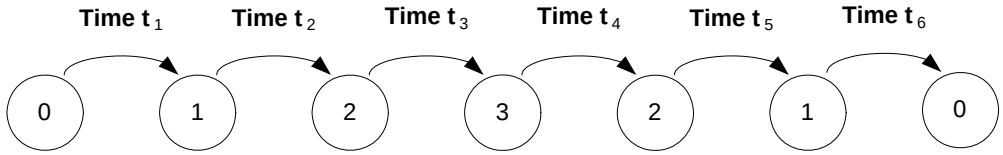
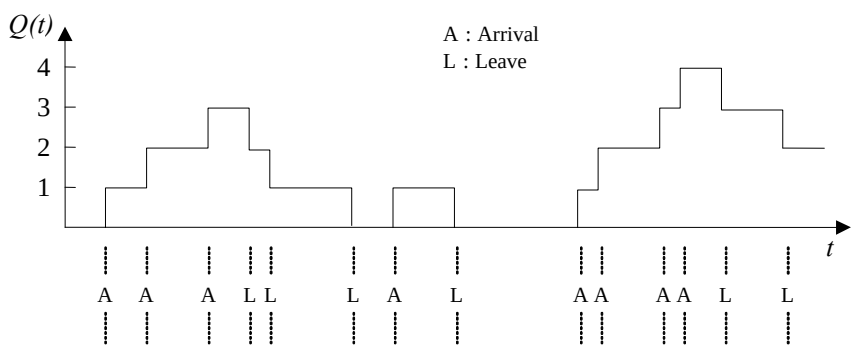
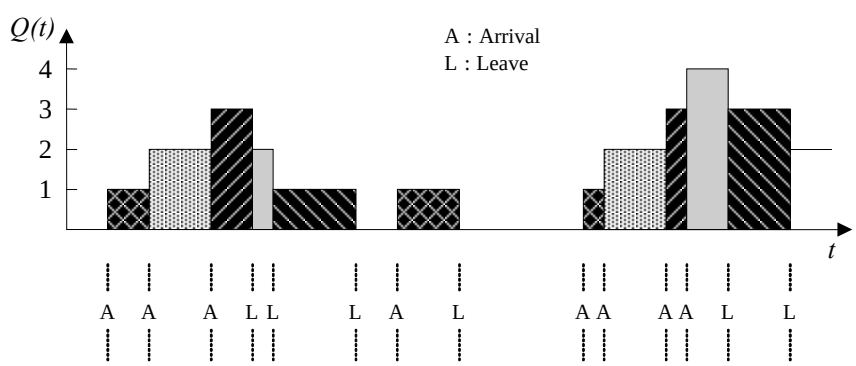


Figure 15 System Status for A Single Lock

Based on the change of system status, Figure 16(a) illustrates a simplified time path of number of tows in system *Q(t)* for a single-chamber waterway lock system. Two major arrival and leave events at locks are indicated by A & L at each discrete time point. Along the time axis, the number of tows in system is increased by one when there is an arrival event, and decreased by one when there is a leave event. The number of tows in the system could be down to zero if all the vessels are served before any new arrivals. If the FIFO control policy is in effect, the arrival event for the 1st vessel is followed by the 1st leave event, the arrival event for the 2nd vessel is followed by the 2nd leave event,…, etc (as indicated in figure). The time between the arrival and leave events is the time the vessel spends in the system, including lockage service time and possible waiting time. The change in the number of tows in a single lock system, including waiting tows *q(t)* outside of lock chambers ($q(t) = Q(t) - 1$).



(a)



(b)

Figure 16 Time Path of Average Number of Tows in System

As time-average data, average number of tows in system is calculated based on the area under the time path (as shown in Figure 16(b)) divided by the overall simulation time.

$$Q(t) = \sum_n (Q_i \times t_i)/T$$

Average Queue Length

In a simple one-server queuing model, the average number of customers in the system *Q(t)* simply equals the average number of customer in queue *q(t)* plus one tow in service. Therefore, average queue length is simply calculated by $q(t) = Q(t) - 1$. DLSIMsimulates a single lock system with two-way traffic. There could be a single measure of average number of tows in system, but two measures of average queue length for the two directions andupstream. Similarly for the average number of tows in system, both performance measurements of average queue length are time-average data and should be calculated with a similar time path, as shown in Figure 17.

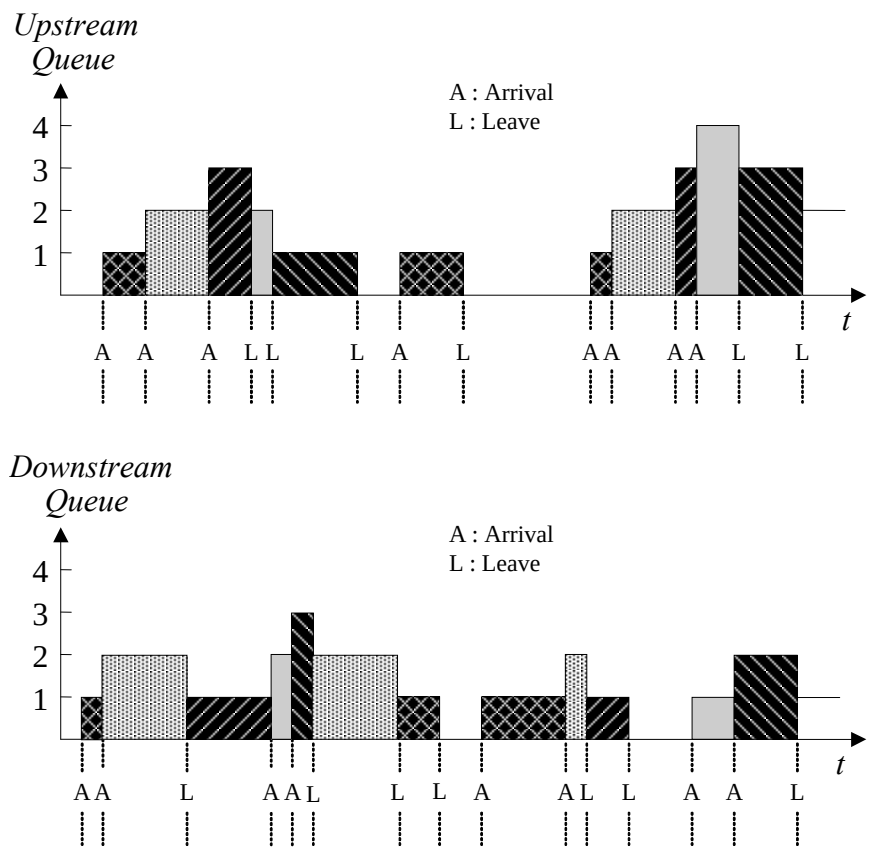


Figure 17 Time Path of Average Queue Length

Similarly, as time-average data, the average number of tows in queue is calculated based on the area under the time path (as shown in Figure 17) divided by the overall simulation time.

$$q(t) = \sum_{m}(q_i \times t_i)/T$$

Lock Utilization Rate

Figure 18 shows the chamber status with 0 (= idle) and 1 (= busy). The area under the functions *Q(t)* and *U(t)* is accumulated from all the rectangular areas ($Q(t) \cdot \Delta t$ and $U(t) \cdot \Delta t$) during the interval between the current time and the previous event time.

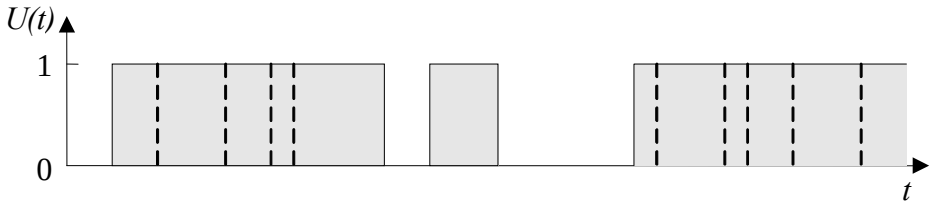


Figure 18 A Realization of Time-Average Variables

Overall Simulation Time

Overall simulation time in this study is defined as the total time required for completing all the lockages of each arriving vessel. That is, at the end of simulation, there is no vessel in the system or waiting in queue. Under normal operating conditions, the greater the traffic, the longer the required simulation time. However, the simulation time will grow exponentially, not linearly, especially when the system is approaching to the capacity.

Model Test

Currently, DLM is designed to run individually for a single-lock system, as well as serve as a module which will be integrated into network simulation model, BasinSym. In order to check all the designed features in DLM, a model test is performed to ensure the correct logic and expected test results.

Study Lock

The Marmet Lock on the Kanawha River, a tributary of the Ohio River, is selected for the test purpose. It is a two-chamber lock located between the London upstream lock and the Winfield downstream lock. From the network definition of NaSS, a single-lock system is designed as a lock reach, which differs from a regular reach, with two nodes at its ends.

Input Data

For integration purposes, DLM and NaSS share the same data structure. Therefore, an SQL database, "BasinSym", which is used by NaSS, is used for testing the DLM with specific information for Marmet.

Lock Information

The current information about the Marmet Lock is shown on Table 6. The IDs used in network configuration are defined in NaSS.

Table 6 2007 Lock Information at Marmet

Lock Characterstics		
Lock ID		54
Reach ID		212
Upstream Node ID		213
Downstream Node ID		214
Number of Chamber		2
Chamber ID	**Main**	83
	Aux	84
Chamber Dimension	**Main**	360×56
	Aux	360×56

Vessel Information

The current shipment list for the Marmet Lock, provided by DAPP, has one year of shipment data for year 2007 (as shown in Table 7). There are 6653 vessels in total with 6601 commercial trips and 52 recreational trips in the data for the year 2007. In the given shipment list no high priority vessels, such as government vessels and passenger vessels, are recorded at the Marmet Lock in the year 2007.

Table 7 2007 Vessel Information at Marmet

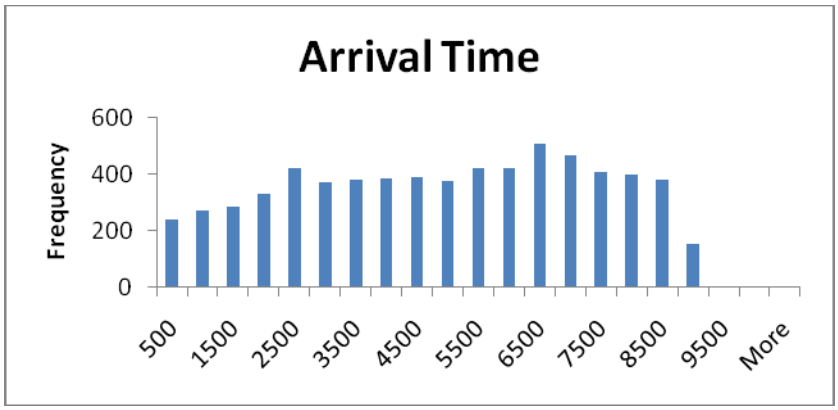
Vessel Type	Total	Upstream	Downstream
Tows (Commercial Vessels)	6601	3300	3301
Recreational Vessels	52	29	23
High Priority Vessels	0	0	0
Other Vessels	0	0	0

Since lock control policies are only applied to the commercial vessels, rather than recreational vessels or high priority vessels, the recreational vessels are screened out from the shipment data. A total of 6601 commercial vessels are tested in the base case for various control policies.

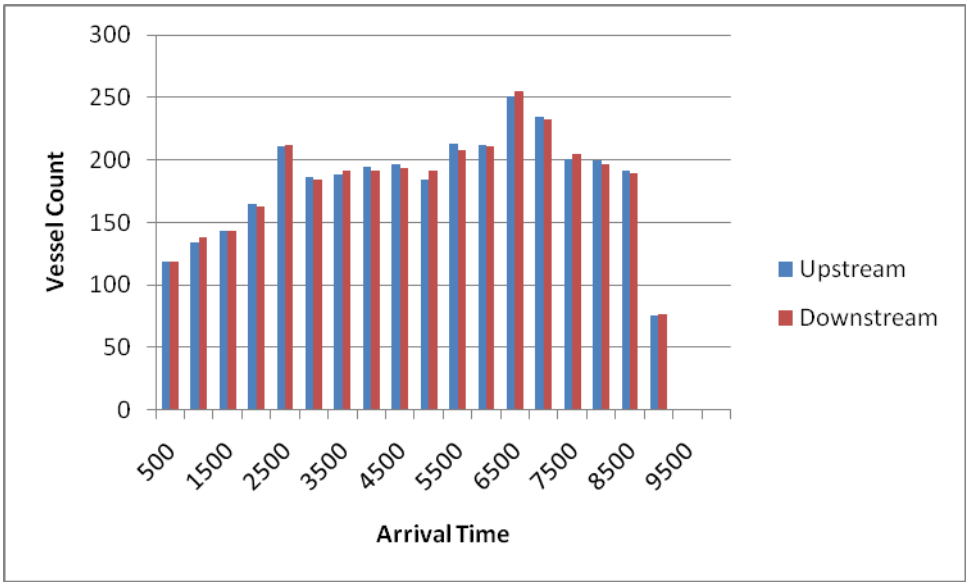
- Arrival pattern

  Figure 19 shows some detailed vessel information in the prepared shipment list. With a year-round data (6601 vessels in total 8760 hours, shown in

Figure 22(a)), there are two small arrival peaks around 2500 hours and 6500 hours, i.e., approximately in April and September, respectively. Traffic is still running during the winter season (ex. Dec. to Feb.) and some seasonal demand variations can be seen in the histogram chart with higher traffic in spring, summer and fall (hour 2000 to hour 8500). In addition, vessels show similar arrival patterns for both upstream and downstream (as shown in Figure 19(b)) with same peak arrivals and seasonal variations. Overall at Marmet in 2007, traffic flows are balanced in the two directions.



(a)



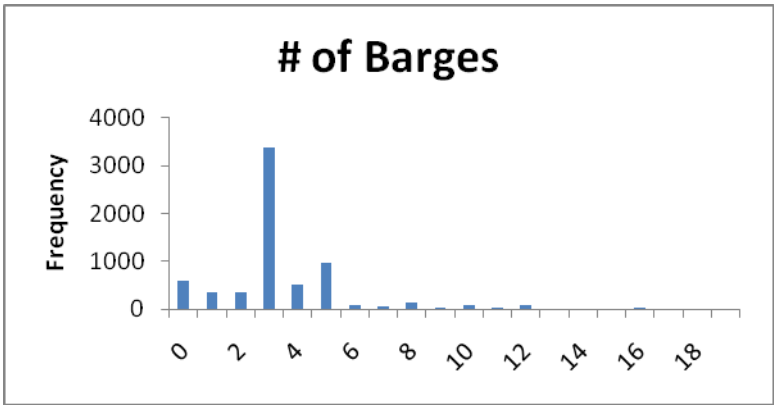
(b)

Figure 19 Vessels' Arrival Pattern

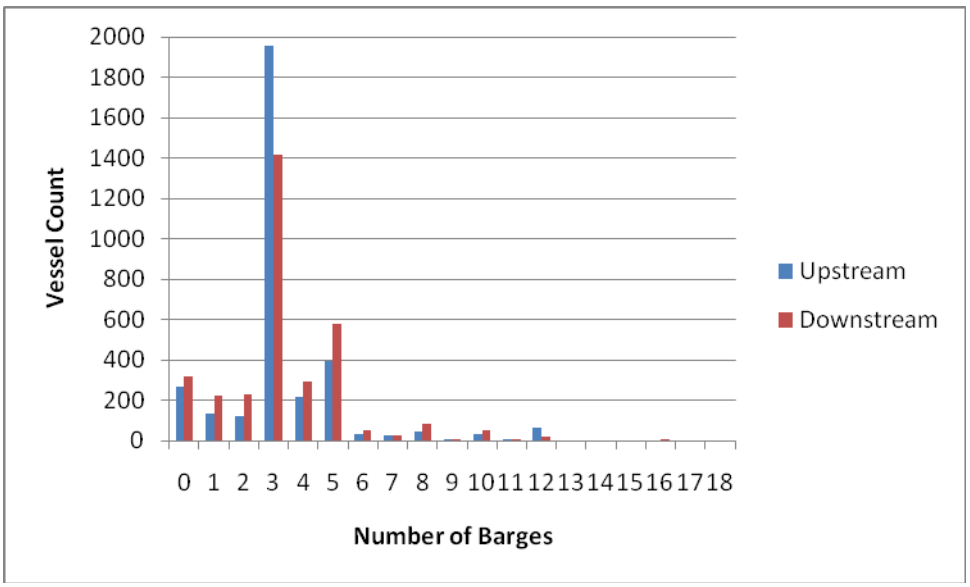
- Configurations

Tows with 3 barges are the most common vessels at lock Marmet (as shown in Figure 20(a)). There are much fewer large tows with more barges, i.e., tows with more than 6 barges. However, if travel directions are considered, more

large tows are found downstream vessels than upstream (as shown in Figure 20(b) and Table 8).



(a)



(b)

Figure 20 Vessels' Configurations

Table 8 2007 Vessel Configuration at Marmet

# of Barges	Total Vessels	Upstream Vessels	Downstream Vessels
0 ~ 2	1289	522	767
3 ~ 5	4858	2567	2291
6 ~ 8	267	105	162
9 ~ 12	103	43	60
13 ~ 15	83	64	19
16 ~ 18	1	0	1

- Required cuts

In DLSIM, the number of cuts is determined based on the dimensional information of carried barges and lock chambers. When a vessel is loaded into the shipment list, the information of required cuts is pre-calculated for each chamber. Since the dimensions of main and auxiliary chambers at the Marmet Lock are identical, 360×56, the required cuts for both chambers are the same. Table 9 and Figure 21 shows the cut information for 2007 shipment list at Marmet. Generally speaking, downstream traffic contains more vessels with higher number of cuts (e.g., 4 or 5 cuts). Only very few vessels require more than 6 cuts for both directions. Among 6601 vessels, there is one very large vessel with 12 cuts. If the cut limit is set at less than 12, this very large vessel will never pass through the chamber and stay in the queue for the whole simulation period.

Table 9 Cut Information of Shipment List

Number of Cuts	Number of Vessels	
	Upstream	Downstream
1	374	504
2	161	290
3	2078	1597
4	288	317
5	388	577
6	7	8
7	0	1
8	2	1
9	1	1
10	0	4
11	0	0
12	2	0

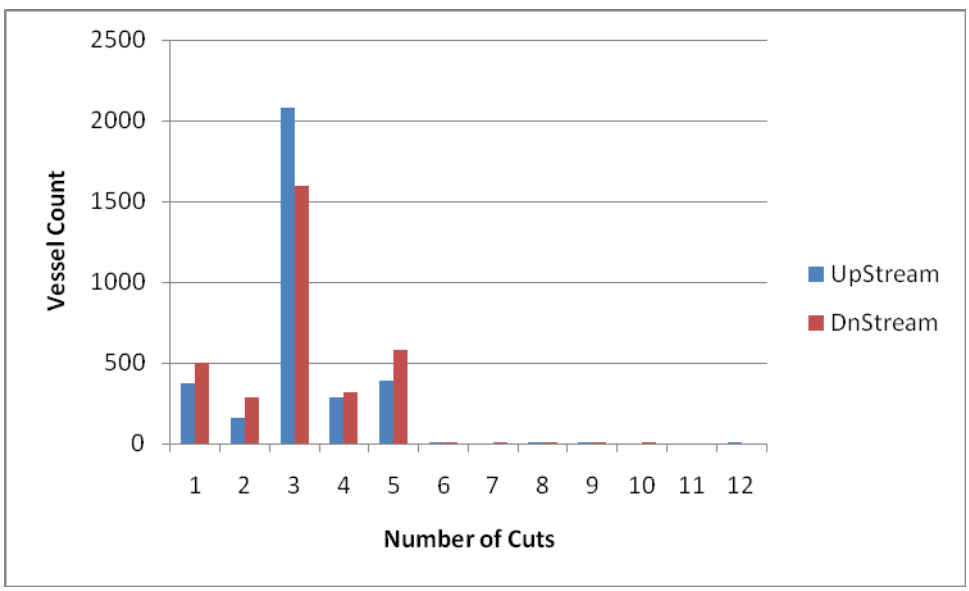


Figure 21 Cut Information for 2007 Shipment List

Operation Information

The current policy adopted at the Marmet lock is FIFO (first come first serve) for both chambers. Detailed lockage information is shown in Table 10. In the Ohio River, since most of the locks are two-chamber locks, recreational craft are not usually allowed in the main chamber and are forced to wait 3 (up to 3) commercial lockages before starting their recreational lockage. Although the developed DLSIM provides various lockage operations (such as multi-vessel lockages, chamber turnback lockage, navigable pass, …, etc.) or operational considerations (such as lock closure, help equipment, …, etc.), those functional options in DLSIM are set off in this test in order to focus on the study of different control policies.

Table 10 2007 Operation Information at Marmet

Lock Operations	Policies	
Control Policy	1. FIFO (for both chambers) 2. N-Up M-Down • 3-Up3-Down • 6-Up6-Down • 12-Up12-Down 3. One Way (mutually for both chambers) • One Way (Up-Down) • One Way (Down-Up) 4. Longest Queue 5. SPF 6. FSPF	
Recreational Craft	Not allowed in main chamber Wait for 3 commercial lockages	
Gate Area Interference	1200 ft (Approach) / 600 ft (Exit)	
Cut Limit	15 (for both chambers)	
Help Equipment	No (for both chambers)	
Assistance	No (for both chambers)	
Lockage	Navigable Pass	No
	Multi-Vessel Lockage (at most 2 vessels)	No
	Mixed-Vessel Lockage (No limit)	No
	Multi-Rec Lockage (one Rec per 100 feet)	No
	Chamber Turnback Lockage	No

Processing Time Distributions

Processing time distributions for detailed lockage components and various vessel types are provided by DAPP, shown in *tblChamberOpsLevel10* in SQL database. The additional time for multi-vessel lockage is assumed in this test.

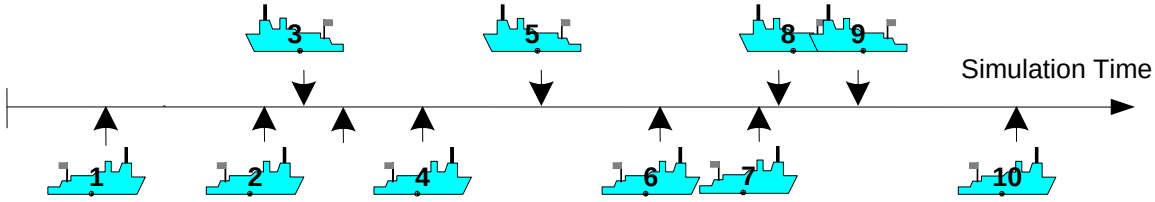
Performance Curves for Different Control Policies

Performance curves for various control policies could be based upon different traffic levels. The sensitivity analysis for each one of the control policies is needed to indicate how the performance changes as traffic changes.

Vessel Demand

In DLSIM, vessels are prepared in a pre-determined shipment list in which the vessels' arrival times at lock are given based on the recorded historical data. In the shipment list, trip information such as arrival times, origin/destination, vessel types, barge sizes and other details (number and types of carried barges, load and unload information from origin to destination as well as visiting points in between) are pre-determined. The number of required cuts could then be determined based on the barge sizes and chamber size. Although loading/unloading and docking/undocking activities might not occur during trips, vessel re-configurations or chamber packing/unpacking maneuvers could occur while locking the vessels.

With a specified shipment list, all vessels are ready as inputs for the model at the beginning of the simulation. They are ordered by their recorded arrival time. That is, in such an event-based simulation model, when simulation starts, each single vessel will be inserted into the modeled system based on the simulation clock (as shown in Figure 24).



	Arrival Time	Barge Information	Origin/ Destination	Load/ Unload	Visits Information	……
Vessel #1						
Vessel #2						
Vessel #3						
Vessel #4						
Vessel #5						
?						

Figure 22 A Shipment List Input for Simulation Model

Therefore, with the given information, it is very difficult to make changes in the given demand, increasing or decreasing the number of vessels as well as all the carrying and loading information in a given simulation time. For planning purposes, it is preferable to generate vessel traffic while running the simulation in order to take into account the future traffic changes.

In addition, control policies are only applied to commercial vessels (e.g., tows with or without barges), but not on recreational vessels and high priority vessels. Recreational craft are not affected by the control policies but follow the operation

designed for recreational vessels. Therefore, the recreational vessels are first taken away from the shipment list. Based on the given traffic data (as shown in Table 7), only 6601 commercial vessels are in the shipment list with balanced traffic of 3300 upstream vessels and 3301 downstream vessels.

In order to overcome the restrictions and difficulties of including varied demand levels into the test scenarios, an alternative of adding more vessels during the simulation is proposed in this study.

Change in Demand Level

The demand level is changed basically by increasing or decreasing the number of vessels for the same period of simulation. In addition to considering the number of vessels, other information, such as arrival times, barges, load/unload, visits…, etc., is also required in determining vessel traffic. Therefore, how to make changes as well as what to change in the existing shipment list are two major issues in the task of preparing various demand levels. In order to simplify the task, changing number of vessels and their arrival times are considered in this test.

It is preferable to have vessels' arrivals as random as possible during the simulation period even though the arrival rate (e.g., number of vessels per year) is the same. That is, currently there are 6601 commercial vessels in the shipment list of year 2007. The demand level can be changed by increasing vessel traffic by a 10% (e.g., 6601*1.1) increment, by 50% (e.g., 6601*1.5) increment, etc. Then the shipment list must be adjusted with the newly increased traffic. Figure 23 shows the logic of changing current demand level as well as the adjustment of new shipment list.

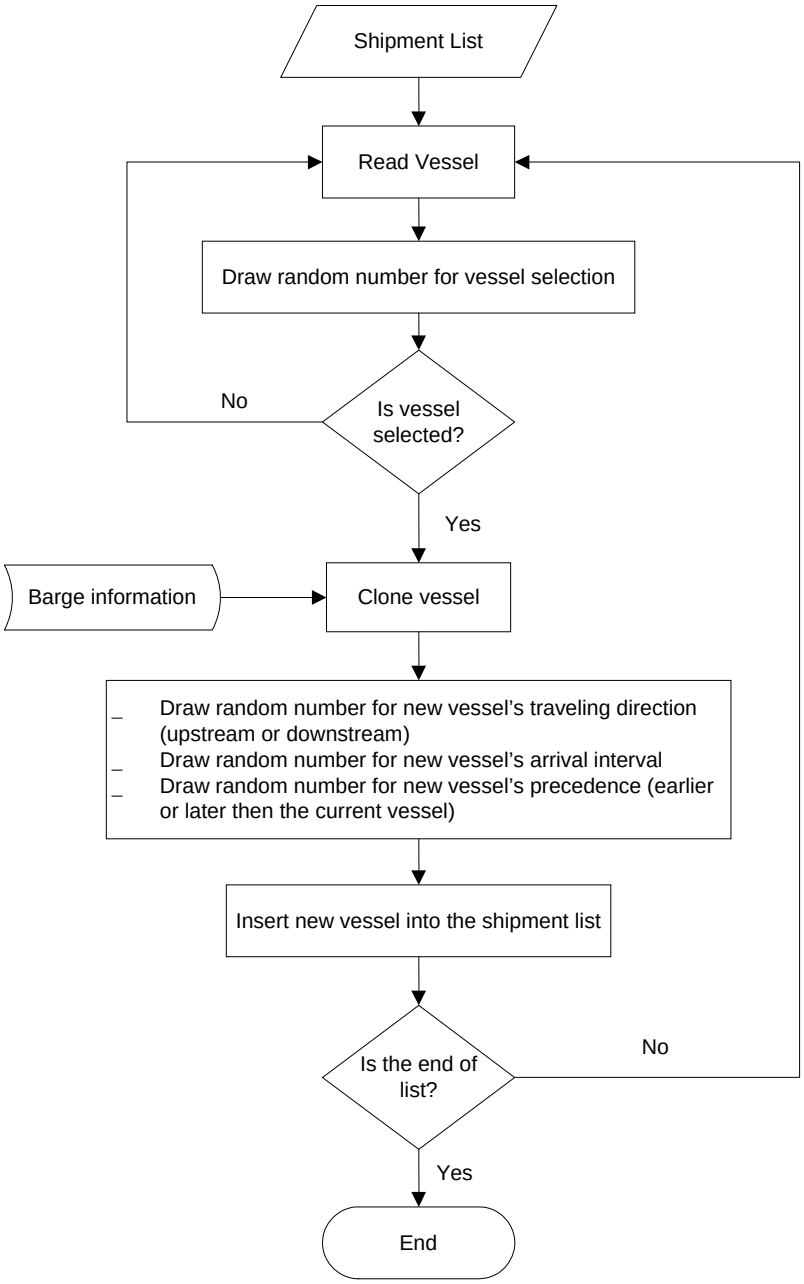


Figure 23 Logic of Changing Vessel Demand Level for Simulation Test

As can be seen, random numbers are drawn several times in the proposed logic for changing demand. First, the shipment list with given vessel information is loaded into the simulation model. From the beginning of vessel list, a random number is drawn to determine if the vessel is selected for "clone" process. If the vessel is selected, the same information is given with vessel type, barge size and other details (number and types of carried barges, load and unload information from origin to destination as well as visiting points in between) in the clone process.

Three more random numbers are drawn afterward for the cloned vessel in order to determine the travel direction (upstream or downstream), arrival interval (normally

distributed with specified mean and standard deviation) and precedence relation with selected vessel (ahead or after). With all the information, the cloned vessel is then inserted into the shipment list according to the assigned arrival time. The overall procedure is repeated until the end of original list. Figure 24 shows the difference between the original shipment list and the new shipment list with the newly cloned vessel.

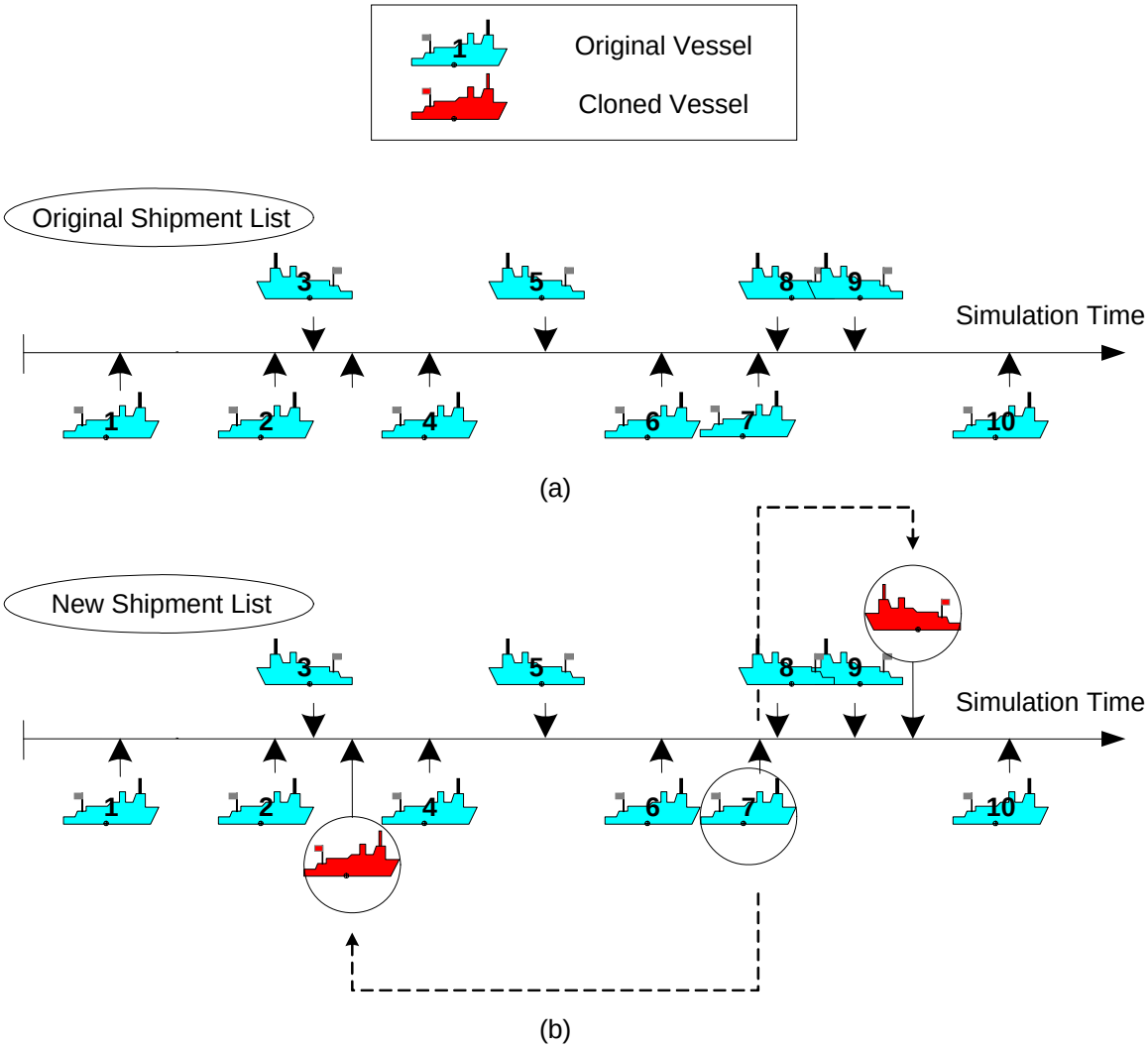


Figure 24 Change of Shipment List Input for Simulation Model

For example, if vessel #7 is selected for the clone process, a random number is drawn to determine the travel direction of new vessel. With arrival time of 07:07:00 for the selected vessel, the time interval for the cloned vessel's arrivalis normally distributed with mean of 10 and standard deviation of 3. If the random number is drawn with value of 17, then the arrival time for the duplicated vessel could be at later at 07:24:00 or earlier at 06:50:00 based on the random number draw for the cloned vessel's precedence relation with the selected vessel.

The different demand levels in the testing scenarios are determined with the first random number which determines if the vessel is selected for the cloning process. If

demand is increased by 10%, then 10% of vessels in the shipment list will be selected for clone process. If demand is doubled, 100% of vessels in the shipment list will be in the clone process. Table 11 shows the different demand levels tested in this study.

Table 11 Tested Demand Level

Scenarios	# of Total Vessels	# of Up. Vessels	# of Dn. Vessels
Base Case	6601	3300	3301
5% increase	6939	3455	3481
10% increase	7261	3629	3632
15% increase	7577	3784	3793
20% increase	7894	3938	3956
25% increase	8226	4082	4144
30% increase	8603	4252	4351
35% increase	8932	4424	4508
40% increase	9302	4596	4706
45% increase	9598	4762	4836
50% increase	9902	4939	4963

Designed Scenarios of Static Control Policies

In current DLSIM, six different control policies are modeled as static control polices. However, there could be various parameters for different control policies, such as direction, UpCount, DownCount and fairness value (e.g., number of vessels in queue) shown in Table 3). Not all parameters are considered in each control policy, but with its own logic concerns. As for the OneWay policy, since it is specifically used at two-chamber locks by providing lockage services from assigned chambers based on vessels' travel directions, it is excluded in this study for general lock systems. Table 12 shows the designed scenarios of static control polices.

Table 12 Designed Scenarios for Static Control Policies

Lock Policy	*Designed Scenarios*	*Direction*	*UpCount*	*DownCount*	*Fairness Value*
FIFO	FIFO	-	-	-	-
N-Up M-Down	3-Up 3-Down	-	3	3	-
	6-Up 6-Down	-	6	6	-
	12-Up 12-Down	-	12	12	-
Longest Queue	Longest Queue	-	-	-	-
SPF	SPF	-	-	-	-
FSPF	FSPF Q3	-	-	-	3
	FSPF Q5	-	-	-	5
	FSPF Q10	-	-	-	10

As can be seen, there are no parameter settings for the FIFO, Longest Queue and SPF policies. Only one scenario is designed for each one of them. For the N-Up M-down policy different values are set for parameters of *UpCount* and *DownCount*. In order to

simplify the analysis, for both serving directions, N is equal to M as 3, 6, or 12, respectively. Therefore, 3 scenarios are designed, 3-Up 3-Down, 6-Up 6-Down and 12-Up 12-Down, for testing policy of N-Up M-Down.

The One-Way policy can only be practiced at two-chamber locks. In addition, based on fairness concerns, there should be two identical chambers with same operational considerations (e.g., preference) and features (e.g., service time). The preferred direction is assigned to each chamber, U or D, for parameter of "*Direction*". The value indicated in parameter of "*Fairness Value*" sets the threshold of changing service direction. That is, if chamber's preferred direction is set as U, chamber is assigned to serve upstream vessels. However, chamber will serve downstream vessels if there is no vessel traveling upstream when chamber is idle but number of vessels in downstream queue exceeds the fairness value. Whenever there is an upstream vessel, the chamber's serving direction will be set back to its preferred direction. Therefore, a total eight scenarios are designed (4 different fairness values and 2 combinations of serving direction) for testing polices of One Way.

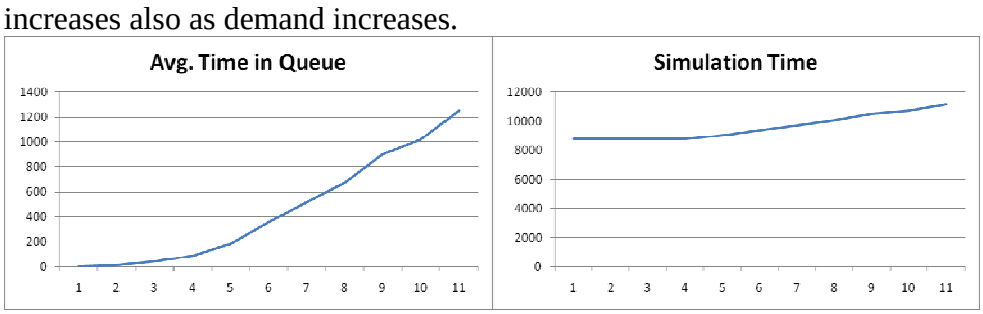
The policies of Longest Queue, N-Up M-Down, and One Way are already practiced at some locks, in addition to the FIFO. Policies of SPF and FSPF are proposed in some waterway studies before, but not yet pilot tested in field study. This study will provide a comprehensive understanding on the performance for various control polices, practiced or not yet practiced, and further help lock operators and decision makers.

Performance Curves for Different Control Policies

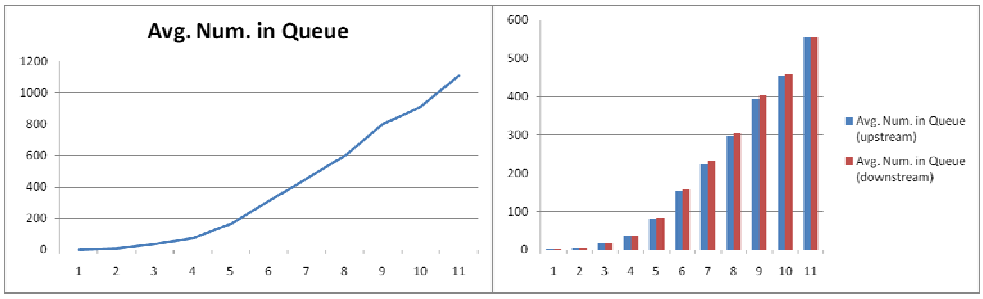
From the discussion in the previous section, four performance measurements are selected to create the performance curves: average time in queue, average queue length for both directions, and overall simulation time. The different performance curves for different control policies are presented below. Demand is increased with 5% of original traffic for each point along the x axis. That is, the demand level at the first x point is the base demand from original shipment list. The demand level has increased 50% at the last x point. The technique of increasing demand for testing purposes in DLSIM is discussed in the previous section.

- FIFO

  Figure 25 shows the performance curves for the most common control policy of FIFO upon increasing traffic demand. As expected, the average time spent in queue, the average queue lengths for both directions are increased as demand increase. The time for dispatching all traffic (e.g., simulation time) increases also as demand increases.



(a) (b)



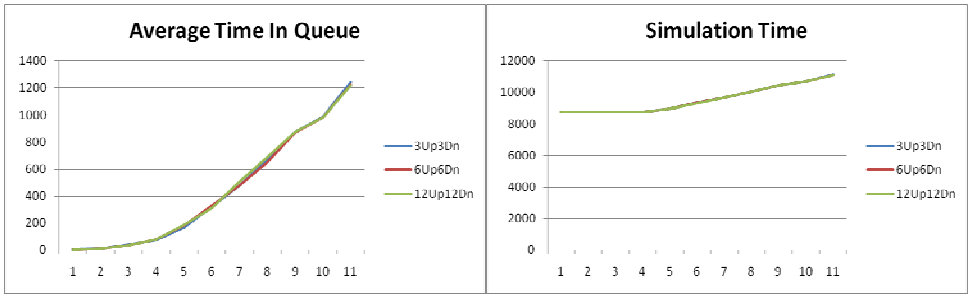
(c) (d)

Figure 25 Performance Curves for FIFO

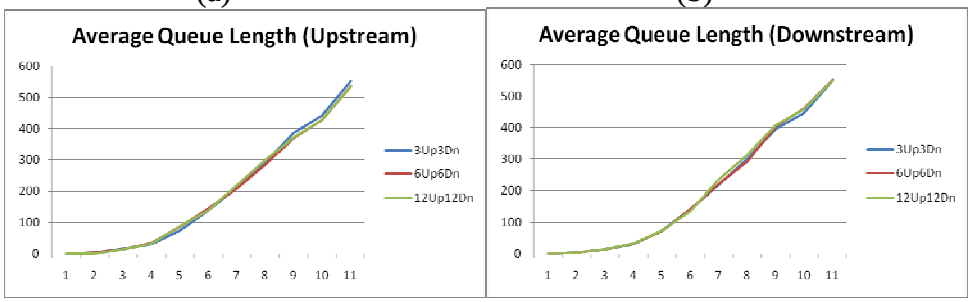
- N-Up M-Down

Usually this policy shows benefits at some locks where approach distances are long from both directions . Since the approach area is only allowed for one-way passing through, with long approach distance, the times for chamber turning back in order to serve vessels from the same direction could be shorter than the times that chamber is idle for approaching vessels from the opposite direction.

Figure 26 shows the performance curves for the policy of N-Up M-Down with different parameter settings. Since the approach distance is not considered yet as part of lock geometry while modeling DLSIM, the benefits from this specific type of control policy do not have significant effect in this test.



(a) (b)

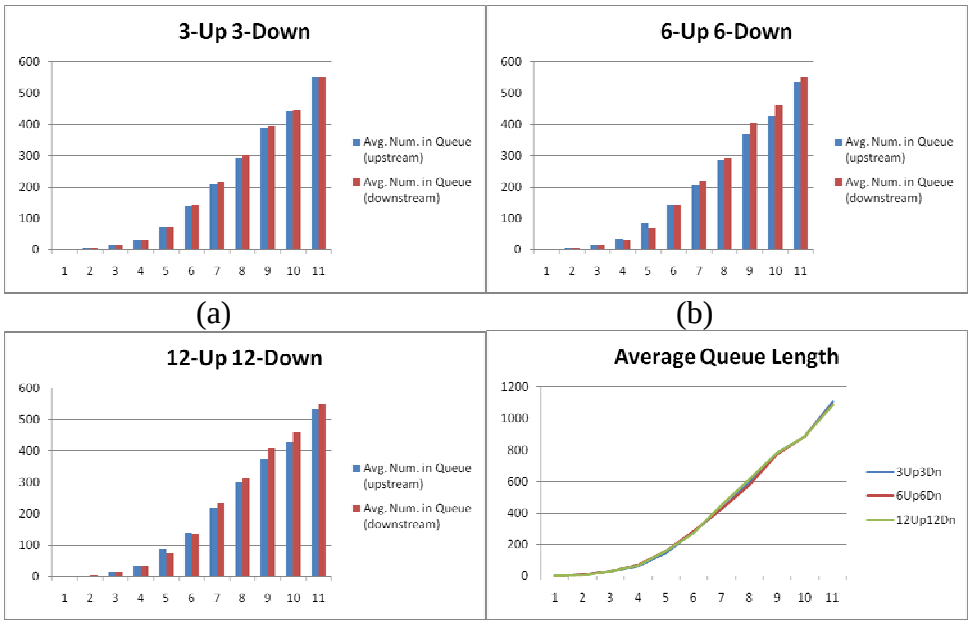


(c) (d)

Figure 26 Performance Curves for N-Up M-Down

The N-Up and M-Down (e.g. the settings for UpCount and DownCount) policy is run only when queues are long enough in both directions. If queue length does not reach the threshold, the FIFO policy is in effect (detail policy

design is discussed in the previous section of "Lockage Policies"). Therefore, even with different parameter settings, the performance results are similar to each other with increasing average time spent in queue, the average queue lengths for both directions while increasing travel demand. Figure 27 further shows the comparison of queue lengths from different travel direction based on three different levels of N-Up and M-Down. There are no significant discrepancies between different threshold counts.



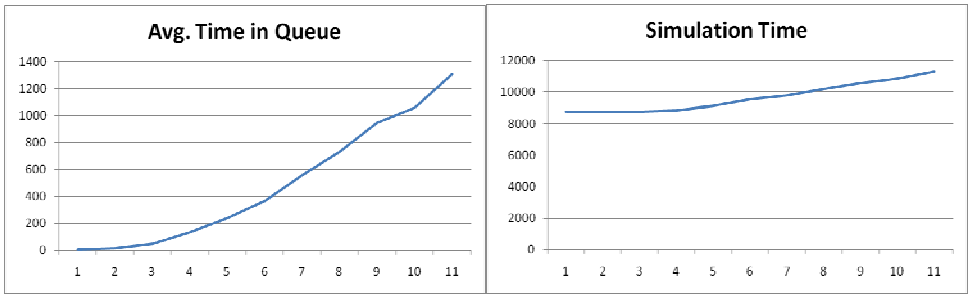
(a) (b)

(c) (d)

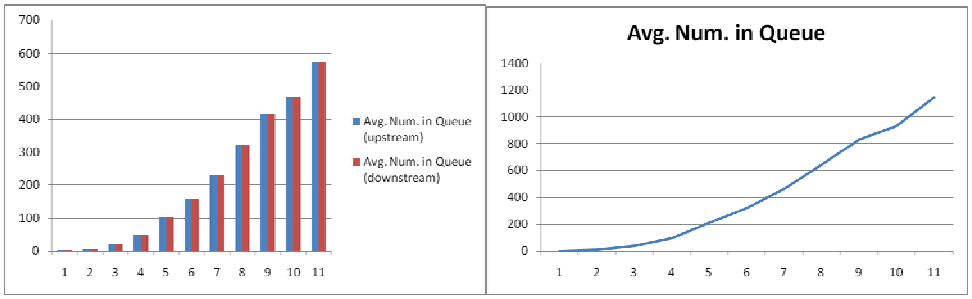
Figure 27 Average Queue Length for Group of N-Up M-Down

- Longest Queue

A simple logic is embedded in the policy of Longest Queue in which a vessel is selected from the longer of the two queues (upstream and downstream) at a lock. Figure 28 shows the performance curves of this policy. Since this policy focuses on balancing the queue length for both directions, i.e., it always selects a vessel from the longer queue among two directions, the average queue lengths are almost the same (as seen in Figure 28(c)) for upstream and downstream at any level of traffic demand.



(a) (b)



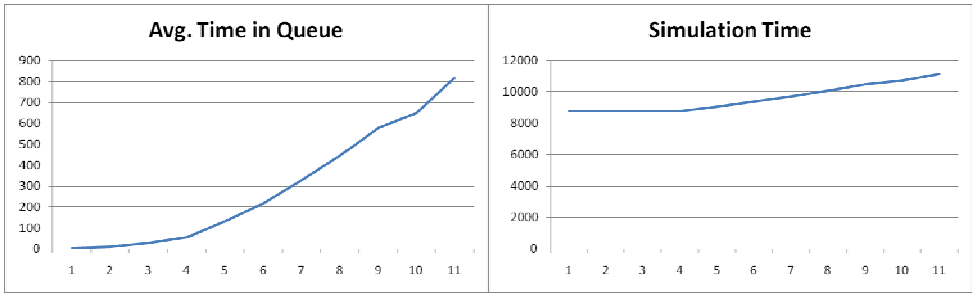
(c) (d)

Figure 28 Performance Curves for Longest Queue

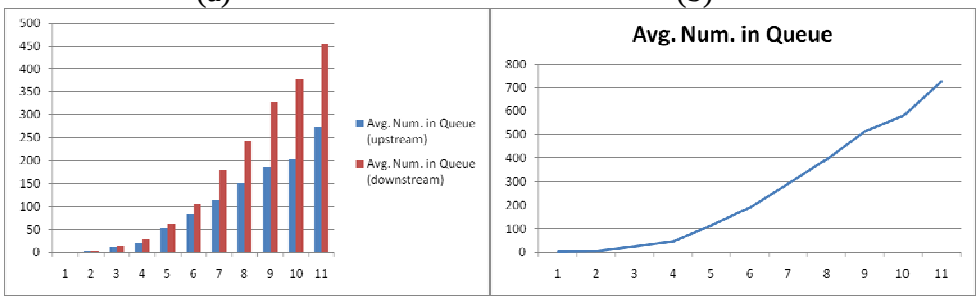
- SPF

Shortest Processing Time First is a proposed ideal policy which has not yet been applied in the lock operation. It favors tows with shorter service time. Currently DLSIM models service time based on the number of required cuts. That is, tows with more cuts require longer service time. Thus, this policy chooses the tows with fewer cuts in the queue and locks them ahead of tows with more cuts, regardless of their arrival times. Generally speaking, with this preference, the average waiting time and average queue length should be reduced compared with FIFO.

Figure 29 shows the performance curves of this policy. Since this policy is closely related to the tow size (e.g., number of carrying barges), rather than travel directions. If the vessel configurations from both directions are different, there are differences in the average queue length for both directions. The longer queue is shown in the direction with more larger tows since they are kept in queue and by past by smaller tows.



(a) (b)



(c) (d)

Figure 29 Performance Curves for SPF

- FSPF

Since SPF favors tows with shorter processing time, larger tows may experience more delays with SPF than FIFO, thus raising concerns on fairness issue. In order to balance the system efficiency and fairness among individual tows, FSPF is proposed to be intermediate between SPF and FIFO. FSPF is a fairer SPF control policy that gives priority to tows which have waited for a certain number of lockages (F*, the fairness value pre-defined in the input file) based on the SPF rules. In this test, FSPF1 stands for FSPF with F* value of 1, FSPF7 stands for FSPF with F* value of 7, …, etc. Different fairness values will influence the system performance. Figure 30 shows the performance curves of FSPF with various parameter settings, wait for 1 lockage, or 5 lockages, or 10 lockages, respectively.

Furthermore, the average number of tows in the queue is a significant indicator for evaluating the system. FSPF with lower fairness value gives larger tows more chances to leave the waiting queues while they keep being passed by smaller tows. Also, in the contrast, the system will be more efficient with lower delays and shorter queues based on the higher F*.

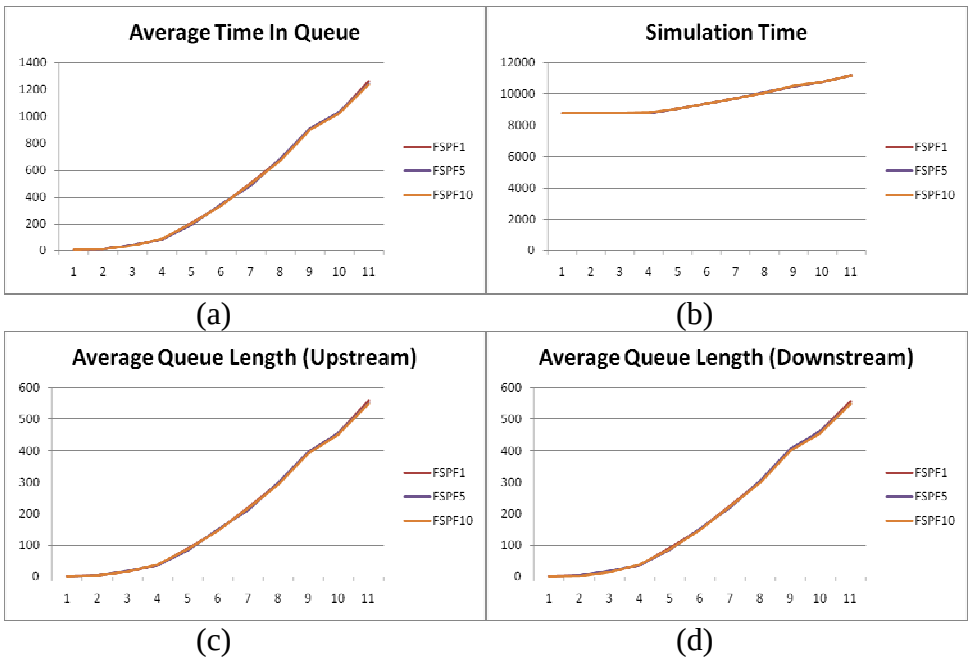


Figure 30 Performance Curves for FSPF

In general, the performance of FSPF depends on the threshold setting for fairness value (i.e., count of queue). With a small F*, FSPF will be close to FIFO. However, with large F*, FSPF should be close to SPF. In the current test, the fairness threshold is set as 1, 5 and 10. Figure 31 further shows the comparison of queue lengths from different travel directions based on three different fairness

levels of FSPF. Only small differences are observed in the performance curves but with a decreasing trend as fairness values increase.

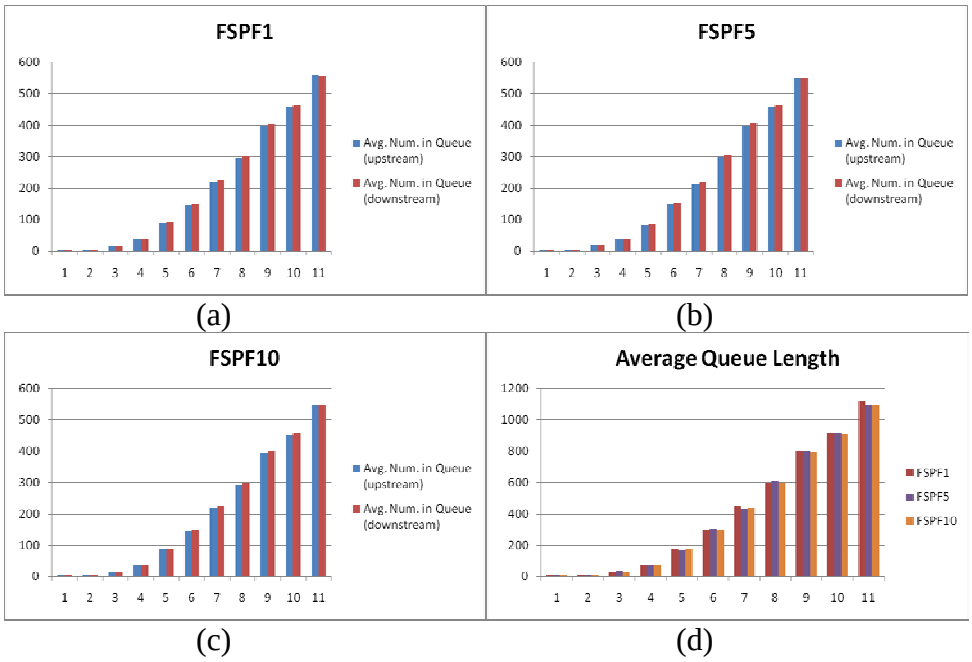


Figure 31 Average Queue Length for Group of FSPF

Comparison of Different Control Policies

Performance curves for individual control policies, or groups of control polices with different threshold settings, have been discussed in the previous section with sensitivity analysis upon various demand levels. However, from a lock operator's point of view, among various control policies, the concerns are which policy can out-perform another ones and help increase overall system efficiency at certain time, certain cost, and certain conditions. Thus, in order to provide such information to lock operator or users to simulation model, the following section provides a pre-analysis of comparing different control policies upon various demand levels. Two major measures of performance is discussed, average time in queue and average queue length.

Figure 32 shows the comparison of average time in queue for different control policies. While other polices do not show substantial differences from FIFO in changing (increasing or decreasing) average time in queue in, SPF does save a lot of waiting time in queue if demand is high.

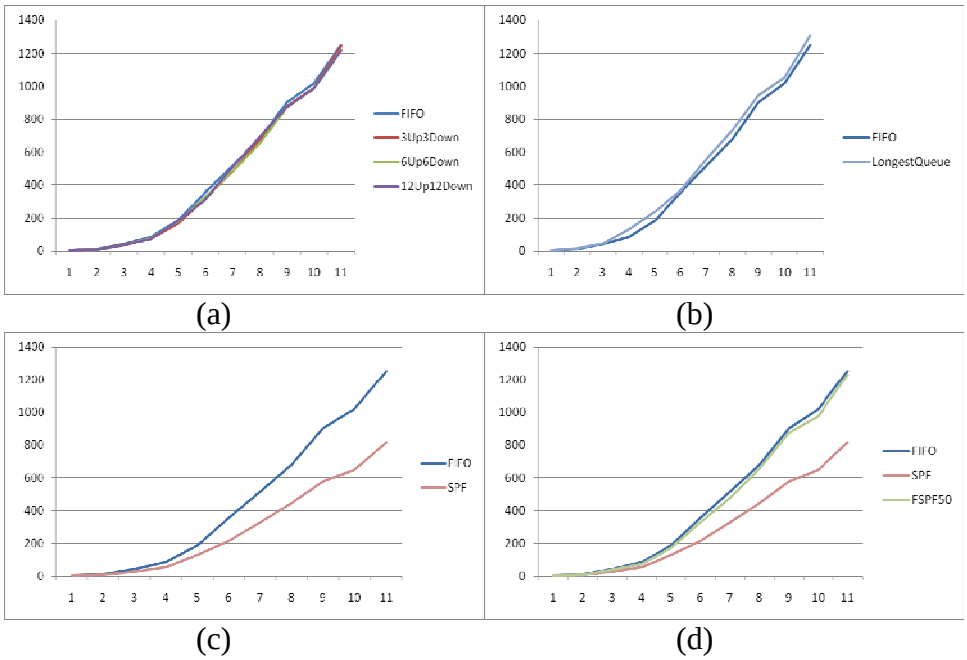


Figure 32 Comparison of Average Time in Queue

Similarly to Figure 32, Figure 33 and Figure 34 show the comparison of average queue length among different control policies. SPF still out-performs other polices with shorter average queue length. As expected, with a fairness value of 50, the performance of SPF50 stands in between FIFO and SPF.

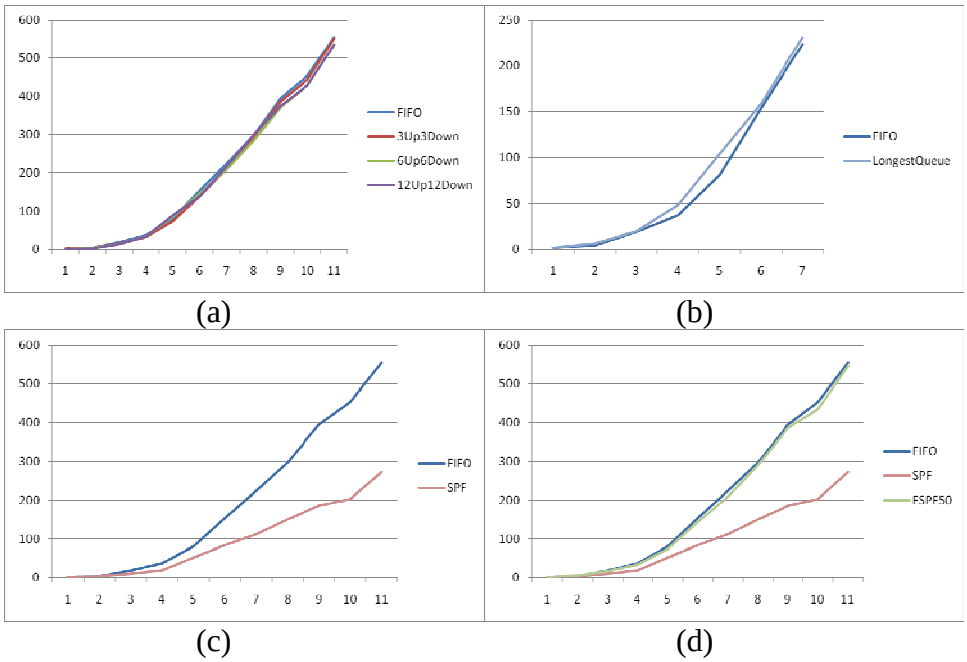


Figure 33 Comparison of Average Queue Length (Upstream)

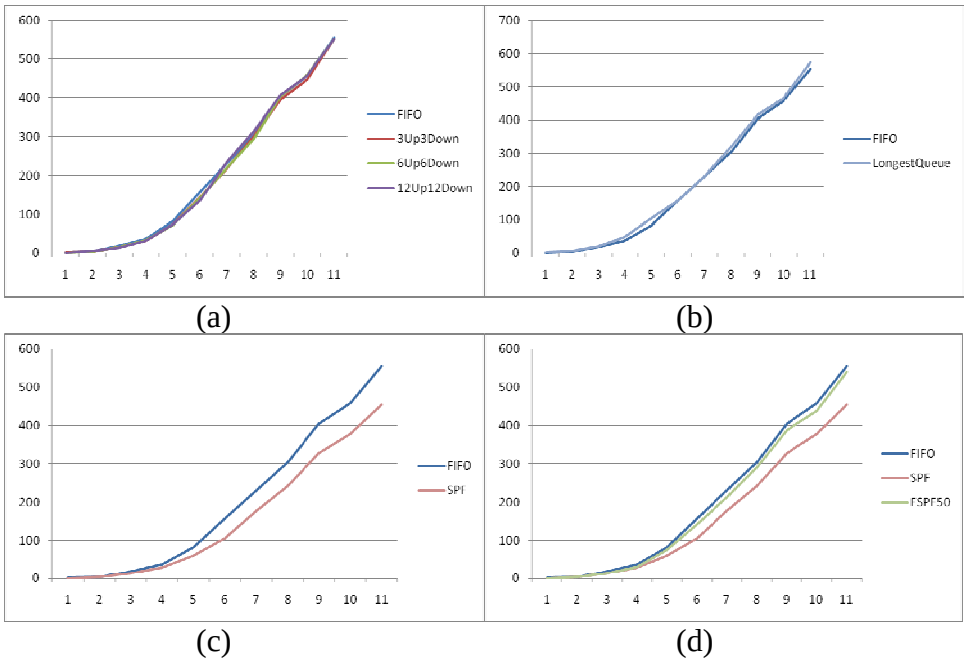


Figure 34 Comparison of Average Queue Length (Downstream)

From the figures shown in the above analysis, no clear thresholds are observed among various policies. That is, for example, SPF always out-performs FCFS, without any crossover, for any traffic level. If that is the case, there should always be a superior control policy at any place, any time, and under any traffic condition. However, that is not practical in real world operation. Compared with FIFO, the fairest rule, there is always some cost of "unfairness", such as switching direction, or bypassing other vessels. With those subtle considerations, some cost and "inconvenience" should be further considered in the modeled policies.

Conclusions and Summary

A Detailed Lock Simulation Model (DLSIM) has been developed and tested. It models in detail the various steps in the lockage process, various vessels and their proper operational rules, and different operational ways in locking and dispatching vessels. The performance of various control policies are tested and evaluated at different demand levels. FIFO is the most common and fairest control policy. The strengths of N-Up and M-Down will be more apparent if one-lane approaching distances are significant. It benefits lockage by serving platoon vessels from the same direction if exit time for the current vessel is much longer than chamber turnback time for the coming vessel with same travel direction. Dispatching vessels based on the longest queue balances the queue length for both directions. SPF out-performs other control policy by saving waiting time and shortening queue lengths. If the cost of making changes from FIFO can be considered, the test results will be more realistic with expected crossing curves among various control policies.

Information Transfer Program Introduction

For the eighth year, the Maryland Water Resources Research Center supported a 1-day symposium on a water issue important to the State.

The Center also published two newsletters. Articles focused on water research taking place in Maryland and information on featured scientists. The newsletter mailing list is approximately 220, with others receiving an electronic version.

Toxics in Maryland Waters

Basic Information

Title:	Toxics in Maryland Waters
Project Number:	2009MD198B
Start Date:	3/1/2009
End Date:	2/28/2010
Funding Source:	104B
Congressional District:	5th
Research Category:	Not Applicable
Focus Category:	None, None, None
Descriptors:	
Principal Investigators:	Allen Davis, Allen Davis

Publications

There are no publications.

Toxics in Maryland Waters

The Maryland Water Resources Research Center sponsored a 1-day colloquium on *Toxics in Maryland Waters* on November 6, 2009. This event consisted of a series of seven (scheduled) presentations and related discussion. Speakers included:

- "***Survey Of Human use Pharmaceuticals in The Chesapeake Bay***", A.S. Pait, National Center for Coastal Ocean Science, NOAA, Silver Spring, MD.

- "***Reproductive Health of Fishes and Association with Complex Chemical Mixtures***", V.S. Blazer, , USGS, Kearneysville, WV.

- "**Road *Salts in the Suburban Landscape: Transport Direct Effects and Interactions with Other Pollutants***", J. Snodgrass, R. Casey, S. Lev, Towson University. E. Landa, , USGS, Reston, VA.

- "***Losses of Arsenic and Phosphorus from Poultry Litter Amended Fields***", C.D. Church, Chemist, ARS, USDA, University Park, PA.

- "***Assessing Biological Effects of Toxic Contaminants in The Chesapeake Bay***", M.J. Hameedi, , NOAA, Silver Spring, MD.

- "***Pfiesteria***", W.K. Vogelbein, Virginia Institute of Marine Sciences, Gloucester Point, VA.

- "***Sources and Sinks of Methylmercury in the Chesapeake Bay and its Watershed***", C.C. Gilmour, Smithsonian Environmental Research Center, Edgewater, MD. (Last minute cancellation due to illness)

Attendance included over 100 faculty, students, and professionals from outside agencies. The Maryland Sea Grant College co-sponsored this event.

USGS Summer Intern Program

None.

Student Support					
Category	**Section 104 Base Grant**	**Section 104 NCGP Award**	**NIWR-USGS Internship**	**Supplemental Awards**	**Total**
Undergraduate	1	6	0	0	7
Masters	7	4	0	0	11
Ph.D.	2	0	0	0	2
Post-Doc.	0	0	0	1	1
Total	10	10	0	1	21

Notable Awards and Achievements

Brand and Snodgrass (2010) received coverage on the front page of Conservation Maven, an online hub for the conservation community (http://www.conservationmaven.com/frontpage/2009/8/17/do-created-wetland).

News Pieces National Public Radio-Living on Earth "Snow and Salt Report"
www.loe.org/shows/shows/htm?programID=09-P13-00008#feature9

Baltimore Sun Bay and Environment Blog "Icy dilemma:Road salt taints streams, reservoirs"
weblogs.baltimoresun.com/news/local/bay_environment/blog/2009/03/icy_dilemma_road_salt_taints_s.html