Hydrologic Modeling as a Predictive Basis for Ecological Restoration of Salt Marshes

CHARLES T. ROMAN*
National Park Service[1]
Cooperative Park Studies Unit
Graduate School of Oceanography
University of Rhode Island
Narragansett, Rhode Island 02882, USA

RICHARD W. GARVINE
Graduate College of Marine Studies
University of Delaware
Newark, Delaware 19716, USA

JOHN W. PORTNOY
National Park Service[1]
Cape Cod National Seashore
South Wellfleet, Massachusetts 02663, USA

ABSTRACT / Roads, bridges, causeways, impoundments, and dikes in the coastal zone often restrict tidal flow to salt marsh ecosystems. A dike with tide control structures, located at the mouth of the Herring River salt marsh–estuarine system (Wellfleet, Massachusetts) since 1908, has effectively restricted tidal exchange, causing changes in marsh vegetation composition, degraded water quality, and reduced abundance of fish and macroinvertebrate communities. Restoration of this estuary by reintroduction of tidal exchange is a feasible management alternative. However, restoration efforts must proceed with caution as residential dwellings and a golf course are located immediately adjacent to and in places within the tidal wetland. A numerical model was developed to predict tide height levels for numerous alternative openings through the Herring River dike. Given these model predictions and knowledge of elevations of flood-prone areas, it becomes possible to make responsible decisions regarding restoration. Moreover, tidal flooding elevations relative to the wetland surface must be known to predict optimum conditions for ecological recovery. The tide height model has a universal role, as demonstrated by successful application at a nearby salt marsh restoration site in Provincetown, Massachusetts. Salt marsh restoration is a valuable management tool toward maintaining and enhancing coastal zone habitat diversity. The tide height model presented in this paper will enable both scientists and resource professionals to assign a degree of predictability when designing salt marsh restoration programs.

The history of diking, draining, and impounding salt marsh–estuarine ecosystems for purposes of mosquito control, land reclamation, wildlife habitat enhancement, and flood protection is well documented (Montague and others 1987, Daiber 1986). These management practices often result in significant changes to a system's physical, chemical, and ecological characteristics. Roman and others (1984) found that restriction of tidal flushing by roads, causeways, and bridges is a major factor contributing to the degradation of salt marshes and small estuaries along Connecticut's Long Island Sound shoreline. The Herring River marsh–estuarine system, located in Cape Cod, Massachusetts, USA, represents a typical case whereby wetland functions and processes have been altered as a result of diking and associated tidal restriction in the early 1900s. Major hydrological and vegetation changes, stream acidification, episodes of stream anoxia, fish kills, mosquito control problems, and other effects have all been attributed to the diking, ditching, and stream channelization of Herring River (Portnoy 1984, 1991, Soukup and Portnoy 1986, Portnoy and others 1987).

Restoration is an appropriate management goal for the now degraded Herring River ecosystem. Studies of other northeastern United States salt marshes under regimes of restricted tidal flow have demonstrated that rehabilitation of ecological functions can be initiated with the reintroduction of tidal flushing (Ferrigno and others 1987, Fell and others 1991, Sinicrope and others 1991, Barrett and Niering 1993, Peck and others 1994). In other regions, numerous case studies show that degraded coastal wetlands and small estuaries are being successfully restored (Wolf and others 1986, Kusler and others 1988, Kusler and Kentula 1990). Fifty percent of the nation's coastal

KEY WORDS: Salt marsh; Habitat restoration; Hydrologic modeling; Massachusetts

[1]Present affiliation is National Biological Service with same address.
*Author to whom correspondence should be addressed.

wetlands have been lost (Tiner 1984, Dahl 1990), and others are significantly degraded; thus, it seems essential that wetland and estuarine restoration should be a priority resource management activity for government and private agencies with resource conservation missions.

With a focus on the Herring River estuary, this study presents an approach for use in evaluating the feasibility of restoring tidal exchange to restricted marsh–estuarine systems. When evaluating sites for restoration, coastal resource managers must have a rationale for balancing the ecological benefits of restoration against potential socioeconomic consequences (e.g., residential and commercial flooding). A one-dimensional numerical model was developed to predict tide height levels for numerous increased openings of the tide-restricting gates at the mouth of Herring River. Knowledge of tide heights over complete tidal cycles and during coastal storms enables prediction of expected ecological changes and identification of flood prone areas. Derivation of the model and its application to other systems are discussed.

The Study Site: Ecology of a Tide-Restricted Marsh Estuary

The Herring River dike, constructed in 1908, is fitted with one sluice gate, allowing minimal tidal exchange, and two flapper-gates allowing only ebb-directed flow (Figures 1 and 2). Mean tidal range measured from a tide staff at the immediate downstream side of the dike is 2.3 m, while immediately upstream the mean range is substantially restricted to 0.5 m. Given eight decades of tidal manipulation, the restricted marsh surface elevation averages 70 cm lower when compared to the adjacent unrestricted marsh. Similar elevation differences between unrestricted and restricted marshes have been noted in other systems (Roman and others 1984) and may be related to decreased contribution of coastal sediments, drying of peat and subsequent compaction and shrinkage, and especially accelerated decomposition under aerobic conditions.

Historically, *Spartina* sp. marsh dominated the Herring River basin. Now, under a regime of reduced tidal flow, woody species, such as *Pinus rigida* (pitch pine), *Acer rubrum* (red maple), *Prunus serotina* (black cherry), and *Rosa virginiana* (rose), are evident throughout (Roman 1987). The only remnants of herbaceous wetlands are confined to stream channel edges and limited expanses of *Typha* sp. (cat-tail) marsh. *Spartina alterniflora* (saltwater cordgrass) bor-

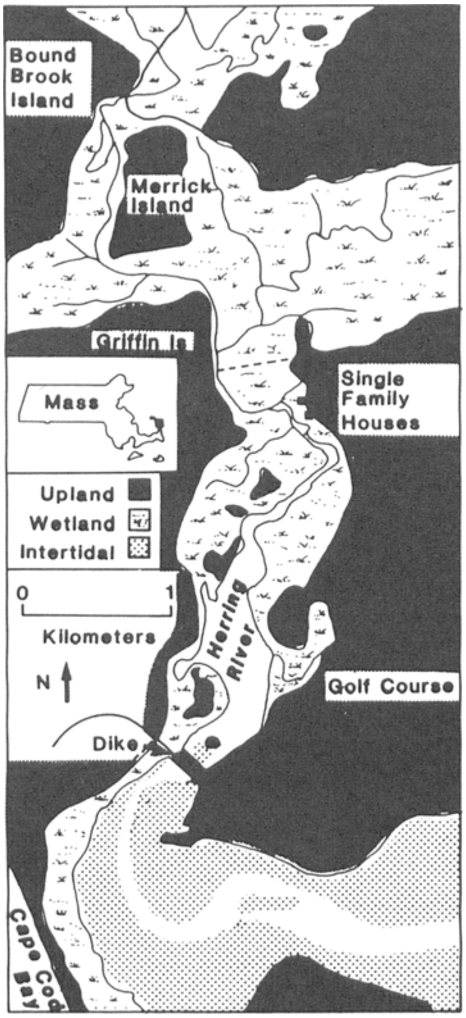

Figure 1. The Herring River salt marsh–estuarine ecosystem in Wellfleet, Massachusetts. Note the roadway and dike crossing the system near the mouth and cultural features (i.e., golf course and single-family dwellings) located within the tidal floodplain.

ders areas of the restricted marsh near the dike, while upstream, fringes of brackish and freshwater species occur (e.g., *Juncus effusus, Glyceria obtusa, Phragmites australis*).

Water quality problems in the system are severe. Periods of summer anoxia commonly occur upstream of the dike and are accompanied by die-offs of juvenile blueback herring *(Alosa aestivalis)* and alewife *(Alosa pseudoharengus)*. Soukup and Portnoy (1986) suggest that with lowered water table levels and aeration of old salt marsh peat, pyrite, abundant in marsh peat, is oxidized, releasing acidic leachate to the river. Accelerated decomposition of the aerated marsh peat

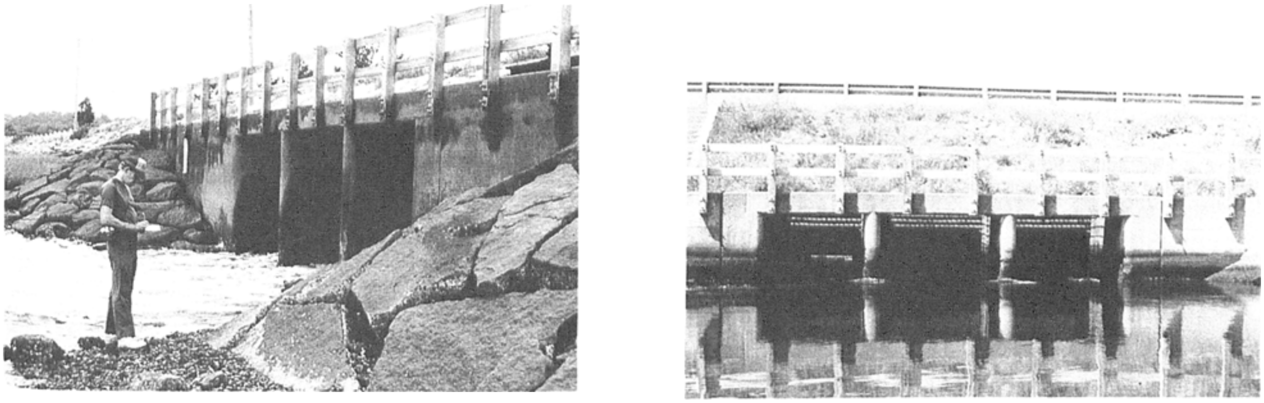

Figure 2. Dike and tide gates near the mouth of Herring River. Tidal range downstream of the dike (left) is 2.3 m and 0.5 m upstream (right). Note the three openings through the dike, fitted with a sluice gate and two flapper-type tide gates.

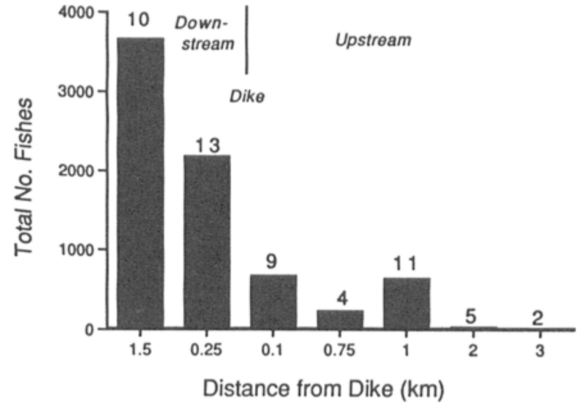

Figure 3. Total number of individual fishes collected at two seine haul stations located downstream of the dike and five stations upstream, within the restricted marsh. Numbers above the bars are number of species collected. Seine haul data from July and September 1984 are combined (N = 4–6/station) (Roman 1987).

leads to increased organic matter input to streams, thus increasing oxygen demand and depleting stream dissolved oxygen. Additionally, the chemical oxygen demand of reduced iron and sulfur minerals may contribute to stream anoxia (Portnoy 1991).

Summertime seine hauls revealed striking differences in the number of fish downstream and upstream of the dike (Figure 3) (Roman 1987). The downstream assemblage is similar to that described by others for New England salt marshes (Teal 1986, Ayvazian and others 1992). Of the resident species, those that spend much of their lives in the Herring River estuary—striped killifish *(Fundulus majalis)*, Atlantic silverside *(Menidia menidia)* and common killifish *(Fundulus heteroclitus)*—were most abundant. Nonresident species found downstream of the dike, or those that use the marsh–estuary as a nursery area, included Atlantic menhaden *(Brevoortia tyrannus)*, blueback herring, alewife, and others. The species composition of fishes collected from the upstream brackish water stations (0.1 km, 0.75 km, and 1.0 km upstream of dike) was similar to downstream; however, abundance was greatly reduced (Figure 3). The freshwater portion of Herring River (2–3 km upstream of dike) was the poorest habitat both with regard to number of species and number of individuals captured. Only three freshwater species were captured in July and September seine hauls, chain pickerel *(Esox niger)*, pumpkinseed *(Lepomis gibbosus)*, and golden shiner *(Notemigonus crysoleucas)*, represented by only seven individuals.

Several factors are contributing to the depauperate fish community upstream of the dike. With a decrease in mean tidal range, there has been a reduction in the area of submerged and intertidal wetland habitat, thereby reducing shelter, forage, and spawning areas. In addition, poor water quality (high acidity, hypoxia/anoxia) has adversely affected the fish community.

Macroinvertebrates collected from the *Spartina* marsh located downstream and upstream of the dike differed dramatically (Table 1) (Roman 1987). Only two species, ribbed mussel *(Geukensia demissa)* and salt marsh snail *(Melampus bidentatus)*, were collected upstream of the dike. The composition and density of species collected from the downstream salt marsh were similar to other New England marshes (Fell and others 1982, Peck and others 1994). Reduced tidal range and salinity, coupled with water quality im-

Table 1. Salt marsh macroinvertebrate densities from sites located 1 km downstream and 1 km upstream of the Herring River dike[a]

	Downstrean			Upstream		
Taxon	Sas	Sat	Sp	Sas	Sat	Sp
Bivalvia						
Crassostrea virginica	0	24	0	0	0	0
Geukensia demissa	16	188	36	0	12	0
Gastropoda						
Ilyanassa obsoleta	0	52	0	0	0	0
Littorina littorea	0	116	20	0	0	0
Melampus bidentatus	12	0	652	0	120	600
Crustacea						
Carcinus maenas	0	0	1	0	0	0
Uca pugnax[b]	32	nd	120	0	0	0

[a]Density estimates are based on 0.25 m^2 (N = 3) quadrats collected in August 1984 (Roman 1987). Sas, short *Spartina alterniflora;* Sat, tall *Spartina alterniflora;* Sp, *Spartina patens*, nd, no data.

[b]Value represents density of burrow holes.

pacts, appear to be the principal factors affecting the upstream salt marsh macroinvertebrate community.

Methods

Model Development

Model Cases. The documented ecological and biogeochemical impacts associated with tidal restriction provided an incentive for developing a program of restoration. However, complete opening of tidal gates was not appropriate because of the potential for flooding residential dwellings and a golf course (see Figure 1). Furthermore, the mean elevation of the wetland surface within the restricted basin was about 0.7 m lower than the downstream *Spartina* marsh. Thus, the reintroduction of tidal flow could excessively flood the restricted wetland, thereby impeding the restoration of herbaceous-dominated intertidal low and high marsh. These problems are common to other potential restoration sites, and they address the need for predicting water levels prior to reintroducing tidal flow.

The Herring River tide gate structure is engineered so that numerous configurations are possible for controlling flow volume. The tide height model developed for the system predicts water levels over complete tidal cycles for different configurations of the structure. When coupled with elevations of the wetland surface and critical flood-prone areas, it becomes possible to make responsible decisions concerning restoration. Three model cases are presented here: (1) the sluice gate opened 130 cm allowing both ebb and flood flow and two tide gates allowing only ebb flow (see Figure 2); (2) sluice gates opened 25 cm in all three dike openings; and (3) no restrictions through the structure (no sluice or tide gates). Even under case 3, considerable restriction of tidal flow by the actual dike structure and roadway across Herring River will still occur.

Model Type and Formulation. There are two principal parts to the model formulation: a momentum balance equation and a volume conservation (continuity) equation, which accounts for the accumulated volume in the reservoir (i.e., tidal basin upstream of the dike). The momentum balance is simply one between the horizontal pressure gradient produced by the water height difference across the dike and opposing friction applied at the channel surface. Consequently, for any given channel through the Herring River dike, the volume flux Q_i (where i denotes one of three channels, i = 1, 2, 3) is simply proportional to the square root of the water height difference $Y_d - Y_u$ (where Y_d is the height at the downstream side of the dike and Y_u at the upstream side). In practice, the proportionality coefficient is determined empirically. Manning's law is used (Linsley and Franzini 1979) such that the momentum balance has the form

$$Q_i = \frac{A_i^{5/3}}{n_i P_i^{2/3}} \left[\frac{|Y_d - Y_u|}{L_d} \right]^{1/2} \frac{(Y_d - Y_u)}{|Y_d - Y_u|} \qquad (1)$$

Here $Q_i(t)$ is the volume flux passing through channel i at time t in cubic meters per second, n_i is Manning's coefficient for the channel, A_i the cross-sectional area in square meters of the flow, P_i the wetted perimeter in meters (length of the intersection in the cross-sectional plane in contact with the solid surface), and L_d the channel length for the dike. A_i and P_i are simple time-dependent functions of the average water depths in the channel, and hence are easily computed for any given Y_d and Y_u. The Manning coefficient, n_i, however, must be determined by calibration and must also be time dependent if a sluice gate is present in the channel.

The second principal component of the model is given by the mass conservation or continuity equation applied to the river.

$$\frac{dV_u}{dt} = A_u \frac{dY_u}{dt} = Q + Q_f \tag{2}$$

Here, V_u *(t)* is the upstream volume of water. Its time rate of change is simply the horizontal area of the water surface A_u *(t)* times the rate of surface rise. The change in volume is produced solely by the total inflow through the dike $Q = Q_1 + Q_2 + Q_3$ and the freshwater inflow Q_f. The latter, however, is neglected, since its typical value, 0.25 m^3/sec, is small compared to typical empirical values of tidal flux, about 5.0 m^3/sec, and maximum flux *(Q)* often above 10 m^3/sec.

For a simple vertical-sided reservoir, A_u is constant. For the Herring River, however, A_u is a strong function of water height, Y_u, with rising levels permitting the water to first rise within the channels of the basin and subsequently to flow over the channels onto the marsh surface. By using bottom profiles for the main channel and topographic relief data for the wetland surface, a piecewise linear function for A_u (Y_u) was developed. This function accounts for the fact that, as the water in the system reaches a level above the channel banks, the area available for water storage (i.e., as sheet flow over the marsh surface) is significantly increased. This increase in A_u sharply reduces the water level, Y_u, that would otherwise occur in confined channels, or a simple reservoir.

Calibration and Testing. Water heights were simultaneously measured (at hourly intervals) from four tide staffs located immediately upstream of the dike to 3 km upstream and from one staff downstream of the dike over three separate semidiurnal tidal periods. These data were used to perform calibration trials for selecting optimal values of the friction parameters (e.g., Manning's coefficient, n_i) and subsequently to test the model predictions for Y_u *(t)* against the measured upstream level. For the three calibration/testing periods, both tide gates were operative but the sluice gate openings differed (130 cm, 61 cm, 51 cm). Figure 4 plots the model computed (Y_m) and measured (Y_u) tide height for the calibration runs. As noted, Y_m and Y_u were close at all phases of the tide, with negligible root mean square deviations for the three runs ranging from 0.06 to 0.09 m. In fact, no objective improvement in this agreement is physically realistic, since wind-induced height fluctuations within the system (which are likely present in the data but not accounted for by the model) would typically be a few centimeters.

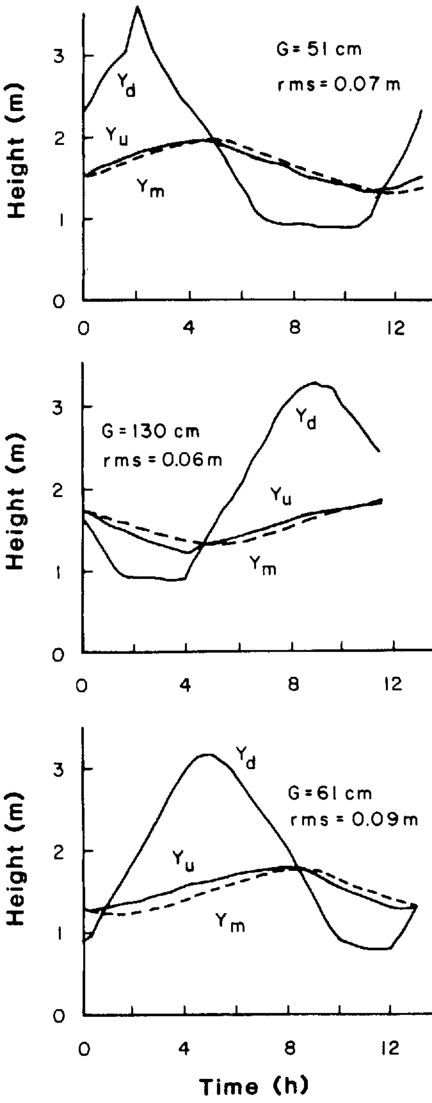

Figure 4. Water levels over semidiurnal tidal periods, relative to mean low water, upstream of the dike from measured field data (Y_u) and model computed (Y_m). Measured water levels downstream of the dike are also shown (Y_d). Sluice opening (G) and root mean square differences (rms) are also indicated.

Model errors for predicting upstream water levels for dike openings greater than those tested would probably increase, yet even a doubling in uncertainty (i.e., 0.18 m root mean square deviation) would be

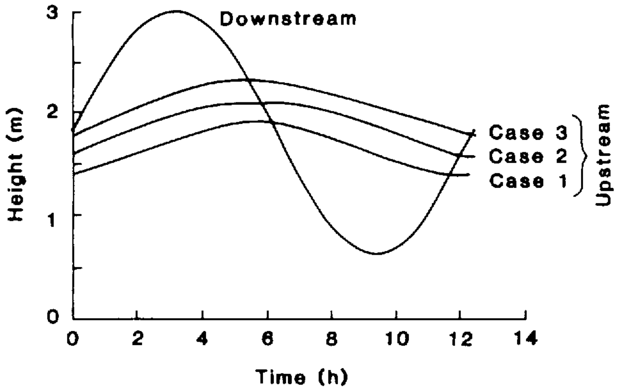

Figure 5. Comparison of tide height model predictions for cases 1–3. A downstream semidiurnal tidal period, with a mean range of 2.4 m is also shown. Note the phase lag between high water downstream and upstream of the dike.

sufficiently small to allow for practical application of the model.

Results and Discussion

Model Predictions

The effect of increasing the dike opening is to increase the water level almost uniformly over the tidal cycle (Figure 5). Tidal range increases little (Table 2). From case 1, with severe restrictions, to case 3 (no restrictions through the dike), the high water level increased by about 0.4 m over the cycle. In contrast, the mean tidal prism increased over twofold. These results follow from the increasingly large horizontal area (A_u) available as the restrictions diminish. As the mean water level over a cycle increased, more water was diverted laterally to begin flooding the system. Between low water for case 1 and high water for case 3, A_u increased by a factor of >4. This increased storage area compensates for the considerable tidal volume increase.

Even with no sluice or tidal gates (case 3), the actual dike structure and roadway across Herring River still restrict tidal exchange. The mean high water level downstream of the dike was empirically estimated at about 3.2 m, almost 1 m greater than predicted for the case 3 configuration.

The model was also used to predict tide heights for the three cases under conditions simulated by a 100-yr storm event (February 1978 storm). With two tide gates operative and the sluice gate open 61 cm (slightly more restrictive than case 1), the river height peaked at 2.08 m; with three sluice gates open 25 cm (case 2), it peaked at 2.34 m; and with no gates (case 3) at 2.76 m. During the February 1978 storm, maximum high water downstream of the dike approached 4.6 m (Boston tide gauge station, National Oceanic and Atmospheric Administration).

The elevations at which the residential dwellings and the golf course will be flooded are 2.10 m and 2.14 m, respectively. It becomes clear that with implementation of any of the cases presented here, flooding during normal tide conditions and especially during a 100-yr storm event will occur. Tide height model predictions were also used in conjunction with groundwater geophysical studies to conclude that reintroduction of tidal flow would not result in contamination of domestic wells adjacent to the Herring River by seawater intrusion (Fitterman and Dennehy 1991). Thus, the model can be successfully used to identify cultural consequences of restoration, and if necessary, provide a basis for recommending appropriate alternatives for protection of those impacted human features.

Model Applications at Other Sites

The Herring River tide height model can be successfully applied at other potential restoration sites. In 1930, about 80 ha of the Hatches Harbor salt marsh (Provincetown, Massachusetts) were diked, dramatically restricting tidal flow in an effort to control mosquito populations. Shortly thereafter an airport was constructed within the Hatches Harbor floodplain, deriving protection from the dike. Tidal flow is restricted through a 60-cm-diameter circular culvert. The tide height model was used to determine an appropriate culvert configuration that will maximize marsh restoration goals, while protecting airport facilities. At Hatches Harbor, the existing earthen dike allowed an unlimited variety of configurations (i.e., circular or rectangular culverts, one culvert or several in series) to be modeled, whereas at Herring River the dike structure is much more substantial, thus limiting the selection of configurations. For Hatches Harbor, a principal design criterion was that during a 100-yr storm the maximum water height upstream of the dike would not exceed 3.05 m above mean low water to protect the airport. As noted from Table 3, several configurations can be modeled to approximate the design criteria.

A major outcome of the Hatches Harbor tide height modeling revealed that restricting culvert height dampened storm tide levels in the marsh because, once sea level exceeds the culvert top, turbulence through the culvert increases exponentially. Restricting culvert height allows maximum flow during normal tides (i.e., with wide culverts, 6 m), while filtering out storm tides that may exceed the 3.05-m

Table 2. Predicted high water, low water, range, and tide volume for three model cases at the Herring River dike[a]

					Volume	
Case	Configuration	High (m)	Low (m)	Range (m)	Ebb ($\times 10^5 m^3$)	Flood ($\times 10^5 m^3$)
1	1 sluice gate at 130 cm, 2 tide gates	1.93	1.38	0.55	1.02	0.99
2	3 sluice gates at 25 cm	2.11	1.60	0.52	1.39	1.36
3	no gates/sluice	2.33	1.80	0.54	2.34	2.25

[a]Predictions are for upstream of the dike. Heights are relative to mean low water. For these model runs tide range downstream of the dike was assumed to be 2.4m, a typical mean.

Table 3. Predicted high water, low water, range, and intertidal volume for various culvert configurations at the Hatches Harbor dike[a]

	Mean tide				Storm tide
Culvert configuration	High water (m)	Low water (m)	Range (m)	Intertidal volume ($\times 10^4 m^3$)	High water (m)
Rectangular culvert					
6.1 m wide, 2 m high	2.74	2.29	0.46	7.11	4.05
6.1 m wide, 0.76 m high	2.67	2.24	0.42	5.18	3.10
Circular culvert					
1 culvert, 0.9 m diameter	2.55	2.45	0.09	1.08	2.74
4 culverts, 0.9 m diameter	2.65	2.35	0.30	4.25	3.09

[a]Predictions are for upstream of the dike. Heights are relative to mean low water. High water level for a stimulated "100-yr" storm are also presented.

threshold to protect the airport facilities. The conventional circular culverts, although able to approximate the threshold criteria, resulted in substantially reduced tidal range and intertidal volume when compared to rectangular configurations (Table 3).

Balancing Ecological and Cultural Considerations

Predictions from the tide height model provide the basis for suggesting ecological changes and cultural changes that may occur within the marsh-dominated ecosystem with implementation of restoration actions. For example, knowledge of tidal flooding characteristics, coupled with data on marsh topography and salinity predictions from a companion circulation model, enabled a well-informed indication of expected trends in vegetation, water quality, fish and shellfish distribution, and waterbird utilization with restoration management. Similarly, the potential for flooding of cultural features that have encroached upon the wetland (i.e., roads, residential dwellings, golf course, airport facilities) can be predicted. Given an appreciation for ecological benefits and identification of cultural impact thresholds, decision makers can effectively assess the feasibility of restoration.

To enhance success, planning for wetland restoration projects must be based on scientific information that enables both researchers and natural resource managers to predict the outcome (National Research Council 1992, Zedler 1992). The tide height model presented in this paper has universal application and provides a well-justified basis for designing tidal marsh restoration projects, and moreover, for predicting ecosystem responses.

Restoring ecologically degraded salt marsh ecosystems is a valuable management tool toward maintaining and preserving habitat diversity in the coastal zone. Ecological restoration will undoubtedly become a critical technique on a global scale as pristine habitats become scarce (Jordon and others 1988, Zedler 1988).

Acknowledgments

This research was funded by the National Park Service, Water Resources Division, and administered through the National Park Service Cooperative Research Unit at Rutgers University and the Cooperative Park Studies Unit at the University of Rhode Island. Thanks are extended to Gudrun Marteinsdottir, Richard Orson, and Raymond Grizzle for their assistance with ecological data collections.

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