Smolders, Sven; Meire, Patrick; Temmerman, Stijn; Cozzoli, Francesco; Ides, S.; Plancke, Yves M. G.

A 2Dh Hydrodynamic Model of the Scheldt Estuary in 1955 to Assess the Ecological Past of the Estuary

TUC 2013, 16-18/10/2013, Karlsruhe - Kopmann, Goll (eds.) - © 2013 Bundesanstalt für Wasserbau ISBN 978-3-939230-07-6

A 2Dh hydrodynamic model of the Scheldt estuary in 1955 to assess the ecological past of the estuary

S. Smolders, P. Meire, S. Temmerman
Ecosystem Management Research Group
University of Antwerp (UA)
Antwerp, Belgium
svensmolders@gmail.com

F. Cozzoli
Spatial Ecology
Netherlands Institute of Sea Research (NIOZ)
Yerseke, The Netherlands

S. Ides
Antwerp Port Authority
Antwerp, Belgium

Y. Plancke
Flanders Hydraulics Research (FHR)
Antwerp, Belgium

***Abstract*—The macrozoöbenthos model of Cozzoli et al. [1] uses maximum flow velocity, inundation time, daily averaged salinity and daily salinity range to assess habitat suitability. A 2Dh hydrodynamic TELEMAC model of the Scheldt Estuary with a bathymetry of 1955 was built and calibrated for water levels. The hydrodynamic parameters and the tracer values for salinity are used as input for the macrozoöbenthos model to assess the ecological situation of the estuary in 1955. This paper discusses mainly the difficulties of retrieving the right boundary conditions and parameter settings for this model. The mesh resolution influences model parameters like tracer diffusivity and velocity diffusivity so they can't be taken from a different calibrated model.**

I. INTRODUCTION

Estuaries worldwide form the transition zone between sea/ocean and the inland rivers; between fresh and salt water. This transition zone knows a unique ecological richness typical for estuaries [2]. Due to human pressure: expanding economical activities, cities and ports, this unique ecological niche has to be protected. Most parts of the Scheldt Estuary in the Netherlands and Belgium are protected as Natura 2000 area. Structural changes to these areas can only be done if the ecology is not harmed or is even benefitting. It is however very difficult to assess the impact of future changes on the ecology. Ecotope maps and habitat maps are made for the ecological evaluation of the system. The creation of such maps is a time consuming task, since a lot of parameters are necessary to create them: depth maps, soil composition, wave action, current velocity, geomorphology, salt and possibly nutrients. For some of these data field samples have to be taken. This method of evaluation has proven its value, but is not useful for scenario analysis, where quick results are needed for the assessment of different possible future scenarios.

Cozzoli et al. [1] developed a model to predict the habitat suitability for macrozoöbenthos. This model uses only four parameters: maximum flow velocity, inundation time, daily averaged salinity and daily salinity range. These parameters are easily available from a hydrodynamic model. This benthos model is an ideal tool for the assessment of ecological changes caused by morphological changes. The macrozoöbenthos fauna is the trophic level between basic nutrients, algae and higher trophic levels like birds and fish. Different morphological scenarios can be simulated using only a 2Dh hydrodynamic model to generate the four parameters necessary for the benthos model. For every point in a models mesh the habitat suitability for macrozoöbenthos species can be predicted and this for different scenarios. In this way maps can be created which can be compared with a reference map. With the help of some guidelines from an ecologist, for example that the total biomass needs to increase in a certain area or a specific species is needed more in another area; a modeler can apply and assess the changes in benthos habitat suitability within minutes.

In the same way that we can assess the changes in benthic habitat suitability for future scenarios, we can assess the changes for a model with the bathymetry and boundary conditions of some past period. Comparing it with the current situation will give new insight in the ecological development of the estuary. In this paper we discuss the building of a model of the Scheldt Estuary with the bathymetry of 1955 and the problems encountered doing this: model boundary conditions and the value for the tracer diffusivity (no measurements available). Model simulation results will be used as input for the macrozoöbenthos model.

II. 2DH TELEMAC MODEL OF SCHELDT ESTUARY IN 1955

A. Scheldt Estuary

The Scheldt Estuary extends from Vlissingen, The Netherlands (km 0) to Ghent, Belgium (km 160). At Merelbeke and Gentbrugge (Ghent area) weirs prevent the tide from penetrating more upstream. The tidal influence reaches to major tributaries (they are included in the model, Fig. 1). Discharges of the Scheldt and tributaries are negligible compared to the tidal volume. The estuary is well mixed, which means that vertical salinity gradients are small or negligible [3]. Tide gauge measurement stations are numerous but only the ones used for calibration of water level data are given in Fig. 1: Cadzand, Westkapelle, Vlissingen, Terneuzen, Hansweert, Antwerpen, Hemiksem, Temse, Sint-Amands, Dendermonde, Boom, Walem, Duffel. In 1955 the average tidal range at the mouth was 3.82 m NAP (Normaal Amsterdams Peil ~ mean sea level) and at Antwerpen it was 4.86 m NAP.

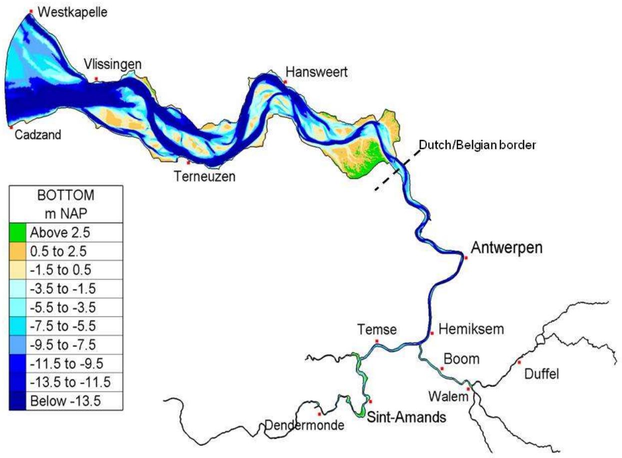

Figure 1. Area and bathymetry of Scheldt Estuary model in 1955. Red dots indicate tide gauge measurement stations used for calibration of the model.

B. Mesh

A new mesh was built for this model using Blue Kenue. The outline was taken from another TELEMAC model of the Scheldt Estuary. It was adapted to use for the 1955 model based on the bathymetry data sets that were available. Upstream Hemiksem (Fig. 1) the mesh of the 2009 model was reused. This area is of less interest for the research and was created with the channel mesher, as the flow direction is pseudo 1D and to save on calculation nodes. Downstream Hemiksem a mesh resolution of 40m was used to have a very high resolution for the benthos model. In the mouth area a mesh resolution of 300m was used quickly decreasing (Edge growth ratio 1.08) towards the 40m resolution just downstream from Vlissingen (Fig. 1).

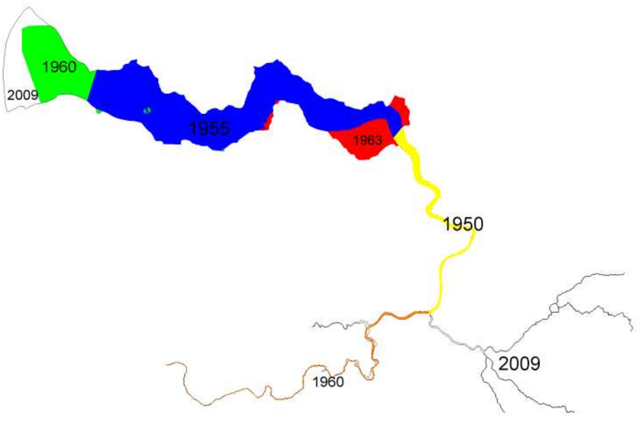

Figure 2. Indication of which bathymetry data of which year is used in what part of the model.

C. Bathymetry

Fig. 2 shows a map of the model with different colours. Each colour represents a different year of which the bathymetry data was used for the model. The mouth area consists of bathymetry of 1960. The main part of the Western Scheldt (Fig. 2, blue colour) consists of bathymetrical data from 1955; it is the area of interest for our research. For the larger marsh areas (Fig. 2, red colour) data from 1963 were used. The Sea Scheldt, downstream Hemiksem, (Fig. 2, yellow colour) consists of a bathymetry from 1950. The Sea Scheldt, upstream Hemiksem, (Fig. 2, brown colour) consists of a bathymetry from 1960. The bathymetry data of the tributaries (Fig. 2, grey colour) was kept the same as in the 2009 model. The resolution of the grid in this part of the model is too large for an accurate bathymetrical representation. Water movement in this upstream part of the model was adjusted using the bottom friction coefficient.

D. Boundary conditions

The downstream boundary condition is a free surface boundary with the water level time series that was recreated for this model. The upstream boundaries are discharges. There are six upstream discharge boundaries.

1) Water level at estuary mouth

No continuous water level measurements of 1955 are available. For the model we need a time series of water levels for 1955 or close to this date with a ten minute interval for Cadzand to implement as boundary on the model. A three hour interval time series of water levels for Vlissingen for 1954 was available together with all low and high water levels. With this information a boundary condition time series with a 10 minute interval could be made following the next steps:

a) Harmonic analysis of the three hour interval time series of Vlissingen for 1954 was executed in Matlab using t_tide [4]. Only the tidal constituents that had a sound to noise ratio higher than 5 were used in the following steps. An offset of -0.056m compared to the 0m NAP level was found in the harmonic analysis.

b) With the found harmonic components a new complete, ten minute interval, time series of the water level of Vlissingen was created using t_predict in Matlab [4], i.e. an astronomical time series.

c) A correction for the offset was made to the astronomical time series by adding +0.056m.

d) For the whole of 1954 the difference between high water level measured and the high water level calculated, i.e. the astronomical tide, was taken. The same was done for the low water levels. These differences were averaged over a 14 day period, i.e. a neap spring tidal cycle, with calculation of the RMSE. We want to find a 14 day period in 1954 were the difference between measured and reconstructed tide is the smallest. The calculation time window was moved with an interval of one week. For one year 49 average differences were calculated. The period from 26[th] of October till the 8[th] of November (week 42) showed the smallest average difference

(0,28m). The RMSE values per week of 1954 are given in Fig. 3.

e) The difference between the astronomical tide time series and the measured high and low water levels is caused by meteorological influences. The astronomical time series is corrected based on the measured values and this is done per high or low water separately. For a certain high water (HW) all points in the time series of that tide, higher than 0m NAP, are multiplied with the following factor: (HW measured/HW astronomical). This is done for every single high and low water in the selected spring neap tidal cycle.

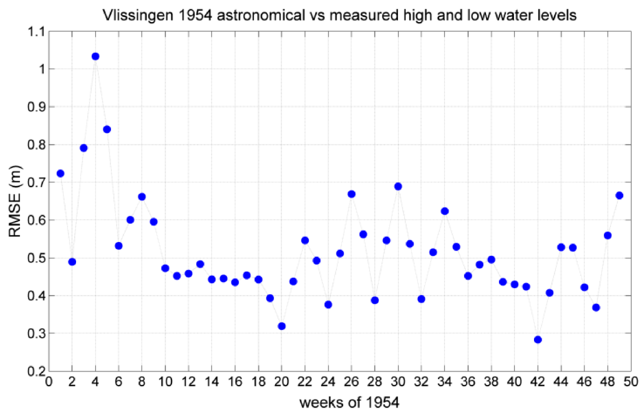

Figure 3. RMSE values for 2 week interval average differences between measured and reconstructed astronomical tide of 1954.

f) If the difference in phase of high or low water between measured and astronomical tide is larger than 10 minutes, the phase is corrected for the particulate high or low water. The time interval will stay at 10 minutes, so if a correction is made, one or more points are removed or added to the same tide. All points (water level every 10 minutes) in between the high and low water, that have the phase difference, need to be corrected and were resampled. The result is a perfectly reconstructed time series of the water levels in Vlissingen in 1954 (Fig. 4).

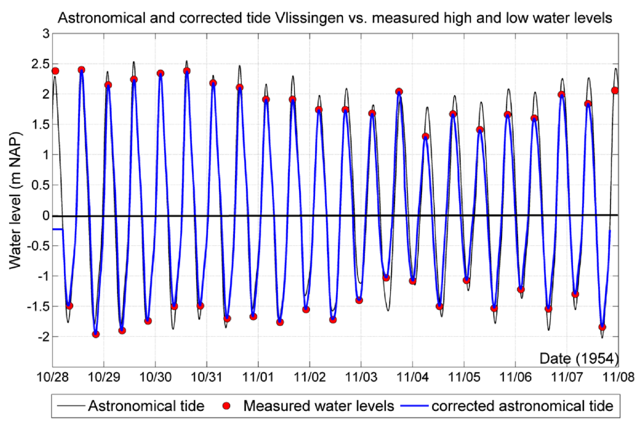

Figure 4. Astronomical tide compared with measured high and low water levels and the corrected tide for water level and phase.

g) For the model boundary we need this time series for Cadzand and not for Vlissingen. So an extra correction has to be made for the water level and phase. A QQplot was made for the water levels at Vlissingen and Cadzand. For 1954 the water level of Cadzand is on average 0.03m higher than the water level of Vlissingen. Applied only on the used spring neap tidal cycle the difference was only 0.005m. So 0.005m was added to the time series of Vlissingen. The same was done for the phase and high water occurred 34 minutes earlier in Cadzand compared to Vlissingen in 1954. The same was done for Westkapelle. The final boundary condition time series was found by taking the average of the Cadzand and Westkapelle time series and add 0.04m to it.

2) Discharges upstream boundaries

The discharges in the Scheldt Estuary didn't change much over the last 60 years. There is a seasonal variation and big influence of individual rain events. No daily averaged discharge data were available for the simulation period October-November 1954, therefore an average discharge per boundary was calculated taking the daily averaged discharges of October and November of the last ten years into account. These values were used as fixed continuous discharge values for the upstream freshwater discharge boundary conditions. For the Scheldt River, Dender, Zenne, Dijle, Grote Nete and Kleine Nete these values are respectively 28m³/s, 8m³/s, 10m³/s, 18m³/s, 4m³/s and 5m³/s.

III. CALIBRATION

The model was calibrated using a Manning bottom friction coefficient for areas in between tide gauge measurement stations. No measurements of flow velocities or discharges were available for 1954. High and low water levels and their time of occurrence were available for Vlissingen, Terneuzen, Hansweert, Antwerpen, Hemiksem, Temse, Sint-Amands, Dendermonde, Boom, Walem and Duffel (Fig. 1). Per station measured high and low water levels are compared with the simulated ones. Changing the bottom friction coefficient between two stations will affect the water levels. A cost function is used to evaluate the calibration effort of a simulation. Since only the part of the model downstream Antwerpen is of major interest, these stations are given a higher weight factor (δ) then the ones upstream Antwerpen. The total sum of all weight factors should be equal to one, as shown in (1). The stations Vlissingen, Terneuzen, Hansweert and Antwerp were given four times the weight of the station more upstream and Hemiksem was given twice the weight of the stations upstream. For the calibration 16 high and 16 low water levels were compared. An average difference (D_{avg}) in water level (WL) was calculated per station for these high and low water levels (n = # high or low water levels used for calibration) separately as shown in (2).

$$\sum_{i=1}^{m} \delta_i = 1 \quad (1)$$

$$D_{avg}(WL) = \frac{\sum_{j=1}^{n}(WL_{meas.}(j) - WL_{model}(j))}{n} \quad (2)$$

Finally the cost function summarizes all stations with their own weight factor, as shown in (3). The cost function gives a value for high and low water separately and for their phase

$$Cost = \sum_{i=1}^{m} \delta_i * D_{avg}(WL)_i \quad (3)$$

To start the first calibration simulation the Manning coefficients of a 2009 model were used in a simplified form. In total five calibration steps were simulated of which the fourth one gave the best results. The cost function for high and low water levels together gave a weighted average difference of 0.05m for the first simulation. The cost function for the Root-Mean-Square-Error (RMSE) on these water level differences gave 0.14m. For the phase the cost functions weighted average delay was 18 minutes with RMSE of 21 minutes. These results improved and the fourth simulation gave a cost function weighted average on the water levels of 0.05m, the same result as the first simulation, but the RMSE dropped to 0.08m. The cost function value for the phase improved to 11 minutes with a RMSE of 14 minutes. The final Manning coefficient values used in the fourth simulation are shown in Fig. 5.

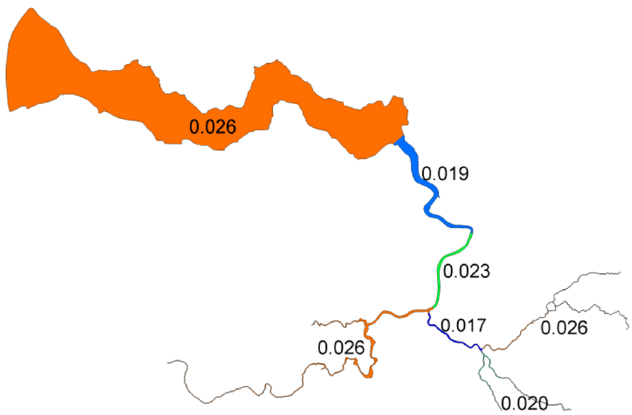

Figure 5. Manning coefficients for the best calibration result.

Although several tide gauge measurement stations are present in the downstream part of the estuary (orange part in Fig. 5), one general Manning coefficient of 0.026 gave the best results for this part. Only results from the calibration period will be used further, so we did not validate it further.

IV. SALINITY

A. Salinity as passive tracer in 1954 model

Salinity was added to the model as a passive tracer. There are no salinity data available for 1954, so the tracer could not be calibrated for this model. Salinity as a passive tracer was already successfully calibrated in another Scheldt Estuary TELEMAC model representing the bathymetry of 2009 [to be added]. The parameter value for the tracer diffusivity, i.e. 0.8m²/s, was taken from this model and applied in the 1955 model. Where we expected the same salinity intrusion or less, because of a smaller tidal range, the simulation results gave a much higher salt intrusion (Fig. 7). Because there were no data to compare the results with, further test were done to confirm or refuse the models salinity results.

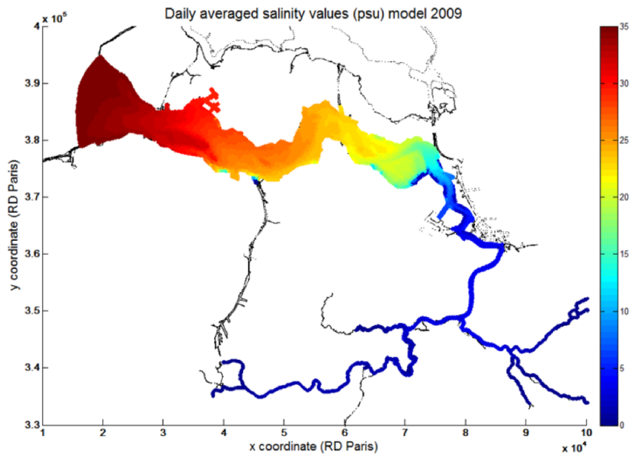

Figure 6. Daily average salinity values (PSU) in model of 2009.

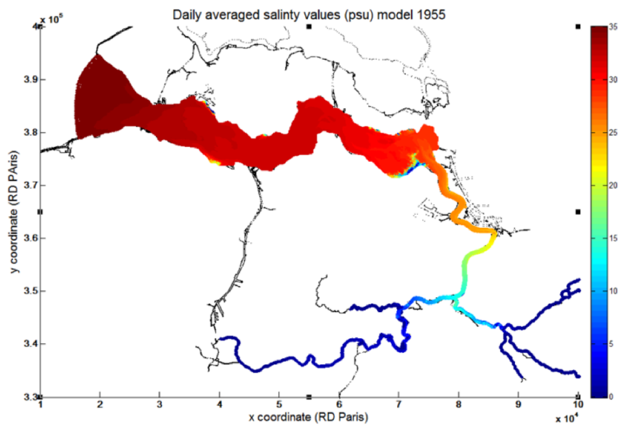

Figure 7. Daily averaged salinity values (PSU) for model of 1955 using the same tracer diffusivity value as the model of 2009.

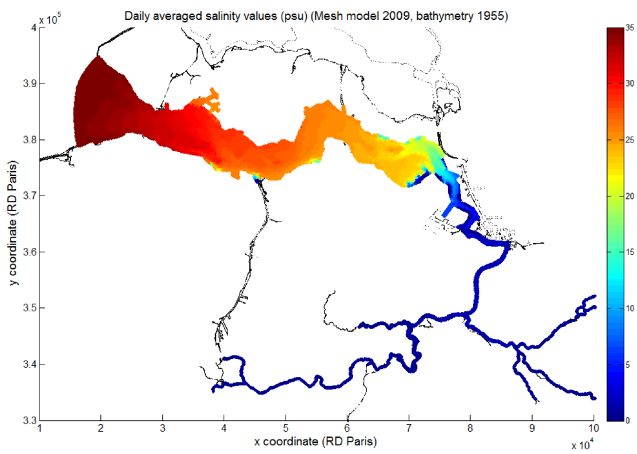

Figure 8. Daily averaged salinity values (PSU) for model with mesh of 2009 model and bathymetry and boundary conditions of 1955.

The total diffusion in the model is always the sum of the natural diffusion, i.e. the tracer diffusivity value given by the user, and the artificial or numerical diffusion in the model. This artificial diffusion is influenced by model parameters like the mesh size and the time step. By changing the mesh resolution, one changes the total diffusion in the model. Since the mesh resolution in the model of 1955 is much finer (40m) than that of the model of 2009 (150m), the diffusion and the salinity results in the model of 2009 (Fig. 6) will not be comparable with the diffusion and results of the 1955 model (Fig. 7). In fact the finer mesh resolution of the 1955 model increased the total diffusion in the simulation, giving a wrong salinity distribution in the estuary. With no calibration data and a different mesh than the 2009 model it was impossible to get the salinity value right in this model.

The solution for the salinity calibration problem was to apply the 1955 models bathymetry on the mesh of the 2009 model, for which the salinity influencing model parameter values were calibrated and known. Using also the 1954 boundary conditions gave salinity results that were in the line of expectations (Fig. 8). Salinity results of this simulation were mapped on to the finer mesh of the 1954 model using Blue Kenue in order to have salinity values for every node of the 1954 model mesh.

This method of using another calibrated and validated model of the same area to simulate tracer values for a period for which no data were available, can also be turned around: the bathymetry and boundary conditions data of 2009 could be used on the 1954 model mesh and then the salinity could be calibrated for this mesh and time step. This way demands a new round of calibrating the salinity data, which is time consuming, and therefore it was chosen to do it the other way. It might still be a good solution if it was absolutely necessary to have the salinity diffusivity parameter value for the 1954 model mesh.

B. Tracer sensitivity

1) Mesh size

Every physical term in the TELEMAC model will have a different reaction to mesh size because of the numerical or artificial diffusion. The fact that mesh resolution plays an important role in the artificial diffusion of advection schemes is shown in the following example: the artificial diffusion of a simple 1D upwind scheme in finite differences is given by (U*dx)/2, where U is the velocity and dx the mesh size. The artificial diffusion may completely mask the values given by turbulence models, especially in free surface flows where the mesh size may be several kilometres [5].

To have an idea of the influence of the mesh size on the salinity distribution in both the 2009 and the 1954 model, all parameters were kept the same in two simulations where only the mesh resolution was altered. The initial tracer value was set to zero. The result for a certain point (the same location in both meshes) is given in Fig. 9. The higher mesh resolution clearly speeds up the diffusion of the tracer. This example shows that an increasing mesh resolution increases the artificial diffusion in the model.

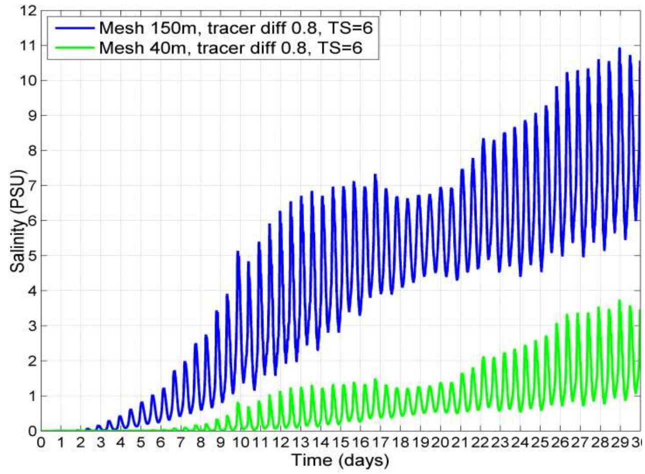

Figure 9. Influence of mesh resolution on tracer (salinity) values while all other parameters were kept constant.

Since all physical parameters are affected by artificial diffusion the result seen in Fig. 9 is thus not only the result of a changed tracer diffusion. It is a result of the influence of a changing mesh resolution on artificial diffusion and thus on all physical parameters in the model. This must explain why the salinity diffusion in the 1955 model, although the mesh resolution had decreased and thus decreasing the numerical diffusion, is higher than in the 2009 model with its coarser mesh.

2) Time step

To meet the Courant criterion the time step will be reduced with reducing mesh size. With a different time step the advection schemes may behave differently. For example the method of characteristics is less diffusive in one step than in two half steps. [5].

The salinity diffusion in the 2009 model was tested for two different time steps, i.e. 6 and 60 seconds. A smaller time step will result in larger salinity diffusion. The effect of the time step on the salinity diffusion in our model was small, i.e. less than half a PSU unit.

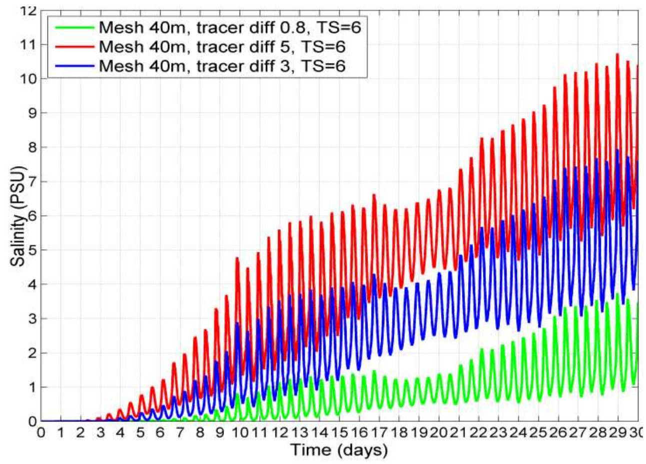

Figure 10. Effect of the tracer diffusivity value on tracer diffusion, while all other parameters were kept constant.

3) *Tracer diffusivity*

Finally, the effect of different values for the tracer diffusivity parameter was tested to compare their impact on tracer diffusion with the effect of especially mesh resolution. All other parameters were kept constant while the tracer diffusivity parameter values altered from 0.8 to 3 to 5m²/s. The results are shown in Fig. 10. In increase in tracer diffusivity from 0.8 to 5m²/s shows the same effect on the salinity distribution as a mesh resolution increase from 40m to 150m.

V. Application of Benthos model

For every node in the mesh of the model the maximum flow velocity, the inundation time, the average salinity and the salinity range are calculated over a single spring tide. These are the input parameters for the macrozoöbenthos model of Cozzoli et al. [1,6]. Without going further into detail about this model, an example is shown in Fig. 11, where two maps of the Western Scheldt are shown, i.e. the situation in 1955 and the one in 2010, giving the bivalve, Macoma baltica, expressed in g/m². The figure also shows the difference in total tons of ash-free dry weight (AFDW) of M. baltica for the entire estuary between 1955 and 2010. Regarding the numbers given in the figure it is important to realize that the output of the benthos model is expressed in potential of biomass (0.95 percentile) because this gives a better estimation of the habitat suitability [7, 8]. More information about the difference in Macrozoöbenthos between 1955 and 2010 and the difference between Eastern and Western Scheldt can be found in [6].

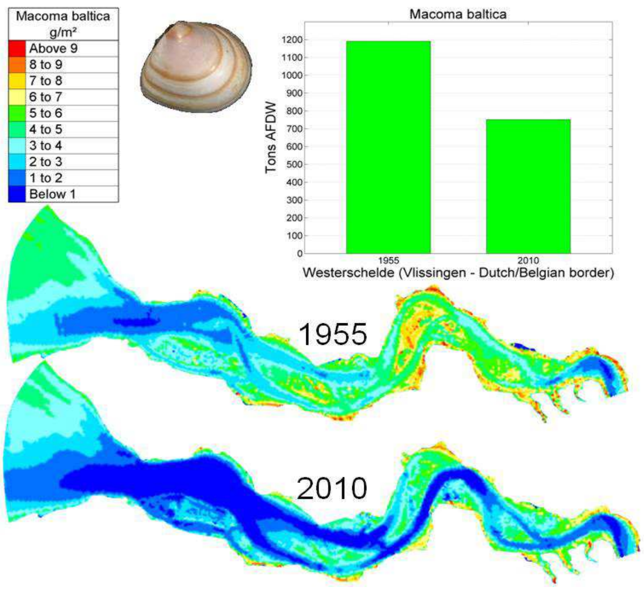

Figure 11. Maps of the Western Scheldt showing the bivalve, M. baltica, expressed in g per m² in 1955 and 2010. The total ash-free dry weight difference for the estuary between 1955 and 2010 is given in the bar chart.

VI. Conclusions

With the knowledge and help of a Scheldt Estuary model of 2009 a new model was calibrated for 1955. A seaward water level boundary time series was re-created using harmonic analysis. Calibration was done with measured high and low water values. Salinity was modeled as a passive tracer on the mesh of the 2009 model using boundary conditions and bathymetry of the 1955 model, because the 2009 model was calibrated for salinity. The tracer diffusivity is the sum of the parameter value and the numerical diffusivity. The mesh size, time step, velocity diffusivity and tracer diffusivity value determine the total diffusion of the salinity in the model.

A reliable calibrated 2Dh hydrodynamic model of 1955 with a complete salinity gradient provides all the information (maximum flow velocities, inundation time, average salinity and salinity range) for the application of a macrozoöbenthos model to compare the present (2009) and past (1955) ecological state of the Scheldt Estuary.

Acknowledgement

This research was financially supported by the Port of Antwerp and the Dehousse scholarship of Antwerp University. The authors like to thank Flanders Hydraulics Research for the cooperation and the Canadian Hydraulics Centre for the Blue Kenue software. Finally our gratitude goes to the TELEMAC consortium for making the TELEMAC software freely available and the great support they offer via the website!

References

[1] F. Cozzoli, M. Eelkema, T. Bouma, T. Ysebaert, V. Escaravage, P.M.J. Herman, 2013. A mixed modeling approach to predict the effect of environmental modification on species distributions. “unpublished”.

[2] R. Costanza, R. d’Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. O’Neill, J. Paruelo, R. Raskin, P. Sutton, M. van den Belt, 1997. The value of the world’s ecosystem services and natural capital. *Nature* 387, 253–260.

[3] W. Baeyens, B. van Eck, C. Lambert, R. Wollast, L. Goeyens, 1998 General description of the Scheldt estuary. *Hydrobiologia* 34, 83-107.

[4] R. Pawlowicz, B. Beardsley, and S. Lentz, 2002. "Classical Tidal Harmonic Analysis Including Error Estimates in MATLAB using T_TIDE", Computers and Geosciences, 28, 929-937.

[5] Jean-Michel Hervouet, personal communication on TELEMAC forum (www.openTELEMAC.org) June, 2013.

[6] F. Cozzoli, S. Smolders, M. Eelkema, T. Bouma, T. Ysebaert, V. Escaravage, P. Meire, S. Temmerman, P.M.J. Herman, 2014. Antropocene in estuaries: effects on macrozoobenthos. “unpublished”.

[7] M.J. Anderson, 2008. Animal-sediment relationship re-visited: Characterising species’ distributions along an environmental gradient using canonical analysis and quantile regression splines. Journal of Experimental Marine Biology and Ecology, 366 (2008) 16-27.

[8] F. Cozzoli, T. Bouma, T. Ysebaert, P.M.J. Herman, 2013. Application of non-linear quantile regression to macrozoobenthic species distribution modelling: comparing two contrasting basins. Marine Ecology Progress Series 475 (2013) 119-133.