Modelling of marine radionuclide dispersion in IAEA☆ MODARIA program: Lessons learnt from the Baltic Sea and Fukushima scenarios

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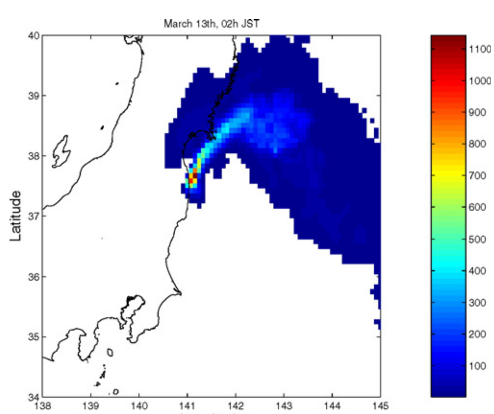
HIGHLIGHTS

- Models applied to simulate ^{137}Cs marine dispersion after nuclear accidents.
- Not good agreement initially found in highly dynamic environments.
- Difficulties in developing models for decision making after emergencies highlighted.

GRAPHICAL ABSTRACT



ABSTRACT

State-of-the art dispersion models were applied to simulate ^{137}Cs dispersion from Chernobyl nuclear power plant disaster fallout in the Baltic Sea and from Fukushima Daiichi nuclear plant releases in the Pacific Ocean after the

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Baltic Sea
Chernobyl
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Hydrodynamics
Dispersion model
Caesium

2011 tsunami. Models were of different nature, from box to full three-dimensional models, and included water/sediment interactions. Agreement between models was very good in the Baltic. In the case of Fukushima, results from models could be considered to be in acceptable agreement only after a model harmonization process consisting of using exactly the same forcing (water circulation and parameters) in all models. It was found that the dynamics of the considered system (magnitude and variability of currents) was essential in obtaining a good agreement between models. The difficulties in developing operative models for decision-making support in these dynamic environments were highlighted. Three stages which should be considered after an emergency, each of them requiring specific modelling approaches, have been defined. They are the emergency, the post-emergency and the long-term phases.

1. Introduction

The International Atomic Energy Agency (IAEA) has organized programmes on radioactivity dispersion model testing since the VAMP (Validation of Model Predictions) program in 1988 (see IAEA, 2000, for the aquatic group work). The most recent effort is the MODARIA[1] (Modelling and Data for Radiological Impact Assessments) project, launched in 2012. Ten working groups were organized in four main topics: Remediation of Contaminated Areas, Uncertainties and Variability, Exposures and Effects on Biota, and Marine Modelling.

Because of recent developments in marine science and marine modelling, as well as the radioactive pollution due to the Fukushima Daiichi nuclear power disaster in March 2011, it was considered worthwhile to carry out a new exercise on dispersion model comparisons for the marine environment. Marine modelling draws special attention after the catastrophic earthquake and tsunami which severely damaged the Fukushima Daiichi Nuclear Power Plant (FDNPP) and resulted in uncontrolled release of radioactivity into the air and ocean. Approximately 80% of the radioactivity released due to the accident in March–April 2011 was either directly discharged into the ocean or deposited onto the ocean surface from the atmosphere (Povinec et al., 2013). ^{137}Cs concentrations in the ocean reached a maximum in mid-April of 2011 and have thereafter quickly declined. However contamination of the bottom remains quite high, showing sign of a slow decrease with time.

Working Group 10 (Modelling of marine dispersion and transfer of radionuclides accidentally released from land-based facilities) was consequently defined within MODARIA. It included experts from the following institutes and countries: Instituto de Engenharia Nuclear (IEN/CNEN, Brasil), Institut de Radioprotection et de Sûreté Nucléaire (IRSN, France), National Technical University of Athens (NTUA, Greece), Japan Atomic Energy Agency (JAEA, Japan), Korea Institute of Ocean Science and Technology (KIOST, Republic of Korea), Korea Atomic Energy Research Institute (KAERI, Republic of Korea), Norwegian Radiation Protection Authority (NRPA, Norway), University of Seville (USEV, Spain), Institute of Mathematical Machines and System Problem (IMMSP, Ukraine) and Ukrainian Centre of Environmental and Water Projects (UCEWP, Ukraine).

State-of-the-art models were assessed in the frame of this project. Models showing different characteristics and levels of complexity, from those based on a box-type approach to those making use of the shallow-water and advection/diffusion equations were tested. The performed exercises provided the opportunity to learn more about the appropriate usage of models for the management of complex environmental problems in view of the uncertainty and, often, of the vagueness of the input data, the uncertainty of the model parameters and the compatibility of different kinds of models applied to a specific contamination scenario.

In particular, two contamination scenarios were investigated: deposition and subsequent dispersion of ^{137}Cs on the Baltic Sea from the Chernobyl nuclear power plant disaster in 1986 and the dispersion of ^{137}s released from Fukushima Daiichi nuclear power plant in the Pacific Ocean after the earthquake and tsunami in March 2011 (originating from both liquid releases into the ocean and from atmospheric deposition on the sea surface). Significants amounts of ^{137}Cs were introduced in the marine environment as a consequence of these accidents. In particular, 4.7 PBq were deposited on the Baltic Sea after Chernobyl (HELCOM, 2013). Regarding Fukushima accident, it was estimated (Kobayashi et al., 2013) that 3.5 PBq of ^{137}Cs were introduced in the Pacific Ocean from March 26th to June 30 due to direct releases and leakages from the plant. Additionally, about 6 PBq were deposited on the ocean surface between March 12th and April 6th (Min et al., 2013; Kawamura et al., 2011).

Although a detailed description of the modelling exercises was given in separate papers [Periáñez et al. (2015a,b) for the Baltic and Fukushima respectively], the objective of this paper consists of providing a discussion on the lessons learnt from both scenarios.

2. Methods

Models which participated in the exercises are listed in Table 1, where appropriate references for detailed descriptions are included. They range from box models to finite difference and finite element numerical models solving simultaneously the Navier-Stokes equations for water circulation together with a sediment transport model and the radionuclide dispersion model including adsorption/release of radionuclides between water and the solid phases (suspended matter in the water column and bed sediments). Also, both Eulerian and Lagrangian dispersion models were tested.

In the case of the Baltic Sea four models were applied. They were two box-models: NRPA and POSEIDON; a 2D depth-averaged model forced by annual mean wind: USEV-2D; and a full 3D model including thermodynamics: THREETOX. In the case of Fukushima box models were not applied. Instead, all models were Eulerian or Lagrangian three dimensional dispersion models. The origin of the water circulation fields

Table 1
Models participating in the exercises.

Institute and country	Model	Scenario[a]	Reference
NRPA, Norway	Box model	BS	Iosjpe et al. (2002, 2009)
IMMSP, Ukraine	POSEIDON	BS	Lepicard et al. (2004)
IMMSP, Ukraine	THREETOX	BS	Maderich et al. (2008)
USEV, Spain	USEV-2D	BS	Periáñez et al. (2013)
IMMSP/KIOST, Ukraine/Rep. of Korea	I/K-E (Eulerian) I/K-L (Lagrangian)	F	Roland et al. (2012)
KAERI, Rep. of Korea	LORAS	F	Min et al. (2013)
IEN, Brasil	SisBahia	F	Lamego (2013)
JAEA[b], Japan	SEA-GEARN	F	Kobayashi et al. (2007)
USEV, Spain	USEV-3D	F	Periáñez et al. (2012)

[a] BS, Baltic Sea; F, Fukushima.
[b] JAEA has applied the model in two configurations: finite differences (JAEA FDM) and particle-tracking (JAEA PT).

[1] http://www-ns.iaea.org/projects/modaria/default.asp?l=116

Table 2
Models participating in the Fukushima exercises, indicating the origin of water circulation.

Model	Circulation model	Dispersion model	Spatial resolution
USEV-3D	JCOPE2[a]	Eulerian	1/12°
Sibahia	Own calculated	Eulerian/Lagrangian	Variable
I/K-E	SELFE[b]	Eulerian	Variable, min 500 m
I/K-L	JCOPE2	Lagrangian	1/12°
JAEA	Univ. of Kyoto[c]	Eulerian/Lagrangian	1/6° to 1/72°
KAERI	JCOPE2	Lagrangian	1/12°
KAERI	NCOM[d]	Lagrangian	1/8°

[a] Miyazawa et al. (2009).
[b] Roland et al. (2012). I/K-E is an Eulerian radioactivity transport module embedded together a sediment transport module in 3D SELFE hydrodynamic model.
[c] Kawamura et al. (2011).
[d] Barron et al. (2006).

required to force them is different for each model (Table 2). All models applied in both scenarios included water/sediment interactions.

Some of the models listed in Table 1 were applied to the corresponding scenario before the present model intercomparison exercise was designed. This is the case with NRPA and POSEIDON models for the Baltic and KAERI, JAEA and USEV-3D models for Fukushima. Such applications may be seen in the corresponding references in Table 1.

Details on the model applications and exercises carried out are given in Periáñez et al. (2015a,b) and are not repeated here. The spatial domains of the exercises may be seen in Fig. 1.

3. Results and discussion

In the case of the Baltic Sea, models were started 6 months after Chernobyl deposition and when the first comprehensive investigation on the fallout distribution in the entire Baltic Sea was carried out, in October 1986. A map of ^{137}Cs in surface water over the Baltic obtained from measurements was used as initial conditions (Fig. 1). This date is $t = 0$, and five year long simulations were carried out. The four models applied to the Baltic Sea produced very consistent results for the temporal evolution of ^{137}Cs inventories in the water column and bed sediments over the Baltic, time series of radionuclide concentrations in water and sediment at some fixed points and time series of averaged ^{137}Cs concentrations in water and sediment over several sub-basins of the Baltic for five years following Chernobyl deposition on the sea surface. These averaged concentrations were also in good agreement with those obtained from measurements and reported in HELCOM[2] database.

The Baltic Sea is a complex semi-enclosed marine environment. We may mention that it presents vertical stratification which reduces towards the north, horizontal density gradients and significant freshwater supplies. It is also partially covered by ice, especially in the north and during the winter. THREETOX model includes all these processes, but they are completely neglected in USEV-2D, for instance. In spite of this, there is a remarkable agreement between these models, as well as between these two models and the box models. Surprisingly, it seems that these processes are not too relevant for radionuclide transport in the Baltic Sea after the Chernobyl accident since similar results are obtained with models which include them and models which do not. Even in the case of bed sediments the agreement between all models is good (Periáñez et al., 2015a), in spite of the fact that each model is using its own description of water/sediment interactions. Nevertheless, it should be kept in mind that radionuclide transport could have been different if the accident occurred in winter, when a significant ice cover would exist.

The situation is completely different in the case of Fukushima releases. A first set of calculations consisting of simulating the dispersion of a perfectly conservative radionuclide (remaining dissolved, without interacting with suspended matter and bed sediments) released at a constant hypothetical rate from Fukushima led to considerable differences in model results (concentrations in surface water some km offshore Fukushima expanding over several orders of magnitude). These results may be seen in Fig. 2 (top panel). A further chain of simulations was then carried out to investigate the reason for these significant model discrepancies. As an example, model results when all of them use the same description of hydrodynamics (from JCOPE2 model), bathymetry and diffusion coefficients are presented in the bottom panel of Fig. 2. It may be seen that outputs from models are now significantly closer than before. The produced signals are similar and results are within the same order of magnitude. From the set of simulations, it could be concluded that the main factor in producing differences between models was the use of different water circulation (Periáñez et al., 2015b). It must be commented that JCOPE2 circulation model was selected after a comparison of sea-surface-temperature (SST) fields produced by the hydrodynamic models with SST fields obtained from satellite images[3] (IAEA, in press). It was concluded (Periáñez et al., 2015b) that a reasonable agreement between observed and calculated SST fields was obtained with JCOPE2 model. It is interesting to note that results from different models which use the same circulation in the top of Fig. 2 (USEV and KAERI-JCOPE2) are closer than results from the same model forced by different circulation (KAERI-JCOPE2 and KAERI-NCOM).

It did not seem possible to achieve a further agreement between the applied models. Differences in model outputs are now attributed to intrinsic differences between them: a) Lagrangian vs. Eulerian models and b) the different numerical schemes which may be applied for each model category mentioned in a). The particular method used to reconstruct radionuclide concentrations from the density of particles in Lagrangian models may be relevant as well.

A comparison of model results with ^{137}Cs measurements in water and sediments was also carried out. Realistic source terms for both direct releases and atmospheric deposition were used. Direct releases were reconstructed by Kobayashi et al. (2013) and are presented in Fig. 1. Atmospheric dispersion modelling was used to obtain ^{137}Cs deposition maps over the ocean (Periáñez et al., 2015b). An example is presented in Fig. 1 as well. Time series of measured and calculated ^{137}Cs concentrations at several points were presented in Periáñez et al. (2015b) and are not shown here; instead additional results, not shown in that paper, are discussed. Maps of calculated ^{137}Cs distributions in surface water, by I/K-E and USEV-3D models as examples, are presented in Figs. 3 and 4 respectively. Corresponding maps obtained from interpolation from measurements are shown in Fig. 5 (from Inomata et al., 2016). The calculated distributions, corresponding to the concentrations of the dissolved fraction of ^{137}Cs, reflect the water circulation used in each model (SELFE and JCOPE2 for I/K-E and USEV-3D respectively, Table 2). The effect of the different spatial resolutions of hydrodynamic models is clearly apparent from Figs. 3 and 4. A large anticyclonic eddy south from Fukushima is apparent in I/K-E results during the entire month of April. This eddy is not so clearly solved by JCOPE2 hydrodynamic model due to its coarser resolution. It should be commented that SST satellite images give evidence of the presence of such eddy (IAEA, in press). Both models agree in the fact that they tend to produce an elongated patch in the north-south direction and, later, leading to offshore transport of radionuclides; but SELFE circulation leads to an accumulation of material in the Bay of Sendai which is not produced by JCOPE2 water currents. Distributions of ^{137}Cs in surface water for the same dates obtained from measurement interpolation (Fig. 5) show a rather isotropic dispersion around Fukushima, without any evidence of eddies or complex circulation.

[2] www.helcom.fi

[3] Obtained from Ibaraki Prefectural Fisheries Experimental Station: NOAA satellite image, available at: http://www.pref.ibaraki.jp/bukyoku/nourin/suishi/gyomusen/noaa/noaa.html

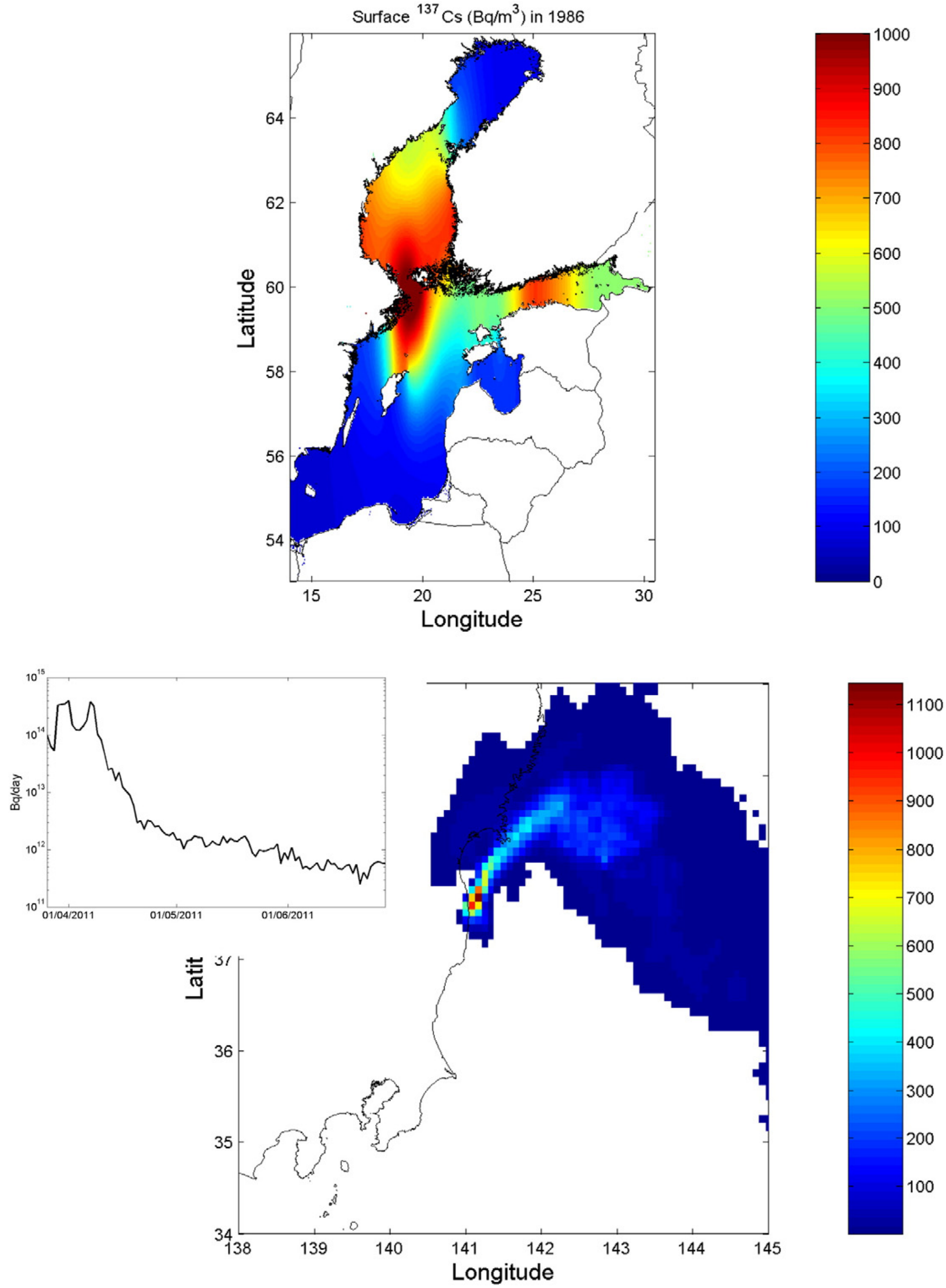

Fig. 1. Top: ^{137}Cs (Bq/m^3) in surface water of the Baltic Sea in October 1986 interpolated from measurements. Bottom: Example of calculated ^{137}Cs deposition (Bq/m^2) on March 13th at 02 h JST over the domain. Data correspond to integrated depositions over a three hour time interval. Japanese Standard Time (JST) is 9 h ahead of UTC. Direct releases into the ocean are shown in the inset.

This has to be attributed to the small number of measurements and different sampling times used to produce these maps by an optimal interpolation method.

The natural question which arises is why models with very different natures (from box to full 3D models) and parameterizations led to very coherent results in a complex marine environment like the Baltic Sea while, on the other hand, a drastic model harmonization was required in Japan coastal waters to have a reasonable agreement between rather similar models. Two marine environments have been studied: a highly dynamic system (Fukushima coastal waters) and a semi-enclosed basin (Baltic Sea). The description of the hydrodynamics is much more critical in the case of a dynamic system, since in the case of the Baltic Sea results of models are in good agreement in spite of the different approaches and simplifications applied by models. In the case of Fukushima, even similar hydrodynamic models lead to different current fields which, in turn, lead to very different radionuclide dispersion patterns. Given the intensity and variability of currents in this area, as well as the presence of unsteady eddies due to current convergence here,

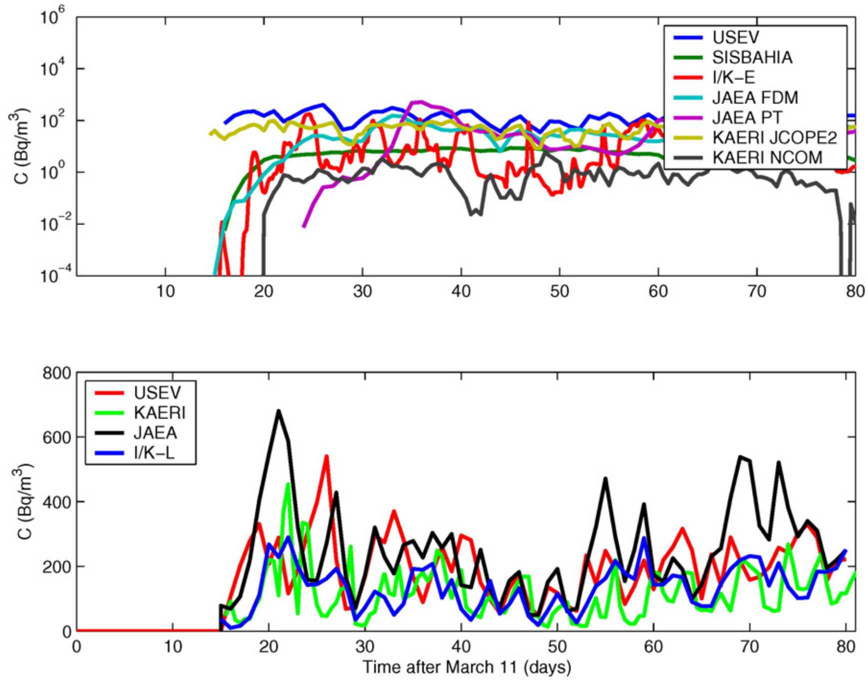

Fig. 2. Temporal evolution of a tracer concentration at the water surface 15 km offshore Fukushima for a constant hypothetical release calculated by some models. Top: each model uses its own water circulation and parameters. JAEA has applied the model in two configurations: finite differences (JAEA FDM) and particle-tracking (JAEA PT). KAERI has run the same model using circulation from two hydrodynamic models: JCOPE2 (Miyazawa et al., 2009) and NCOM (Barron et al., 2006). Bottom: all models use the same water circulation (JCOPE2) and parameters (diffusion coefficients, bathymetric data). The particle-tracking configuration has been used by JAEA for this exercise. Concerning I/K, I/K-L is a stand-alone model which was linked with JCOPE2 for this exercise. I/K-E is an Eulerian radioactivity transport module embedded with a sediment transport module into 3D SELFE model, which was used in the first set of simulations. Both models, however, describe the transport processes in a similar way.

small differences in the hydrodynamics produce different dispersion patterns. These differences tend to be amplified with time.

It must be mentioned that marine dispersion models have been incorporated into computerized-decision-systems (CDS) for supporting the management of nuclear emergencies. For instance, box models were incorporated in MOIRA-PLUS CDS when it was modified to be able to deal with seas and coastal areas (Monte, 2014) and POSEIDON was integrated into JRODOS CDS (Bezhenar et al., submitted for publication). More sophisticated marine models have also been used: THREETOX has been recently integrated into JRODOS CDS (Maderich et al., submitted for publication). Other nuclear emergency response tools have been developed for specific areas of the world, as Korea (Min et al., 2014), France (Duffa et al., 2016), Japan (Kobayashi et al., 2002) or Spain (Periáñez and Pascual-Granged, 2008). Very similar methods are also applied to other pollutants as chemicals (Havens et al., 2009) and oil spills (Sotillo et al., 2008), and to rescue operations in the sea (Jordi et al., 2006).

However, results from the present exercises highlight the difficulties of developing operative modelling systems for supporting decision-making in cases of emergencies in highly dynamic environments: the output of the system will be very dependent on the ocean model which has been used for the prediction of currents. Further research on this type of emergency models is clearly required.

It seems evident that dispersion models are robust tools, providing consistent results. But the problem is located in the hydrodynamic forcing in energetic regions characterized by strong current variability, like Fukushima waters. Although applied hydrodynamic models may be providing a coherent general picture of water circulation in the area of interest, small differences in current magnitude and/or direction in the area of release cause that initial transport pathways are different.

The question then is how to proceed to develop a reliable model to be used to support decision-making after an emergency. A multi-model approach, as described by Monte et al. (2008), may be of interest when environmental processes are complex. Through this approach, the conclusions that obtain the greatest degree of consensus among modellers are made evident and the aspects that are subject to dispute and which should therefore be handled carefully also become clear. Nevertheless, a multi-model application is not the perfect choice when an emergency is involved and a rapid response from the model is required. In any case, it may help (in the model development stage) to select the most adequate characterization of water circulation to be used in the operational dispersion model. Care should be taken in any case: there may be cases when an "outlier model" is closer to observations than the "consensus". An example is provided in IAEA (1995) - pages 26–28.

We could define three stages, characterized by increasing spatio-temporal scales, after a nuclear accident in a coastal facility, each of them requiring a specific kind of model to give a response to decision makers. It must be noted that an "ideal" model which could be applied for all spatio-temporal scales does not exist. Of course physical-chemical processes are the same, but depending on the scales in which we are interested the numerical realization and involved simplifications are different. This leads to the different modelling approaches: from box models to full 3D coupled hydrodynamic-dispersion models.

The three stages which should be considered after an emergency are the following:

1. Emergency phase: The temporal scale of the simulation extends from hours to a few days and the spatial scale to be solved from tens to a few hundred km. In this case a very rapid response (in matter of seconds to a few minutes) should be given by the model to decide, for instance, if swimming must be immediately banned in a beach, or the area where fishing should be banned. This rapid response may be achieved using data on forecast of currents and diffusivity from operational marine models and using Lagrangian models to predict the transport of radioactivity. The temporal horizon of such water current prediction is limited by the temporal scale of weather forecasts, which is about 7–10 days. Examples of this approach are

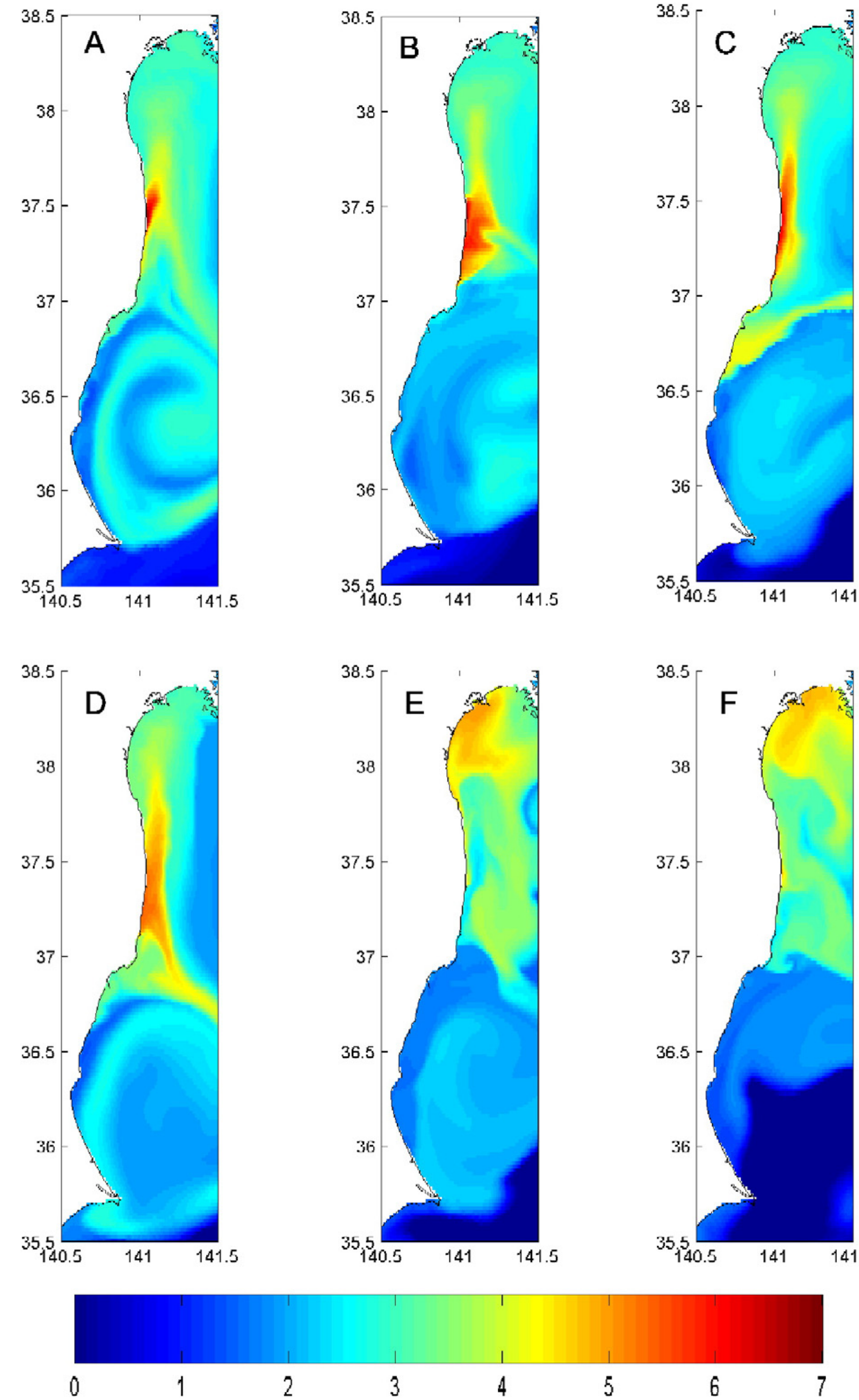

Fig. 3. ^{137}Cs concentrations (Bq/m^3) in surface water calculated by I/K-E model. Logarithms of the concentrations have been drawn. A to F: April 1, April 6, April 11, April 21, May 1, May 11.

given by Periáñez and Pascual-Granged (2008), Duffa et al. (2016) and Maderich et al. (submitted for publication). The marine product contamination can also be estimated using biota dynamic models, as it was done by Duffa et al. (2016). In this initial stage, the model output would also help to develop sampling strategies for monitoring.

2. Post-emergency: the temporal scale extends to a few weeks and the spatial one to the order of 10^2–10^3 km. We may imagine that a desalination plant produces fresh water for irrigation a few hundred km from the nuclear facility. It should be decided if taking sea water should be stopped. Now there is more time to provide an answer than in the first stage. In this phase the use of short-term ocean forecasts is not viable. The potential solution is using data from analogous periods of previous years and formation of ensemble of radioactivity predictions to estimate future contamination of water, sediments and biota. With respect to the dispersion model, both Lagrangian and Eulerian approaches could be used (for instance Kawamura et al., 2011; Periáñez et al., 2012, respectively).
3. Long-term: this phase would imply the assessment of the long-term consequences of the accident, including transfers of radionuclides to sediments and biota, as well as evaluating the potential role of

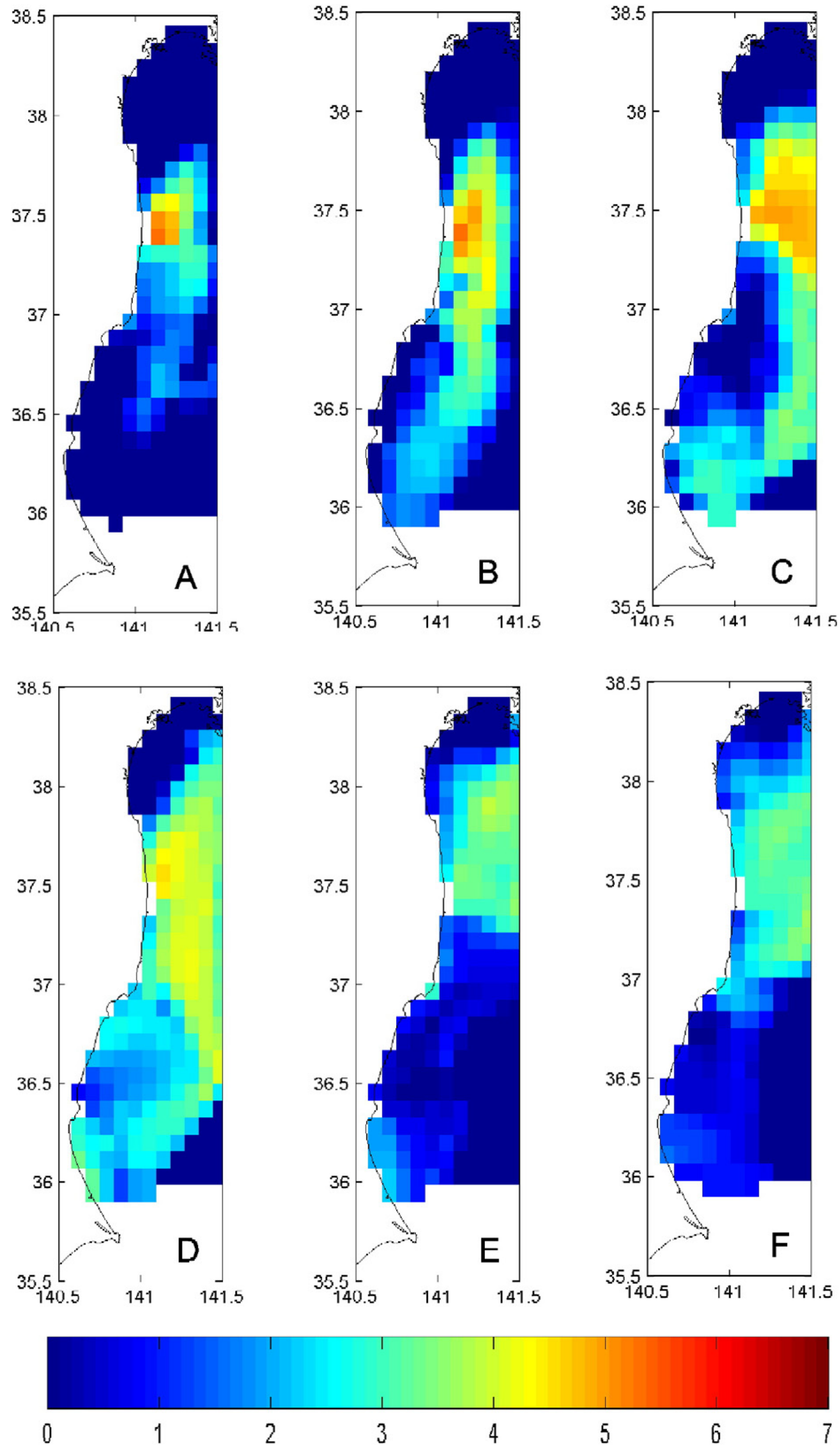

Fig. 4. Same as Fig. 3 but for USEV-3D model.

sediments as a source of contamination once radionuclide concentrations in seawater have decreased (Periáñez, 2003). This assessment may be carried out with Eulerian models, in which these complex processes are more easily included than in Lagrangian ones, and coupled dynamic biota models (Vives i Batlle et al., 2016). Water current fields are obtained from time-averaging of ocean circulation model outputs. Simulations over several months may be carried out for spatial scales of some hundred km. For even longer-term assessments (years to decades and thousand km), box models should be used (Lepicard et al., 2004; Iosjpe et al., 2009).

In any case, for highly dynamic environments, we have found that model output is extremely sensitive to the ocean model which is used to obtain circulation. Thus, the ocean model should be selected with

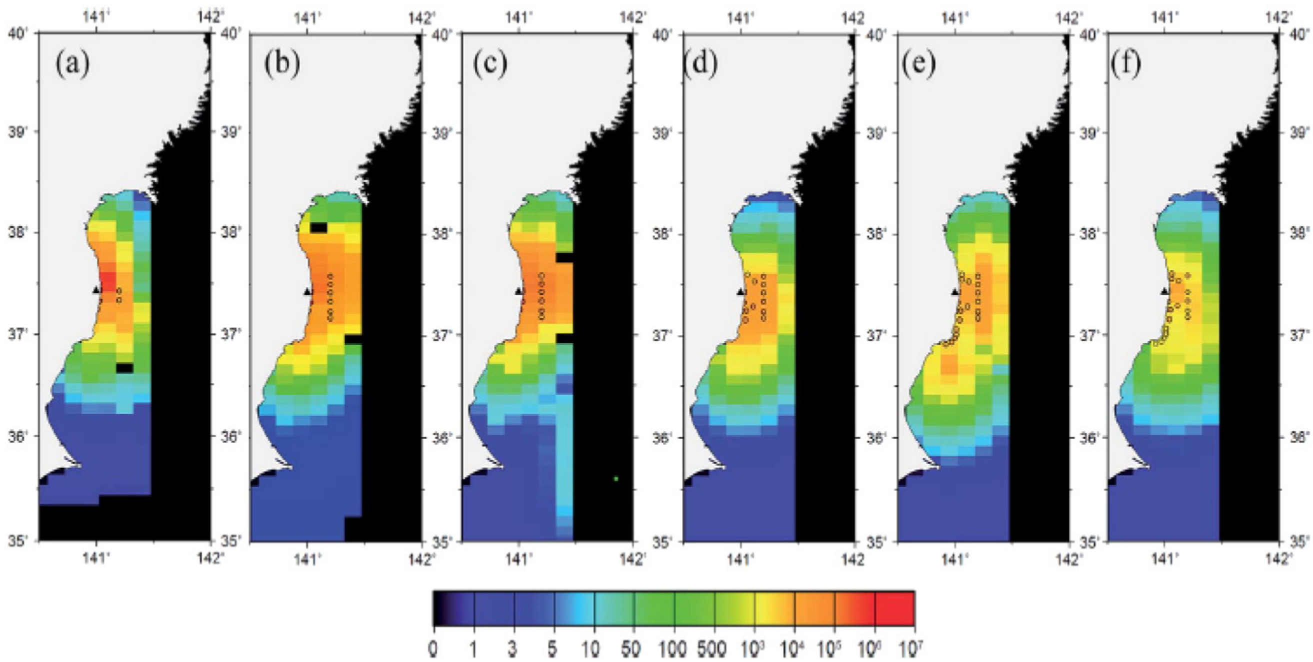

Fig. 5. Distributions of ^{137}Cs in surface water (Bq/m^3) interpolated from measurements. A to F: April 1, April 6, April 11, April 21, May 1, May 11. Figure from Inomata et al. (2016).

great care and after a detailed comparison with local measurements of currents. In this sense, Duffa et al. (2016) indicated that local forecasts of marine circulation should be used for emergency modelling. Although global ocean models produce realistic pictures of the general circulation in the ocean, their outputs differ in the local scale in dynamic environments, as it has been found. This may be, at least in part, attributed to their relatively coarse spatial resolution.

Overall, models to be used for emergencies in the marine environment should be carefully tuned for each particular location, i.e., for each nuclear facility for which it is decided to have a modelling tool to support decision-making after a potential emergency occurring there. In other words, we cannot be a priori confident in generic models which import ocean forecasts of currents if a highly dynamic environment is involved.

4. Conclusions

State-of-the-art models were applied to simulate the dispersion of ^{137}Cs coming from Chernobyl fallout in the Baltic Sea and Fukushima releases in the Pacific Ocean. The basic components of models which solve the transport of radionuclides in the abiotic compartments of a marine system are the hydrodynamic sub-model, which provides water circulation, and the dispersion sub-model, which includes advection/diffusion as well as a description of water/sediment interactions.

It was found that the energetics of the considered system (magnitude and variability of currents) is essential to obtain a good agreement between different models. Good agreement can be achieved between models of very different nature in environments characterized by weak currents. However, even similar models lead to rather different results in highly dynamic systems characterized by strong and variable currents.

This fact highlights the difficulties in developing operational models for emergency management and decision-making support (which is one of the main application of numerical modelling) in these dynamic environments. For this purpose, coastal nuclear facilities should put a significant effort in selecting the most appropriate hydrodynamic model for their specific location after a detailed and careful comparison of model results and observations.

Three stages which should be considered after an emergency, each of them requiring specific modelling approaches, have been defined. These have been denoted as the *emergency phase*, the *post-emergency* and the *long-term*.

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