



**FRAM STRAIT STUDIES (FRAMZY 2002 AND ACSYS-ABSIS 2003)
ICE DRIFTER DATA EVALUATION FOR SEA ICE KINEMATICS AND
DYNAMICS STUDIES – TECHNICAL REPORT**

Milla M. Johansson, Jouko Launiainen, Gerd Müller & Burghard Brümmer

REVIEW OF OIL SPILL EFFECTS ON ARCTIC MARINE ECOSYSTEMS

Johanna Ikävalko

MERI – Report Series of the Finnish Institute of Marine Research No. 54, 2005

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Cover photo by Milla Johansson. A drifter equipped with meteorological sensors is being deployed on an ice floe in the Fram Strait in March 2002.

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FRAM STRAIT STUDIES (FRAMZY 2002 AND ACSYS-ABSIS 2003) ICE DRIFTER DATA EVALUATION FOR SEA ICE KINEMATICS AND DYNAMICS STUDIES — TECHNICAL REPORT

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ABSTRACT

Arrays of satellite-reporting ice drifters were deployed on ice floes in the Fram Strait (80°N, 0°E) in February-March 2002 and March-April 2003, as a part of joint projects between the University of Hamburg and the Finnish Institute of Marine Research. The drifters reported position, air pressure and temperature. The position data yielded ice drift trajectories. A few buoys were equipped with additional meteorological sensors. In this report, the data and basic analysis are described. The position data were interpolated to hourly values by the method of optimum interpolation (kriging). The air pressure and temperature data were quality checked and the pressure data compared to ECMWF analyses.

In 2002, the drifters generally moved south-southwest, at a speed of 0.1 - 0.4 m/s. Most of the drifters survived for at least two weeks; one of them reported data up to the beginning of August. The air pressure data reported by the drifters were found to agree well with the ECMWF analysis fields.

In 2003, the drifters were deployed in a more easterly location, north of Svalbard. They moved generally south-southwest, at a speed of 0.01-0.1 m/s, which was clearly lower than the speed of the drifters in the previous year. Most of the drifters survived for at least one month, some of them reporting data up to July. The air pressure data reported by the drifters again agreed well with the ECMWF analyses.

Key words: Fram Strait, sea ice, ice drift, satellite drifters, satellite buoys, FRAMZY 2002, ACSYS-ABSIS 2003

1. INTRODUCTION

FRAMZY 2002 and ACSYS-ABSIS 2003 were carried out as joint projects between the University of Hamburg (UHAM) and the Finnish Institute of Marine Research (FIMR). The projects consisted of marine meteorological, sea ice, sea ice drift and remote sensing studies in the Fram Strait (80°N, 0°E). The purpose was to gather observations to quantify the transfer processes between the atmosphere, the ice and the sea. Regional forcing controls the sea ice transport in the Fram Strait and the resulting ice export from the Arctic Ocean. The field expeditions were conducted in March 2002 and April 2003 with the Finnish research vessel *Aranda* and the German Falcon research aircraft (of the Deutsches Zentrum für Luft- und Raumfahrt, DLR). In April 2003, *Aranda's* expedition was coordinated with the expedition of the German research icebreaker *Polarstern* (Alfred Wegener Institute for Polar and Marine Research).

The sea ice drift during these expeditions was tracked with drifters deployed on ice floes. The drifters' positions were determined in real-time with the Argos satellite location system. This report describes the data obtained from the drifters, and the error corrections as well as the basic analysis conducted.

1.1 FRAMZY 2002

During the FRAMZY 2002 (Fram Strait Cyclone Experiment) expedition, a total of 14 drifters were deployed on ice floes. Twelve of the drifters were parachute-launched to the ice from an aircraft operated by the UHAM. These were deployed on 27.2.2002 in a rectangular grid inside the area 80.5°N

- 83°N and 5°W - 13°E, with an approximate grid spacing of 80-90 km. The drifters were equipped with pressure and temperature sensors, and identified with the Argos ID-numbers 9430-9440 and 9492. The ice conditions during the drifter deployment are shown in Fig. 1.

Two more drifters (Argos IDs 1154 and 3333) were deployed from RV *Aranda* during her field expedition; ID 3333 on 8.3.2002, and 1154 on 11.3.2002. These drifters were equipped with meteorological sensors.

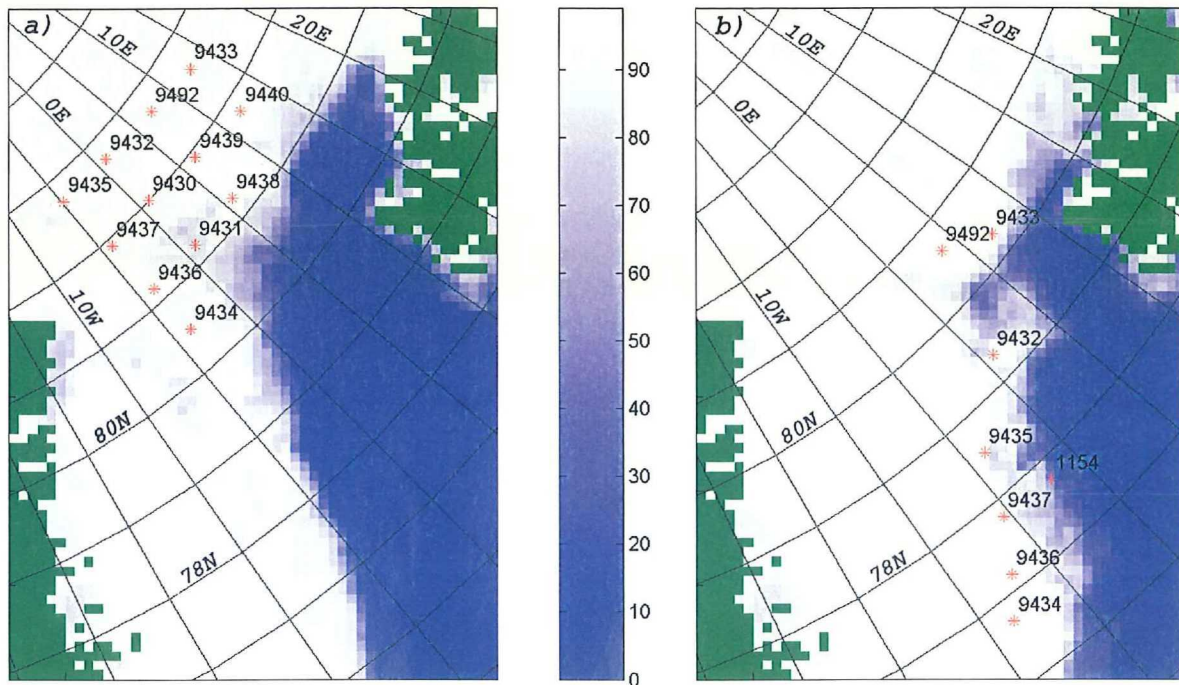


Fig. 1. a) Ice concentration and the deployment locations of the parachute-launch drifters (9430 to 9492) on 27.2.2002. Drifters 1154 and 3333 were deployed from RV *Aranda* in March 2002. b) Ice concentration and locations of the remaining drifters at the end of the RV *Aranda* expedition on 25.3.2002. The ice concentration maps are based on NOAA satellite data, processed by the University of Bremen (Kaleschke & al. 2001).

1.2. ACSYS-ABSIS 2003

During the ACSYS-ABSIS 2003 (Arctic Climate System Study - Arctic Atmospheric Boundary Layer and Sea Ice Interaction Study) expedition, a total of 11 drifters were deployed. Nine of them were parachute-launched to the ice, as in 2002, from an aircraft operated by the UHAM. They were deployed on 27.3.2003 in a rectangular grid inside the area 81°N - 83°N and 2°E - 18°E, with an approximate grid spacing of 95-105 km. They were thus deployed one month later than the FRAMZY 2002 drifters. The deployment area was also further east. These drifters, equipped with pressure and temperature sensors, were identified with the Argos ID numbers 20601, 20626, 20627, 20751, 20755, 20756, 20811, 20835 and 20842. The ice conditions during the drifter deployment are shown in Fig. 2.

Two drifters (Argos IDs 1097 and 3335), equipped with meteorological sensors, were deployed from RV *Polarstern* during her field expedition; 3335 on 4.4.2003, and 1097 on 5.4.2003.

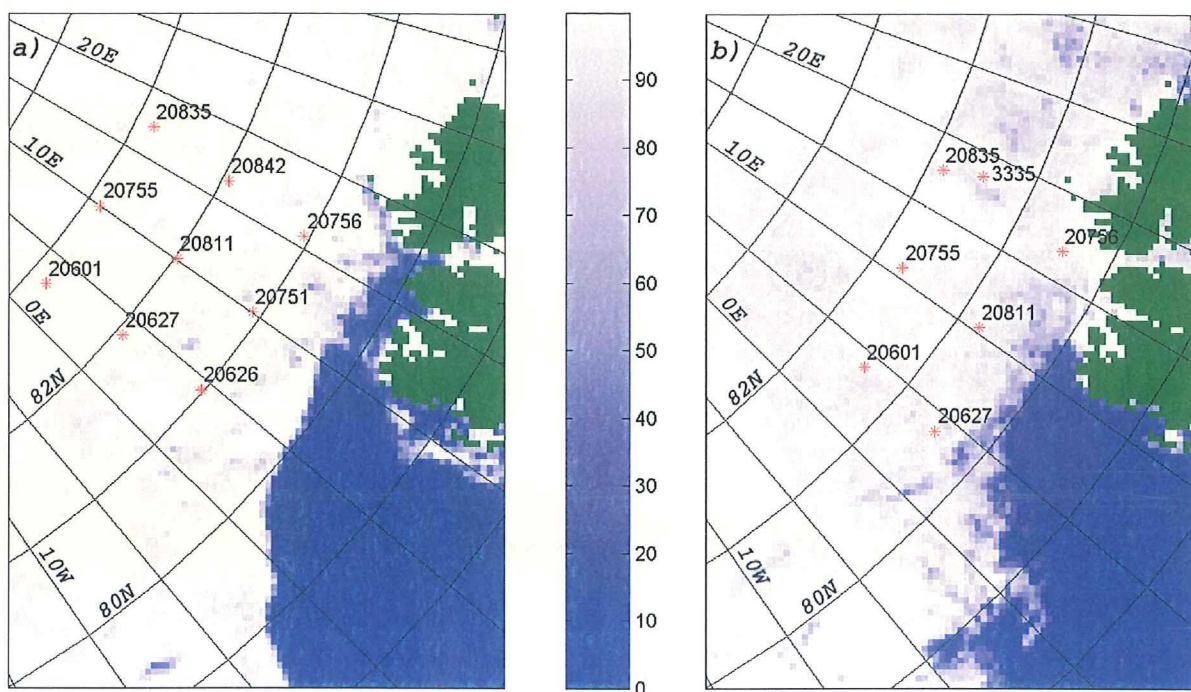


Fig. 2. a) Ice concentration and the deployment locations of the parachute-launch drifters (20601-20842) on 27.3.2003. Drifters 1097 and 3335 were deployed from RV *Polarstern* in early April 2003. b) Ice concentration and the locations of the surviving drifters at the end of the RV *Aranda* expedition on 21.4.2003. The ice concentration maps are based on NOAA satellite data, processed by the University of Bremen (Kaleschke & al. 2001).

2. DATA PROCESSING AND STORAGE

The raw Argos data were transferred to the FIMR in real-time and stored. During the field expeditions of RV *Aranda*, the drifter data were also transferred to the ship, and were used together with satellite images to aid in the estimation of ice movement and navigation. After the expeditions, the complete datasets for each drifter were extracted from the raw data and reformatted for further processing.

The Argos data sets consist of two principally different parts. The drifter locations are determined by CLS Argos, based on the transmission Doppler shift of the messages received, and are included in the data set along with a time stamp. The other part contains the messages transmitted by the drifter. The sensor data is included in this message, as well as a time stamp. These two parts of the drifter data were stored in separate data files prior to further processing. The processing of the Argos location data is described below in Section 2.1., and that of the sensor data in Section 2.2. The raw data and the corrected data are available from the authors upon request.

2.1 Location data

The Argos system provides three quality classes for the satellite-based position fixes. The accuracies of the classes are, as advertised by CLS Argos (Argos User's Manual, 2004):

- Class 3 < 150 m
- Class 2 150 – 350 m
- Class 1 350 – 1000 m

The time intervals between successive drifter position fixes are irregular, depending on the timing of satellite passes over the experiment area. To simplify further analyses and to smooth out the effect of inaccuracies in the position fixes, we interpolated the locations to regular time intervals. On the average, 19 locations per day were available in 2002, and 29 locations per day in 2003. It was thus

relevant to interpolate the locations to a 1-hourly interval. Drift velocities were calculated from the hourly and 6-hourly interpolated positions for each drifter.

There were some obviously erroneous location fixes in the data, identified as distinct peaks in the trajectories and/or exceptionally high drift speeds. These erroneous points were deleted from the data sets prior to further analyses.

2.1.1 Optimum interpolation (kriging)

The method of optimum interpolation (kriging; Cressie, 1991) was applied to the location data. In this method, an individual interpolated value is estimated as a linear combination of the neighbouring values in time. The neighbouring values are given different weights based on a variogram. The variogram is calculated from the data, and describes the variability of the quantity - the location in this case - as a function of time. The calculations were performed using the Matlab Kriging Toolbox (Gratton & Lafleur, 2001), available on the Internet. The latitudes and longitudes were transferred to rectangular coordinates, and the interpolation method was applied to the time series of the x and y coordinates separately.

A semivariogram was first calculated for the time series, and a power function fitted to it:

$$\gamma(\Delta t) = c\Delta t^b \quad (1)$$

where b and c are the parameters to be fitted. $\gamma(\Delta t)$ is the semivariogram, Δt is the time interval between the points. This kind of semivariogram is applicable to drifter movements (Hansen & Poulain, 1996). The physically meaningful range for the parameter b is $1 \leq b < 2$. If $b=1$, the motion is Brownian, or pure diffusion. The limit $b=2$ corresponds to rectilinear motion. Sea ice drift lies generally between these two extremes. An example of a semivariogram is shown in Fig. 3.

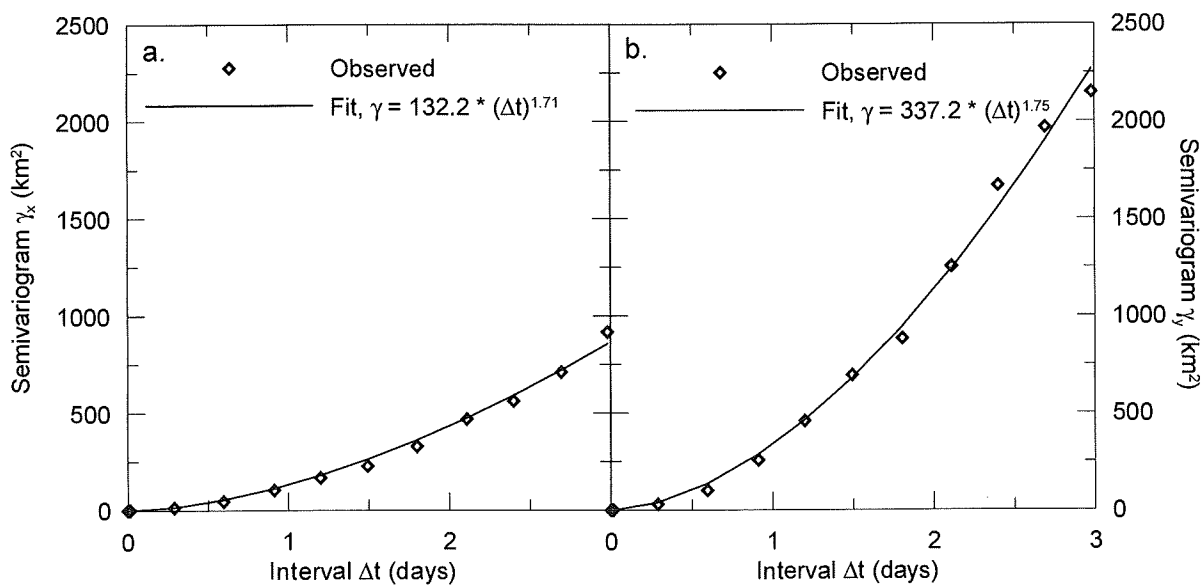


Fig. 3. Semivariograms for the time series of the x coordinate (a) and the y coordinate (b) of drifter 1154, together with the power function fitted to the observed points.

A property of all power functions of the type of Eq. 1 is that $\gamma(0) = 0$. This is physically feasible for an ideal ice drift, since drifter positions with an arbitrarily short time interval should converge. In particular, two simultaneous observations should result in identical coordinates. In practice, observation errors cause deviations from this. The behaviour of the semivariogram in the case of small time intervals is described with a "nugget effect" (Cressie, 1991):

$$\gamma(0) = c_{mg} = c_{MS} + c_{ME} \quad (2)$$

where the nugget coefficient c_{mg} consists of the microscale variation c_{MS} , and the measurement error c_{ME} . The microscale variation c_{MS} describes the variations on a time scale that is smaller than the resolution of the sampling time intervals. In our case, $c_{MS} = 0$. The measurement error c_{ME} can be estimated from the overall accuracy of the Argos locations.

The nugget effect was estimated using the data of the FRAMZY 2002 drifters. The semivariogram was calculated for very small time intervals, and extrapolated down to a time interval of 0. This results in an estimate of $0.1 - 0.5 \text{ km}^2$, which corresponds to a position accuracy of $0.3 - 0.7 \text{ km}$. This is of the same order as the values given by Argos. Accordingly, we chose to use the value $c_{mg} = 0.49 \text{ km}^2$ in our calculations. Generally, a large nugget coefficient c_{mg} results in a smooth interpolated trajectory, while a small coefficient forces the trajectory to follow the observed - possibly erroneous - points more closely.

The positions were interpolated to a 1-hour time interval using the semivariograms. Each point was interpolated using ten observed points - five points before and five points after the hour in question. Points that were further away than 24 hours in time were not used, however. There were some gaps in the time series that were too long to be filled with interpolation. In general, the interpolated trajectories fit the observations well (Fig. 4).

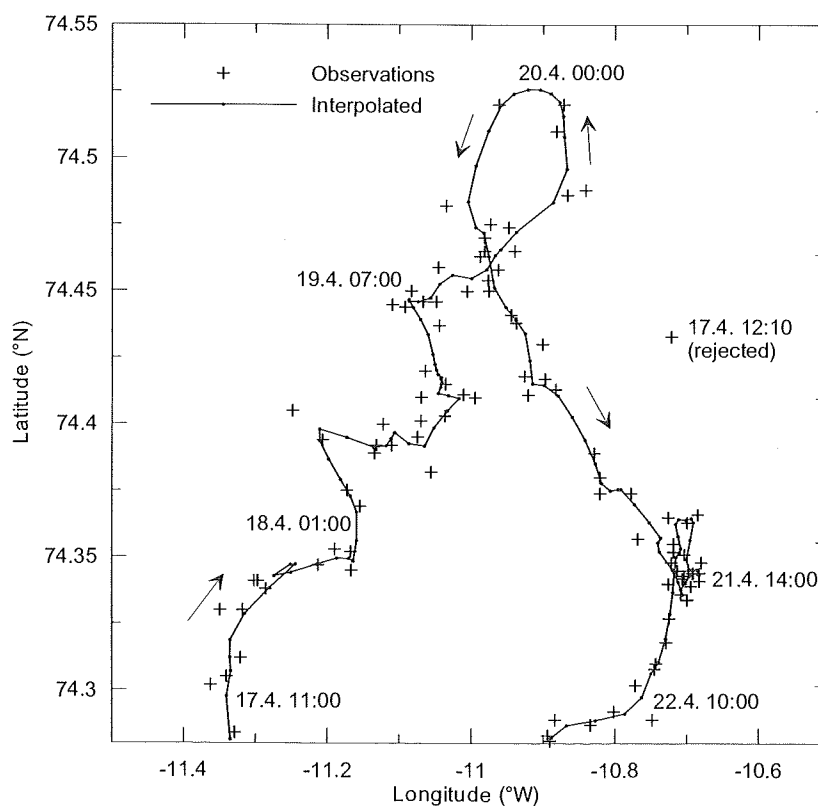


Fig. 4. A part of the trajectory of drifter 1154, together with the hourly positions obtained by optimum interpolation. The obviously erroneous point was rejected before interpolation.

2.1.2 Polynomial interpolation

For comparison, another interpolation method was tested. The method was that used in an earlier Fram Strait study (Hoerber, 2000), based on fitting a second-order polynomial to five observed points closest to the hour considered. An example of a comparison between these two interpolation methods is given in Fig. 5.

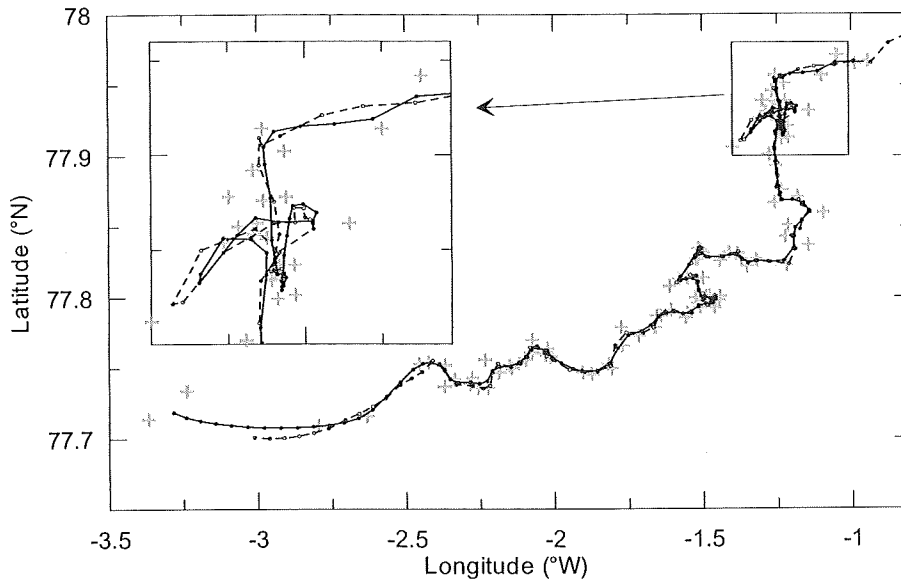


Fig. 5. Comparison between optimum interpolation (solid line) and polynomial interpolation (dashed line), for the trajectory of drifter 1154. The original observations are marked with crosses. Part of the trajectory for 21.-26.3.2002 is shown.

On the average, the mutual difference between the interpolated locations is of the order of 0.3 km. This is of the same order as the inaccuracy of the Argos positions discussed above. However, individual points may differ considerably more, especially in cases where the observed locations are sparse. In some cases, this may yield a rather significant difference in estimating the discrete drift velocities and trajectories based on the hourly data.

The trajectories obtained by optimum interpolation are generally smoother than those obtained by polynomial interpolation, thus resulting in smoother estimates of the drift velocities. The method of optimum interpolation also behaves better in cases where the time intervals between observations are irregular - such as two nearly simultaneous observations with differing locations, or long gaps in the time series.

2.2 Meteorological data

The drifter transmissions were quite noisy, and the raw data messages contain a large amount of erroneous values (see example in Fig. 6). No specific reason for the poor message quality was identified, but the possibility of simultaneous transmissions of several mutually closely-located drifters interfering with each other was considered.

2.2.1 Parachute-launch drifters

The drifters with Argos IDs 9430-9492 and 20601-20842 transmitted data messages containing air pressure, temperature and battery voltage. The values were converted to physical units by CLS Argos.

The satellites often receive several transmissions of the same message from a drifter. The count of identical repetitions is expressed as a “compression index” for each message. As the first step in our data analysis, the message with the highest compression index was chosen for each time considered.

The compression index correlates clearly with transmission errors - erroneous transmissions were usually received only once. Thus, an easy and effective method for error removal is to discard messages with a compression index of one. Naturally, this also removes some correct data. The method resulted in 27-40 % of discarded data messages for drifters 9430-9492 and 23-55 % of discarded messages for drifters 20601-20842. However, the remaining time series of pressure and temperature were considered to be adequate for further analysis. This method, in addition to being simple and efficient, is

independent of the content of the data messages themselves. Thus, it is less dependent on somewhat subjective algorithms for determining an acceptable pressure value, for instance.

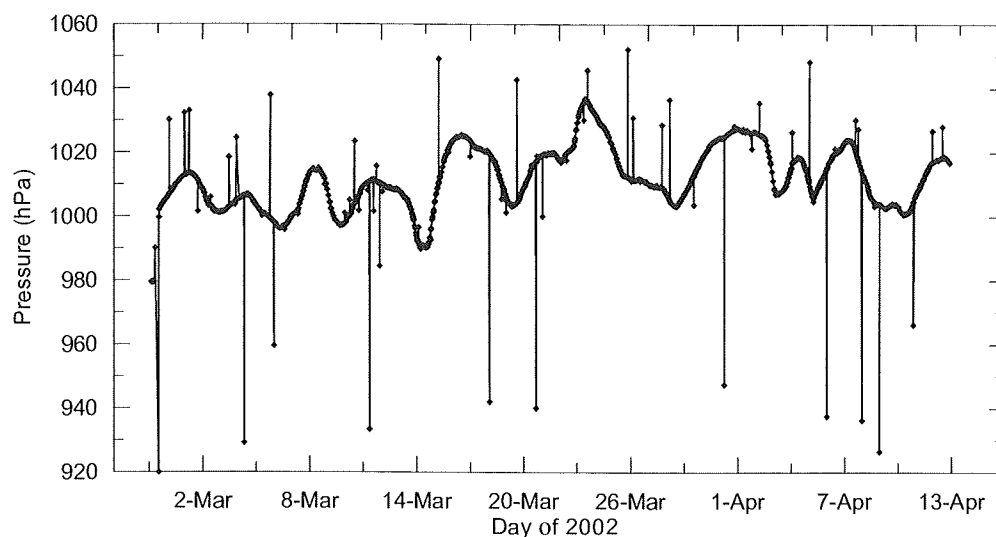


Fig. 6. The raw air pressure time series transmitted by drifter 9432 in 2002.

2.2.2 Meteorological drifters

The meteorological parameters measured by drifters 1154, 3333, 1097 and 3335 are given in Table 1. The data messages of drifters 1097, 3333 and 3335 were converted to physical units by CLS Argos. Instead, the messages of drifter 1154 were received from Argos as raw bytes, and were converted to physical units at FIMR, using drifter-specific conversion formulae. These drifters stored data values once an hour, and the hour of observation was included in the data messages. Originally, the clock of drifter 1154 was not set correctly, and it was also reset during the experiment, possibly because of a battery failure. The timestamps were corrected by comparing the hours transmitted by the drifter with those determined by Argos.

The identical data messages were first combined as above. Clearly erroneous transmissions, containing only zero values for drifter 1154, were removed. The failure of the various sensors of drifter 1154 was determined from the data, and data values from the failure onwards were removed. After these corrections, one data set for each hour was selected. Whenever there were different data sets resulting from transmission errors, that with the largest compression index was considered to be the most reliable. Finally, messages with a compression index of one were discarded. The percentage of discarded data sets for these drifters was around 20 %.

Table 1. Parameters measured by the meteorological drifters.

| Drifter 1154 (FIMR) | | Drifters 3333, 3335 and 1097 (UHAM) | |
|----------------------------|------------------------------|--|------------------------------|
| Sensor | Height above ice surface (m) | Sensor | Height above ice surface (m) |
| Pressure | | Pressure | |
| Air temperature | 4.0 | Pressure tendency | |
| Temperature difference | 4.0 / 1.0 | Ice temperature | - |
| Relative humidity | 4.0 | Air temperature | 0.6 |
| Wind speed | 4.5 | Relative humidity | 0.8 |
| Wind gust | 4.5 | Wind speed | 2.1 |
| Wind direction | 4.5 | Wind direction | 2.1 |
| Drifter heading | - | Drifter heading | - |

2.2.3 Comparison with weather analyses

The pressure data measured by the drifters were compared with model analyses. The European Centre for Medium-Range Weather Forecasts (ECMWF) analyses for the area 70°N - 85°N, 25°W - 20°E, and having a resolution of 0.5°, were used. The drifter observations were linearly interpolated to 6-hour intervals and compared to the analysis values interpolated to the drifter locations.

3. RESULTS FOR FRAMZY 2002

3.1 General drift properties

Most of the drifters survived for at least two weeks (Table 2), drifting generally southward through the Fram Strait (Fig. 7). Drifter 1154 survived up to the beginning of August 2002. At that time, it was located somewhat east of Cape Farewell, failing presumably due to exhausted batteries. The trajectories of all 14 drifters are given in Fig. 7. Unfortunately, the meteorological sensors of drifter 1154 generally operated for only a few days after deployment. The wind sensor failed on 15.3.2002, the air temperature and humidity sensors on 18.3.2002. Only the pressure sensor operated up to August 2002.

Table 2. The first and last data messages received from the drifters in 2002.

| Drifter | First message (UTC) | | Last message (UTC) | |
|---------|---------------------|-------|--------------------|-------|
| 1154 | 11.03.2002 | 10:01 | 03.08.2002 | 23:52 |
| 3333 | 08.03.2002 | 12:45 | 16.03.2002 | 09:39 |
| 9430 | 27.02.2002 | 16:10 | 18.03.2002 | 14:04 |
| 9431 | 27.02.2002 | 12:49 | 20.03.2002 | 05:32 |
| 9432 | 27.02.2002 | 16:12 | 12.04.2002 | 21:07 |
| 9433 | 27.02.2002 | 16:04 | 07.04.2002 | 04:33 |
| 9434 | 27.02.2002 | 12:48 | 13.04.2002 | 20:45 |
| 9435 | 27.02.2002 | 16:10 | 22.05.2002 | 17:37 |
| 9436 | 27.02.2002 | 13:08 | 19.04.2002 | 14:59 |
| 9437 | 27.02.2002 | 14:26 | 10.05.2002 | 08:03 |
| 9438 | 27.02.2002 | 14:19 | 11.03.2002 | 08:55 |
| 9439 | 27.02.2002 | 17:54 | 18.03.2002 | 00:48 |
| 9440 | 27.02.2002 | 17:53 | 13.03.2002 | 16:03 |
| 9492 | 27.02.2002 | 16:12 | 13.04.2002 | 17:26 |

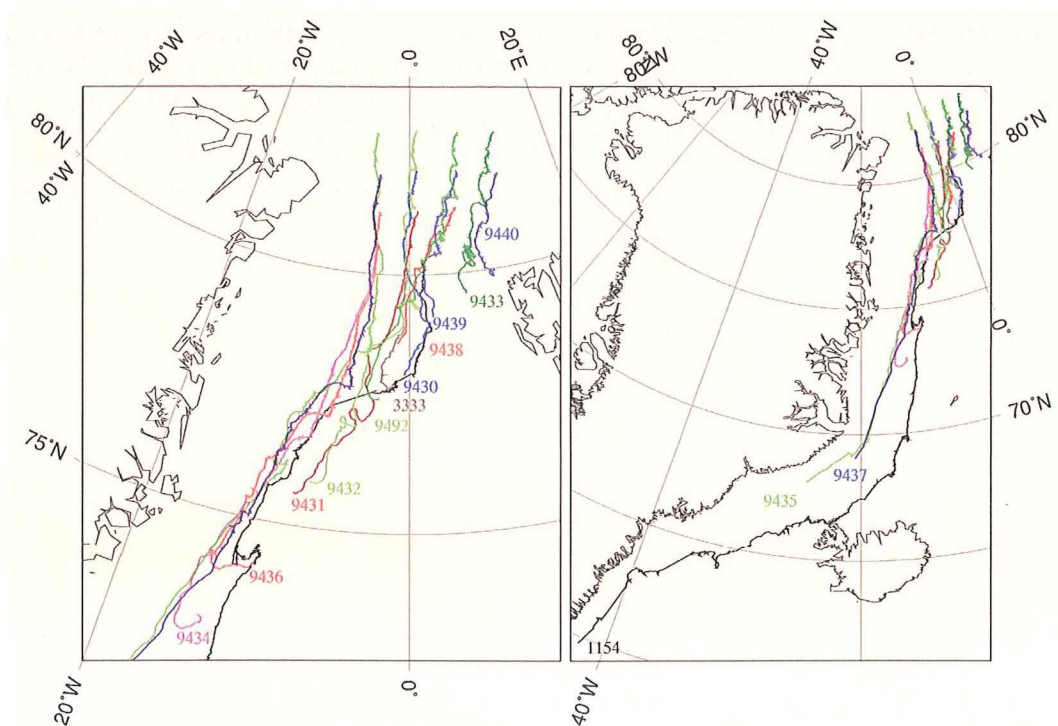


Fig. 7. Drifter trajectories in 2002. a) Trajectories in the Fram Strait during Feb.-Apr. b) The complete trajectories of the three longest-lived drifters.

3.2 Data evaluation

3.2.1 Drifter positions

In total, from 159 to 2 213 position fixes for each drifter were obtained, resulting in 15-25 fixes per day, on average. The location class distribution for the drifters is given in Table 3.

Table 3. Observed number of locations of the drifters, length of the period, and the percentages of the different Argos location classes.

| Drifter | Locations, total | Days of operation ^{*)} | Locations/day, average | Class 3 (%) | Class 2 (%) | Class 1 (%) |
|---------|------------------|---------------------------------|------------------------|-------------|-------------|-------------|
| 1154 | 2 213 | 143.5 | 15.4 | 12 | 25 | 63 |
| 3333 | 159 | 6.7 | 23.9 | 13 | 33 | 54 |
| 9430 | 433 | 18.9 | 22.9 | 20 | 38 | 42 |
| 9431 | 373 | 20.3 | 18.4 | 21 | 36 | 43 |
| 9432 | 812 | 44.3 | 18.3 | 21 | 35 | 43 |
| 9433 | 955 | 38.5 | 24.8 | 26 | 34 | 39 |
| 9434 | 723 | 45.3 | 16.0 | 31 | 37 | 32 |
| 9435 | 1 711 | 84.0 | 20.4 | 35 | 34 | 32 |
| 9436 | 1 039 | 50.8 | 20.4 | 27 | 36 | 38 |
| 9437 | 1 273 | 64.2 | 19.8 | 29 | 36 | 35 |
| 9438 | 259 | 11.8 | 21.9 | 20 | 37 | 43 |
| 9439 | 346 | 18.3 | 18.9 | 18 | 34 | 48 |
| 9440 | 304 | 14.0 | 21.7 | 22 | 40 | 38 |
| 9492 | 1 097 | 45.1 | 24.3 | 28 | 36 | 37 |
| Total | 11 697 | 605.7 | 19.3 | 24 | 34 | 42 |

^{*)} Time between the first and last position fix.

The obtained locations were not distributed evenly over the different hours of the day. Especially in the case of the long-lived drifters, more locations were obtained during daytime than during the night (Fig. 8). An uneven distribution of location fixes during the day was also observed in 1999 in the Fram Strait (Hoeber, 2000). The reason for this behaviour is not well understood.

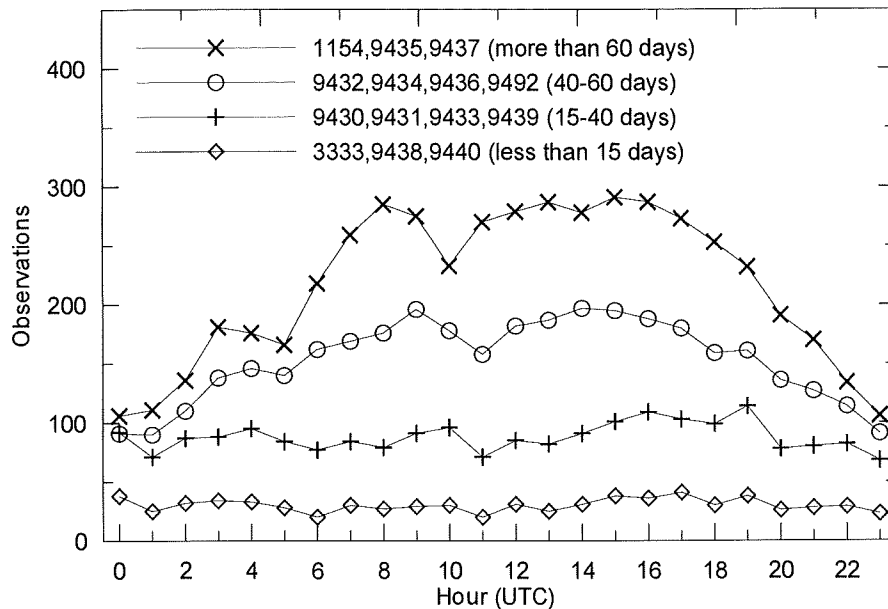


Fig. 8. Distribution of the number of locations as a function of the hour of day. The drifters are grouped in four groups based on their lifetime.

3.2.2 Interpolation of the trajectories

Semivariograms were calculated for each drifter separately. The b values (Eq. 1) obtained for the drifters vary between 1 and 2, without exception. The values for b_y are generally somewhat larger than the b_x values. This implies that the movement in a north-south direction is somewhat more rectilinear than the movement in an east-west direction. This is a consequence of the generally southward direction of the sea ice drift in the Fram Strait.

Table 4. Semivariogram parameters for the FRAMZY drifters.

| Drifter | b_x | c_x | b_y | c_y |
|---------|-------|-------|-------|-------|
| 1154 | 1.71 | 132 | 1.75 | 337 |
| 3333 | 1.25 | 90.0 | 1.74 | 835 |
| 9430 | 1.35 | 44.6 | 1.74 | 364 |
| 9431 | 1.44 | 152 | 1.72 | 758 |
| 9432 | 1.33 | 45.3 | 1.67 | 269 |
| 9433 | 1.10 | 27.6 | 1.43 | 97.9 |
| 9434 | 1.45 | 59.7 | 1.68 | 281 |
| 9435 | 1.69 | 99.0 | 1.74 | 370 |
| 9436 | 1.74 | 46.1 | 1.74 | 205 |
| 9437 | 1.72 | 51.2 | 1.73 | 293 |
| 9438 | 1.69 | 82.3 | 1.73 | 422 |
| 9439 | 1.36 | 32.0 | 1.56 | 178 |
| 9440 | 1.46 | 37.8 | 1.70 | 170 |
| 9492 | 1.37 | 41.8 | 1.65 | 185 |

3.2.3 Drift velocities

The drift was oriented to the south-southwest on average, with a speed of 0.1-0.4 m/s (Table 5, Figs. 9-10). Drifter 1154 drifted considerably longer than the others, eventually turning to the southwest along the Greenland coast. Thus its overall mean drift direction differs from the others. If only the drift up to the end of April is considered, a direction of 197° is obtained for drifter 1154, which is in accordance with the others. The maximum velocities were generally of the order of 0.7 - 1.3 m/s. In one exceptional case in June, drifter 1154 reached a speed of 1.9 m/s (from hourly data, and thus not included in Table 5). The frequency distributions for the speed and drift direction for two representative drifters are given in Fig. 11.

It is apparent that there still remain peaks in the velocity time series, especially in the hourly velocities. These peaks are probably errors caused by inaccurate location points. These individual peaks have no significant influence on the mean velocities. The maximum values, however, might be affected. The maximum velocities given in Table 5 were individually checked to ensure that they were not caused by apparently erroneous location data.

Table 5. Mean and maximum velocities of the drifters, calculated from locations interpolated at 6-hour intervals. The mean values calculated from 1-hourly locations are practically identical. The maximum velocities calculated from 1-hourly data would be somewhat larger than those given in the table, but since they are very sensitive to errors and gaps in the original location data, they are not given.

| Drifter | u_{mean} (m/s) (west-east) | v_{mean} (m/s) (south-north) | Mean speed (m/s) | Direction (°) | Max. speed (m/s) |
|---------|---------------------------------|-----------------------------------|---------------------|------------------|---------------------|
| 1154 | -0.13 | -0.18 | 0.22 | 216 | 1.33 |
| 3333 | -0.05 | -0.42 | 0.42 | 187 | 0.66 |
| 9430 | -0.02 | -0.27 | 0.27 | 184 | 0.67 |
| 9431 | -0.13 | -0.36 | 0.38 | 200 | 1.05 |
| 9432 | -0.05 | -0.20 | 0.21 | 194 | 0.81 |
| 9433 | -0.03 | -0.10 | 0.10 | 198 | 0.73 |
| 9434 | -0.06 | -0.22 | 0.22 | 196 | 1.04 |
| 9435 | -0.09 | -0.22 | 0.24 | 203 | 1.19 |
| 9436 | -0.04 | -0.18 | 0.18 | 192 | 0.57 |
| 9437 | -0.08 | -0.24 | 0.25 | 199 | 1.03 |
| 9438 | -0.13 | -0.28 | 0.31 | 204 | 0.67 |
| 9439 | -0.04 | -0.19 | 0.20 | 193 | 0.85 |
| 9440 | -0.03 | -0.17 | 0.17 | 191 | 0.50 |
| 9492 | -0.05 | -0.15 | 0.15 | 198 | 1.12 |

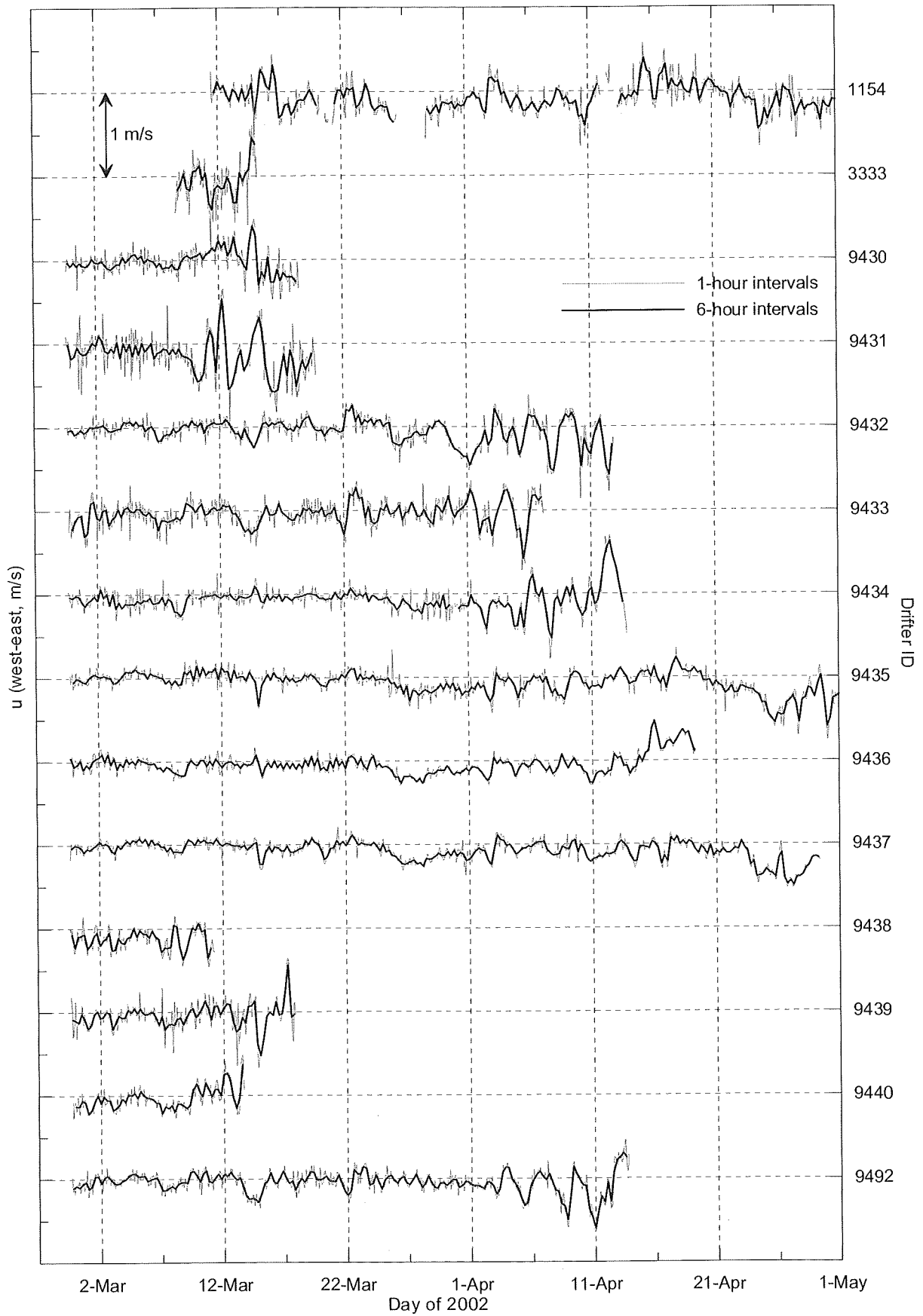


Fig. 9. West-east velocity component for the various drifters, up to the end of April.

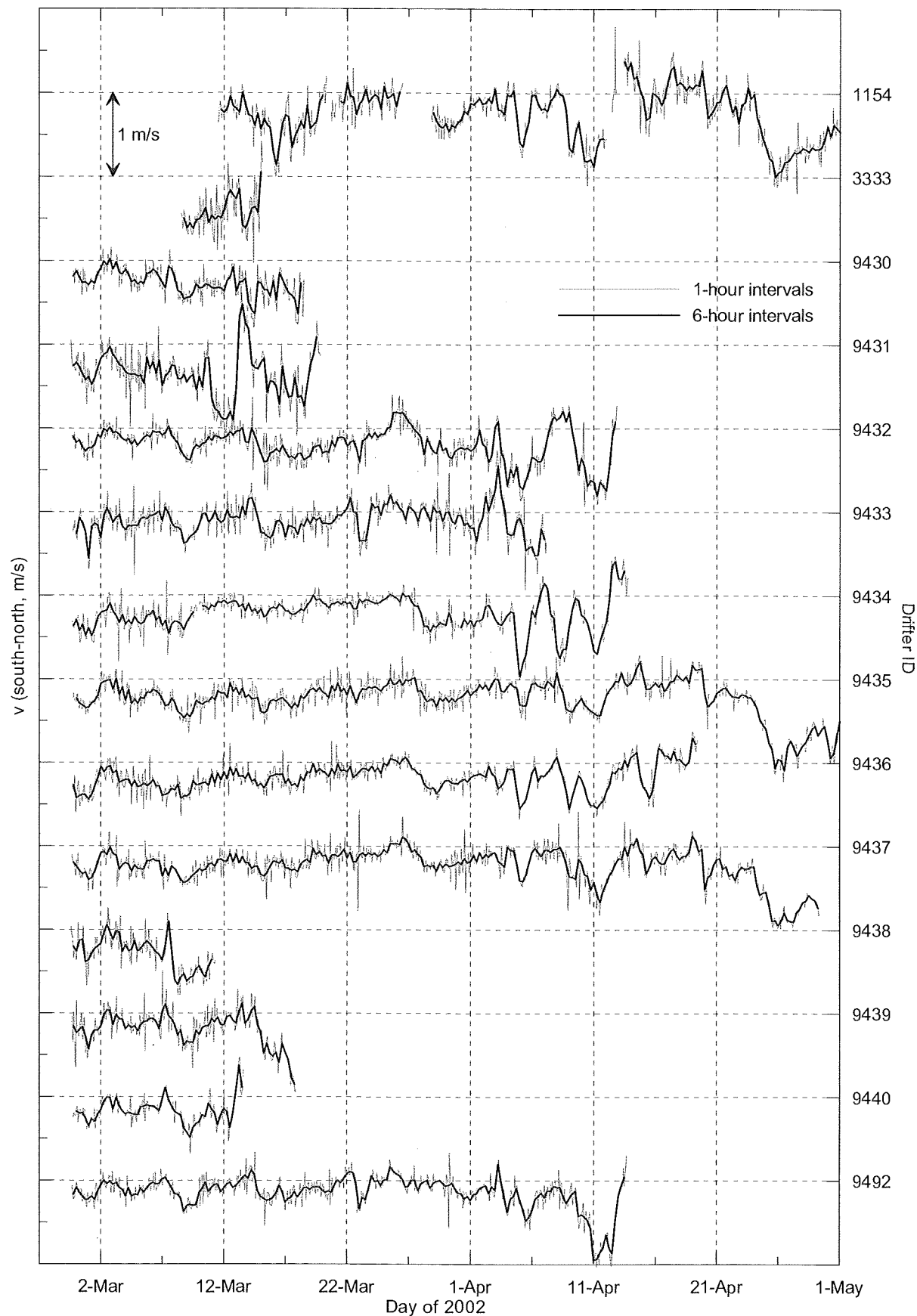


Fig. 10. South-north velocity component for the various drifters, up to the end of April.

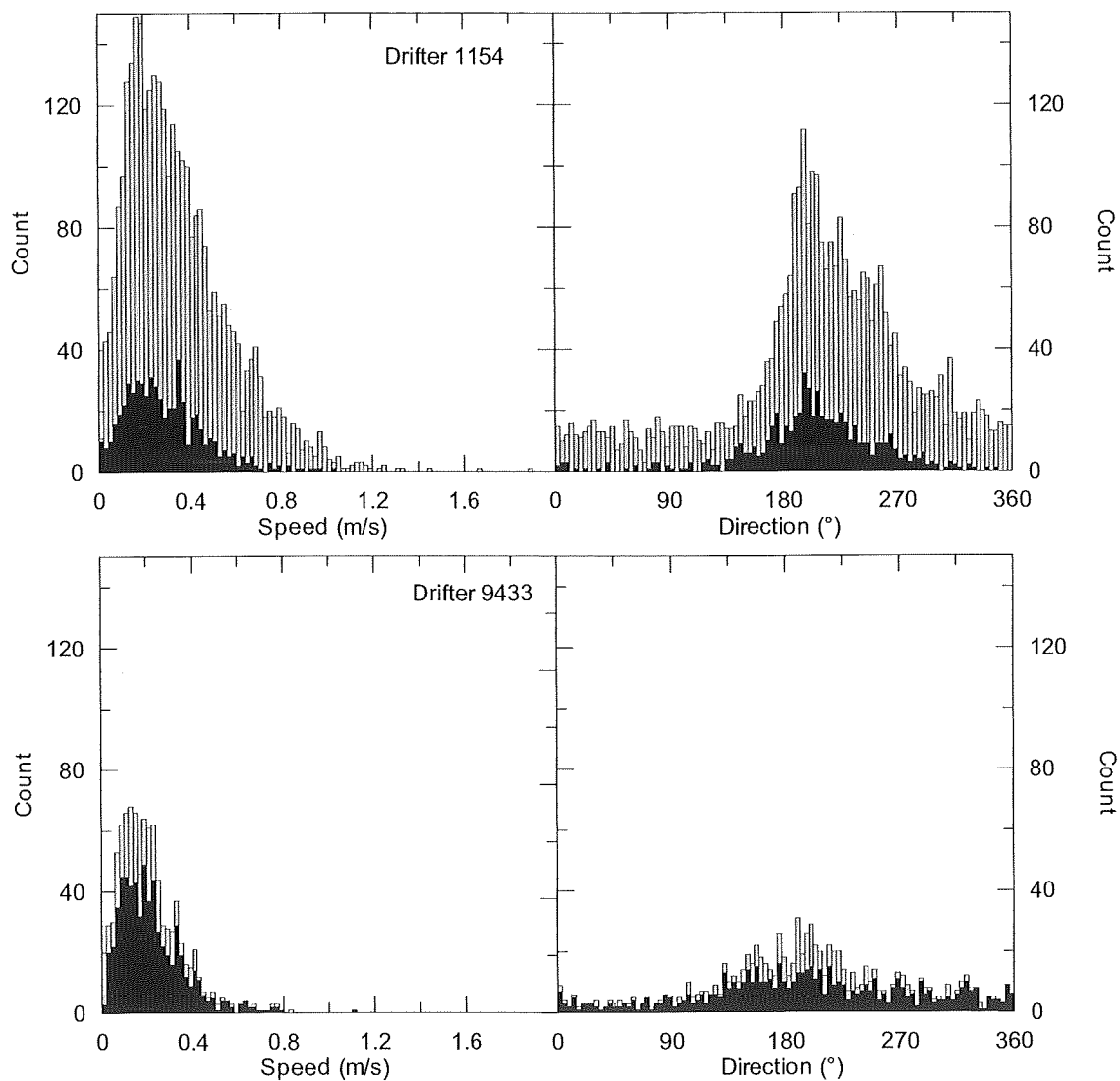


Fig. 11. Distributions of the drift speed and direction for two representative drifters, calculated from hourly velocities. Drifter 1154 survived for a long time and drifted relatively freely to the southwest. Drifter 9433 was stuck in a dense ice field for most of the time (Fig. 7). The black bars represent the distribution during the period when both drifters were operational (see Table 2). The grey bars represent the total distribution for each drifter.

In general, it is interesting to see that during the early stage of FRAMZY (up to 17.3.2002), the east-west movements (Fig. 9) were most pronounced for drifters 1154, 9430, 9431, 9438, 9439 and 9440, i.e. those drifting nearest the open water-sea ice boundary in the middle of the Fram Strait (Figs. 1 and 7).

It is evident that during the period 11.-17.3.2002, drifter 9431 moved more dramatically than any of the other drifters. For this reason, we briefly examined the case. The trajectories of the drifters for these days are depicted in Fig. 12. Drifter 9431 was located near the ice edge, and was thus able to move more freely than the others. The weather forcing possibly caused it to drift rapidly in the low-concentration ice field, while the more dense ice conditions restricted the movement of the other drifters.

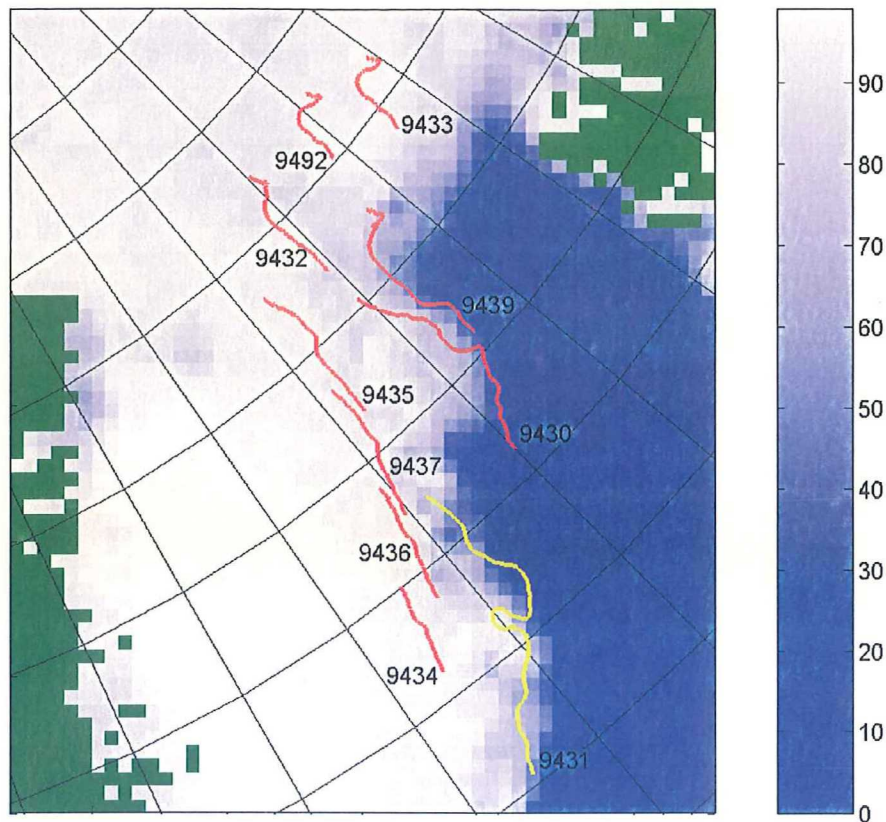


Fig. 12. The trajectories of the drifters over the period 11.-17.3.2002 (only those drifters are included that were operational throughout the whole period). The path of drifter 9431 is drawn in yellow. The background shows the ice concentration on 17.3.2002, based on NOAA satellite data processed by the University of Bremen (Kaleschke & al. 2001).

3.3 Buoy meteorological data

3.3.1 Data time series

Figs. 13 and 14 show the time series of the corrected air pressure and the “buoy-hull” measured ambient (air) temperature for drifters 9430-9492. The time series of pressure, temperature, humidity and wind for drifters 1154 and 3333 are presented in Figs. 15 and 16. In Fig. 17, the parameters measured by RV *Aranda* are given for comparison.

3.3.2 Comparison with weather analyses

Comparisons of the pressure values from the drifters and the ECMWF analyses are shown in Figs. 18-20. The numerical results of the comparison are given in Table 6. The NCEP analyses were also briefly studied. In general, the ECMWF analyses correspond to the observed values more closely. It is to be noted that the FRAMZY 2002 drifter data were not fed to the real-time ECMWF observation data assimilation.

Generally, the ECMWF analysis values agree very well with the observations. However, an overall tendency for ECMWF to slightly underestimate the air pressure of the area is suggested by the measurements.

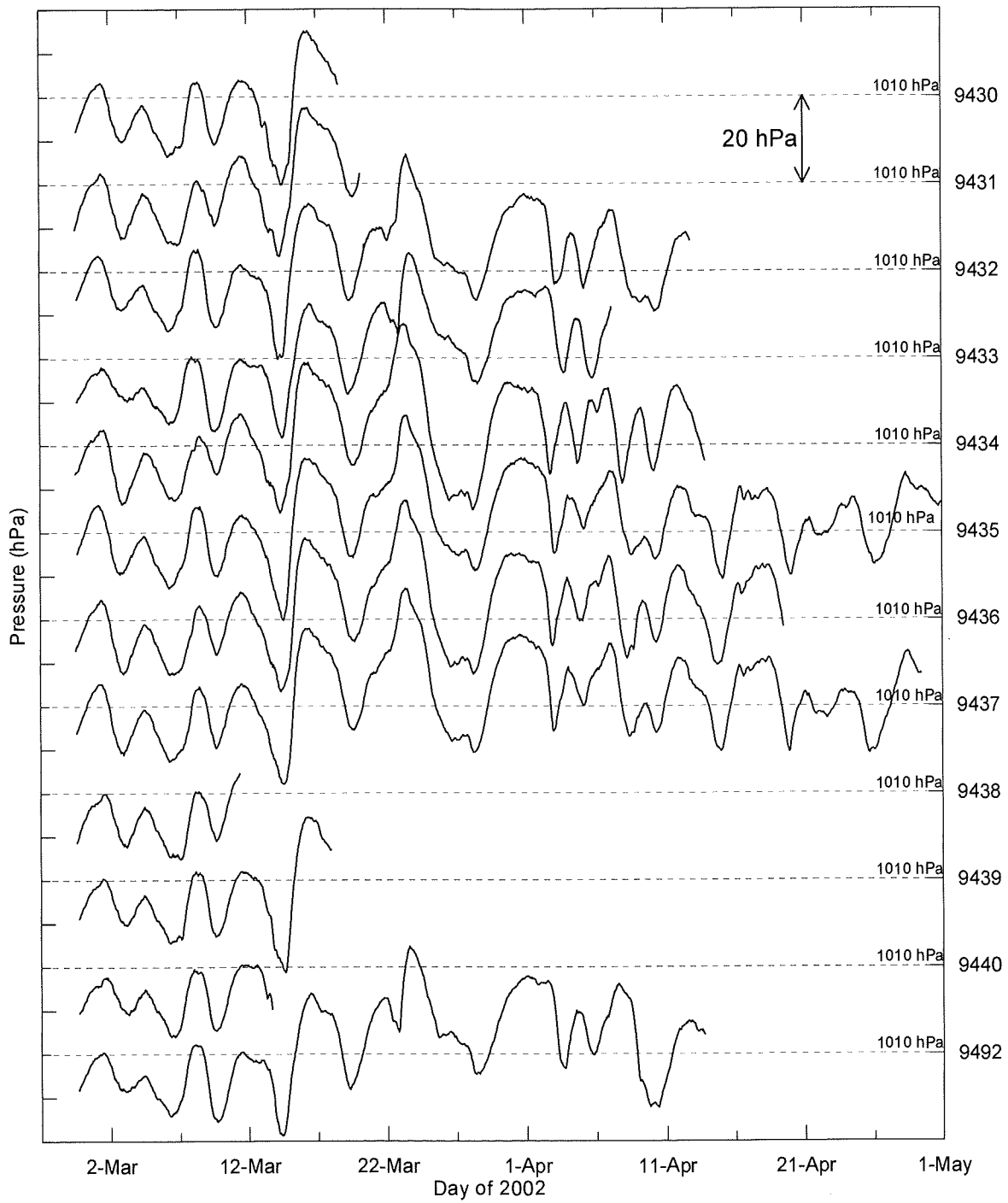


Fig. 13. Air pressure measured by drifters 9430-9492, corrected time series (see text) up to the end of April.

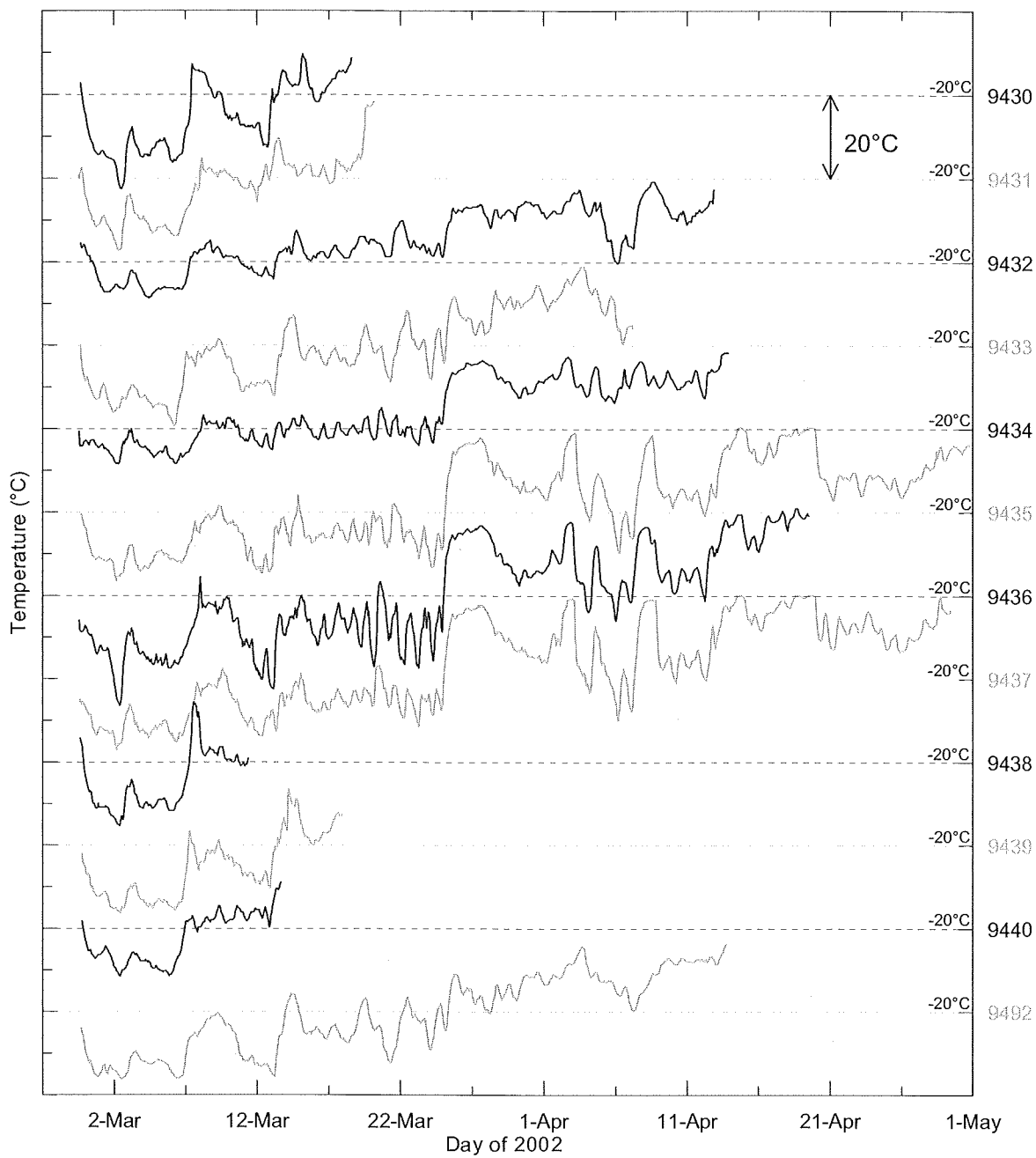


Fig. 14. "Buoy-hull" measured ambient temperature from drifters 9430-9492, corrected time series up to the end of April.

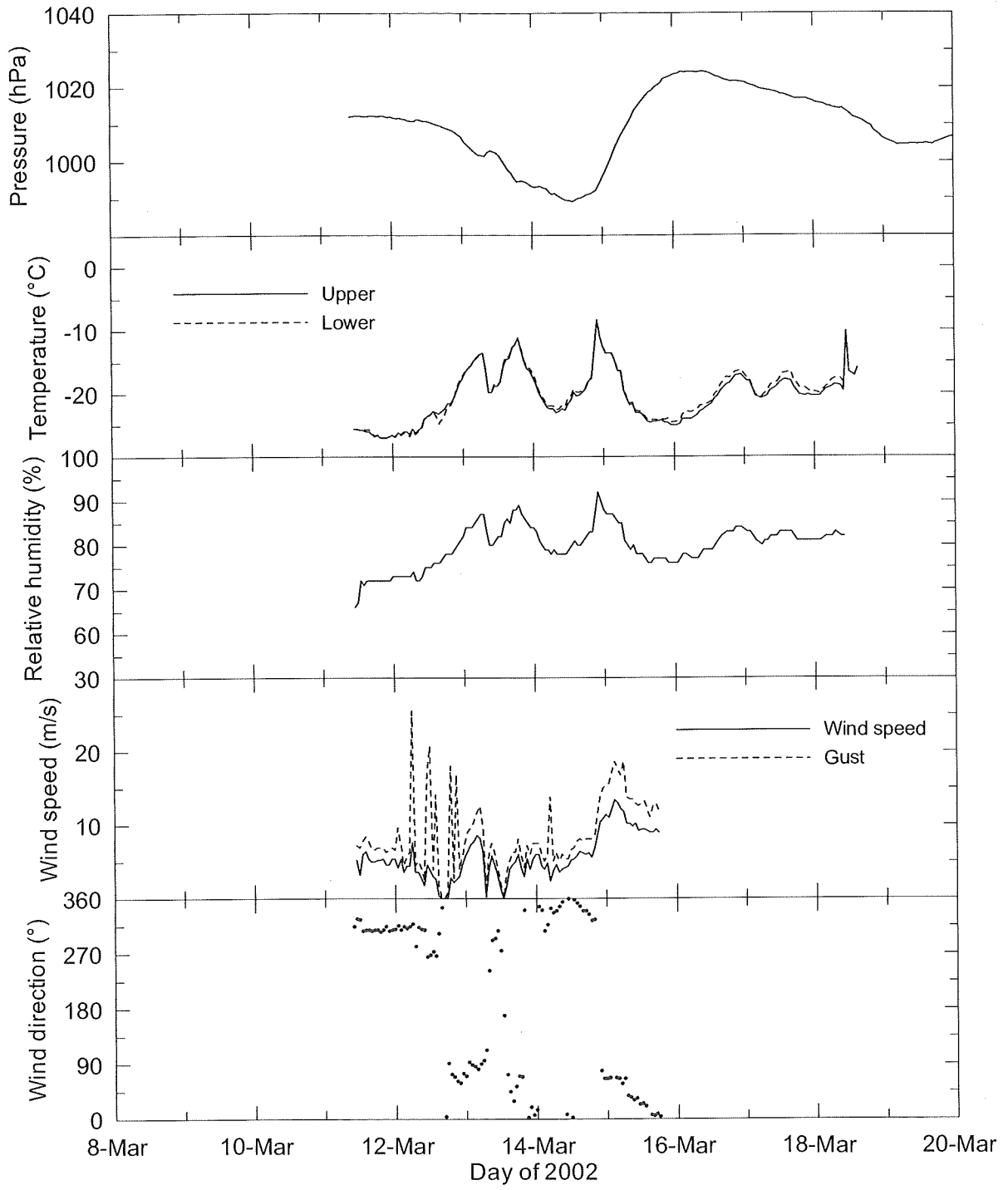


Fig. 15. Meteorological parameters measured by drifter 1154 up to 20.3.2002.

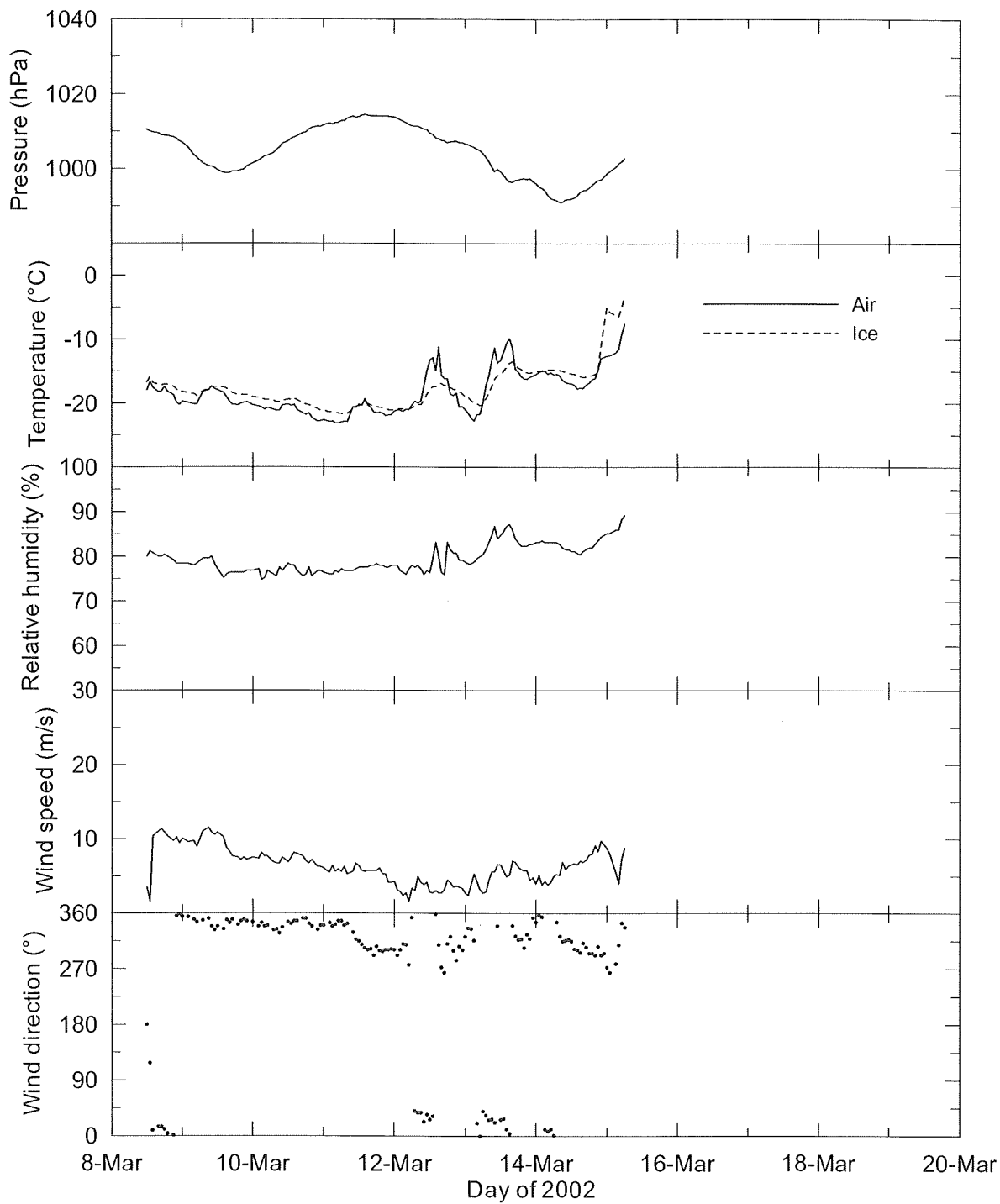


Fig. 16. Meteorological parameters measured by drifter 3333.

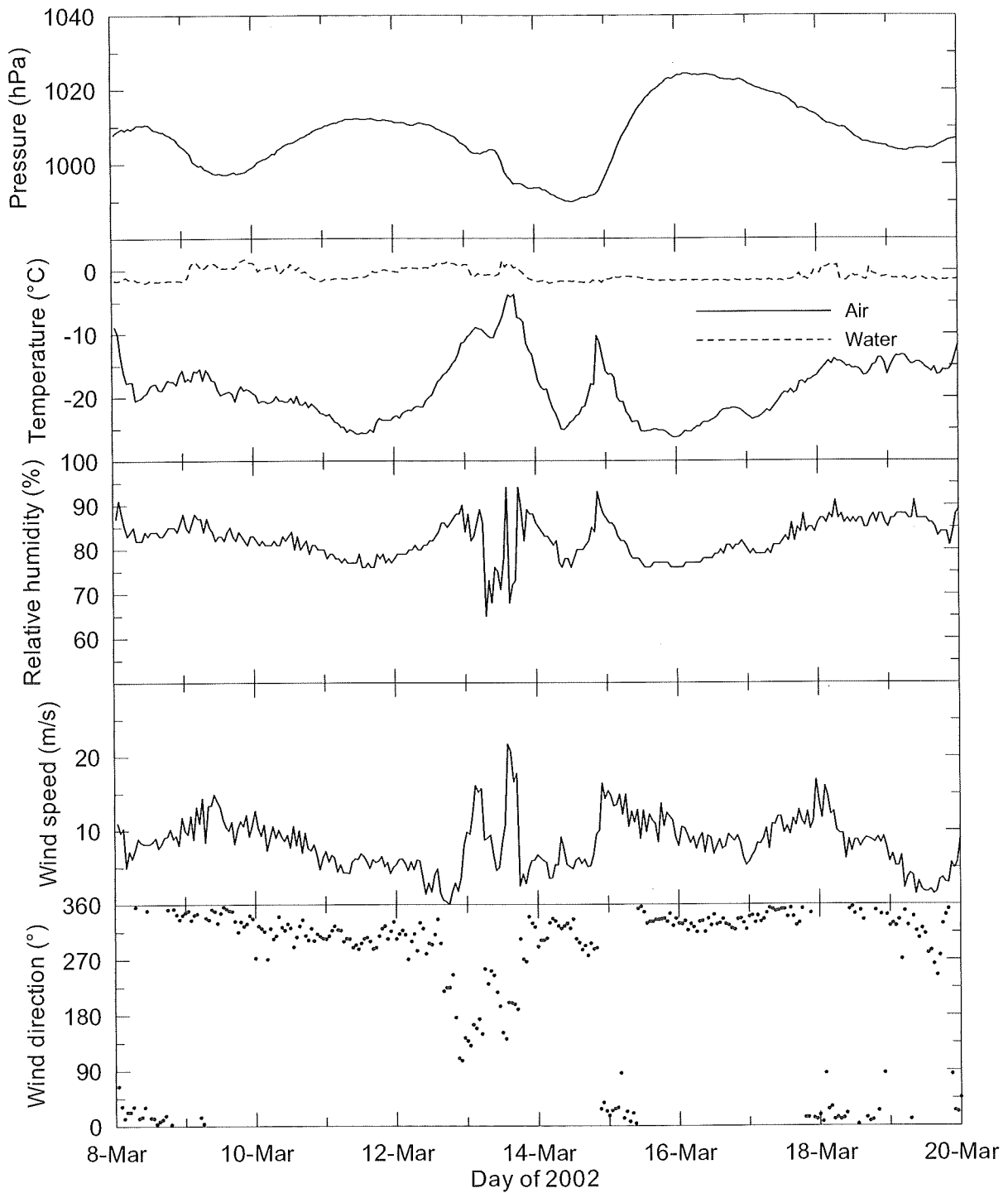


Fig. 17. Meteorological parameters measured by the weather station of RV *Aranda*. Over the period March 8-19, RV *Aranda* operated in the area 78.9°N - 80.4°N, 1°W - 9°E. The air temperature is measured at a height of 13.8 m, the wind speed and direction at 19 m.

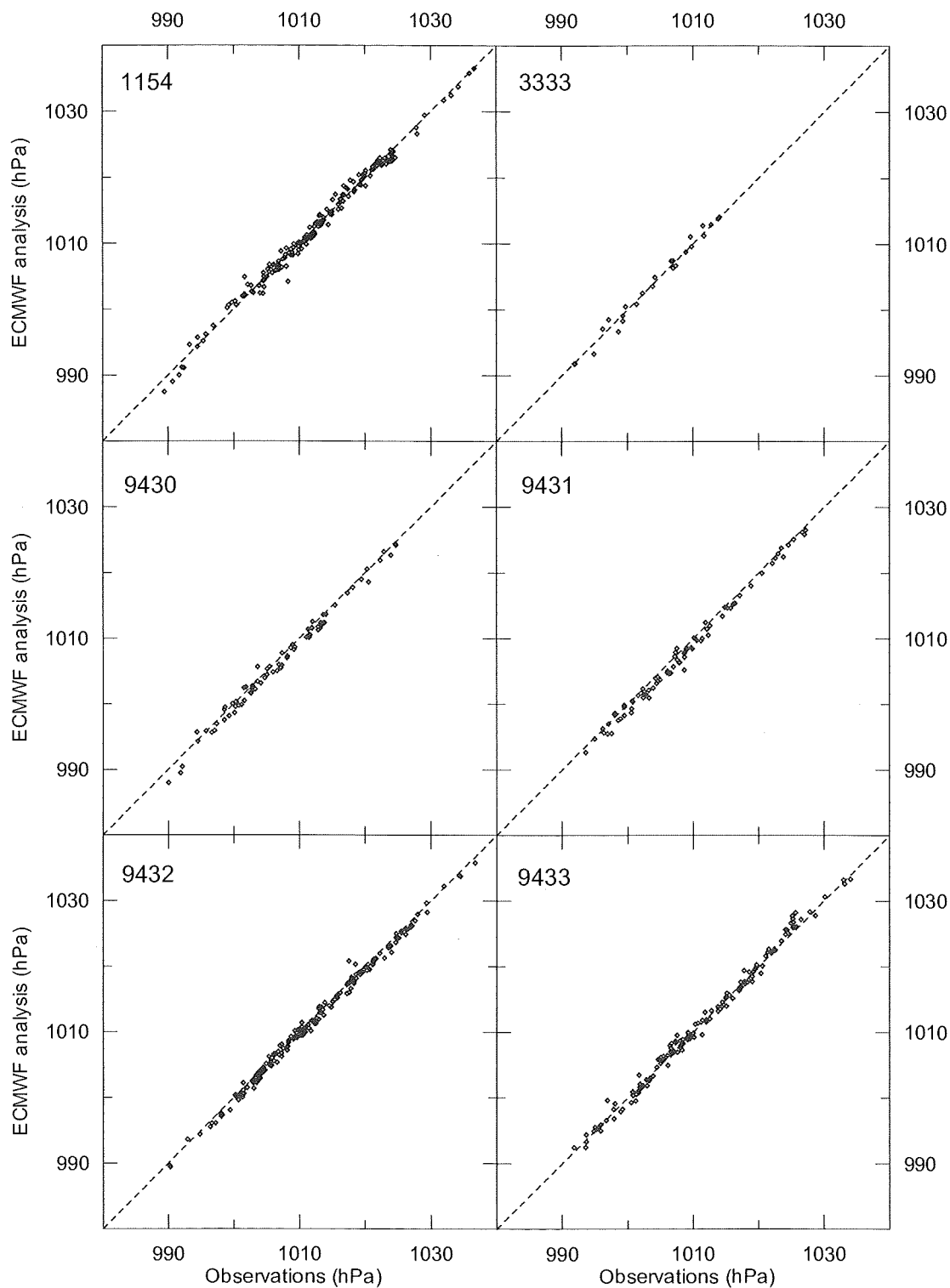


Fig. 18. Atmospheric pressure measured by drifters 1154-9433, in relation to the pressure interpolated for the site from the ECMWF analysis, at 6-hour intervals. The dashed lines denote the ideal 1:1 relationship.

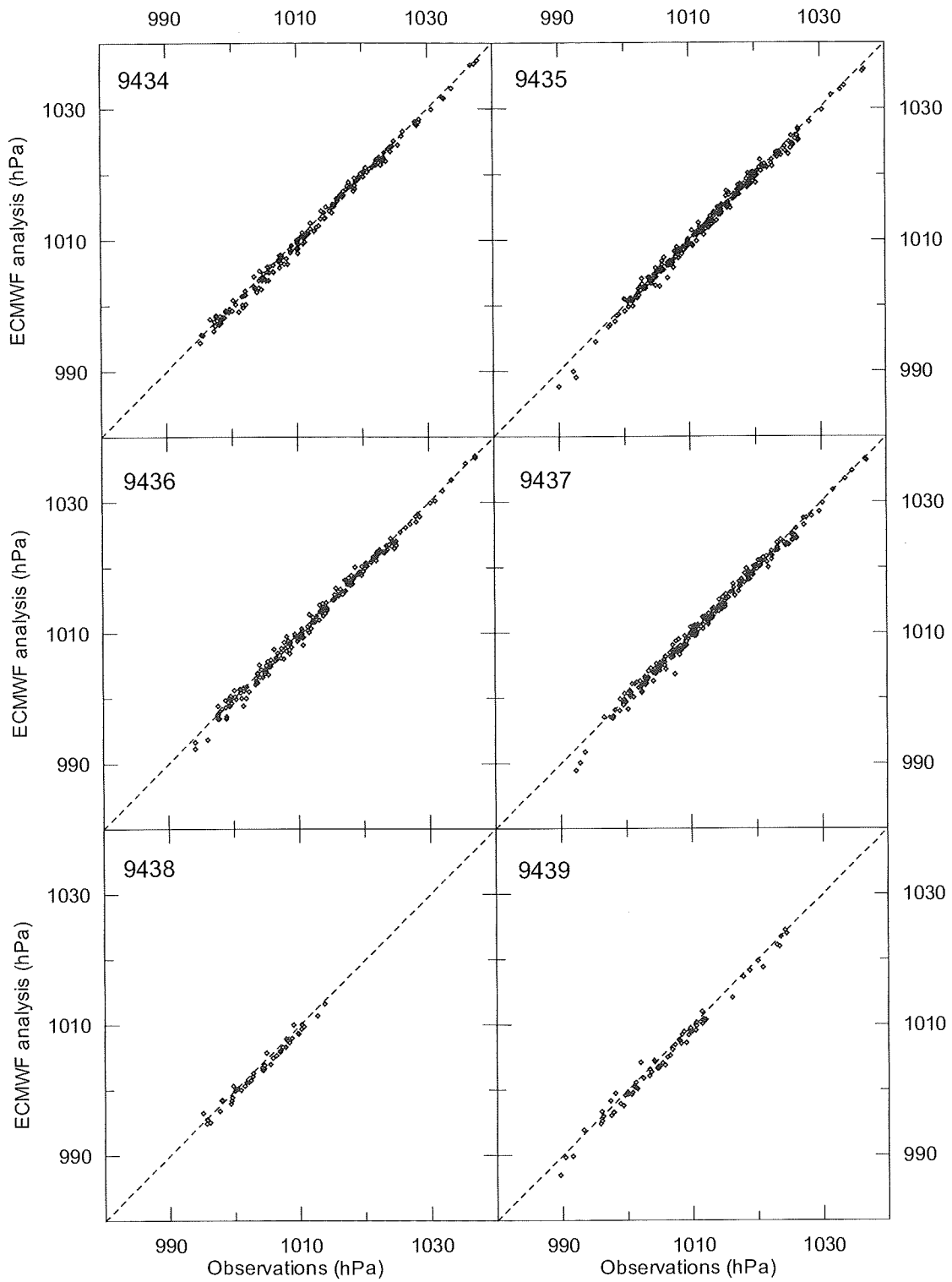


Fig. 19. As Fig. 18, for drifters 9434-9439.

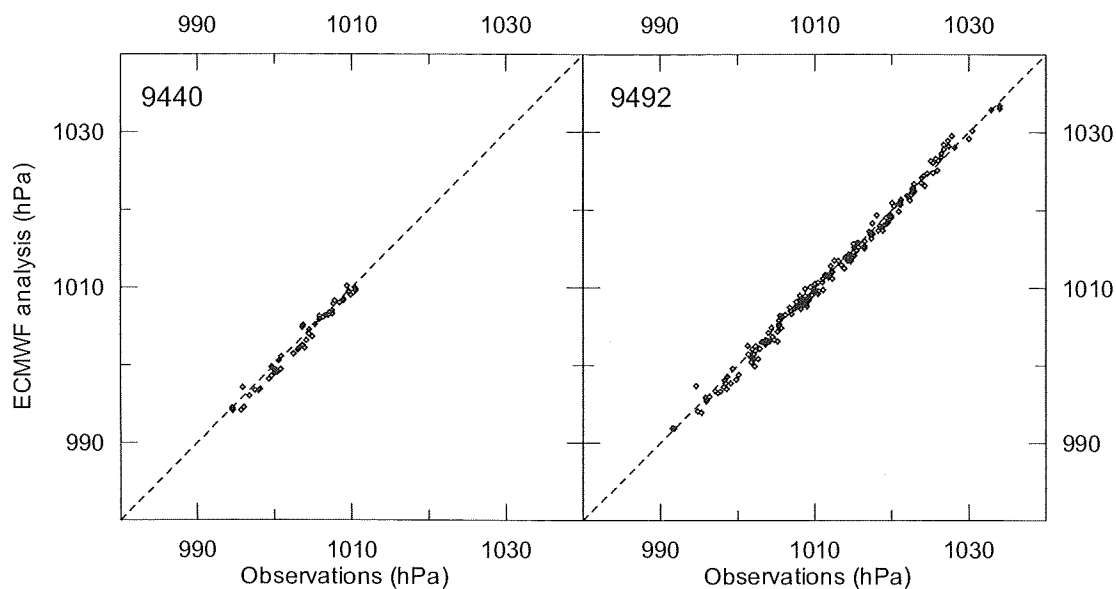


Fig. 20. As Fig. 18, for drifters 9440 and 9492.

Table 6. Comparison of the atmospheric pressure from drifter observations and ECMWF analyses, at 6-hour intervals, up to 30.4.2002. The difference between observations and analyses is denoted by $\Delta p = p_{\text{observed}} - p_{\text{analysis}}$.

| Drifter | Number of points | $(\Delta p)_{\text{mean}}$ / hPa | $(\Delta p)_{\text{max}}$ / hPa | $(\Delta p)_{\text{min}}$ / hPa | $\sqrt{(\Delta p^2)_{\text{mean}}}$ / hPa | R ² |
|---------------|------------------|----------------------------------|---------------------------------|---------------------------------|---|----------------|
| <i>Aranda</i> | 140 | -0.16 | 2.27 | -2.37 | 0.77 | 0.995 |
| 1154 | 187 | 0.06 | 4.05 | -3.29 | 0.89 | 0.990 |
| 3333 | 26 | -0.02 | 1.96 | -1.49 | 0.82 | 0.987 |
| 9430 | 76 | 0.59 | 2.32 | -2.11 | 0.98 | 0.991 |
| 9431 | 81 | 0.75 | 3.30 | -1.00 | 1.05 | 0.993 |
| 9432 | 177 | 0.36 | 1.87 | -3.27 | 0.75 | 0.995 |
| 9433 | 154 | -0.18 | 1.68 | -2.81 | 0.84 | 0.993 |
| 9434 | 180 | 0.45 | 2.23 | -1.25 | 0.81 | 0.995 |
| 9435 | 249 | 0.31 | 3.60 | -1.74 | 0.79 | 0.993 |
| 9436 | 203 | 0.30 | 2.57 | -1.48 | 0.79 | 0.994 |
| 9437 | 243 | 0.31 | 3.80 | -1.33 | 0.79 | 0.993 |
| 9438 | 47 | 0.51 | 1.42 | -1.43 | 0.84 | 0.982 |
| 9439 | 73 | 0.65 | 2.62 | -2.33 | 1.04 | 0.990 |
| 9440 | 56 | 0.47 | 1.67 | -1.47 | 0.85 | 0.979 |
| 9492 | 180 | 0.24 | 2.29 | -2.83 | 0.82 | 0.994 |

4. RESULTS FOR ACSYS-ABSIS 2003

4.1 General drift properties

In 2003, most of the 11 drifters survived for at least one month, some of them surviving up to July (Table 7). Drifter 1097 was recovered from the ice field at the end of the expedition by RV *Polarstern*. Generally, the drifters did not travel as far south as those in the previous year. Instead, most of the drifters remained in the area north of Svalbard. The drifter trajectories are given in Fig. 21.

Table 7. The first and last data messages received from each drifter.

| Drifter | First message (UTC) | | Last message (UTC) | |
|---------|---------------------|-------|--------------------|-------|
| 1097 | 05.04.2003 | 16:47 | 19.04.2003 | 14:58 |
| 3335 | 04.04.2003 | 15:30 | 31.07.2003 | 22:39 |
| 20601 | 27.03.2003 | 13:40 | 11.07.2003 | 02:05 |
| 20626 | 27.03.2003 | 12:02 | 12.04.2003 | 06:07 |
| 20627 | 27.03.2003 | 12:22 | 23.05.2003 | 05:03 |
| 20751 | 27.03.2003 | 15:18 | 07.04.2003 | 09:23 |
| 20755 | 27.03.2003 | 13:38 | 11.07.2003 | 06:01 |
| 20756 | 27.03.2003 | 15:14 | 24.05.2003 | 02:42 |
| 20811 | 27.03.2003 | 15:18 | 10.06.2003 | 19:05 |
| 20835 | 27.03.2003 | 13:40 | 21.07.2003 | 22:03 |
| 20842 | 27.03.2003 | 14:11 | 31.03.2003 | 03:43 |

4.2 Data evaluation

4.2.1 Drifter positions

In total, from 55 to 3 793 position fixes for each drifter were obtained, resulting in 15-32 fixes per day for most drifters (Table 8). Only two of the drifters behaved differently, yielding only 5-7 position fixes per day. The data quality seems to be slightly better than during the previous year, the most accurate class 3 being the most common one observed for all the drifters, except for the abovementioned two. Contrary to the uneven daily distribution of location fixes observed during FRAMZY in 2002 (Fig. 8), the drifters in 2003 did not exhibit such behaviour. Instead, the location fixes were more or less evenly distributed among the hours of the day.

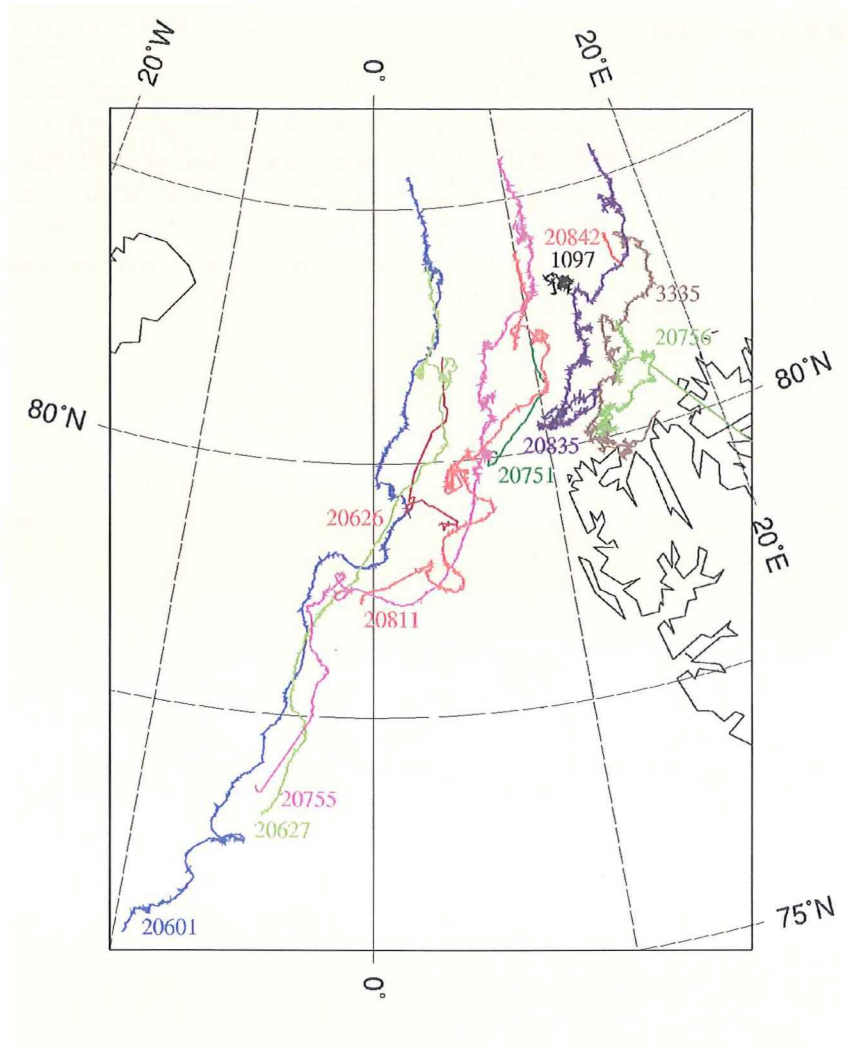


Fig. 21. Drifter trajectories in 2003.

Table 8. Observed number of locations for each drifter, length of the period, and the percentages of the different Argos location classes in 2003.

| Drifter | Locations, total | Days of operation ^{*)} | Locations/day, average | Class 3 (%) | Class 2 (%) | Class 1 (%) |
|---------|------------------|---------------------------------|------------------------|-------------|-------------|-------------|
| 1097 | 380 | 13.9 | 27.3 | 43 | 37 | 20 |
| 3335 | 3 790 | 117.6 | 32.2 | 49 | 35 | 16 |
| 20601 | 3 155 | 105.3 | 30.0 | 46 | 34 | 20 |
| 20626 | 71 | 15.5 | 4.6 | 28 | 35 | 37 |
| 20627 | 1 509 | 56.5 | 26.7 | 44 | 36 | 20 |
| 20751 | 76 | 10.7 | 7.1 | 24 | 41 | 36 |
| 20755 | 3 226 | 105.6 | 30.6 | 45 | 36 | 19 |
| 20756 | 1 665 | 57.4 | 29.0 | 47 | 33 | 21 |
| 20811 | 2 073 | 75.0 | 27.6 | 41 | 36 | 22 |
| 20835 | 3 575 | 116.3 | 30.8 | 50 | 33 | 16 |
| 20842 | 55 | 3.5 | 15.6 | 44 | 40 | 16 |
| Total | 19 575 | 677.3 | 28.9 | 47 | 35 | 19 |

^{*)} Time between the first and last position fix.

4.2.2 Interpolation of the trajectories

Semivariograms were calculated for each drifter. As was the case in 2002, the b values vary between 1 and 2, with two exceptions (Table 9). These exceptions were possibly a result of quite short trajectories in the cases of drifters 1097 and 20842. Drifter 1097 has a b_x value slightly less than one. Despite the non-physical nature of such a value, the interpolated trajectory seems to be acceptable, and thus no correction was made. The b_y value of 2.03 for drifter 20842 clearly disturbs the interpolation, however. Thus, the value was replaced by $b_y = 2.0$ before interpolation. The corrected value results in an acceptable interpolation.

Table 9. Semivariogram parameters for the ACSYS/ABSIS drifters.

| Drifter | b_x | c_x | b_y | c_y |
|---------|-------|-------|-------|-------|
| 1097 | 0.97 | 7.46 | 1.03 | 14.1 |
| 3335 | 1.18 | 11.8 | 1.38 | 12.5 |
| 20601 | 1.52 | 24.3 | 1.68 | 67.2 |
| 20626 | 1.66 | 51.8 | 1.87 | 88.9 |
| 20627 | 1.53 | 30.9 | 1.72 | 108 |
| 20751 | 1.92 | 84.0 | 1.76 | 116 |
| 20755 | 1.55 | 43.1 | 1.62 | 52.9 |
| 20756 | 1.15 | 17.3 | 1.15 | 14.5 |
| 20811 | 1.42 | 47.8 | 1.48 | 51.9 |
| 20835 | 1.37 | 12.6 | 1.67 | 15.1 |
| 20842 | 1.22 | 4.84 | 2.03 | 59.7 |

The coefficients b_x and b_y are slightly smaller for these drifters than for those in 2002. This reflects the less rectilinear nature of the movement of the drifter array in 2003. The coefficients c , which represent the amplitude of the variability on a certain time scale, are smaller for these drifters than for those in 2002. This reflects - in addition to the abovementioned difference - the generally smaller drift velocities in 2003.

4.2.3 Drift velocities

The drift was oriented towards the south-southwest on average, except for the two short-lived drifters 1097 and 20842 (Table 10, Figs. 22-23). The average speed was of the order of 0.01 - 0.12 m/s, which is clearly lower than the average speeds observed in 2002. The maximum speeds are also generally lower. The maximum speeds were checked for possibly erroneous location data, as in the earlier case. The frequency distributions of speed and drift direction for two representative drifters are given in Fig. 24.

It is evident that tidal forcing had an effect on the drift (Figs. 22-23). Especially in the periods 21.-26.4.2003 and 3.-15.5.2003 the diurnal component is clearly visible in the velocity components of some drifters. Some tidal forcing is also present in the velocities of the FRAMZY drifters (Figs. 9-10), but it is not as strong and evident as in this case.

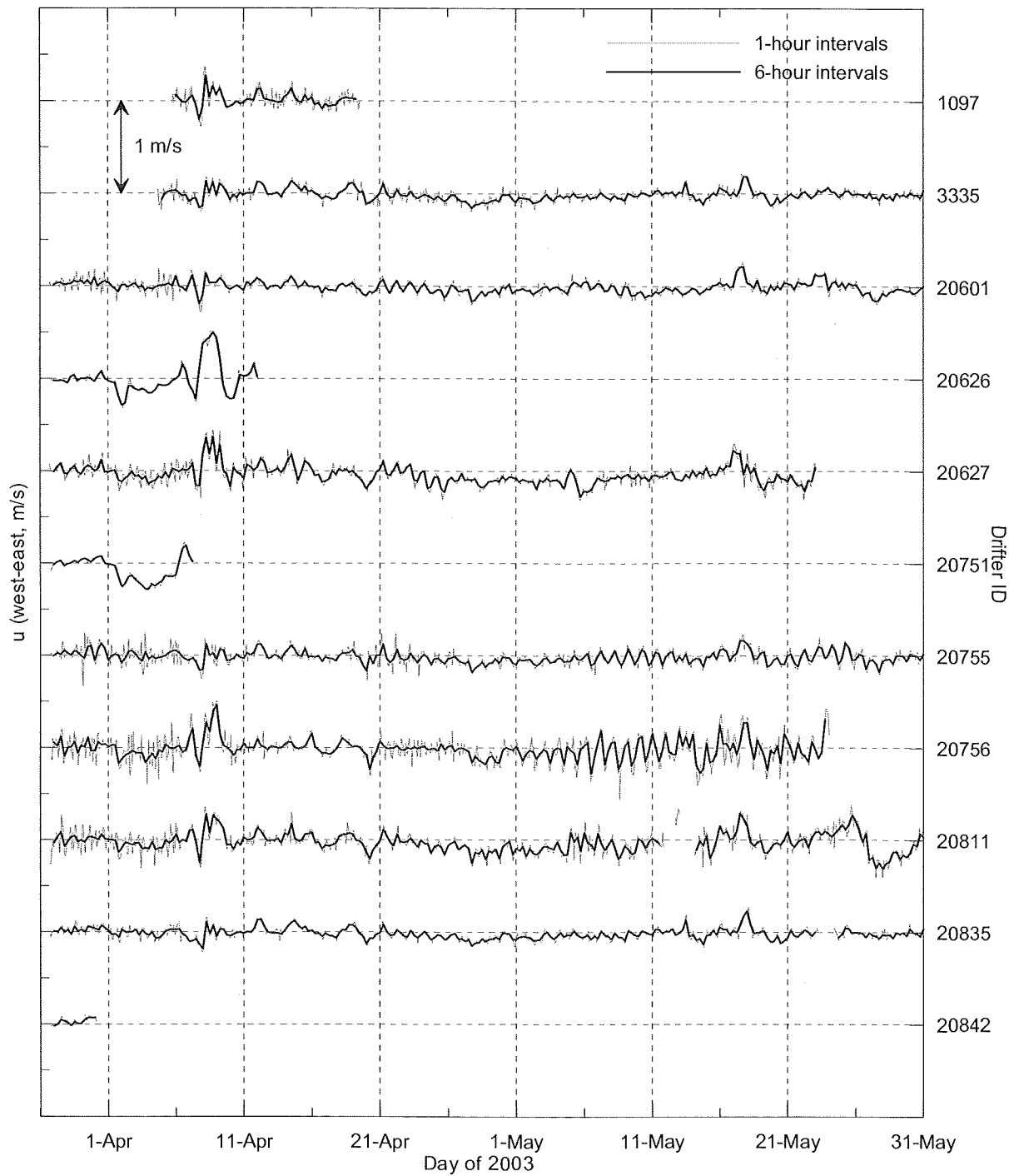


Fig. 22. West-east velocity component for the various drifters, up to the end of May.

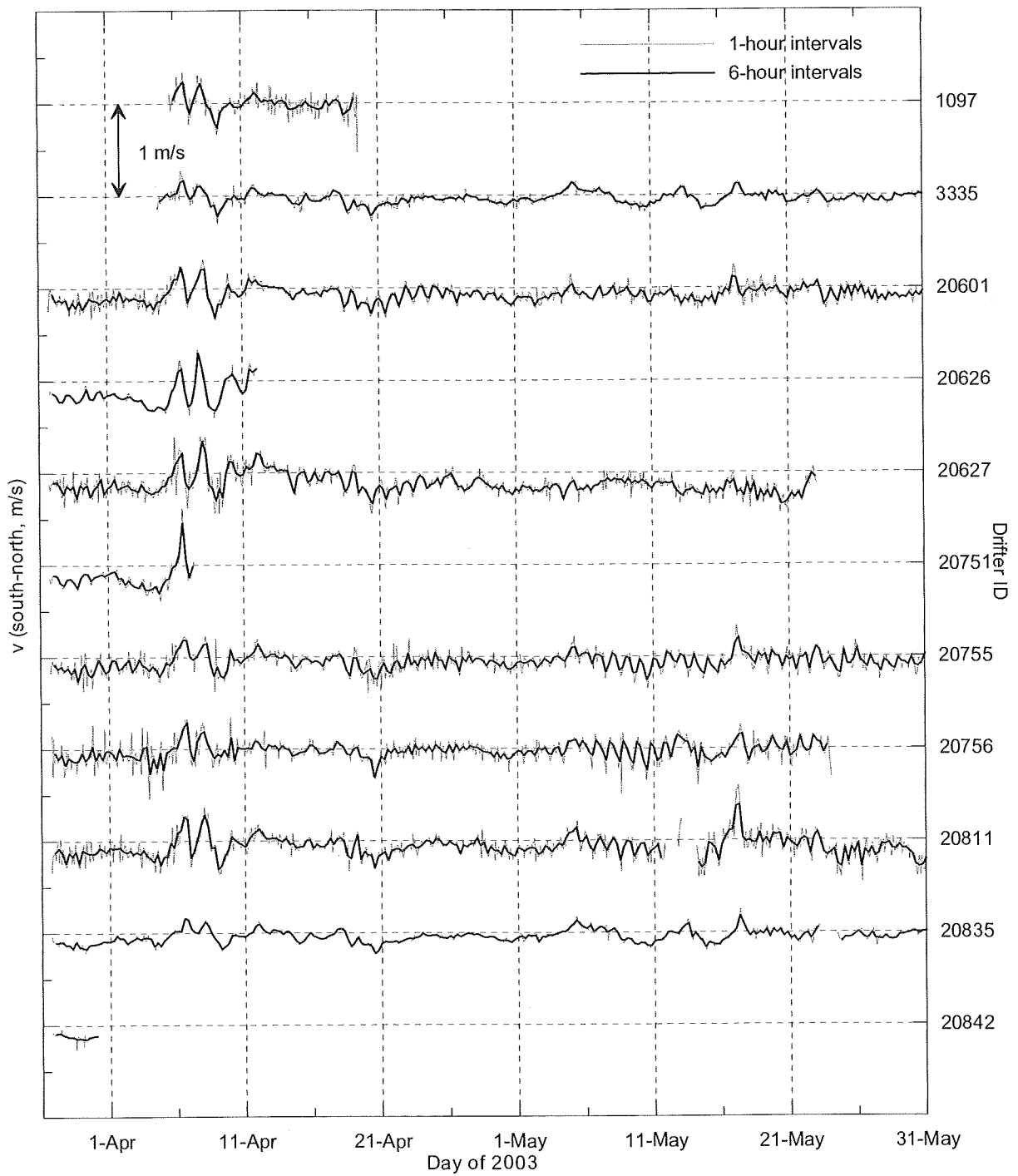


Fig. 23. South-north velocity component for the various drifters, up to the end of May.

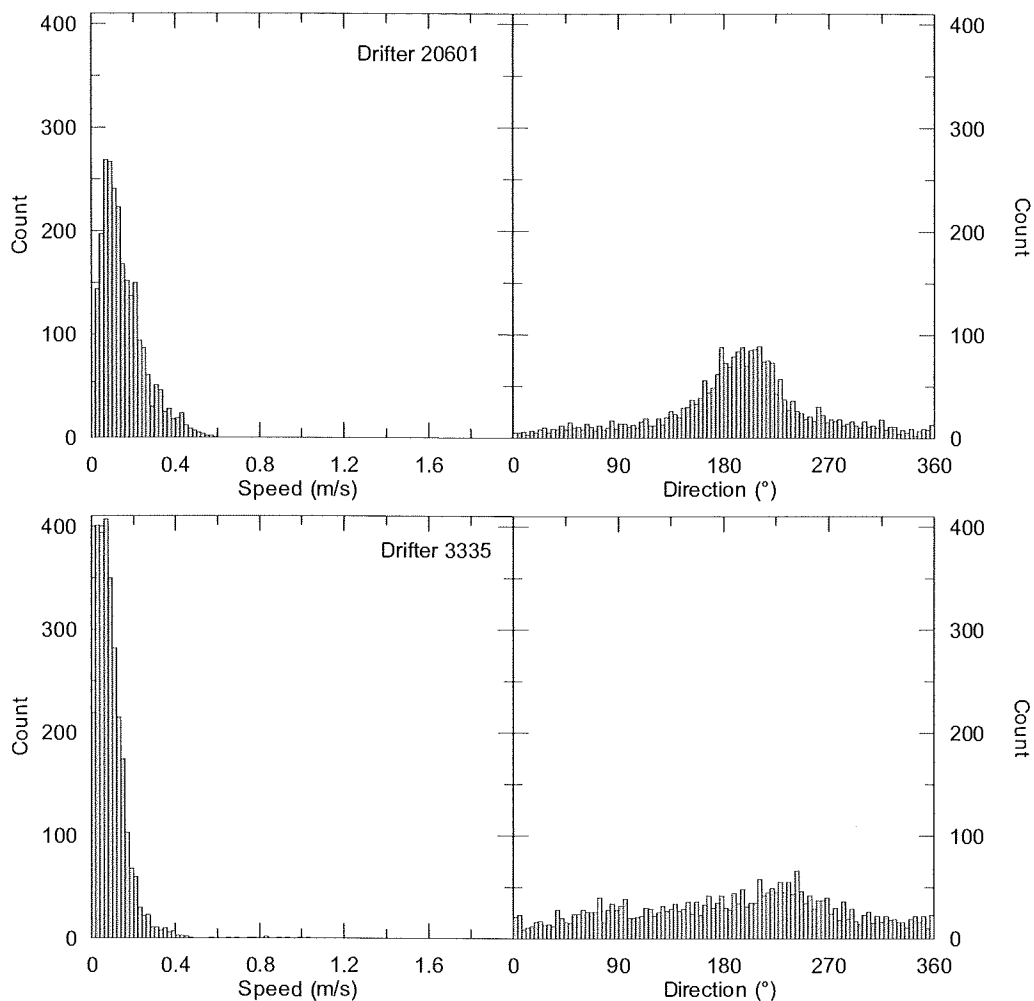


Fig. 24. Distributions of drift speed and direction for two representative drifters, calculated from hourly velocities. Drifter 20601 drifted on a long path southwestward, while drifter 3335 was stuck inside a limited area north of Svalbard (Fig. 21).

Table 10. The mean and maximum velocities of the drifters in 2003, calculated from locations interpolated at 6-hour intervals. The maximum velocities calculated from 1-hourly data would be somewhat larger than those given in the table, but since they are very sensitive to errors and gaps in the original location data, they are not given.

| Drifter | u_{mean} (m/s) (west-east) | v_{mean} (m/s) (south-north) | Mean speed (m/s) | Direction (°) | Max. speed (m/s) |
|---------|---------------------------------|-----------------------------------|---------------------|------------------|---------------------|
| 1097 | 0.01 | 0.00 | 0.01 | 87 | 0.30 |
| 3335 | 0.00 | -0.02 | 0.02 | 187 | 0.87 |
| 20601 | -0.03 | -0.09 | 0.10 | 199 | 0.51 |
| 20626 | 0.00 | -0.14 | 0.14 | 181 | 0.59 |
| 20627 | -0.04 | -0.12 | 0.13 | 197 | 0.42 |
| 20751 | -0.06 | -0.13 | 0.14 | 206 | 0.49 |
| 20755 | -0.03 | -0.07 | 0.08 | 204 | 0.64 |
| 20756 | -0.01 | -0.02 | 0.02 | 203 | 0.47 |
| 20811 | -0.03 | -0.06 | 0.07 | 208 | 0.56 |
| 20835 | -0.01 | -0.03 | 0.03 | 204 | 0.42 |
| 20842 | 0.03 | -0.12 | 0.13 | 167 | 0.16 |

4.3 Buoy meteorological data

4.3.1 Data time series

The time series of the corrected air pressure and the "buoy-hull" measured ambient (air) temperature for drifters 20601-20842 are shown in Figs. 25 and 26. The time series of pressure, temperature, humidity and wind for drifters 1097 and 3335 are presented in Figs. 27 and 28. The parameters measured by RV *Aranda* are shown in Fig. 29.

4.3.2 Comparison with weather analyses

Scatter plots of the observed pressure values in relation to those obtained from the ECMWF analyses are given in Figs. 30-31. Numerical results of the comparison are given in Table 11. As was the case in 2002, the data in 2003 were not fed to the real-time ECMWF observation data assimilation, either.

The atmospheric pressure given by the ECMWF tends to be slightly smaller than the value measured by the drifters, in accordance with a similar result for the 2002 data.

Table 11. Comparison of the sea level pressure from drifter observations and ECMWF analyses, at 6-hour intervals, up to 31.5.2003. The difference between observations and analyses is denoted by $\Delta p = P_{\text{observed}} - P_{\text{analysis}}$.

| Drifter | Number of points | $(\Delta p)_{\text{mean}} /$ hPa | $(\Delta p)_{\text{max}} /$ hPa | $(\Delta p)_{\text{min}} /$ hPa | $\sqrt{(\Delta p^2)_{\text{mean}}} /$ hPa | R ² |
|---------------|------------------|-------------------------------------|------------------------------------|------------------------------------|--|----------------|
| <i>Aranda</i> | 104 | -0.01 | 3.15 | -2.97 | 0.91 | 0.993 |
| 1097 | 56 | -0.44 | 1.64 | -1.74 | 0.85 | 0.996 |
| 3335 | 231 | 0.50 | 2.64 | -1.80 | 0.85 | 0.995 |
| 20601 | 262 | 0.44 | 1.94 | -1.61 | 0.82 | 0.995 |
| 20626 | 50 | 0.23 | 1.78 | -1.64 | 0.83 | 0.995 |
| 20627 | 226 | 0.40 | 3.94 | -1.66 | 0.86 | 0.995 |
| 20751 | 40 | 0.57 | 1.55 | -1.18 | 0.93 | 0.992 |
| 20755 | 262 | 0.51 | 1.94 | -1.51 | 0.83 | 0.996 |
| 20756 | 230 | 0.38 | 3.05 | -2.62 | 0.92 | 0.994 |
| 20811 | 254 | 0.84 | 3.16 | -1.13 | 1.10 | 0.995 |
| 20835 | 257 | 0.44 | 2.09 | -1.43 | 0.75 | 0.996 |
| 20842 | 14 | 0.40 | 1.65 | -1.21 | 0.80 | 0.945 |

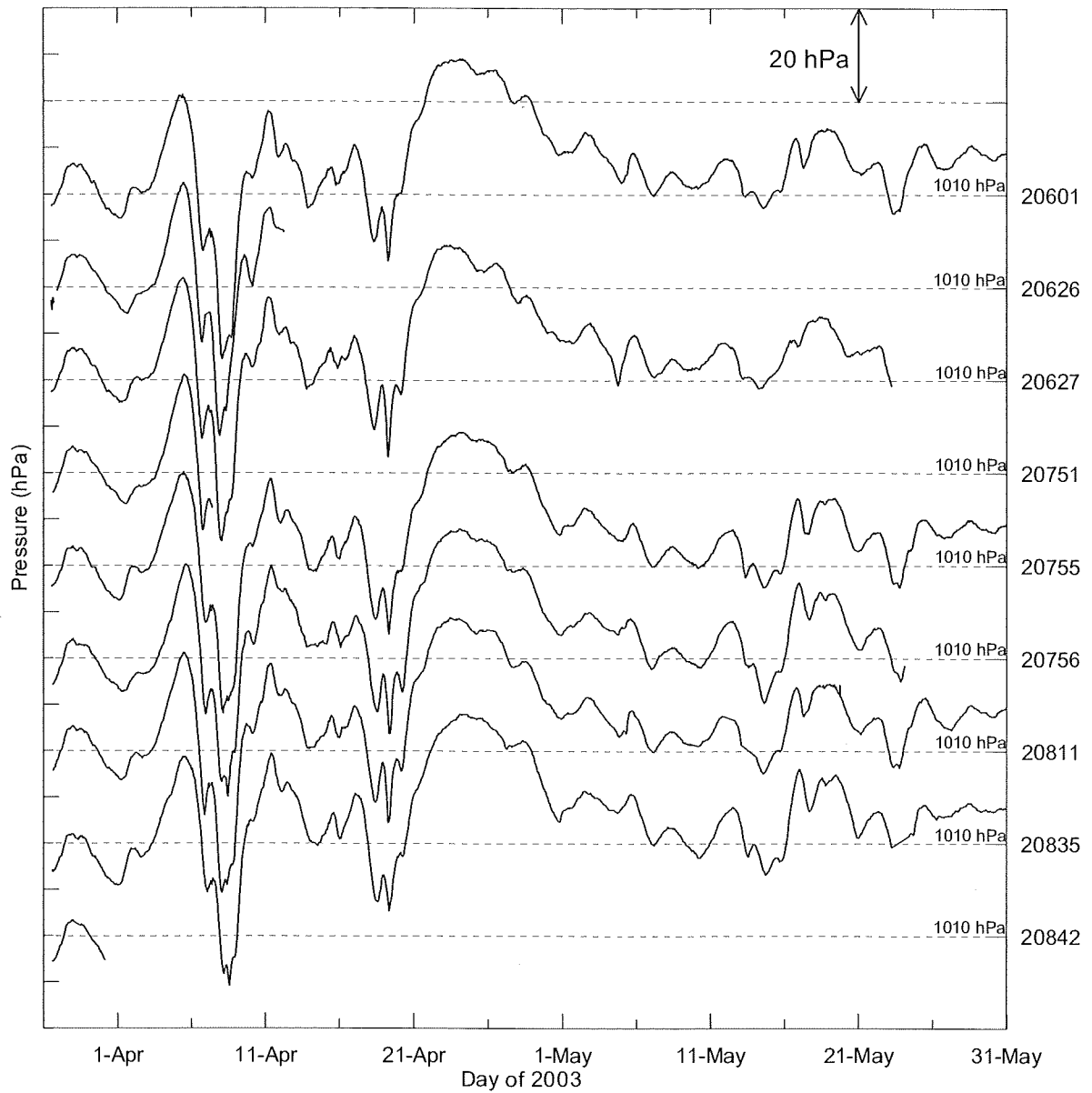


Fig. 25. Air pressure measured by drifters 20601-20842 in 2003, corrected time series (see text) up to the end of May.

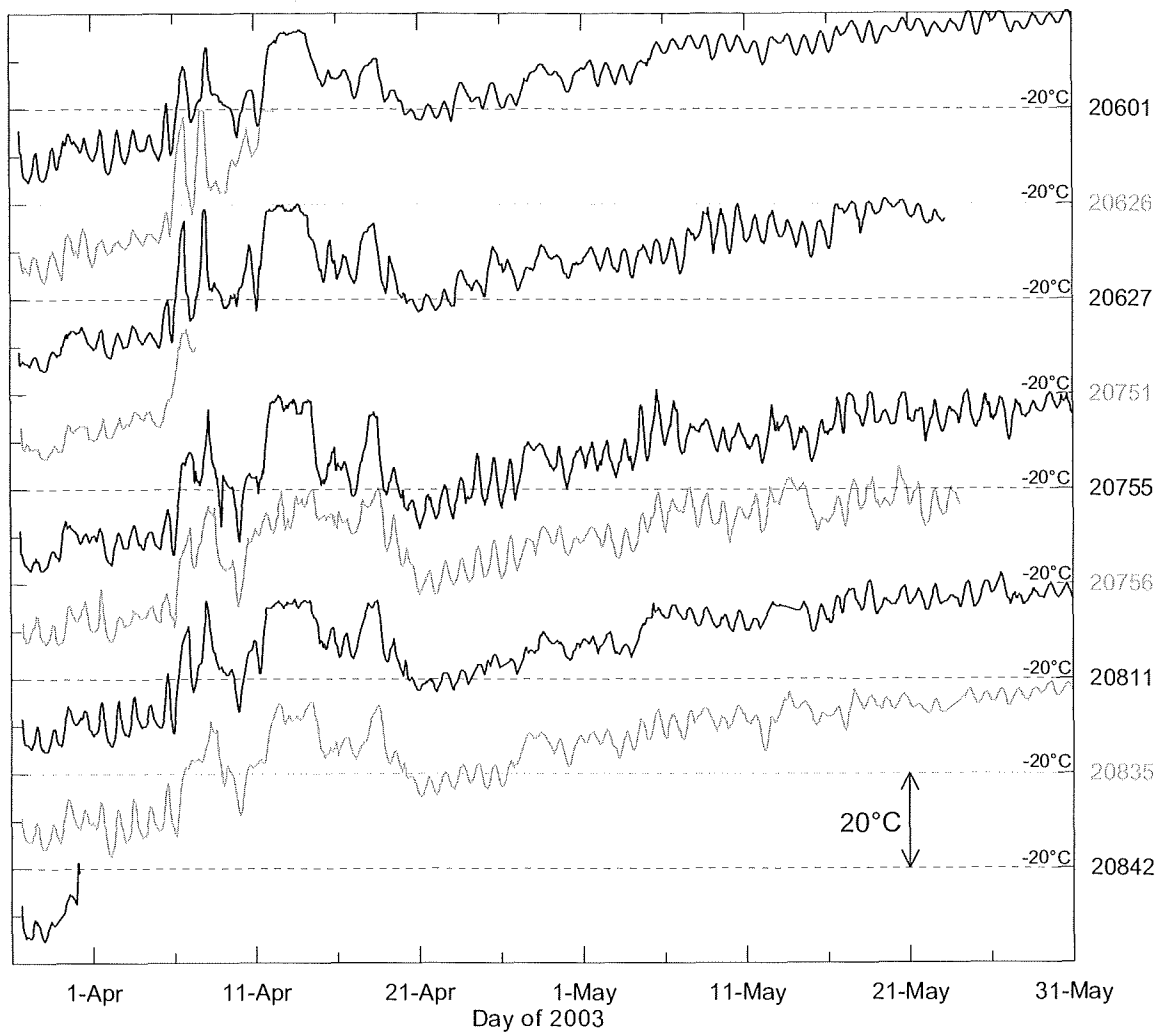


Fig. 26. "Buoy-hull" measured ambient temperature from drifters 20601-20842, corrected time series up to the end of May.

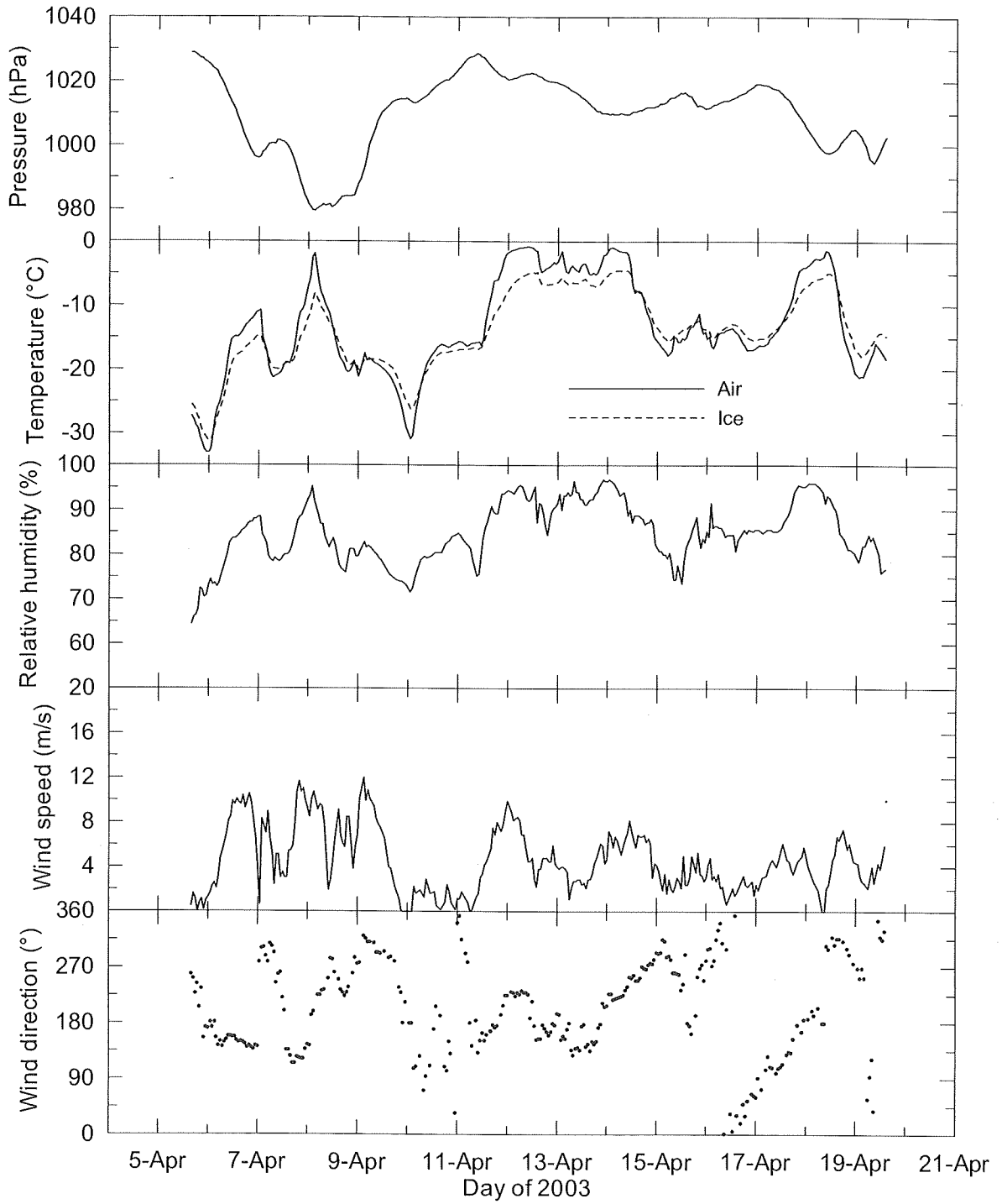


Fig. 27. Meteorological parameters measured by drifter 1097.

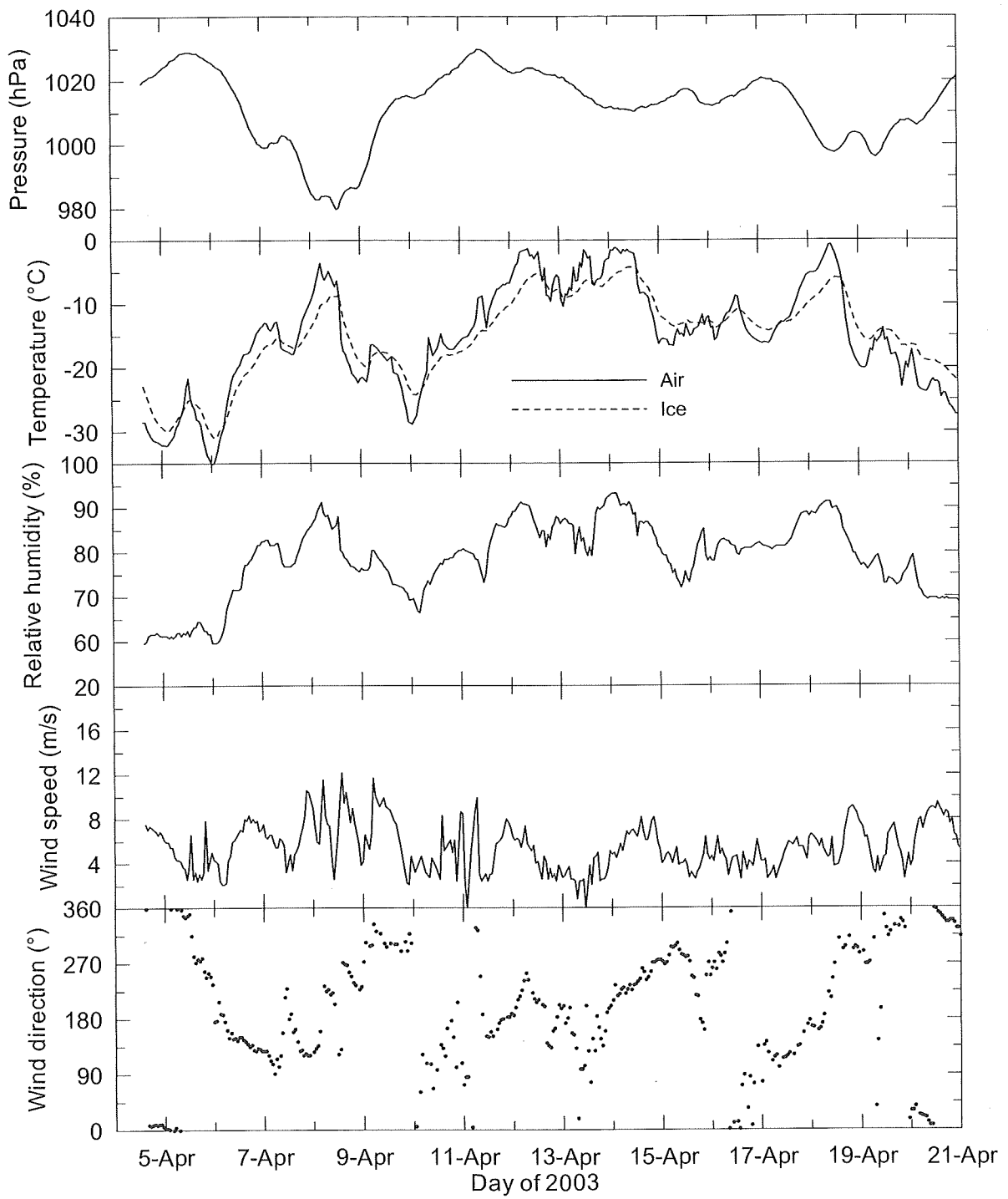


Fig. 28. Meteorological parameters measured by drifter 3335, up to 21.4.2003.

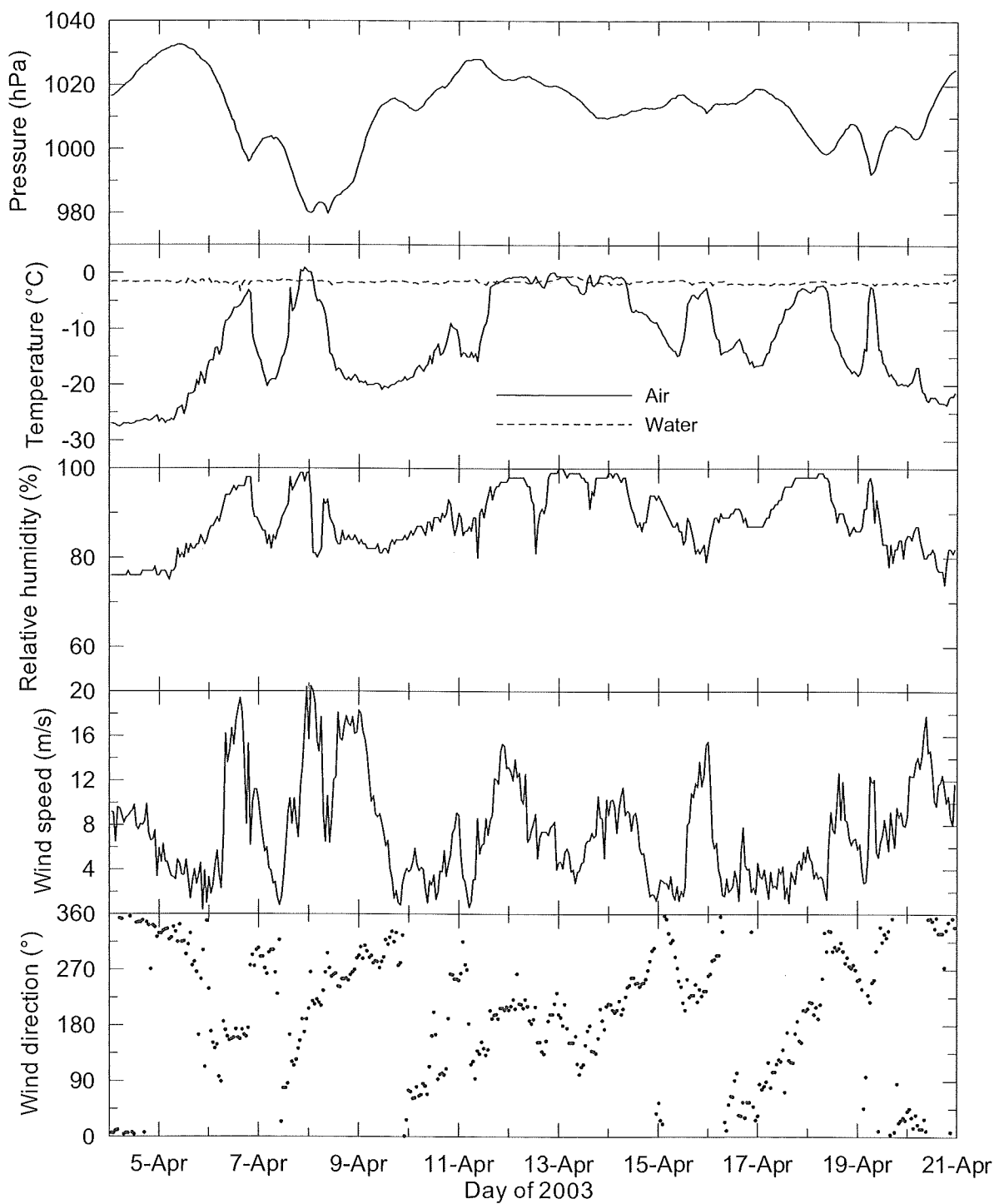


Fig. 29. Meteorological parameters measured by the weather station of RV *Aranda*. Over the period April 4-20, RV *Aranda* operated in the area 79.6°N - 81.0°N, 0°E - 11°E.

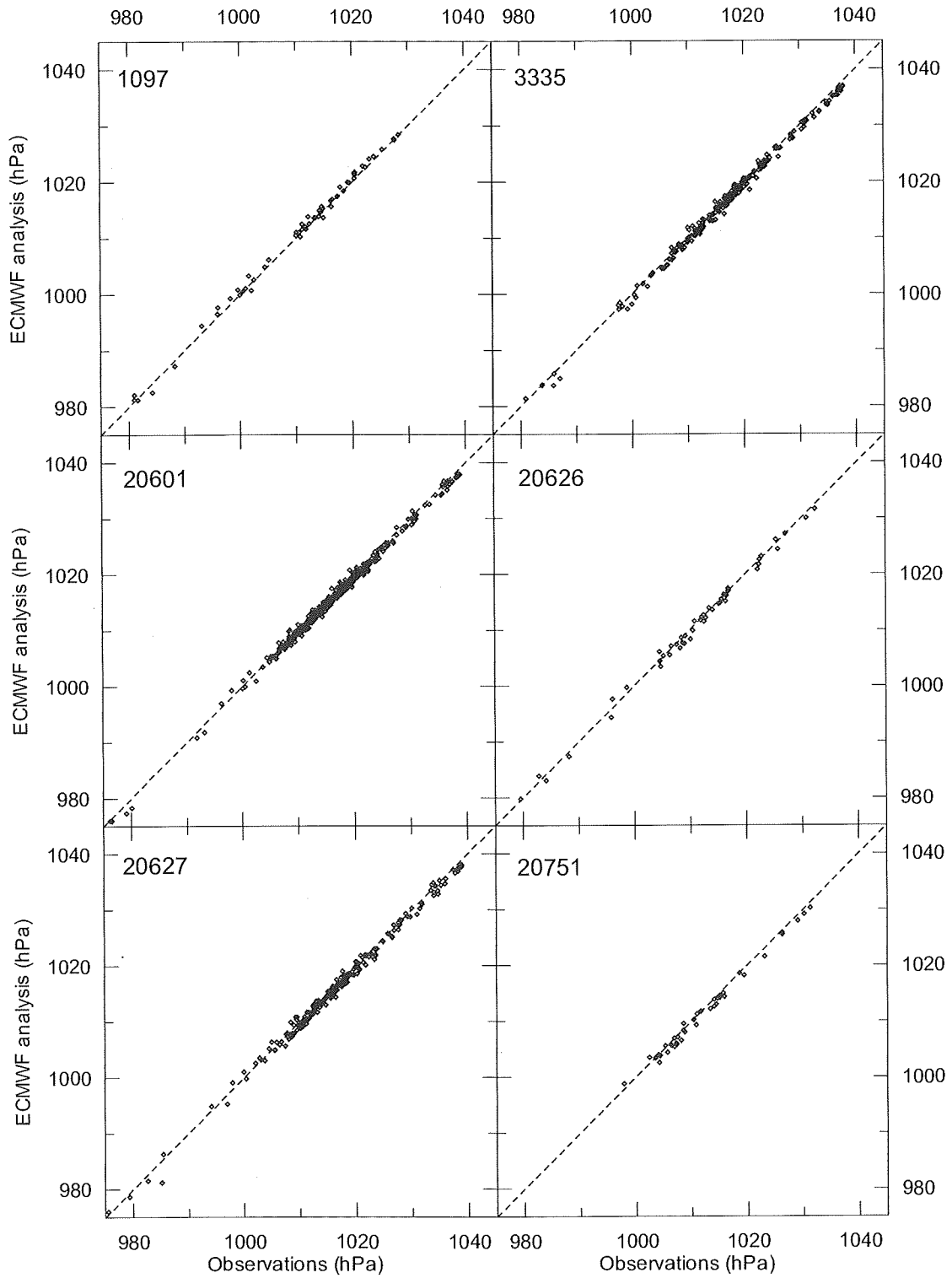


Fig. 30. Atmospheric pressure measured by drifters 1097-20751, in relation to the pressure interpolated for the site from the ECMWF analysis, at 6-hour intervals. The dashed lines denote the ideal 1:1 relationship.

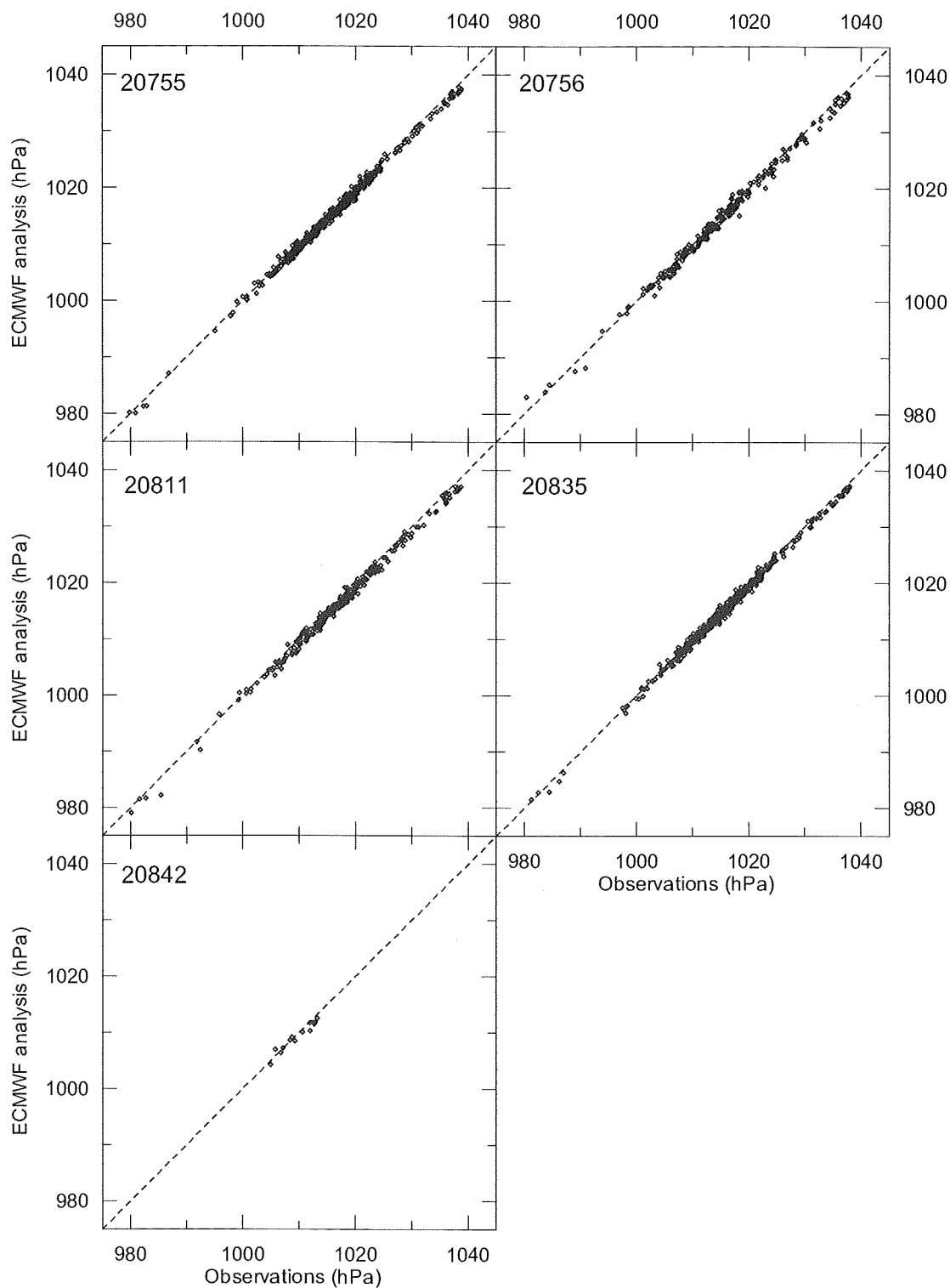


Fig. 31. As Fig. 30, for drifters 20755-20842.

Acknowledgements

We thank the captain and crew of RV *Aranda*, as well as those of RV *Polarstern*, for successful cooperation during the expeditions. The European Centre for Medium-Range Weather Forecasts (ECMWF) is acknowledged for providing their analysis data. The NCEP Reanalysis data was provided by the NOAA-CIRES ESRL/PSD Climate Diagnostics branch, Boulder, Colorado, USA.

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- (Ice concentration data from <http://www.seaice.de/>)

REVIEW OF OIL SPILL EFFECTS ON ARCTIC MARINE ECOSYSTEMS

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ABSTRACT

This review was prepared for the EU DG TREN-funded project "Arctic Operational Platform" (ARCOP), and is based on earlier published literature of oil effects on arctic marine organisms. The project description and all public reports can be found from www.arcop.fi.

The effects of oil on arctic marine organisms are very variable, and depend on several factors, such as the species, life habit and the developmental state of organism that becomes subjected to oil, type of oil, time of exposition, and abiotic environmental factors (e.g. water temperature, presence of ice). The oil effects may be acute or long-term, and the oil and its degradation products may either bioaccumulate into a given organism or biomagnify within the food web. The causes of an oil spill may be locally very dramatic and cause non-recoverable changes in the ecosystem. This is true particularly for benthic and littoral communities, while pelagic (open water) systems typically have a better potential for recovery.

Key words: ARCOP, oil, arctic marine ecosystems, plankton, benthos, littoral, birds, mammals, fish

1. INTRODUCTION

Oil in the Arctic marine ecosystem originates mainly from three sources: drilling activity, leakage from ships, and oil spills during transportation. Drilling activity and oil leakage cause long-term exposure and thus chronic effects on Arctic marine biota, such as changes in species composition, dominance and biomass, while oil spill effects are acute and can cause severe damage locally. In the scope of the ARCOP project this report concentrates on the latter type of contamination.

Oil in the marine environment affects organisms on all systematic levels: microscopic plankton (phyto- and zooplankton), invertebrates such as crustaceans, molluscs and benthic worms, and vertebrates (e.g. fish, birds, seals, polar bear). To suffer from the negative effects of oil, a given organism does not need to become directly into contact with the medium: biomagnification is a process where oil is transported within the food web from a lower level to the next one.

Bioaccumulation of oil (concentration of oil into one organism) leads to more pronounced effects than when the organism is in contact with oil only for a short period of time. Petroleum effects on a given organism can vary from mere physical nuisance (such as oil clinging onto the body) to pathological. Oil can affect different organs and physiological functions, and thus lead to changes in e.g. behaviour (feeding, activity and motility, avoidance reactions etc.), growth, and reproduction. The developmental stage of an organism is often crucial; generally, larval and juvenile stages are more vulnerable than adult individuals.

In this review, examples of oil effects on life are given for all Arctic marine habitats: plankton (chapter 2.1.), littoral and benthic communities (chapter 2.2.), and vertebrates (fish, birds, otters, seals, whales and the polar bear; chapter 2.3.). Information is gathered through times mainly by performing experiments in the laboratory and on field (in particular in the 1970's and 1980's), and by sampling the biota during an oil spill.

2. BIOLOGICAL EFFECTS OF OIL, IN PARTICULAR PAHS

Oil consists of a wide variety of compounds that are toxic to organisms, the worst being the polynuclear aromatic hydrocarbons (PAHs). The effect of PAH compounds on a given marine organism is dependent on numerous factors, which can be both abiotic and biotic (GESAMP 1993, Mosbech 2002a). In the arctic environment, important abiotic factors that affect in particular oil spreading and weathering processes include e.g. the presence or absence of ice and snow cover, water temperature, light conditions, and vertical and horizontal water currents (e.g. Mackay & al. 1975, Clark & Finley 1982, Mackay 1985 (and references therein), Payne & al. 1991, Sydnes 1991, Singaas & al. 1994). Biological events in the Arctic marine ecosystem are strongly linked to specific seasons, thus the possible consequences of a relatively short-lived oil spill depend on the time of the year. Should an oil spill take place after the ice break-up in spring, it would affect the vernal bloom of ice algae and phytoplankton, and thus the rest of the pelagic food chain. Migratory birds would get disposed to oil, which in turn would be transported to coast and thus affect the littoral communities too.

Biochemical processes affecting oil, in particular PAHs in marine ecosystems, and bioavailability of PAH to aquatic organisms are discussed in depth in the works of McElroy & al. (1989) and GESAMP (1993), and are illustrated in Figure 1. Oil is transformed and transported within the marine ecosystem by e.g. metabolism, excretion, incorporation into fecal matter, microbial transformation and degradation, and bioturbation. Numerous laboratory studies have showed that aquatic organisms can accumulate PAH from the water column, from sediments, and from their diet, but that the bioavailability of PAH from these sources is not equivalent (Varanasi & Malins 1977, Neff 1979, McElroy & al. 1989, and references therein). Direct uptake of PAHs from the water column (dissolved PAHs) is the major pathway of bioaccumulation. Trophic transfer of hydrocarbons (biomagnification) is important in aquatic organisms, which have been shown to be capable of accumulating PAH via their diet (McElroy & al. 1989, and references therein). Thus, another important pathway is via food: in cases where uptake from food vs. sediments has been compared, dietary route appears to be more efficient (Corner & al. 1976, Varanasi & Malins 1977 and references therein, McElroy 1985). A good example of this are amphipods, which feed on arctic phytoplankton from the underside of the ice, and can thus be exposed to toxic components of oil trapped under the ice. Amphipods also consume massive amounts of dead plant and animal material, and in turn serve as an important food source for Arctic cod and other fishes, several species of birds, seals and some whales. Busdosh & Atlas (1977) suggested that oil spills in the Arctic region are likely to cause large-scale local mortality of important amphipod species, resulting in serious ecological changes in detritus decomposing processes and food-web relationships.

Bioaccumulation of oil (accumulation of oil in a single specimen) is positively correlated with physical/chemical properties of the PAHs, such as the molecular weight, and octanol/water partition coefficients (McElroy & al. 1989). By knowing the physical/chemical properties of spilled oil, the degree at which the organism would bioaccumulate PAHs can be predicted. The bioaccumulation factor has been studied mainly for fish and tends to increase with increasing molecular weight (McElroy & al. 1989).

In toxicity tests, LD₅₀ concentrations of the tested oil have yielded much information about the tolerance of a given organism to oil and/or dispersant disposition. Typically, oil with a high aromatic content (2- and 3-ringed PAHs, such as naphthalenes, fluorenes, phenantrenes and anthracenes) and fresh crude oil have a greater acute toxicity to marine organisms than oils with more aliphatic nature or weathered oil (e.g. Anderson & al. 1977, Lee & al. 1978). High angular configurations are more carcinogenic than linear or highly condensed ring arrangements (Neff 1979).

Petroleum and its products may have either a mechanical effect, as it is able to penetrate, has a tendency to cling to surfaces and form coverings on objects, or a chemical one, where toxic components (mainly aromatic hydrocarbons of low molecular weight) affect the organism (e.g. Nelson-Smith 1982, Wells and Percy 1985, Robertson 1998). Organisms can become affected by oil by filtration or ingestion, penetration of oil through cell membranes, or through becoming smothered by oil. The sensitivity of a given organism to oil is largely dependent on the severity of contamination, the organism's physiological state, and life cycle. Reproductive and juvenile stages are particularly sensitive to PAH

effects. Also, organisms on a higher systematic level generally have a better PAH metabolism, and consequently tolerance, than organisms at lower levels (e.g. Rice & al. 1977a).

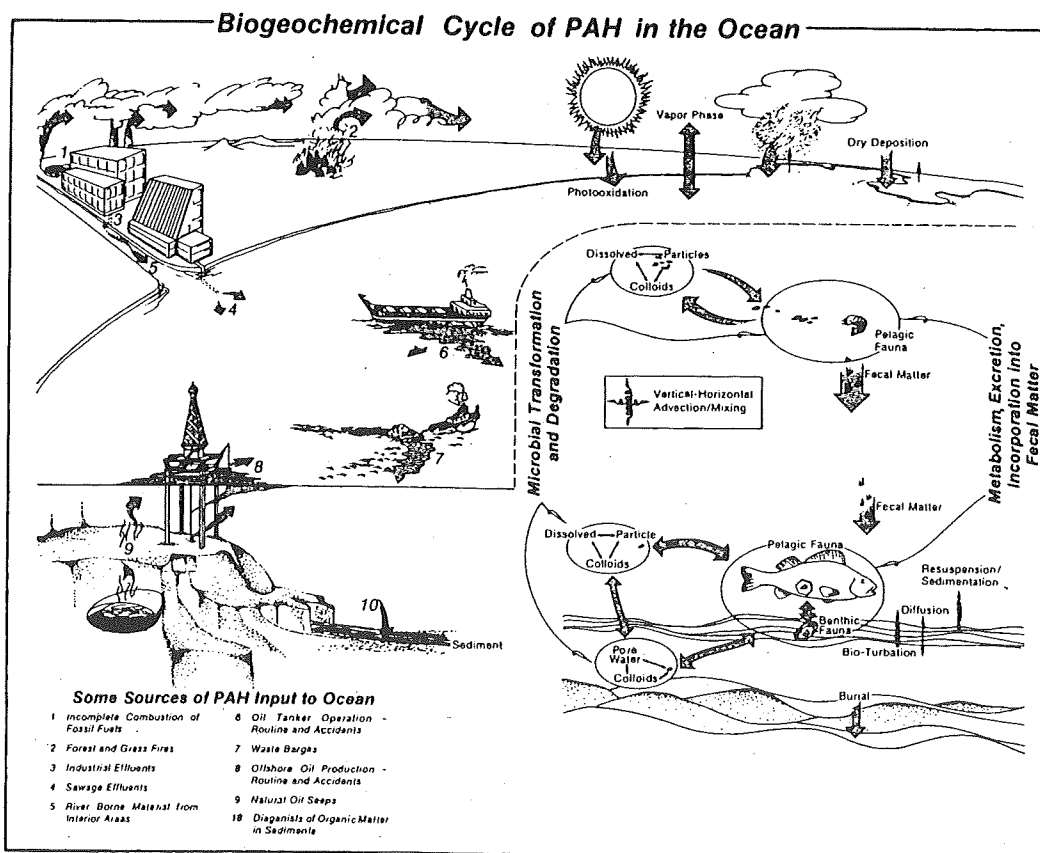


Fig. 1. Biochemical processes affecting PAHs in marine ecosystems. Source: McElroy & al. (1989).

In addition to acute toxicity - virtually causing death of an organism, which is tested using LD_{50} concentrations of a given compound - PAHs can also cause a multitude of sublethal effects. These may become visible almost instantly after exposure to oil, or during later life cycle stages of the organisms (e.g. during maturation or reproduction). Sublethal effects can be expressed in various ways, such as changes in feeding or other behaviour, growth, reproduction capacity, or in the organism's offspring (such as organ abnormalities in the developing embryo). On a cellular level, PAHs can either bind to lipophilic sites in the cell, or affect the DNA creating covalently bound products called adducts (Robertson 1998). Adducts are thought to be one of the initial steps in tumor development caused by carcinogenic PAHs. Metabolic activation is a prerequisite for carcinogenic effects of PAHs in mammals and fish (Robertson 1998) and is introduced later in this report.

In the following chapters, petroleum effects on marine organisms, with particular emphasis on oil spills and arctic species are presented with examples from literature. The report proceeds from plankton (open water) communities to littoral (shorelines), benthos (sea bottom) and, finally, fish, birds and marine mammals.

2.1 Plankton

Plankton organisms live in the open water, and comprise of bacteria, fungi and viruses, primary producers (photosynthesising microscopic algae), heterotrophic consumers (e.g. ciliates, rotifers, fish eggs, medusae) and mixotrophic organisms (from several systematic ranks) (e.g. Valiela 1984). Arctic zooplankton consumes phytoplankton and ice algae for food (Runge & al. 1990, Werner 1997).

Plankton in general is the foundation of the marine open water food web, and is the primary food source for several macroscopic open water organisms, such as fish and whales. Fisheries, for example is very sensitive to changes in plankton quantity and quality.

In many cases, the comparison of results from oil experiments is difficult, questionable or impossible (for discussion on this issue see e.g. Craddock 1977). Particularly older literature with oil experiments shows that tests were made in various ways, and common guidelines for testing oil effects on plankton were generally missing. This is true not only for plankton, but is common for littoral and benthic studies as well. OECD guidelines for e.g. testing of chemicals have later been prepared for some aquatic organisms, i.e. algae (growth inhibition test), zooplankton (*Daphnia magna* reproduction test) and several for fish (e.g. acute and prolonged toxicity tests, and juvenile growth test) (OECD 1984a, b, 1992, 1998, 2000). The plankton data available derive from three sources: field studies in oil-contaminated areas, studies with enclosed systems - such as mesocosms - with test organisms, and laboratory experiments.

Information about the effects of oil on plankton is much sparser than for littoral or benthic organisms. One reason for this is that oil is regarded to affect sea shores and bottoms more than the open water ecosystem (e.g. Robertson 1998). However, as plankton is the food source for a large variety of organisms even on the top level of the marine food chain, it is vital to know petroleum effects on open water communities. Plankton is generally thought to remain rather unaffected by oil, mainly for its capability to escape the contaminated water area. This is partially true for in particular larger zooplankton and in case of a small scale spill. Also, phytoplankton cells reproduce mainly asexually by cell division, therefore enabling a short recovery time. Local and short-term effects on plankton communities (decreased photosynthetic rates, oiled zooplankton) are likely during even a small oil spill, but have only a temporary dampening effect on plankton productivity. In case of a larger spill, the effects last notably longer and are spread to a larger area (e.g. Teal & Howarth 1984). An oil slick in the water column or associated with sea ice has an effect on the organisms therein, and thus the type and composition of algal material (phytoplankton vs. sea ice algae) sinking to the bottom may affect the food quality of benthic animals (Clough 2005).

2.1.1 Bacterioplankton and fungi

Hydrocarbon utilising bacteria (such as *Vibrio*, *Pseudomonas*, *Micrococcus*, *Nocardia* and *Acinetobacter* in the Arctic) are long known to be widely distributed in the world's oceans, also in cold marine ecosystems (e.g. Bunch & Harland 1976, Austin & al. 1977, Atlas 1978, Roubal & Atlas 1978, Atlas & al. 1982). Among fungi, *Penicillium* and *Verticillium* spp. from the northern Canada oil-producing areas were capable of growth on one or more crude oils tested by Davies & Westlake (1979), and oil-degrading strains of a few other genera were isolated. Several genera such as the truly marine *Corollospora*, *Dendryphiella*, *Lulworthia* and *Varicosporina* are known (Kirk & Gordon 1984), but their function as hydrocarbon utilisers in the Arctic is still an issue. Microbial degradation of oil by bacteria (bioremediation) and fungi is dependent on several physical, chemical, and biological factors which are extensively studied and discussed by e.g. Leahy & Colwell (1990). A dramatic rise in concentration of planktonic hydrocarbon utilising bacteria in response to acute input of petroleum hydrocarbons has been documented in some cases (Horowitz & Atlas 1978, Johansson 1980, Dahl & al. 1983), but not as a rule. The biomass increase of hydrocarbon degraders can also be extremely slow and take place within months or even years after oil introduction (Haines & Atlas 1982). As for the diversity of bacteria, it may either decrease or increase due to oil exposure (Atlas & al. 1982, Atlas 1983). In sediments, the biomass of oil degraders may be notably higher than in the water column, like in the case of the *Arrow* oil spill in the Chedabucto Bay, Nova Scotia in 1970 (Stewart & Marks 1978). Generally, hydrocarbon utilisers in sediments seem to increase relatively slowly after the actual spill (e.g. Eimhjellen & al. 1982, Bunch 1987). Hydrocarbons can also cause a temporary change in the behaviour (chemotaxis) of bacteria as evidenced by Mitchell & al. (1972) and Walsh & Mitchell (1973).

2.1.2 Phyto- and zooplankton

Good evidence of effects on plankton derives from field studies of the *Tsesis* spill in the Baltic Sea in 1977 (Johansson 1980). Phytoplankton species composition was not changed by the oil (concentration

50-60 µg/l after 2-5 days of the accident): microflagellates dominated the community before and after the spill. Phytoplankton biomass and productivity increased after the accident, which was probably largely due to depressed zooplankton grazing. Although planktonic bacteria biomass increased notably after the oil spill, the populations of their grazers (ciliates, rotifers) did not (Johansson & al. 1980). Larger zooplankton abundance changed shortly after the accident, probably due to narcosis effects and/or avoidance reactions. No notable change in phytoplankton communities were detected after the platform *Bravo* spill either (concentration of aromatic hydrocarbons up to 8 µg/l) in the North Sea in 1977 (Rey & al. 1977), while an obvious retardation of phytoplankton growth, and a considerable mortality of zooplankton was recorded for several weeks in the vicinity of the *Amoco Cadiz* accident on the NW coast of France in 1978 (Cabioch & al. 1981). Further away, phytoplankton growth was elevated, which was interpreted as a consequence of nutrient release from dead organisms. Zooplankton was contaminated with oil in proportion to their distance from the wreck. Also the groundings of *Arrow* in Nova Scotia in 1970, and *Argo Merchant* on Nantucket shoals in 1976 lead to copepod contamination (Conover 1971, Polak & al. 1978): oil droplets had incorporated into their guts. The *Potomac* spill off Western Greenland in 1977 caused external contamination of plankton (primarily the copepod *Calanus hyperboreus* and the amphipod *Themisto libellilula* (syn. *Parathemisto libellilula*) in the vicinity of the spill, but no oil was found in either copepod or amphipod guts (Maurer & Kane 1978), and thus no severe damage to the zooplankton communities was detected. Thus, field observations at numerous accidental spills show that negative biological effects can occur after a spill, but that the consequences appear rather mild and short-lived.

More detailed information of the effects of oil on plankton organisms originates from laboratory and other experimental studies. The older literature is reviewed extensively by O'Brien & Dixon (1976) and Johnson (1977). Microscopic algae from different systematic ranks may show different responses and tolerance to oil. Pulich & al. (1974) used six phytoplankton species to study experimentally the effects of different crude oils on the growth and photosynthetic rate of microscopic algae. All were inhibited either fully or partially, but significant differences between algal groups were discovered: *Thalassiosira pseudonana* (a diatom) showed the least tolerance to oil, while blue-green algae (*Agmenellum quadlupcanum*, *Nostoc* sp.), green algae (*Dunaliella tertiolecta*, *Chlorella autotrophica*) and the dinoflagellate *Glenodinium halli* were several times more tolerant to one or all types of oils tested. Hsiao (1978) found that exposure to various crude oils generally inhibited arctic phytoplankton growth, but also some signs of stimulated growth were documented after several days of exposure to 10 ppm of a range of crude oils. The degree of inhibition was temperature dependent: at +15 °C the oils were generally less toxic than at 0 or +10 °C. Temperature dependence of oil effects is important in particular in the case of an oil spill accident in the Arctic. Hsiao (1978) further speculated that a major oil spill could cause a change in phytoplankton species composition (from diatoms to microflagellates) and therefore an alteration in the zooplankton communities (species, biomass) that feed on microalgae. Effects of oil on plankton algae may vary even between clones of an algal species as was evidenced for the diatom *Skeletonema costatum* by Mahoney & Haskin (1980) (an important food source for the eastern oyster *Crassostrea virginica*). In the same experiment, other algae (chrysophytes *Monochrysis lutheri* and *Isochrysis galbana*, chlorophyte *Dunaliella euchlora*, and the eustigmatophyte *Nannochloris oculata*) showed generally better tolerance to oil than the diatom. Evidence of differences in oil effects on different phytoplankton groups is published also by Davenport (1982) and Dahl & co-workers (1983).

Studies on zooplankton responses to oil are reviewed e.g. by Wells & Percy (1985) and Robertson (1998). Most zooplankters appear to be very sensitive to in particular dispersed and dissolved oil. The acute lethal toxicity of dispersions and water soluble fraction (WSF), usually expressed as 4-day LD₅₀ values using initial measured concentrations, ranges between 0.05-9.4 mg/litre (Wells & Percy 1985). The major routes of contamination are direct uptake from the water, uptake from food (important for in particular copepods), or ingestion of oil particles that may be the size of the food item (Wells & Percy 1985). The capability of detoxifying hydrocarbons varies between different organisms. Oil particles taken up alone, or with food items, seem to pass chemically unchanged through the gut of for example copepods and pelagic barnacle larvae, and may become discharged in fecal pellets. This, in turn, may lead to biomagnification of oil in the arctic foodweb: the transfer of ingested oil to higher predators or coprophages (organisms that eat feces).

Low concentrations of hydrocarbons can cause sublethal effects in zooplankton, such as changes in behaviour, physiology, development, growth and reproduction (Wells & Percy 1985). Further studies are, however, required as there has been much variation in e.g. exposure conditions, life stages of test animals, and oil types used in earlier experiments. Field observations are made during oil spills and in chronically exposed areas like in the vicinity of oil platforms. Biological effects seem to be detectable but short-lived. Organisms at spills have suffered from direct mortality (copepods, fish eggs, plankton in general), external oil contamination (crustaceans, fish eggs), tissue contamination by aromatic compounds, abnormal development of fish embryos, altered feeding behaviour in copepods, and changes in metabolic rates of zooplankton. Mesocosm experiments by Vargo (1981) showed several negative effects of chronic low concentrations of fuel oil on temperate zooplankton, in particular changes in respiration and excretion rates.

Wide distribution of zooplankters and rapid change of water masses in the open waters promote the recovery of zooplankton communities after oil contamination, while in enclosed water bodies, such as estuaries and bays, the recovery may take notably longer (Wells and Percy 1985, and references therein).

Dahl & al. (1983) studied the effects of Ekofisk crude oil on a planktonic ecosystem using a simplified mesocosm set-up. While diatoms and copepods suffered from the addition of oil, the rapid stimulation in growth of planktonic bacteria (for which low molecular-weight fractions served as energy sources) was observed. Due to increased food availability (i.e. bacteria) their grazers (mainly heterotrophic choanoflagellates and tintinnid ciliates) increased as well. Very low concentrations of petroleum hydrocarbons (470 µg/l) were considered to be toxic to diatoms. Clear range of oil sensitivity of arctic freshwater zooplankton was detected in experimental studies by O'Brien (1978) and Atlas & al. (1978). As branchiopods (fairy shrimp *Branchionecta paludosa*) and amphipods seem particularly sensitive, cladoceran *Daphnia middendorffiana*, the calanoid crustacean *Heterocope septentrionalis* and isopods in general showed better tolerance to oil.

Copepods are good test organisms in oil experiments: they are easy and inexpensive to access and maintain, and show rapid responses to treatments. Petroleum is acutely toxic, but has also numerous sublethal effects (narcosis, paralysis, decreased feeding and defecation rates, disrupted phototaxis and altered swimming activity) on copepods. The effects of petroleum hydrocarbons on copepods have thus been extensively studied in laboratory set-ups, outdoor enclosure experiments, and at various spilled sites, of which only a few examples are given here (for a careful review of older literature see Wells & Percy 1985). Cross & Martin (1987) examined effects of untreated, solidified and dispersed oil on under-ice meiofauna during the Baffin Island Oil Spill (BIOS) project (for a description of the project, please see Sergy & Blackall 1987). Harpacticoid copepods and polychaete worms showed high sensitivity to oil, in particular the dispersed type, while cyclopoid and calanoid copepod nauplii (juvenile stages) were more tolerant to it. Untreated and solidified oil did not affect nematode, polychaete and copepod densities. The growth of adult harpacticoid copepods and their copepodite (juvenile) stages, and cyclopoid nauplii was, in fact, slightly stimulated by untreated and solidified oil (Cross & Martin 1987). Negative effects of oil on arctic copepods have been evidenced by several researchers. Melbye & al. (2001) studied the effect of low oil concentration on the copepod *Calanus finmarchicus*, which is an important species in the arctic pelagic food web. Oil with very low water-soluble component (and low content of aromatics) had very weak acute toxicity towards the test organism. Furthermore, another arctic copepod, *Calanus hyperboreus* is considered very resistant to crude oil (Percy & Mullin 1975, Foy 1979) when compared to e.g. amphipods and isopods. Short-term exposures of high concentrations of aromatic hydrocarbons did have profound effects of the copepod *Eurytemora affinis* as documented by Berdugo & al. (1977): significant reduction in subsequent length of life, total number of eggs produced, mean brood size, and the rate of egg production was evident.

The sensitivity of planktonic cod eggs (*Gadus morrhua*), and sea urchin eggs and embryos (*Strongylocentrotus droebachiensis*) to naphthalenes and Ekofisk crude oil was documented by Falk-Petersen & al. (1982, 1983). The locomotory motion and thus swimming of the arctic medusa *Halitholus cirratus* is negatively affected by crude oil disposition (Percy & Mullin 1975).

2.1.3 Sea ice microbial communities

Sea ice associated (epontic) communities are characteristic to polar regions. They consist of a wide variety of microscopic organisms (size ranges from picoplankton (0.2-2 μm to mesoplankton (20-200 μm) (e.g. Ikävalko 1997, 2004, Werner 1997, Thomas 2004). Primary producers, grazers, predators and, finally, degraders are present within sea ice. Sea ice communities consist largely of the same groups of organisms (but often different species) as the plankton, and live trapped in brine channels within ice. Thus they are not capable of escaping oil contamination, and the effects of oil on ice biota can be much stronger than on free-floating plankton in the open water. Ikävalko & co-workers (2005) made experiments on the effects of Statfjord crude oil on sea ice biota by exposing ice algae to oil for 63 days. Oil was practically lethal for dinoflagellates and chlorophytes, while some diatoms survived by forming thick-walled resting stages which are relatively resistant to environmental changes. Vegetative diatom cells survived better in the interior and close to the underside of the ice than in the ice surface where oil was distributed. No negative effects of oil on ice diatom growth or photosynthesis were detected in in-situ experiments by Cross (1987). Conflicting results from experiments on sea ice algae may be explained by several factors, such as differences in study methods (oil type and concentration, duration of predisposition to oil, laboratory vs. field experiments), microalgal species studied, and possibly the physiological state of algae (not measured).

2.2 Littoral and benthic communities

The marine benthic habitat can be divided into two areas, the intertidal (here also referred to as the littoral zone) and the sea floor (benthos). Characteristic for the intertidal zone are strong variations in water level, notable stress caused by wave action and, in the Arctic, ice scouring and summertime reduction in surface salinity due to melting of ice and snow. Thus, the marine organisms in the intertidal must be tolerant to e.g. exposition to air and direct sun (desiccation), and fresh/brackish water. The High Arctic intertidal is regarded inhospitable to colonisation (Menzies & al. 1973). In the Eastern Canadian Arctic it is typically colonised by the rough periwinkle *Littorina saxatilis* and the barnacle *Balanus balanoides* (Ellis 1955, Ellis & Wilce 1961). Below the intertidal zone is the benthos, which light never reaches. The upper part is the so called barren zone, that typically extends to a depth 3-5 metres, sometimes even 15 metres), which, due to ice scavenging and low surface water salinities, is devoid of infauna (animals partly or completely buried into the substrate) and sessile epifauna (animals attached onto substrates) (Ellis 1960). In the subarctic, ice scouring effect is infrequent and the intertidal communities become notably more diverse. High intertidal zone is inhabited by a variety of invertebrates and smaller macroalgae, such as green and coralline (red) algae. Where ample light for photosynthesis, macroalgae such as kelps (giant brown algae) flourish in deeper intertidal, and offer habitats for diverse invertebrate communities – amphipods, barnacles, mussels, echinoderms, nematodes et cetera (George 1977, Wells & Percy 1985). Much of the polar basin lies beneath the barren zone and thus the permanent ice cover. These zones are called the shelf zone, slope zone and the abyssal (deep sea) (Fig. 2. (from Wells and Percy 1985)). The vulnerability of littoral and benthic communities to spilled oil varies due to the environmental factors introduced above, and thus the vertical and horizontal distribution of biological communities (Fig. 2.). Oligomixity (high population densities of a single species) is characteristic to the Arctic Ocean in general and particularly true for the benthic communities (George 1977). The most abundant and diverse groups in the arctic benthos include bivalve molluscs, polychaete worms, amphipods and isopods (Marshall 1982). Benthic populations in the Arctic tend to show less fluctuation in abundance than those of warmer seas (Ellis 1960). This is partially due to slower growth and longer life span, but also their altered reproduction strategies: many benthic species have shortened or eliminated the vulnerable pelagic larval stage, and larvae are produced in brood chambers.

Shoreline and shallow subtidal communities are most affected during a coastal oil spill, and oil impacts on sedimentary shorelines have been reported from several accidental and experimental spills at lower latitudes. Also laboratory experiments have been made on various littoral and benthic organisms, and some of them are introduced as examples in this report. In nature, the impacts of an oil spill are dependent largely on the amount of dispersed oil and the type of substratum on the shoreline (e.g. Robertson 1998). On rocky coasts, wave action may remove the oil rather quickly and transport it to the

open sea or the benthos, while in sheltered estuaries and on muddy shores the oil effects in the littoral zone are more pronounced. Oil can reach the bottom by various mechanisms: 1) direct mixing of oil with sediments by wave action in shallow water, and consequently transport to deeper water by density currents, 2) sorption onto particulate matter suspended in water column and subsequent sinking, 3) uptake by zooplankton, release in and subsequent sinking of pellets, and 4) take-up of non-volatile aromatic hydrocarbons by phytoplankton, and further sedimentation (Conover 1971, Mackie & al. 1978, Sanders & al. 1980, Teal & Howarth 1984).

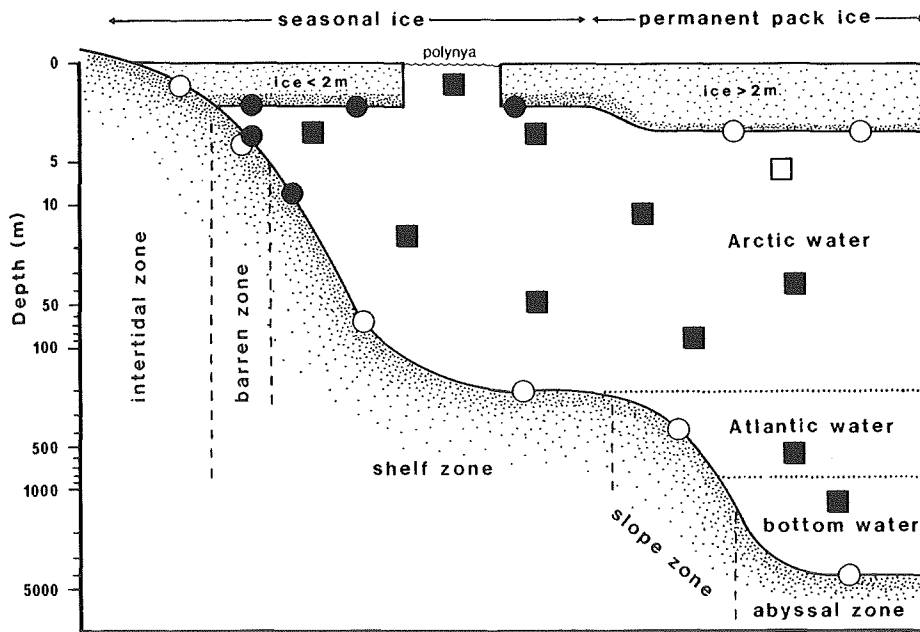


Fig. 2. Habitats of Arctic marine invertebrates and their vulnerability to spilled oils (filled circles = possible impacts on benthos, filled squares = possible impacts on zooplankton, open circles = no anticipated impacts on benthos, open squares = no anticipated impacts on zooplankton. Source: Wells & Percy (1985)

2.2.1 Macroalgae

Attached algae (seaweeds) and their responses to oil have not been given much attention in the Arctic, although vegetations of macroalgae (in particular green and brown algae) live attached to rocky shores. Production estimates for arctic kelp *Laminaria solidungula* communities, for example, vary between 7-20 gC/m²/year (Dunton & al. 1982, Chapman & Lindley 1980, 1981). The rate of photosynthesis by marine phytoplankton and macroalgae may be stimulated or repressed depending upon the concentration of hydrocarbons and method of exposure (Johnson 1977, and references therein). Oil can form coatings on algae, and thus decrease CO₂ uptake and water loss. This has been documented for *Laminaria digitata* and *Fucus vesiculosus* (Phaeophyceae, brown algae), *Porphyra umbilicalis* (Rhodophyceae, red alga) and *Enteromorpha* sp. (Chlorophyceae, green alga) (Schramm 1972). Intertidal macroalgae are considered relatively resistant to oil due to mucus production, as evidenced e.g. for the giant kelp (*Macrocystis* sp.). Mucus prevents contact between the petroleum and kelp tissue (Mitchell & al. 1970). During the BIOS project macroalgae were exposed to untreated and dispersed oil (Cross & al. 1987a). Biomass, number of species and reproductive condition of the most dominant algae did not seem to be adversely affected by neither type of oil. The lack of major effects was explained partially by the mortality of herbivores and thus decreased grazing pressure on macroalgae, and the vegetative mode of reproduction, which is not as sensitive to environmental disturbances as sexual reproduction (Cross & al. 1987a). A heavy oil pollution can cause retarded growth and even death of seaweeds as was evidenced after the *Torrey Canyon* accident (Ranwell 1968) and a spill of 1000 tons of bunker oil on the Arctic coast of Norway (Wikander 1982). Even in the latter case, in

lightly polluted areas the new sprouts of contaminated algae looked healthy the following spring (Wikander 1982). In mesocosm experiments, oil caused growth reduction in macroalgae *Ascophyllum nodosum* and *Laminaria digitata* (Gray 1987). Hydrocarbons can also affect reproduction of some fucoid macroalgae (*Fucus serratus*, *F. vesiculosus*, *F. edentatus*) by hindering gamete release, fertilisation, or by altering the behaviour of motile gametes as PAHs produce similar attraction to *Fucus* sperms as do the egg cells (Cook & Elvidge 1951, Steele 1977).

2.2.2 Invertebrates

Sublethal biological effects of PAHs on benthic invertebrates in general are numerous, and include changes in behaviour, physiology, growth and reproduction of coelenterates (corals, sea anemones, hydrozoa, medusae), annelid worms, adult and juvenile arthropods (marine crustaceans such as mysids, and amphipods, isopods, shrimps, prawns, crabs), molluscs (gastropods such as limpets, and bivalves such as oysters, mussels and clams), and echinoderms (e.g. starfish, asteroids, sea urchins) (Straughan 1976, Johnson 1977, Wells & Percy 1985). Generally, long-term effects of petroleum include the development of tumours, neoplasms, diseases caused by bacteria and viruses, and mycoses in invertebrates (Hodgins & al. 1977, Wells & Percy 1985 and references in both works). Benthic invertebrates may also suffer from ostial closure, loss of responsiveness to mechanical stimuli and narcosis (Mageau & al. 1987) which are discussed later in this chapter. Chronic oil predisposition and long-term responses of benthic fauna to hydrocarbons has been monitored in the vicinity of oil platforms by e.g. Menzie 1982, Addy (1987), Kingston (1987), Moore & co-workers (1987a), Gray & al. (1990) and Olsgard & Gray (1995), but are not discussed here further. To summarize, the responses of benthic communities to offshore oil exploration are measurable, complex, often irreversible, and results are presented mainly on the community level (species dominance, competition, and succession).

2.2.2.1 Bivalves and gastropods

The two classes of marine molluscs which have been studied extensively with respect to petroleum pollution, Gastropoda and Bivalvia, consist of animals with distinctly contrasting adult habits. Most gastropods (e.g. limpets, periwinkles and snails) are free-living epibenthic animals, while most bivalves (mussels, oysters and cockles) are either sessile or sedentary burrowing forms (Fish & Fish 1989).

The literature lists a multitude of oil effects on bivalves and gastropods. They include e.g. mortality, abnormal larval development, and various molecular, biochemical, cellular and physiological responses such as enzymatic hydrocarbon detoxification or elimination, changes in enzymatic activity in general, atrophy of the epithelium, changes in oxygen consumption, feeding, excretion, growth and, finally, ecophysiological consequences like the general reduction in population fitness leading to changes on an ecosystem level, i.e. affecting biological interactions (e.g. Haranghy 1956, Bayne & al. 1982, Lewis 1982, Southward 1982, Stickle & al. 1985, Gray 1987, Moore & al. 1987b, Neff & al. 1987, Giessing & al. 2002). During the BIOS project Neff & co-workers (1987) reported several histopathological changes in bivalves *Mya arenaria* and *Macoma calcaria* caused by untreated and/or dispersed oil; digestive track necrosis, increase in the number of mucus cells in the digestive track epithelium, granulocytomas, invasive neoplasia (probably cancer), vacuolisation of the digestive tubule epithelium, increased parasitism, and hemocytic infiltration. The stress level (indicated by concentrations of glucose, glycogen, trehalose, total lipid and free amino acids) was lower in animals exposed to dispersed oil than for animals receiving oil alone (Neff & al. 1987).

Documented sublethal effects of oil in gastropods only range from simple narcotisation to loss of chemosensitivity, and have been reviewed earlier in depth by Johnson (1977, and references therein). Narcosis caused symptom is for example the detachment from substrate. The loss of chemosensitivity may change motility and alter the direction of movement, and thus affect food capture. Reduced filtration rates, possibly due to direct inhibition of the cilia by hydrocarbons (Johnson 1977), affect several functions in bivalves and gastropods, including reduced feeding. This has been documented for e.g. the oyster, *Crassostrea virginica* (Stegeman & Teal 1973), blue mussel *Mytilus edulis* (Phelps & al. 1981, Widdows & al. 1982), the hard clam *Mercenaria mercenaria* (Keck & al. 1978), and an arctic bivalve, *Yoldiella arctica* (Percy & Mullin 1975).

High concentrations of oil can cause shell closure and narcotisation of ciliary surfaces in bivalves, and consequently affect respiration and feeding rates negatively (Johnson 1977, Bayne & al. 1982 and references therein, Mageau & al. 1987). At low concentrations of oil, rates of oxygen consumption are first increased in bivalves, such as the soft-shelled clam (*Mya arenaria*), blue mussel (*Mytilus edulis*), Baltic telling (*Macoma balthica*) and the gastropod *Littorina littorea* (edible periwinkle) (Bayne & al. 1982 and references therein). Metabolic rates increase due to hydrocarbon association in the body tissues, and mucus secretion and excretion increase. As a result, energy expenditure increases while less energy (reduced carbon flux) is available for growth and reproduction (Stainken 1978, Widdows & al. 1982, Bayne & al. 1982, and references therein). Other effects are manifested in the structure and development of the eggs and embryos, like anomalies in the gonads of the Baltic telling *Macoma balthica* (Stekoll & al. 1980). The BIOS project showed different uptake dynamics among species (filter-feeding bivalves *Mya truncata* and *Serripes groenlandicus*, and deposit-feeding green sea urchin *Strongylocentrotus droebachiensis*), but in all cases the effect was immediate, short-lived and resulted in temporary accumulation of hydrocarbons (Mageau & al. 1987). As a consequence of oil exposure, the bivalves suffered from ostial closure, retraction of the siphon (decreased filtration rate and growth), the loss of responsiveness to mechanical stimuli, narcosis, increased enzymatic activities and accumulation of hydrocarbons in tissues.

2.2.2.2 Molluscs

Low levels of petroleum hydrocarbons can also affect the behaviour of molluscs. The production of the byssus by juvenile and adult mussels may be reduced, leading to weakened attachment on substrate. In mesocosm experiments, oil affected negatively the recruitment of the edible periwinkle *Littorina littorea*, a common inhabitant on rocky coasts in subarctic, and consequently populations declined over time (Gray 1987). The burrowing behaviour of infaunal bivalves such as *Macoma balthica* may be impaired, and clams can be stimulated to leave the contaminated area (Lindén 1977, Taylor & Karinen 1977, Stekoll & al. 1980). The crawling rates of gastropods *L. littorea* and *Theodoxus fluviatilis* may change (Hargrave & Newcombe 1973, Lindén 1977). During the BIOS project Cross & Thomson (1987) noted that the use of untreated and dispersed oil had very distinct effects on macrobenthic infauna (bivalves *Mya truncata*, *Macoma calcarea*, *Serripes groenlandicus*, *Astarte borealis*). While untreated oil had practically no effect, dispersed oil caused marked acute effects on infauna, including emergence from the substrate, narcosis and progressive decrease in condition. Neither type of oil release caused any large scale mortality of benthic infauna, neither were significant changes in community structure detected (Cross & Thomson 1987).

2.2.2.3 Bristle worms

Annelids, such as bristle worms (Polychaeta), are common on the shore under stones and rocks, and buried in mud and sand. While the adult worms seem rather resistant to oil pollution (Johnson 1977, and references therein), hydrocarbons can cause narcosis, immobilisation and death of their larvae (Chia 1973, Carr & Reish 1977). Once in sediments, hydrocarbons are taken up by benthic organisms with greater uptake of the heavier relative to the lighter molecular weight aromatic compounds. Uptake from water may occur more readily than from sediments in carnivores and filter feeders, while deposit feeders such as polychaetes with more intimate contact with porewaters could be expected to show a more rapid uptake from substrate (Anderson & al. 1978). Exposure of the polychaete *Nereis succinea* to oil in a laboratory experiment resulted in a decrease in growth rate and an increase in mixed function oxygenase (MFO) activity relative to unexposed individuals (Lee & al. 1981). MFO is responsible for the metabolic modification of foreign organic compounds in vertebrates, such as fish and cetaceans and has been detected also in marine crabs and polychaetes (e.g. Johnson 1977, and references therein, Lee & al. 1981, Rice 1985, Lockhart & Metner 1991, George & al. 1995). Studies on annelids and hydrocarbons are few. Some monitoring on the succession of polychaetes has been made during oil spills. Several years after the *Amoco Cadiz* and *Arrow* spills, the lugworm *Arenicola marina* was very common, in case of *Arrow* even more abundant than prior to the accident (Gordon & al. 1978, Gundlach & al. 1981). Hydrozoa, corals, and anemones are typical benthic organisms in lower latitudes,

where hydrocarbons are known to affect the behaviour, growth and reproduction of these sessile animals (Johnson 1977, and references therein).

2.2.2.4 Echinoderms

Echinoderms include intertidal and benthic, slowly moving invertebrates such as sea urchins, brittle-stars and starfish. Fuel oil and gasoline interfere with the development of boreoarctic sea urchins (Falk-Petersen 1979). In sea urchins oil may stimulate the oxygen consumption, weaken their adherence to the substrate, cause retarded fertilisation, and interfere with the development of the embryo (Johnson 1977, and references therein). A range of behavioural changes in the green sea urchin *Strongylocentrotus droebachiensis* and the starfish *Leptasterias polaris* was recorded during the BIOS project caused by a short-term exposure to dispersed oil by Cross & co-workers (1987b) and Mageau & co-workers (1987); unnatural postures (animals upside down), narcosis (i.e. loss of responsiveness to mechanical stimuli), and for sea urchins impairment of the tube foot (resulting in substrate detachment), changes in spine attitude, and frequent shedding of gametes was evident. Furthermore, in starfish the chemoreception system may become partially inhibited, thus affecting food particle sensing and capture (predation) (Johnson 1977, and references therein).

2.2.2.5 Littoral crustaceans

The effects of oil on littoral crustaceans are extensively studied, and earlier literature is carefully reviewed by e.g. Johnson (1977) and Wells & Percy (1985). Evidence shows that these organisms may possess a variety of responses to hydrocarbons. Physiological responses include changes in e.g. respiration rate (increases and decreases), hormone production (thus reproduction), molting and hatching, development of larvae, chemoreception-mediated behaviour (affecting reproduction and feeding) and disruption of osmoregulation. Behavioural changes are mainly due to narcosis, which in turn affect locomotor activity (and thus feeding and escape reactions), burrowing behaviour, and reception of chemical signals (food particle capture), even shell evacuation. Also physical nuisance is caused: when oil is adhered on the locomotory parts of the crustacean, swimming and feeding is affected. Also, oily coatings on hard substrates hinder the settlement of e.g. pelagic motile barnacle larvae (Straughan 1971). Ingestion of oil droplets tends to decrease feeding rates (Blackman 1972). The arctic marine amphipod *Gammarus oceanicus*, for example, has showed several negative physiological responses to crude oil (Aunaas & al. 1990). Water soluble fractions of crude oil increased respiration, sodium in haemolymph, and thus water content of the organism. Water emulsions reduced respiratory rates, causing oxygen deficiency due to oil droplet adherence to gill membranes. The use of dispersants reduced the mortality of the amphipods. Low concentrations of crude oil and oil fractions significantly depressed respiration rates of arctic amphipods *Boeckosimus affinis* and *Anonyx nugax*, while with high concentrations the depression was reversed (Percy 1977, Busdosh 1978). Baden & Hagerman (1981) and Baden (1982a, b) exposed the shrimp *Palaemon adspersus* to water soluble fraction of North Sea crude oil. As a result, the ventilatory behaviour was disturbed, osmoregulation impaired, and a significant increase in oxygen consumption was detected. A slight stimulation of metabolism at low hydrocarbon concentration followed by a decrease with increasing concentrations was detected for the littoral mysid *Mysis littoralis* (Wells & Percy 1985). In a behavioural study, arctic amphipods *G. oceanicus* and *Onisimus affinis* did not show avoidance of contaminated area in the presence of even high concentrations of crude oil (Percy & Mullin 1975, Percy 1976).

Petroleum hydrocarbons interfere also with reproductive processes (e.g. reduced precopulation frequency and number of larvae, premature shedding of eggs) of crustaceans such as *Gammarus oceanicus* and *Boeckosimus affinis* (Lindén 1976, Busdosh 1978). Differences in response rates to various oil-water mixtures were documented also by Riebell & Percy (1989), by exposing the arctic littoral mysid, *Mysis oculata*, to oil-in-water dispersions and water-soluble fractions of crude oil. The latter fraction was significantly more toxic, and the species was regarded as exceptionally sensitive to crude oil. Opposed to sessile and slowly moving animals in the littoral and benthos motile invertebrates, like littoral amphipods, are generally capable of avoiding oil slick by escaping it (Bonsdorff & Nelson 1981, Gulliksen & Taasen 1982).

2.2.2.6 Case studies

Massive kills and long-term effects can occur when oil reaches the littoral and the benthos in sufficient quantity, as has been evidenced for several oil spills in the past. Within 12 hrs after the *Florida* spill in West Falmouth, Massachusetts in 1969, the macrobenthos was nearly eradicated at most heavily oiled sites (Sanders & al. 1980). Opportunistic species typically play a vital role in the initial recolonisation of an eradicated area (Teal & Howarth 1984). After the *Florida* spill the annelid worm *Capitella* sp. and the nematode *Mediomastis* sp. increased greatly in abundance, monopolising the otherwise defaunated sediments for months after the accident. The *Arrow* spill in Nova Scotia in 1970 caused the decline of the bladder wrack, *Fucus vesiculosus*, for five years, while the spiral wrack, *Fucus spiralis*, disappeared and had not reappeared even six years after the spill (Thomas 1978). Rocky shore animals such as barnacles and periwinkles, however, did not change in abundance or distribution except in areas where their habitat changed. While the *Tsesis* oil spill in the Baltic Sea in 1977 had virtually no effect on the bladder wrack *Fucus vesiculosus* (dormant at the time of the accident) it caused a dramatic acute reduction in biomasses of the sediment dwelling amphipod *Pontoporeia affinis* and the polychaete *Harmothoe sarsi* (Notini 1980, Elmgren & al. 1983). Although heavily contaminated with oil, the Baltic teller, *Macoma balthica*, and nematodes were more tolerant and showed only little mortality. Recovery in the littoral zone began within two months, but the speed depended on the degree of exposure to oil and the species involved. One year later the animals had returned to their pre-spill condition, except at the most heavily contaminated stations. Abundance of amphipods, *H. sarsi* and harpacticoids began to increase, and hydrocarbon concentration in *M. baltica* decrease during the second summer after the spill. Three years after the accident *Pontoporeia* and *M. baltica* biomasses had remained depressed, while *H. sarsi* showed normal abundance. The recovery of species with long life span, such as *M. baltica* in this example, require considerably more time than short-lived species (Elmgren & al. 1983). Biological consequences of also other oil tanker wrecks in the Baltic have been monitored. The accidents of *Eira* in 1984, *Antonio Gramsky* in 1979 and 1987, and *Baltic Carrier* in 2001 caused notable effects in particular in the littoral and benthos (The Baltic oil spill 1979, Hirvi 1989, Pécseli & al. 2004). Increased hydrocarbon concentrations in the tissues of e.g. *Macoma balthica*, *Mytilus edulis*, *Lymnea palustris* and *Gammarus* spp. were recorded, and in some cases decrease in the population sizes of these organisms. Furthermore, oiled specimens of zooplankton and dead birds (e.g. black guillemots *Cephus gylle*, and eiders *Somateria mollissima*) were collected. Some fish suffered from morphological abnormalities after exposure to oil. Also, the growth of fish was dampened and skin diseases were documented (The 1979 Baltic Oil Spill, Hirvi 1989). In the case of *Baltic Carrier* spill off the Danish coast, hydrocarbon concentration in flounder remained elevated long after the accident (Pécseli & al. 2004).

Oil from the *Amoco Cadiz* on the northwest coast of France spread over a large area, with highest concentrations found in muddy sediments (Cabioch & al. 1981). Amphipods were virtually eliminated, and in the intertidal massive mortality of e.g. heart urchins and razor clams were observed. The *Amoco Cadiz* spill caused permanent changes in the shallow water eelgrass (*Zostera marina*) community, i.e. the total disappearance of filter feeders, and the very diverse amphipod community has been replaced by the dominance of only two species, one of them new to the area (den Hartog & Jacobs 1980). The spill had little effect on polychaete annelids (Chasse 1978). In the intertidal, the knotted wrack, *Ascophyllum nodosum* (Phaeophyceae, brown alga) was replaced by the much more tolerant bladder wrack, *Fucus vesiculosus* at sites where it grew in the vicinity (Gundlach & al. 1981). Populations of bivalves, periwinkles, and limpets in the intertidal, heart urchins in the benthos, copepods in the pelagial, and sea birds were most severely affected (Conan 1982). The *Exxon Valdez* oil spill in the Prince William Sound in 1989 had relatively mild effects on the littoral communities (Stoker & al. 1992). The survivors were seaweeds (fucoïds), barnacles, mussels and periwinkles, which after two years of the accident had re-established themselves and were in a state of pre-spill condition. A long-time survey of oil effects on sublittoral fine-sand macrobenthic community was made after the *Aegean Sea* oil spill off the NW coast of Spain in 1992-1996 (Gómez Gesteira & Dauvin, 2005). A short period of high mortality of in particular amphipods was followed by a period of low species diversity and low abundance. Recovery began 3 years after the spill.

2.3 Vertebrates: fish, birds, otters, seals, whales and the polar bear

2.3.1 Fish

Fish are generally more sensitive to hydrocarbons than invertebrates (Rice & al. 1977a, 1979). Arctic cod (*Boreogadus saida*), for example, is among the most sensitive fish species to oil that were studied by Rice & co-workers (1979). Most pelagic fish also show response relatively quickly to toxicants, while sedate bottom species react slower. Stress, such as fluctuations in water salinity, temperature, food abundance, disease and parasites depress the fitness of fish and thus reduce its ability to tolerate pollution (e.g. Moles 1980, for a review see Rice 1985). Water temperature affects the toxicity of hydrocarbons; in cold water aromatic hydrocarbons persist longer (decreased biodegradation and evaporation), and extreme temperatures (both low and high) may affect the ability of fish to metabolise or excrete aromatic hydrocarbons and their metabolites (Rice 1985). Fish also possess the ability to "learn" to tolerate hydrocarbons; previous sublethal exposure to hydrocarbons induces higher levels of hydrocarbon metabolising enzymes in fish (Egaas & Varanasi 1982).

Fish take up oil through ingestion of contaminated food and directly from water (for a review, see Rice 1985). The rate and quantity of hydrocarbon uptake depends on exposure concentration, the molecular weight of the compounds tested, and the amount of lipid in the fish (which, again, is related to fish species, age, season and reproductive stage). Once hydrocarbons are accumulated in fish, many compounds will be metabolised or excreted. Enzymatic metabolising takes place in liver by the mixed function oxygenase systems MFO and the enzyme CYP1A (Rice 1985, Lockhart & Metner 1991, George & al. 1995). The hepatic enzyme CYP1A can be used as a biomarker for petroleum hydrocarbon exposure in the polar cod (*Boreogadus saida*) (George & al. 1995). Most metabolites are probably less toxic than the parent compound and will be excreted. The effects of oil can be targeted to one or several organs in the fish: liver, gut, pancreas, vertebrae, eye lens, stomach, brain and olfactory (odour sensing) organs (studies are reviewed e.g. by Rice 1985) – all sensed as odd fish flavour or smell by us humans. Tainting of commercial fish by oil has been studied on field and experimentally, and in many cases fish show relatively fast purification after light exposure, i.e. the flavour and odour caused by oil disappear (e.g. Ackman & Heras 1992, Lochart & Danell 1992).

Fish exposed to sublethal concentrations of petroleum in the environment show several behavioural, physiological, biochemical and various long-term effects, which are reviewed in depth by Patten (1977), Rice (1985), and in the extensive text book edited by Varanasi (1989). Fish can detect hydrocarbons at different threshold levels (Patten 1977, and references therein) as they have excellent olfaction, which most fish use for detecting hydrocarbons. However, short-term exposures can damage the olfactory epithelia and render these tissues useless (e.g. Solangi & Overstreet 1982). The detection and avoidance behaviour varies among species and life stages, and fish larvae for example are not always capable of avoiding the contaminated area. There are conflicting records of the avoidance behaviour of fish during oil spills; sometimes fish seem to actively avoid the contaminated area, while in other cases they seemed rather affected to it (discussed by e.g. Rice 1985). Other physiological responses include alteration in metabolism and activity, such as decreased rate of heartbeat, cough responses or convulsive respiratory reactions of fish (possibly due to the aromatic compounds in the petroleum), altered respiration, changes in blood parameters and ion concentrations, and decreased energy reserves (Rice & al. 1977b, reviewed in depth by Rice 1985). Oxygen consumption may either increase or decrease, depending on the type of oil and fish species (Patten 1977, and references therein). Narcosis and consequently cessation of movement and feeding, and changes in activity patterns (swimming movements, gulping at the water surface, erratic motion, hyperactivity) is reported for several fish species and their larvae, and are reviewed by Patten (1977) and Rice (1985).

Effects on fish reproduction is little studied, but evidence shows that oil affects the survival (mortality) of in particular fish eggs and larvae, hatching, and the development of the embryo (Patten 1977, and references therein, Whipple & al. 1981, Teal & Howarth 1984). Eggs and larvae are easily affected by temperature, salinity and pollutants because they have fewer structures and organs capable of detoxifying oil, are intimate with the environment, their mobility is restricted, and many develop at or near the water surface where oil spill can be expected (Rosenthal & Alderdice 1976, Rice 1985). The earlier the juvenile is exposed to oil the more severely it is damaged. When the embryo approaches

hatching it is more sensitive to oil than after hatching. The sensitivity again increases until its yolk is absorbed and it begins to feed on its own (Rice 1985, and references therein). There is also a negative correlation between hydrocarbon concentration in water and growth, food uptake and thus weight, percent fat and caloric content of adult and juvenile fish (Korn & al. 1976, Moles & Rice 1983, and the review by Rice 1985).

2.3.2 Birds

The published literature concerning bird mortality due to oil is large, and reviewed in depth by e.g. Bourne (1968, 1976), Vermeer & Vermeer (1975), Holmes & Cronshaw 1977, Leighton & al. (1985), Robertson (1998) and Mosbech (2002b). In many cases the mortality has been substantial, but it is not always clear how estimates were made, as discussed by Leighton & co-workers (1985). *Torrey Canyon* accident in 1967 caused death of at least 20-30 000 birds, but the actual number may exceed even the estimated one (Holmes & Cronshaw 1977). Even very small oil spills can cause similar mass mortality of sea birds (Barret 1979, Robertson 1998). Divers are at high risk, because they spend much of their time sitting on water. King & Sanger (1979) ranked sea birds in terms of vulnerability to oil pollution, and identified Atlantic puffin (*Fratercula arctica*), common murre (*Uria aalge*), thick-billed murre (*U. lomvia*), razorbill (*Alca torda*) and northern gannet (*Sula bassanus*) as particularly vulnerable. Sea birds are at considerable risk because of their social behaviour (Robertson 1998). Large aggregations of birds occur in connection with breeding, molting, overwintering, and preparation for migration. Oil kills birds in many ways, but the main way is by breaking down the bird's waterproofing and thermoregulation. An oiled bird will response by preening itself, and consequently inhale and swallow toxic compounds that damage its liver, lungs, kidneys, intestines and other internal organs. Such poisoning is as lethal as the loss of waterproofing, so hypothermia is the actual cause of death (Holmes and Cronshaw 1977, Leighton & al. 1985).

The embryotoxic effects of oil have been studied experimentally to some extent (e.g. Couillard & Leighton 1989, for review see Leighton & al. 1985). Oil from the feathers may pass through the pores in eggshells and either cause death of embryos, abnormalities or affect hatching success. Ingested petroleum may cause ovarian dysfunction (affecting fecundity) and thus delay the onset and rate of lay (Holmes & Cronshaw 1977, Harvey & al. 1982). Depression of growth rate in young birds ingesting oil is a commonly reported phenomenon for a variety of species (reviewed by Leighton & al. 1985). Marine birds must constantly excrete sodium chloride through nasal glands, as they receive salt in excess through their diet. Such osmoregulation can be affected by oil (e.g. Peakall & al. 1983), lead to ion imbalance in body fluids and thus affect e.g. the function of muscles and the nervous system. Anaemia has been reported by e.g. Leighton and collaborators (1983). Exposed birds are also more likely to suffer from other pathological effects such as irritation of the gastrointestinal mucosa, lipid pneumonia, fatty degeneration of liver, atrophy of pancreas, toxic nephrosis, enteritis, aspergillosis, and infective arthritis (for review see Holmes & Cronshaw 1977). Other physiological changes in birds caused by oil are e.g. increased basal metabolic rate, changes in hormone production, body temperature and water flux (Harvey & al. 1982, Hughes & al. 1990).

2.3.3 Mammals

The Arctic seas are a habitat for a large proportion of marine mammals in the world, in particular whales, seals, sea otters and polar bears (Engelhardt 1985, GESAMP 1993). All marine mammals need to remain in contact with the air-water interface as they are dependent on air breathing, and the polar bear feeds on fish and seals it catches from the surface water. Thus, they may come in contact with a surface oil slick, in particular in sea ice covered areas where the open surface (such as breathing holes) is limited (Engelhardt 1985, Robertson 1998). Oil causes problems to mammals through coating by oil and inhalation of volatile hydrocarbons. Inhalation can be life threatening in the case of prolonged exposure (Geraci & St. Aubin 1980). Certain marine mammal species may additionally be vulnerable through the food vector, e.g. bivalves and crustaceans that have a potential to bioaccumulate hydrocarbons (Engelhardt 1985).

The information of effects of oil on whales is quite limited, but generally whales are anticipated to remain unharmed by contact with oil (Engelhardt 1985, and references therein, Robertson 1998). There

are no records of oil fouling of the skin of free-living whales, suggesting either that oil may not stick to the skin surface due to its quality, or that contact with oil is rare because whales avoid slicks. Whether active avoidance occurs remains uncertain, but observations in spill situations suggest that whales do not take notice of oil spills (Engelhardt 1985, and references therein, Robertson 1998). However, in experiments exposure of cetacean skin to oil has caused cell damage in epidermis (Geraci & St. Aubin 1982). The unique structure of cetacean skin and the fact that it contains lots of vitamin C may serve to protect against harmful effects of oil (Geraci & St. Aubin 1980). There is some suggestion that whales may take up petroleum derived hydrocarbons (Engelhardt 1985). The presence of MFO and a hydrocarbon marker enzyme P-450 has been demonstrated in the liver of several cetacean species (Geraci & St. Aubin 1982), indicating that cetaceans should be capable of detoxifying oil. Spilled oil may interfere with feeding behaviour, in particular filtering efficiency, through effect on baleen function as oil becomes trapped onto baleen hairs (Geraci & St. Aubin 1982, Braithwaite & al. 1983). However, the development of offshore petroleum and gas resources seems to present more threats to marine mammals than accidental oil spills in the Arctic (Geraci & St. Aubin 1980). Seismic activities, noise (affecting physiology and behaviour), and long-term accumulation of petroleum fractions through the food chain in the vicinity of oil platforms are thus more of concern.

Behavioural consequences and thermal effects of oil fouling have been noted for pinnipeds and sea otters (Geraci & St. Aubin 1980). Davis & Anderson (1976) noted reduced growth rate in oiled seal pups, but could not detect changes in nursing behaviour as a result of oiling. Experimentally oiled sea otters spend more time underwater trying to clean themselves, and seals show variable signs of aggression and arching of the back (Geraci & Smith 1976a, Geraci & St. Aubin 1980). After removal from oiled experiment tanks, animals' behavioural and physical signs disappeared rather quickly. Thermal effects are noted for sea otter pups (*Enhydra lutris*), and the Weddell seal (*Leptonychotes weddellii*) (Kooyman & al. 1977). Most true and phocid seals, sea lions and walrus, however, have a relatively coarse and short fur and a thick blubber. The danger of heat loss due to oil fouling is therefore small, but the fouling may cause a physical hindrance to swimming (Davis & Anderson 1976, Kooyman & al. 1977, Robertson 1998).

Sea otters are peripheral in their occurrence in the Arctic and may therefore be more sensitive to oil effects, in particular thermal effects, than the "true" arctic mammals (Kooyman & al. 1977). Oiling and consequent washing of sea otter furs caused oxygen consumption increase and weakened thermoregulation (Costa & Kooyman 1981). The vulnerability of sea otters was documented clearly during the *Exxon Valdez* oil spill in Prince William Sound in 1989, where estimated 2000-3000 animals perished (Waldichuk 1990). Some anticipated effects of surface contact with oil are irritation and inflammation of eyes, skin, and sensitive mucous membranes (Geraci & Smith 1976a). In experiments acute organ damage has not been recorded even after ingestion of relatively large quantities of oil in ringed seals (*Phoca hispida*) (Geraci & Smith 1976a), but such findings cannot be extrapolated to greater quantities of oil and other marine mammals. Ringed seals rapidly absorbed crude oil hydrocarbons to body tissue and fluids, ultimately excreting the compounds via bile and urine (Engelhardt & al. 1977). Experimental exposure of adult ringed seals to hydrocarbons increased the mixed function oxygenase (MFO) activity, indicating the enzymatic break-down of hydrocarbons in these animals (Engelhardt 1981). Furthermore, studies in ringed seals showed that volatile hydrocarbons and likely to become absorbed through respiratory tract (Geraci & Smith 1976b). Kidney and liver lesions were observed, but no associated lung pathology. Effects of prolonged inhalation may cause disturbance of the central nervous system, pneumonia and death (Carpenter & al. 1978). Oiling of grey seal pups (*Halichoerus grypus*) did not affect the body weight development as the oil typically disappeared when the white lanugo fur was moulted (Jenssen & al. 1991).

Regarding polar bears, severe heat loss and elevated compensatory metabolism have been observed after experimental oiling of their fur (Hurst & al. 1982). Extensive grooming causes ingestion of oil leading to tissue elevation of hydrocarbons and gradual development of dysfunction, and lethal damage in several internal organs. Renal failure may be the ultimate cause of death (Øritsland & al. 1981).

3. DISCUSSION

Petroleum hydrocarbons affect organisms in a variety of ways, ranging from death to biomolecular, pathological and cellular effects to merely physical nuisance. The effect on organisms, and whether it is reversible or not, depends on numerous physical and biological factors, that also affect oil spreading, weathering (including biodegradation), and absorption of toxic compounds into organisms. These are for example the volume and type of spilled oil, water temperature and water currents, the presence of sea ice and snow cover, season (in particular in the Arctic), location of the spill (open water vs. shore line) and oil combating measures taken. At least some effects seem temperature dependent and are more pronounced in warmer water, thus not likely in the Arctic. For long-lived organisms, such as kelps and vertebrates, the developmental stage can be crucial; reproductive organs, embryos and juveniles are at most risk. Thus, we cannot nominate only a single species in the Arctic that could be determined as particularly sensitive to oil and consequently an indicator species of the severity of oil pollution. Sublethal effects are numerous and can become manifested in various ways in the organisms' reproduction, behaviour (e.g. feeding, mating) and thus growth, and physiology (general fitness). The apparent complexity of e.g. the metabolic response to petroleum is hardly surprising in view of the broad range of physiologically active compounds present in crude oil, and physiological processes that are undoubtedly affected in different aquatic species.

An evaluation of the consequences of the environmental contamination requires an understanding of the extent to which it is responsible for changes in individuals and populations in the affected area (discussed in depth by e.g. Clark 1982, Jones 1982). Population change is not solely related to mortality which may be observed, but depends also on the population dynamics (e.g. migration, age structure and reproduction patterns of a given population), stock size, survival strategy of the species affected, and other possible disturbances in the area. Furthermore, while individual organisms may die to oil effects, on a population level the outcome may not be that dramatic. At open sea, plankton is likely to be transported from one area to another by wave action and water currents, and thus the contaminated part of the plankton population may become substituted with "fresh" material from elsewhere. In such an example, the actual sufferers may be found in the benthos; organisms whose food quality and uptake is dependent on the quantity and quality of settling material from above (here dead, oil contaminated plankters).

In the littoral, most severe effects of oil spills are documented for soft bottoms and sheltered bays. Oil penetrates and readily remains between the tiny pores of mud and silt, where it will have more time to stay in contact with benthic organisms. On hard bottoms, such as rocky shores, wave action is usually stronger, and thus capable of transporting even large masses of oil to the open sea. The clinging of oil onto hard surfaces is much less than oil penetration into soft substrates. In general, field studies of the biological consequences of oil spills show good agreement with the experimental data: intertidal and subtidal benthic communities are notably affected and can take a long time to recover, undergoing slow and subtle changes.

Oil does not need to affect directly all organisms on every level of the food web, and yet with time the consequences of an oil spill may be discernible in the entire system (Fig. 3). Such is a consequence of biomagnification; when littoral and benthic crustaceans for example are contaminated by oil, the negative effects will soon be observed in their predators, such as polar cod, seals and, finally, the polar bear, i.e. animals that migrate to the contaminated area from elsewhere. Another example is from the open water ecosystem. Divers and whales will bioaccumulate hydrocarbons by consuming pelagic fish and invertebrates exposed to oil, and consequently develop various hydrocarbon related symptoms (Fig. 3).

Fishery is always one of the main concerns when an oil spill happens. Field studies after oil spills have generally failed to document the widespread effects of oil on fish. In *Argo Merchant*, *Ekofisk Bravo* and *Amoco Cadiz* accidents effects on fish stocks were negligible (e.g. Rice 1985 and references therein, Teal & Howarth 1984). Although adult fish can be killed by oil spills, this probably poses less of a threat to commercial fisheries than do damage to eggs and larvae, or changes in the ecosystem supporting the fishery (Teal & Howarth 1984). Ecosystem changes in the lower levels of the food chain and thus long-term effects on fish are more likely than acute effects. However, such interactions are poorly studied, as is discussed e.g. by Vandermeulen (1982), and Teal & Howarth (1984). One of the

reasons for this is that long-term spill effects are generally local phenomena. Fish have been exposed to oil in many studies, but the methods have varied considerably among studies. Thus, direct comparison between studies is not usually valid.

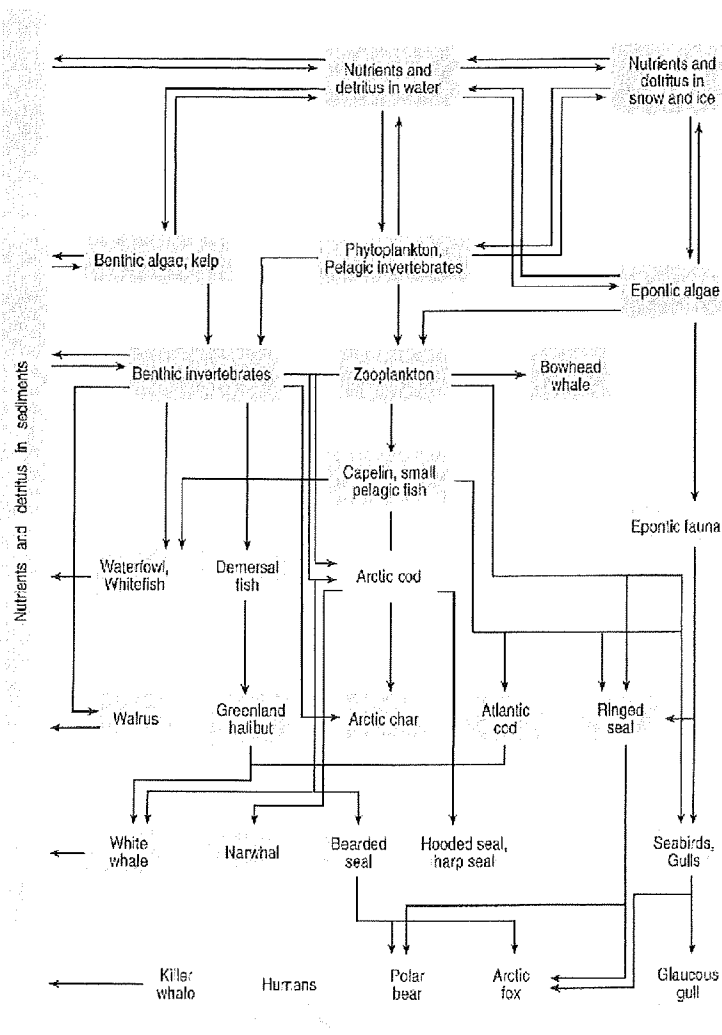


Fig. 3. A generalized marine food web in the Arctic. Source: de March & al. (1998).

There is a general impression of the exceptional sensitivity of arctic ecosystems to oil. Sensitivity of arctic zooplankton and benthic organisms to oil contamination, for example, is discussed by Wells & Percy (1985), and it seems to be a consequence of several biotic and physical factors. In cold water zooplankton has lower uptake, metabolic and excretion rates, possibly leading to lower inputs of oil into organisms, but also to longer detoxification and depuration times. Low temperature can influence the availability of oil by changing its solubility, physical form and the stability of different fractions, or by extending the exposure time by retarding the loss of hydrocarbons from the medium, and extending the predisposition time of organisms to oil. It can also delay the onset of e.g. criteria of death like immobilisation. Based on extensive experimental data set Rice & co-workers (1976, 1977a) concluded, that Alaskan marine invertebrate species may be slightly more sensitive than similar species in lower latitudes. However, our knowledge of the species specific sensitivity of arctic organisms to oil is still patchy. In the past a number of arctic benthic species have been subjected to acute lethal tests with petroleum hydrocarbons (Wells & Percy 1985), but the taxonomic coverage has been uneven: of the 24 species listed by Wells & Percy (1985) almost 90% are crustaceans and at least 50% are amphipods. Also, comparisons of results are complicated by differences in oil types, methods of preparation, exposure regimes and criteria of death. In general, amphipods and decapods seem relatively sensitive to

oil, while isopods are more resistant. As noted earlier, the polar cod seems to be particularly sensitive to oil (Rice & al. 1979).

Avoidance is one way of minimising the negative effects of oil. However, avoidance of oil by motile organisms is not a universal type of behaviour, as was evidenced for the isopods, and fish such as the juvenile English sole (*Parophrys vetulus*, syn. *Pleuronectes vetulus*) by Percy (1976), and Weber & co-workers (1979). Isopods did not seem to exhibit any particular attraction or repulsion responses when confronted by oil masses. The effects that the oil in the sediments may have on fish can depend on overall amount, composition (age) of the pollutant, and the season (activity level of the fish) (Fletcher & al. 1981). Whether fish in nature avoid, ignore or are attracted to an area with an oil spill remains largely unanswered. There are several cases where field observations have been conflicting, as evidenced during past oil spills, where both avoidance behaviour and attraction of in particular fish has been recorded (Teal & Howarth 1984). Non-avoidance behaviour of animals when confronted with crude oil may play a crucial role in determining the subsequent impact upon organisms and thus populations. Burrowing species, like the much studied arctic species, the Baltic telling *Macoma balthica*, rarely venture onto the surface, although most extend their siphons into the overlying water for feeding and respiration. The impact of oil on burrowing species depends largely upon how it is introduced to the habitat. The response of the animal to dispersed oil in the overlying water and to oil within the sediment is to emerge onto the surface, not to burrow deeper into the substrate, and usually in proportion to the dose (Taylor & Karinen 1977, Wells & Percy 1985). Although the sediment depth may affect the surface speed, chronic contamination will force animals to eventually surface (Wells and Percy 1985, and references therein). Not all animals would become killed by oil, at least in the short term, but high concentrations (3 mg/l) and long-term exposure would no doubt be lethal. Reburying is not possible for as long as the oil prevails (Taylor & Karinen 1977, Engelhardt & al. 1983).

Altogether, it seems that long-term effects of oil spills on open water ecosystem, including cetaceans may be less important than generally assumed, and restricted to a relatively small area. However, there is uncertainty about the effect of oil in restricted environments, such as in polynyas and ice leads. In the littoral and benthos, where organisms are often either sessile or slowly moving, and the motility of the spilled oil is weaker than in the open water the effect is likely to be more pronounced, including consequences on an individual and population level, and expanding the time of disturbance. Observed effects of long-term and chronic discharges in marine ecosystem have not been discussed in this report. However, to summarise the existing literature, effects of e.g. oil drilling activities are sooner local than widespread, but the chronic exposure of organisms to oil and drill muds, for example, can cause pronounced, permanent changes in species composition, fitness and diversity of benthic communities (e.g. Sharp & Appan (1982), Addy (1987), Kingston (1987), Moore & co-workers (1987a), Gray & al. (1990) and Olsgard & Gray (1995). The temporal changes seen after oil spills are comparable with the spatial changes observed around chronic discharges, in particular in case of an ecosystem with dominance of only a few species.

4. SUMMARY AND CONCLUSIONS

Oil spill effects on arctic marine organisms and ecosystems can be locally dramatic. The severity of contamination is not only dependent on the type of organism exposed to oil, but also on the type and volumes of spilled oil, the speed of oil weathering processes (e.g. evaporation, dispersion, degradation by bacteria), oil combating measures taken, and the location of the spill. Large spill of oil with a high aromatic fraction is worse to marine life than oil with less aromatic compounds. Weathering of large oil spills require more time than that of smaller spills, and the use of oil dispersants (not discussed in depth in this report) has usually more severe consequences on organisms than the mechanical and/or biological oil combating. Organisms in the littoral and benthos, particularly sessile and slowly moving invertebrates such as molluscs, snails, and crustaceans suffer the most from oil exposure. Seaweeds (macroalgae) attached to hard substrates seem to have a fairly good potential for recovery after oil exposure. In the open water, contaminated plankton can be replaced by "fresh material" transported from sea areas outside the spill site. Avoidance of oil is typical for fish and larger vertebrates (seals, whales), but sea birds, in particular divers are at risk in the pelagial. Oil associated with sea ice may have a notable effect on the ice biota, particularly as oil tends to become trapped between ice platelets

and floes, and within brine channels in ice, and thus lengthens the contamination time. Recovery of marine life may be complete, nearly complete or only partial. Complete recovery is more likely in the open water ecosystems where water currents disperse oil both horizontally and vertically. Partial recovery is typical of littoral and benthic ecosystems, and is dependent not only on the factors described above, but also on the type of the original community (species composition, dominance, biomass), the organisms' potential of adjusting to changes (abiotic and biotic) in the habitat, and thus the ability of recolonising the deserted area. Even after several years of an oil accident, the recovery process of a littoral or benthic ecosystem may still be incomplete and, in the worst case, may not ever reach the original state.

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