

# Particles in relation to water quality deterioration and problems in the network

*State-of-the-art review*

# TECHNEAU

*D 5.5.1 + D 5.5.2*



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## **Title**

Particles in relation to water quality deterioration and problems in the network

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# Executive Summary

## Introduction and Importance

A considerable part of water quality problems in the network is associated with accumulation of particles in the network. Particles in networks have the following two main origins: (1) the ingress with water from the treatment plant, and (2) the corrosion of unprotected iron mains. Re-suspension of sediments may lead to discoloured water at the consumer's tap, which is a major cause of customer complaints. However, not only the aesthetics of discolouration are of concern, as sediment dispersal is accompanied with a release of microorganisms into the water along with undesirable components, then triggering possible threats to the human health. The formation as well as the resuspension of the deposits are not understood in detail. Therefore, a general approach describing all of the important processes is not available. The aim of WP 5.5 is, based on extensive literature review and investigations in networks and pilot plants, to develop a mathematical approach which describes the relevant processes of deposits behaviour in the network. In the next step the mathematical approach should be linked to a hydraulic model. This model will be used to enable a better understanding of the most important processes and, hence, to open the way for efficient operation and maintenance measures (WP 5.6).

## Approach

The aim of this report is to summarize the actual knowledge about deposits in the network and water quality modelling based on an extensive literature review.

## Results

The processes of sediment formation in the network as well as different methods to investigate the build up of deposits are described. Furthermore, microbiological problems associated with sediments are shown. At last, the actual state of water quality modelling is summarized.

## More information

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# 1 Introduction

## 1.1 General approach

The objective of work package 5.5 is to develop an integral model of water quality processes in distribution systems. The model should enable to minimise deterioration of water quality, such as discolouration or taste and odour problems. Models will be employed to enable a better understanding of the most important processes and hence open the way for efficient operation and maintenance measures. The examinations will consider particles and biofilms as well as their interactions with the pipe material in networks. This WP is divided into three components, which are (1) particles and sediments, (2) biofilms and biodegradable compounds and (3) water quality modelling.

This report describes the actual knowledge of the different processes determining the formation and mobilisation of sediments from particles in drinking water distribution systems. The report contains the deliverables D5.5.1 and D 5.5.2 of the TECHNEAU project.

## 1.2 Water quality in the distribution system

The quality of water changes as it passes through the water distribution system. The changes depend on many factors, such as the finished water quality produced at the treatment plant (especially suspended particles content, iron, manganese, aluminium and NOM compounds), material, condition and age of pipes, travel time in the network, temperature, the effect of storage tanks, etc.

A considerable part of water quality problems in the network is associated with accumulation of particles in the network. Particles in networks have the following two main origins:

- (1) the ingress with water from the treatment plant, and
- (2) the corrosion of unprotected iron mains.

Re-suspension of sediments may lead to discoloured water at the consumer's tap, which is a major cause of customer complaints.

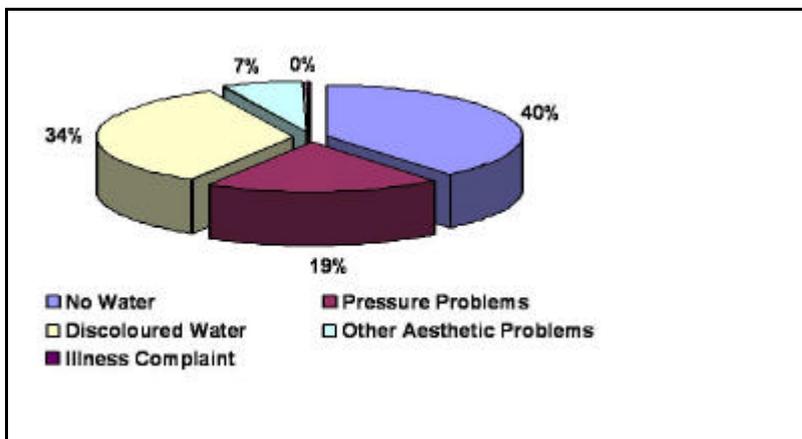
Sediments provide circumstances favourable for microorganisms and even vertebrates to grow within them. Thus, not only the aesthetics of discolouration are of concern, as sediment dispersal is accompanied with a release of microorganisms into the water along with undesirable components, then triggering possible threats to the human health.

Beside the sedimentation of particles from the treatment plant and from corrosion, the transport of sediments influences the network sediment budget. Accordingly, a more detailed knowledge about the processes of sediment formation and sediment mobilisation in a network are necessary to minimise or to avoid deterioration of water quality, such as discolouration or taste and odour, as well as microbiological problems. Models have to be used to enable a better understanding of the most important processes and, hence, to open the way for efficient operation and maintenance measures.

## 2 Accumulation of particles in the transport and distribution system and contribution of the finished water

### 2.1 Perception of particles as discolouration

A large proportion of the customer contacts that drinking water supply companies across the world receive stem from the occurrence of discoloured water in the drinking water distribution system (DWDS). Fig 2-1 shows a typical breakdown of customer contacts for a UK water company (Vreeburg *et al.*, 2007). Fig 2-2 shows some examples of discoloured water supplied to customers, that have led to the complaints.



*Fig 2-1 Typical breakdown of reasons for customer contacts for a 5-year period for a UK water company.*

The pictures in Fig 2-2 show the black to brownish or red nature of discolouration and that it is not covered by the expression 'brown water'. The different appearances of the discolouration suggest that there is not just one reason for the problems, but probably a mixture of different processes that can lead to discolouration in a broad sense.



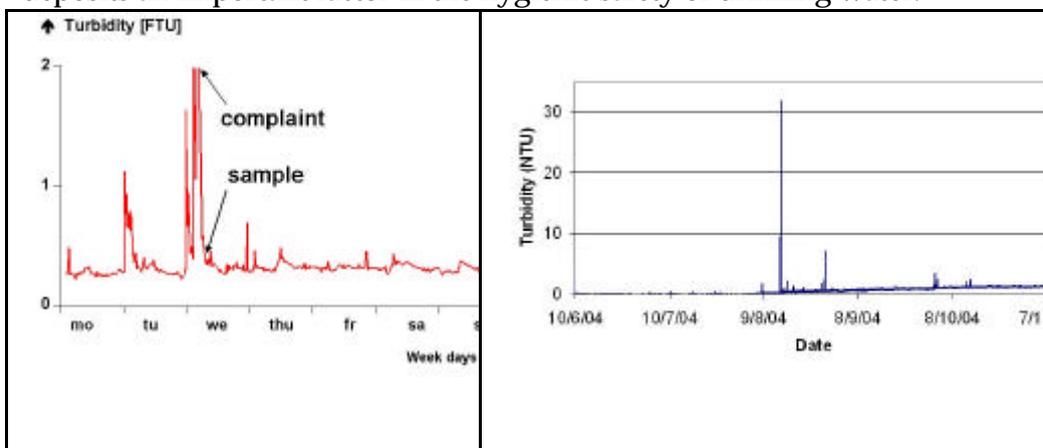
*Fig 2-2 Examples of discoloured water*

Discoloured water incidents as shown in Fig 2-2 greatly affect customers' confidence in tap water quality and the quality of service provided by water companies. Although good customer perception is a major driver for water companies (van Dijk *et al*, 2005), a thorough understanding of the mechanisms and processes that lead to discolouration are currently lacking or at least not widely applied. Hence, water companies can only respond to discolouration complaints in a reactive manner. Within modern customer focussed water companies such reactive maintenance is no longer acceptable, particularly within a strict regulatory framework as in the UK. Water companies urgently need a practicable understanding of the processes and mechanisms leading to discolouration incidents and need to develop management tools and techniques.

## 2.2 Nature of discoloured water

Although referred to as discolouration, the visual effect observed by customers is rarely colour in a strict water quality sense, defined as dissolved contaminants. Typically, if a 'discoloured' water sample is left to stand for a prolonged period (over night) it will clear and material will deposit (Fig 2-2). Hence, it can be concluded that it is particulate matter that the customer experiences as 'discolouration'. The measurable parameter requiring investigation is therefore turbidity. However, different particles have significantly different effects on perceived turbidity, or discolouration. A combination of factors including obscuration, reflection, refraction, diffraction and scatter contribute, although scattering usually dominates. Peak scattering occurs for particles at around half a micron in diameter with a rapid fall off for suspensions of larger or smaller sizes (Russell, 1993). New developments in measuring equipment have made more sophisticated particle counters available that are not dependent on the scattering of light and therefore can give greater detail for a better understanding of the volume of particles involved.

Particulate accumulations are also known to have a relation with biological activity (Gauthier *et al*, 1999). One to 12% of the organic matter in the particulate accumulations may consist of bacterial biomass, making the deposits an important factor in the hygienic safety of drinking water.



*Fig 2-3 Typical discolouration events; left measured in the Netherlands, right in the UK (Vreeburg 2007)*

Discoloured water events are difficult to study in real systems because they often occur over short periods seemingly for unpredictable reason. Fig 2-3 shows some typical short duration events captured by turbidity instruments installed in systems in the Netherlands and the UK, respectively. The figures show that discolouration events have the same characteristics: a sharp rise in turbidity that reduces within a few hours, despite considerable differences between the Dutch and UK networks with respect to the materials used. Similar results are found in monitoring turbidity and velocity at different locations in the Melbourne (Australia) drinking water distribution system (Prince *et al.*, 2001).

The largest growth period for network in the Netherlands was from 1945 – 1980, hence, the average age of the network is 42 years and the predominant pipe materials are PVC and AC. Conversely in the UK the networks have not experienced such intensive investment meaning their systems are still dominated by cast iron pipes dating back over the last 100 years or more. The Australian network is more recent than the Dutch network and has concrete or cement mortar lined cast iron and PVC as dominant pipe materials (Prince *et al.*, 2003). The treatment histories of the systems are also different with systems in the Netherlands having long adopted a very high standard of treatment and a policy of no chlorination, while the UK has seen a variety of levels of service. The Australian network is supplied with unfiltered water, and is dosed with chlorine, fluoride and lime (Verberk, 2007).

These historic factors are key to understanding the levels of service and the processes leading to the occurrence of discolouration events as shown in Fig 2-2 and Fig 2-3. This difference is also manifest in the reactive trigger levels that companies use to initiate cleaning in response to discolouration, typically around 4 contacts per 1000 properties in UK compared with 0,5 to 1 contact per 1000 in the Netherlands and 6 contacts per 1000 properties in Australia. This shows that, despite obvious differences in systems, the nature of discolouration problems is the same though the number of problems varies. Intuitively discolouration in the Dutch systems should be almost non-existent considering its history of good treatment, very low leakage and a network with a limited number of cast iron pipes, but the figures show that discolouration does occur and other processes are involved besides corrosion. This may also highlight the inconsistent nature of customers, with a propensity to make contact predominately when the quality of the water changes from what is perceived as ‘normal’.

### **2.3 Particle-related processes in the drinking water distribution systems**

Discolouration is associated with the mobilisation of accumulated particles from within distribution networks. Such particles have different sizes and

densities and, hence, probably different origins, often characterised as either external sources or from processes occurring within the system. Particles can enter the distribution network as background concentrations of organic and inorganic material from the source water (Lin *et al.*, 1997; South-East-Water, 1998; Kirmeyer *et al.*, 2000; Slaats, 2002; Ellison, 2003), due to incomplete removal of suspended solids at the treatment plant (Gauthier *et al.*, 2001; Vreeburg *et al.*, 2004) or be added to the water by the treatment plant itself, such as carbon and sand particles, alum or iron flocks and bio particles originating from bio filters (Alere *et al.*, 1997). The distribution system itself can also produce particles, such as from pipe and fitting corrosion and lining erosion (Stephenson, 1989; Ruta, 1999; Gauthier *et al.*, 2001; Clement *et al.*, 2002; Slaats, 2002; Boxall *et al.*, 2003), biological growth (Le Chevallier *et al.*, 1987; Stephenson, 1989; Clark *et al.*, 1993; Meches, 2001) and chemical reactions (Stephenson, 1989; Sly *et al.*, 1990; Walski, 1991; Lin *et al.*, 1997; Kirmeyer *et al.*, 2000); or external contamination that may occur during operations such as pipe repairs (Gauthier *et al.*, 1996; Slaats, 2002), intrusion (Gauthier *et al.*, 1999; Kirmeyer *et al.*, 2000; Prince *et al.*, 2001) and backflow. Possibly the most common and significant biological process is biofilm formation which can result from the presence of assimilable organic carbon in the water or the pipe wall (van der Kooij, 2002). The effects of these complex and interacting processes are further complicated by exposure to various different physical and chemical conditions during passage through distribution systems including contact with a range of different pipe materials and ages and different hydraulic conditions. The formation and growth of particles is a very complex process which is currently poorly understood. Factors such as contact times, contact surfaces and hydraulic conditions are likely to play important roles in controlling these processes (Huck *et al.*, 2004). These sources, external and internal, rarely contribute directly to discolouration events but facilitate the gradual accumulation of material within the distribution system.

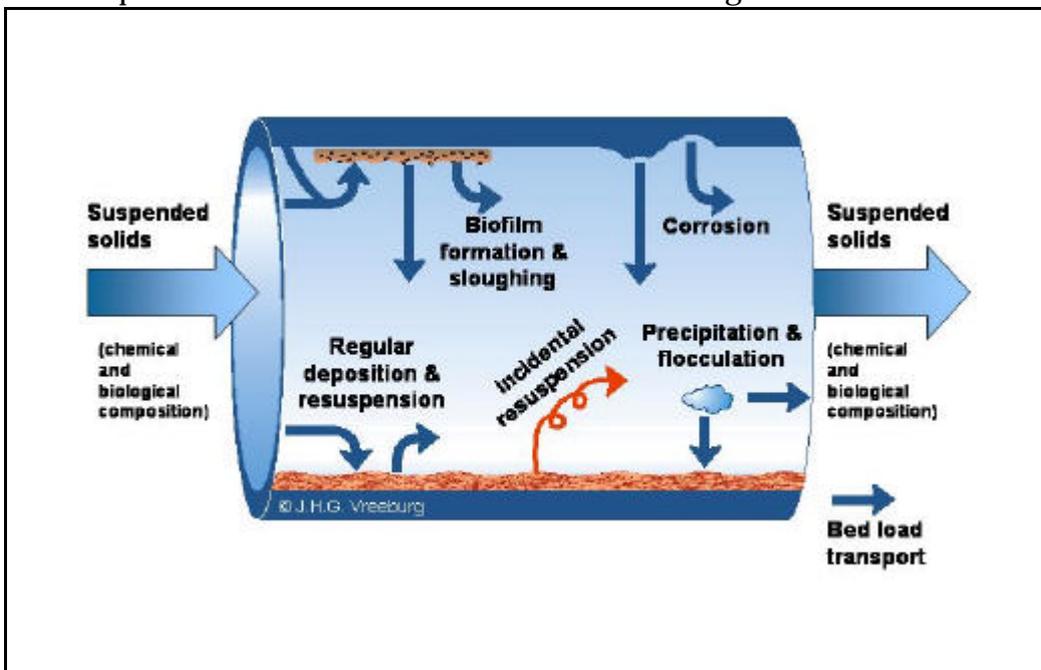
Next to the sources and growth of particles, it is important to understand the hydraulic behaviour of the particles to determine the fate of the particles in the network. Boxall *et al.* (2001) presented results for the distribution of particle sizes found in discoloured water samples, suggesting a reproducible distribution of particle sizes irrespective of network conditions, source water, etc. They suggested that the size range of the particles was predominately less than 0.050 mm, with an average size of around 0.010 mm and a significant number of particles in the sub 0.005 mm range. (Boxall *et al.*, 2001) went on to show that it is unlikely that gravitational settling alone will be a sufficient force for accumulation of such particles since turbulent forces generated by even the lowest flows within a distribution system are likely to be sufficient to overcome gravity settling forces. This is particularly valid for the smaller sized particles found within discolouration samples which will dominate discolouration due to their light scattering properties. Fig 2-4 shows material accumulation due to corrosion processes around the complete circumference of pipe samples and a lack of invert deposits, consistent with these concepts. Samples such as these have been installed in a laboratory facility and significant discolouration was generated by exposing them to flushing flow

rates, despite the disturbance of weakly adhered material caused by obtaining the samples. The discolouration could also be caused by the release of soft material that is normally contained by hard corrosion scales. Rupture of these scales due to the flushing forces could have generated the discolouration.



*Fig 2-4 Material accumulation around the complete perimeter of cast iron pipe samples*

Transport of particles will not only occur through the liquid phase as suspended solids, but can also take place as bed load transport: particles rolling over the pipe wall. Though not mentioned in literature, bed load is a distinct possibility for particle transport. All the aforementioned particle-related processes in a network and are visualised in Fig 2-5.



*Fig 2-5 Particle-related processes in a network. The direction of the arrows indicate the path particles follow in the pipe. The vertically aimed arrows indicate particles settling on the pipe wall; the horizontally aimed arrows indicate particles moving with the water as suspended solids.*

Overall it can be concluded that the mechanism leading to discolouration events are complex, poorly understood and interactive. However, the processes may be understood through the model presented in Fig 2-5. The underlying cause of discolouration is particles attached by some means to the pipe wall, irrespective of their origin, either imported from outside the network by the treatment of produced within the network itself. In normal flow the particles regularly deposit and partly resuspend without affecting the aesthetic quality of the water. If flows are increased above normal values in an hydraulic incident, scouring forces and shear stress increase consequently and then the particles may be mobilised. This incidental resuspension may lead to customer complaints.

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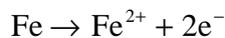
# 3 In-pipe corrosion and production and accumulation of particles

## 3.1 Discolouration as a result of corrosion

The most common process associated with the phenomenon of discolouration is historically the corrosion of cast iron pipes as is suggested on the site of the Drinking Water Inspectorate in the UK (DWI, 2007). Many authors who studied the corrosion of cast iron in potable water systems conclude that this is a major cause of discolouration (Böhler *et al.*, 2005; Smith *et al.*, 1997; McNeill *et al.*, 2001). Corroding pipes, which will be subsequently oxidised, release Iron(II) which dissolves in the bulk water where it is oxidized and forms iron particles and suspended red-brown Ferrous(III)-hydroxide flocks. These may settle within the pipe and/or be transported with the water to other parts of the network, where they settle and form sediments.

## 3.2 Corrosion process

The primary process of corrosion takes place on the pipe's wall, as the anodic release of ferrous iron from iron metal occurs:



Complementarily, the released electrons reduce the acceptors available in the water, such as oxygen, chlorine or hydrogen ion. When dissolved oxygen (DO) is available, its reduction is the predominant cathodic reaction (Kuch, 1988; Sarin, 2004):

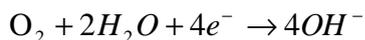
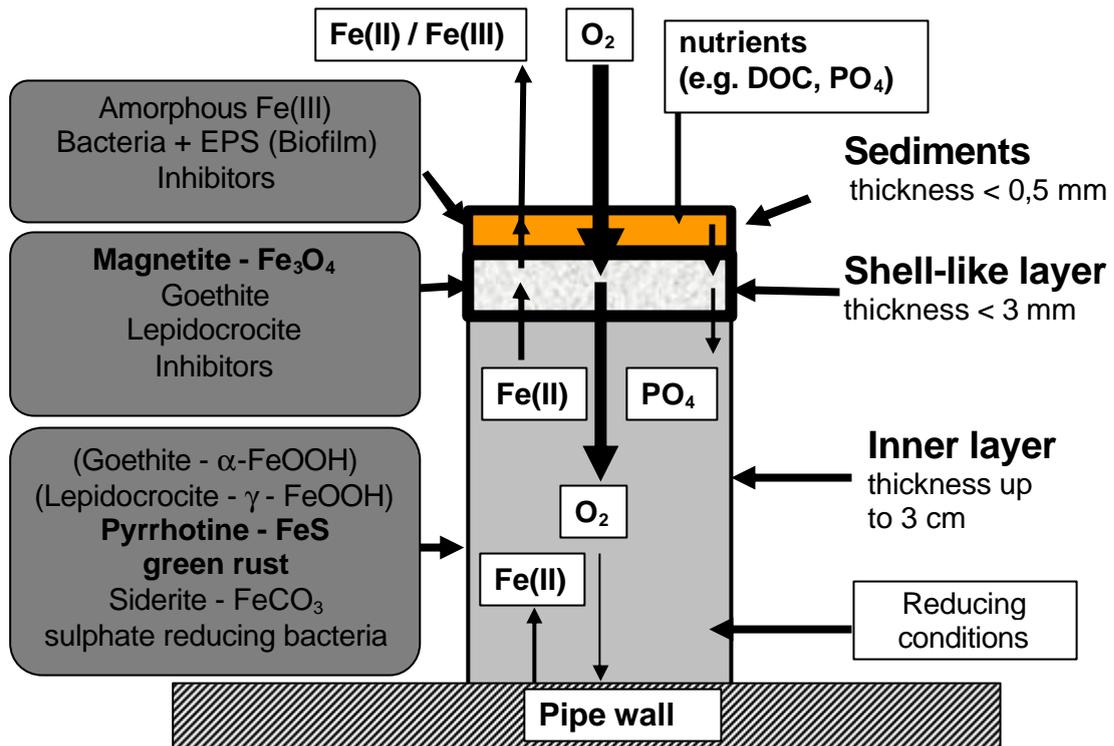


Figure 2 shows the typical layered structure in aged unprotected cast iron and steel pipes' surface after a long term flow with well buffered, sulphate containing drinking water.



**Figure 2:** Schematic layer structure in aged unprotected cast iron and steel pipes (Böhler *et al.*, 2005).

As result of corrosion of the pipe wall in presence of oxygen a scale layer of the formed corrosion products “grows” into the water. This shell-like layer inhibits the diffusion of oxygen from the bulk water to the pipe wall and can so decrease the corrosion rate. Depending on that formation conditions it consists of different oxidised iron phases such as Magnetite ( $\text{Fe}_3\text{O}_4$ ), Goethite ( $\alpha\text{-FeOOH}$ ), Lepidocrocite ( $\gamma\text{-FeOOH}$ ), Hematite ( $\alpha\text{-Fe}_2\text{O}_3$ ) and Siderite ( $\text{FeCO}_3$ ). The type of mineral phase formed is determined by the (1) rate of oxidation, (2) the pH of the solution, (3) the concentration of the  $\text{Fe}^{2+}$  ions, (4) the concentration of other species in solution (Sarin, 2004). The chemical and physical structure of the shell-like layer plays an important role in the further corrosion processes, because of the flux of oxidants, ions and electrons between bulk water and the inner layer. On the other hand it is subject to transformation, precipitation and redissolution processes.

Between the shell-like layer and the pipe wall a porous mass, made up of agglomerates of small particles with different morphologies, is found. Depending on the flux of oxidants and the reaction rates, the conditions within that inner layer are more or less reducing. It contains reduced iron phases such as ferric hydroxides, green rust and Pyrrhotin but also chemically less stable oxidised forms (Sontheimer, 1982, Kuch, 1988, Sarin *et al.*, 2001). The inner layer serves as large source of soluble  $\text{Fe}^{2+}$  (Lytle *et al.*, 2005).

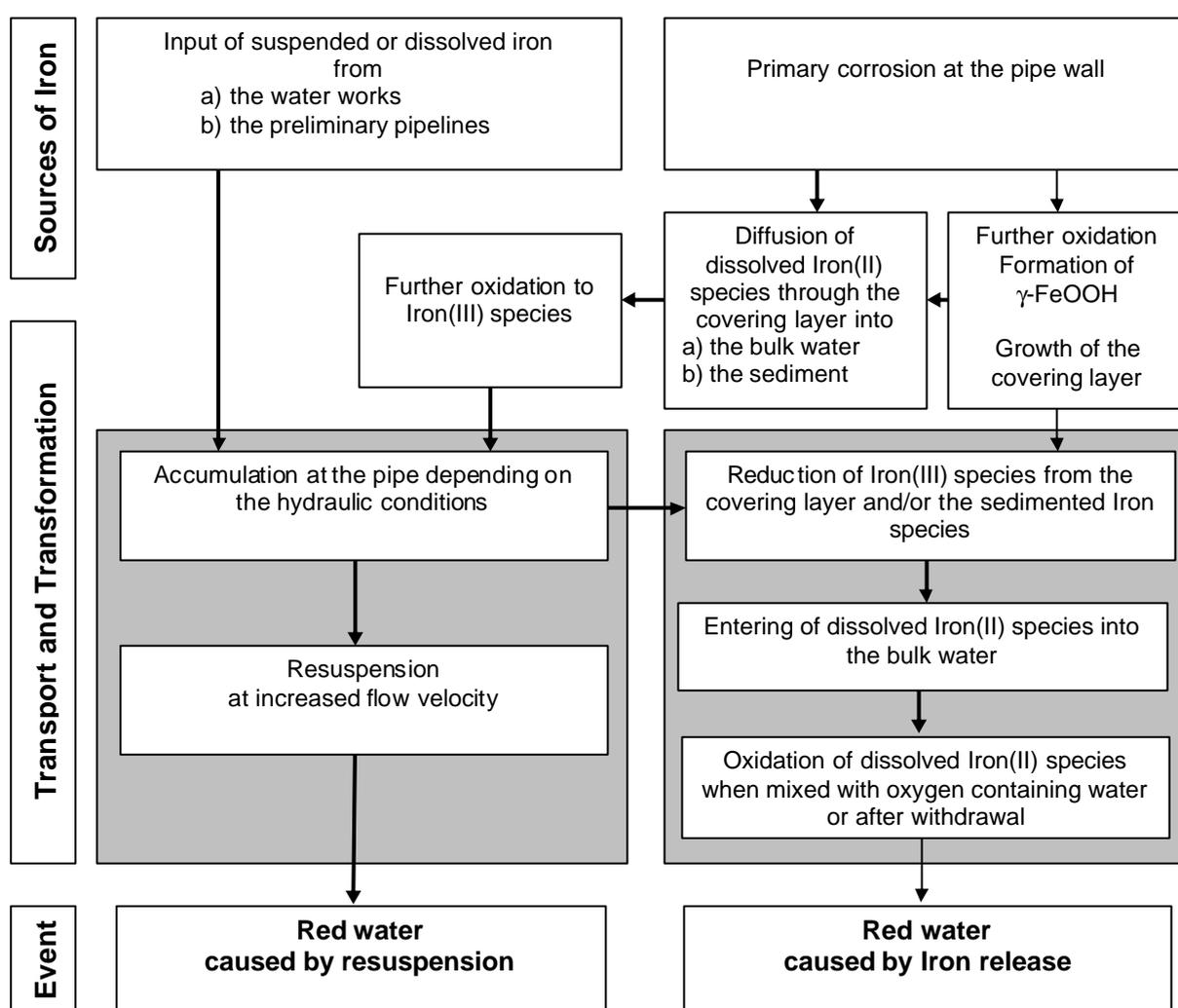
Under operational conditions the presence of oxygen as oxidant plays an important role on the scale composition and its influence on the corrosion process. If the scale surface, where ferrous iron oxidation occurs, is supplied sufficiently with oxygen, ferric iron precipitates mainly within the scale. The porosity of the scale is believed to be reduced, thus leading to lower corrosion rates and iron release (Sontheimer, 1988; Sarin, 2003; Sarin 2004). If oxygen availability is lower than its consumption via the reaction with ferrous iron, an anoxic zone is created at or near the scale surface. The rate of formation of this zone depends on the water chemistry and the nature of the scale. With prolonged period of these conditions ferric phases will be reduced (e.g. by the Kuch mechanism) and Fe(III)-Ions are released to the bulk water (Kuch, 1988; Sarin, 2004; Lytle *et al.*, 2005). As a result the scale porosity can increase again, allowing more ferrous iron to diffuse in the bulk water. The “Kuch mechanism” is described in figure 2.

<b>Continuous flow under non-limiting oxygen concentrations:</b>		
$\text{Fe} + \frac{1}{2}\text{O}_2 + \text{H}_2\text{O} \longrightarrow$	$\text{Fe}^{2+} + 2\text{OH}^-$	primary oxidation at the pipe wall
$\text{Fe}^{2+} + \frac{1}{4}\text{O}_2 + 2\text{OH}^- \longrightarrow$	$\text{FeOOH} + \frac{1}{2}\text{H}_2\text{O}$	further oxidation of Iron(II) species
<b>Stagnation after oxygen depletion:</b>		
$\text{Fe} \longrightarrow$	$\text{Fe}^{2+} + 2\text{e}^-$	anodic Iron release at the pipe wall (on the former cathode)
$2 \text{FeOOH} + 2\text{H}_3\text{O}^+ + 2\text{e}^- \longrightarrow$	$2\text{Fe}(\text{OH})_2 + \text{H}_2\text{O}$	cathodic reduction of corrosion products
<b>Restart of Flow, upon increase of oxygen concentration:</b>		
$2 \text{Fe}(\text{OH})_2 + \frac{1}{2}\text{O}_2 \longrightarrow$	$2 \text{FeOOH} + \text{H}_2\text{O}$	oxidation of dissolved iron

**Figure 2:** Non-steady corrosion – Process scheme (Kuch, 1984)

In this context, the role of loose deposits which accumulate on top of the shell-like layer on the corrosion process has come in the focus of discussion recently (Sarin *et al.*, 2004, Lytle *et al.*, 2005). On-site it can be formed by precipitation, when Fe(II) diffuses out from the scale and encounters higher anion concentrations in the bulk water, by oxidation to Fe(III). These deposits can be originated from the corrosion process itself or have been transported from other pipeline sections. They are assumed to consist mainly of oxide phases such as Lepidocrocite and amorphous Fe(III) in addition to precipitates of silicates, phosphates and carbonates (Sarin *et al.*, 2004). The deposits on top of the shell-like layer might have a direct impact on the corrosion process. Since it covers the protective shell-like layer, it forms an additional barrier for both the oxidants diffusing from the bulk water and, reversely,  $\text{Fe}^{2+}$  from the inner layer to the bulk water. The corrosion products in the corrosion film can themselves be subject of oxidation/reduction and precipitation/dissolution processes.

As mentioned above the deposits can be resuspended easily, then leading to a “secondary” discolouration event. The necessary shear stress for resuspension is given between 0.5 and 1.0 N/m<sup>2</sup> (Powell & Brandt, 2003, AWWA RF, 2003). To deal with discolouration risk due to resuspension different operational approaches are applied. They range from avoiding sudden hydraulic changes which can lead to resuspension (Powell & Brandt, 2003) to the consequent assessment and flushing of fouled pipes (Vreeburg *et al.*, 2004). Following the above described and commonly anticipated conceptual models of corrosion processes in drinking water pipes, the latter approach is more sustainable but also is related to an increased operational effort. A schematic summary of the above explained processes is drawn in Figure 3.



**Figure 3:** Scheme of the processes leading to discoloured (“red”) water in drinking water pipelines

A lot of research has been done to understand the influence of the water composition on the formation and chemical and physical structure of the

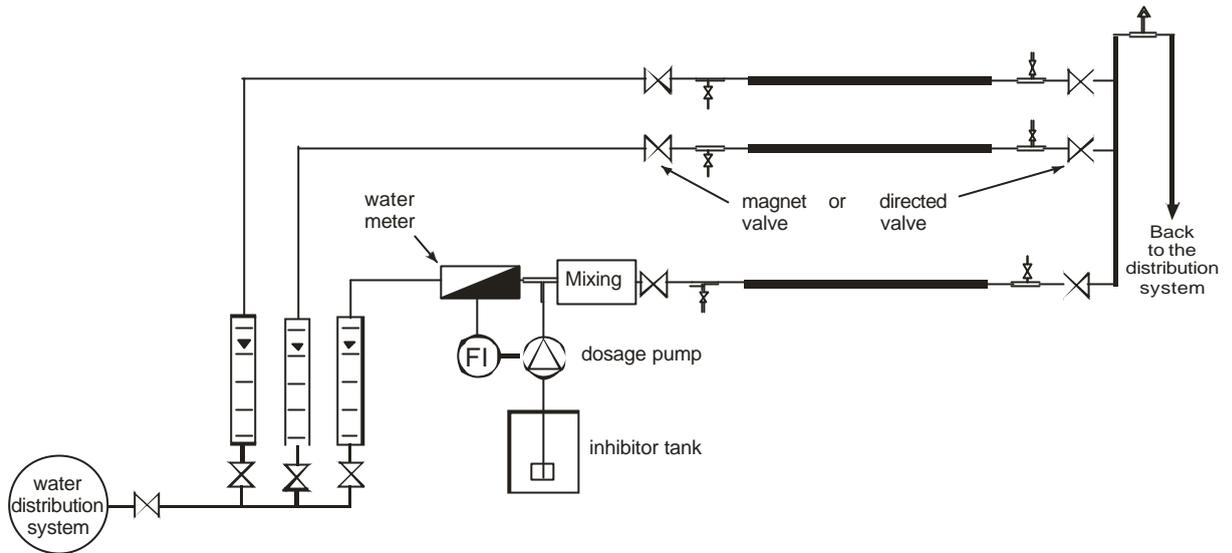
covering layer (e.g. Sontheimer, *et al.*, 1981; AWWA, 1985; Sarin *et al.*, 2003). Summarising, it can be stated that well buffered water with a low corrosion index  $((2 \cdot c(\text{SO}_4^{2-}) + c(\text{Cl}) + c(\text{NO}_3^-)) / \text{alkalinity})$  leads to well established corrosion scales, which inhibit the further corrosion process efficiently. Furthermore, organic matter, pH, phosphate and other parameters influence corrosion and iron release in a complex manner.

Until now it is not possible to estimate the iron release from the material and the formation of loose deposits by the corrosion just by using the relevant water quality parameters. To get information about the interaction of the water and the material, test rigs are used. In the following, two test rigs are described.

### **3.3 Test rig used by TZW**

Corrosion test rigs are used to deduce the cause of red water events in dependence on the water quality and the hydraulic conditions. The used test rig was constructed accordingly to the described system in AWWA (1985). The test rig consists of several 1 m long DN 20 steel (number 1.0035) pipes. The pipes are fed with the same water distributed in the network. Before the beginning of the investigations, the test pipes are treated with hydrochloric acid to remove any residual layers to create starting conditions which are as consistent as possible.

Figure 4 shows the scheme of a testing rig with 3 pipes. The upper pipe is run continuously with a flow of 300 L/h. The middle test pipe is run with an adapted flow pattern of stagnation and flow, by using a time controlled magnetic valve. This control will be replaced in future by a valve-controlled 24h timed line. The pipes are then operated by a 24 h timed line which is adjusted to the conditions in the network area to be tested. In the lower pipe an inhibitor is added to investigate the influence on corrosion. All of the test pipes are investigated in comparison to the reference pipe for assessing the results of changing conditions (flow behaviour, water parameter, water contents).



**Figure 4:** Scheme of a test rig with 3 test pipes DN 20, 1m long

### 3.3.1 Measuring methods

After installation of the test rig, the pipes are operated with a constant flow of 300 L/h and without adding inhibitors until a stable corrosion layer has formed. To define this moment, the following measurements should be realised at constant intervals, which are determined during the first measuring phases. The start interval should be shorter than 6 weeks. If the corrosion rate between two measurements is approximately stable, the program can start.

### 3.3.2 Specific iron discharge

To determine the corrosion rate it is necessary to detect the iron discharge at first. Therefore, seven flush steps are applied to the pipes, accordingly to the algorithm described by Böhler *et al.*, (2005). The water from each flush step is examined in regard to the iron and manganese concentration, as well as the turbidity (Table 1).

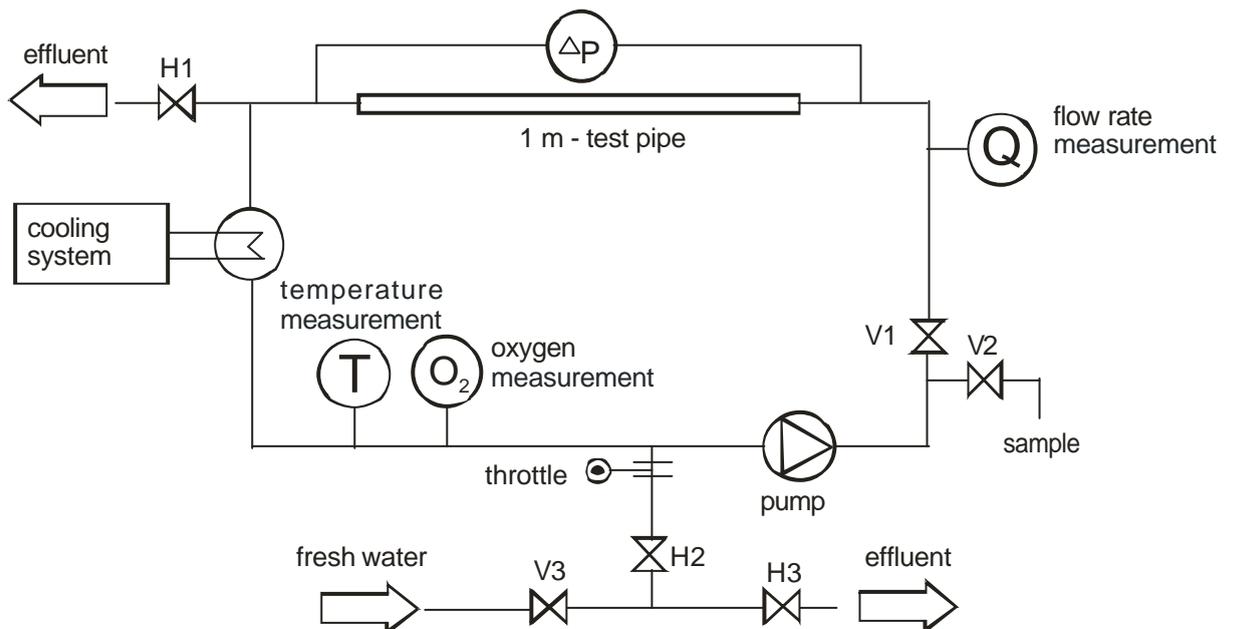
**Table 1:** Table of flushing speeds and duration (Böhler *et al.*, 2005)

Flush step	Flow	Flush speed	Flush duration	Flush water volume
	L/h	m/s	sec	L
1	300	0.27	120	10
2	450	0.40	80	10
3	600	0.53	60	10
4	800	0.71	45	10
5	1,000	0.89	36	10
6	1,250	1.11	29	10
7	1,500	1.33	24	10

### 3.3.3 Corrosion rate

Following flushing, the pipes are taken out of the testing rig one by one and placed into a test stand of corrosion stable materials (Figure 5), identical to that described by Kuch *et al.* (1983). Previously to testing, the pipes are flushed with fresh water for 3 minutes at 1,000 L/h. Afterwards, the rig is adjusted and the measurements for the determination of the corrosion and iron discharge rates start.

The warming of the water due to the pump circulation is compensated by the integrated cooling system, so that the temperature of the system is kept stable.



**Figure 5:** Scheme test loop device (Kuch *et al.*, 1983)

As described by Werner (1976) and Rudek (1979), the corrosion rate can be calculated by the oxygen consumption. It equals the slope of the straight line obtained by plotting the continuous oxygen measurements against a measuring period of 15 minutes.

The background oxygen consumption (zero) is obtained by performing the test with a PVC pipe before each one of the measuring series. Under this condition, no oxygen consumption should occur. If there is significant oxygen consumption, its cause must be identified and eliminated. In case that small decreases in dissolved oxygen are apparent, which may result from a comprehensible increase of the water temperature during measurements, the measurements for corrosion rates estimation must be corrected correspondingly.

### **3.3.4 Iron release rates**

Concomitantly to recording the corrosion rate in the circulating operation, parallel measurements of the iron release rates are carried out. During the 15-minute monitoring period, samples are taken at intervals of 2 minutes in the first 10 minutes, followed by intervals of 5 minutes.

The sample volume taken for the determination of the iron concentration is replaced by fresh water. A correction of the results is not necessary, as the operating volume of the rig is much larger than the sampled volume.

The iron content of the samples is determined with the Inductively Plasma Mass Spectrometry (ICP) and plotted against the measuring time. From the slope of the obtained straight line, the iron release rate can be estimated on a sqm basis, which can then be calculated for a 24-hour period.

## **3.4 Test rig used by SINTEF**

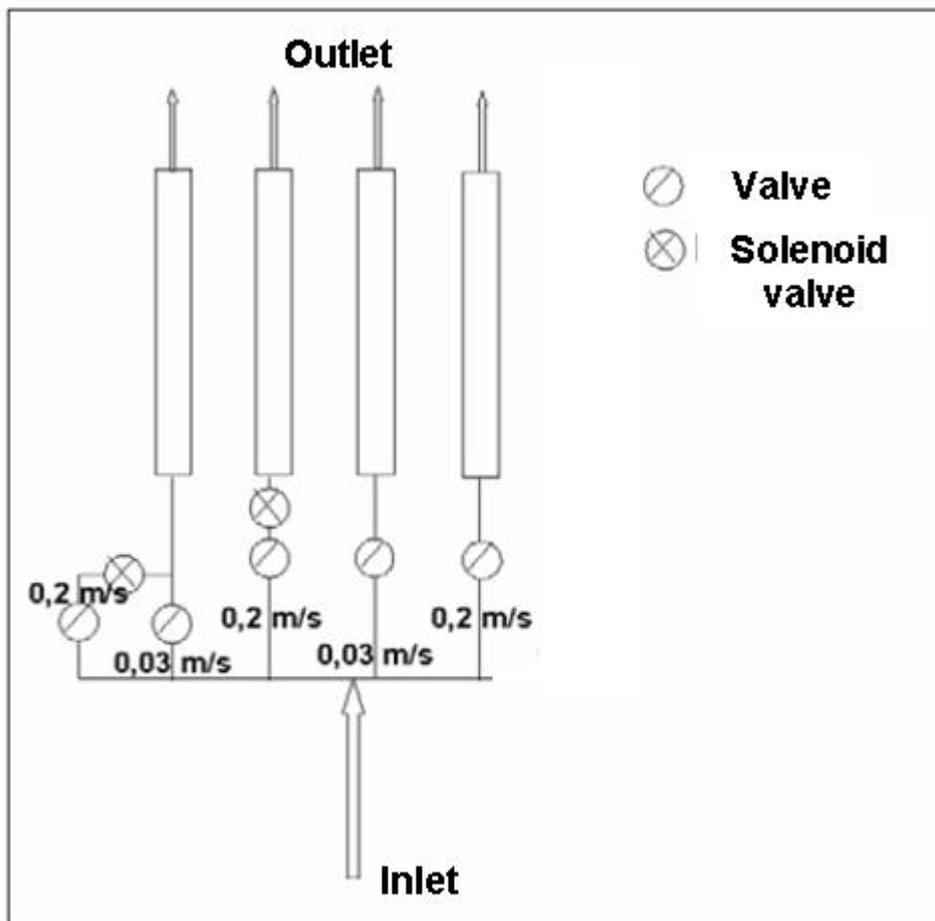
### **3.4.1 Introduction**

The corrosion test rig described below is used for corrosion research in drinking water in several projects. In these projects the focus has been on corrosion rate (with reference to pipe life-time) and protection by corrosion products. However, by a minor change in focus and some additional calculations, the procedure should be well suited for estimating the potential for particles generation due to corrosion at different conditions.

### **3.4.2 Experimental set-up**

The corrosion experiments are conducted in a test rig consisting of four parallel lines with different flow conditions as follows; 1) constant high flow

rate (0.2 m/s), 2) constant low flow rate (0.03 m/s), 3) alternating high flow rate (0.2 m/s for 13 hrs) and complete stagnation (for 11 hrs), and 4) alternating high flow rate (0.2 m/s for 13 hrs) and low flow rate (0.03 m/s for 11 hrs). The flow rate and alternating flows are controlled by adjustment valves, solenoid valves and timers. A schematic sketch of the set-up is shown in figure 6.



**Figure 6:** Schematic sketch of the test rig used by SINTEF for corrosion measurements.

In each line 12 steel coupons (St 37) are mounted and separated by a material which is an electrical insulator (Bakelite). The size of the coupons is 15 mm times 20 mm with a thickness of 1 mm. Both sides of the coupons are exposed to the water. A close up photo of a line with the coupons and insulating Bakelite is shown in figure 7.



**Figure 7:** Close up photo of a line with the steel coupons and insulating Bakelite.

Before the coupons are inserted to the rig they are degreased (using methanol or ethanol), rinsed (in distilled water), dried (at 105 °C for 2 hrs) and weighted (the weight is called  $W_1$ ). Coupons are extracted from the coupon rack at various exposure times. Normally three coupons are extracted from each line at each time, i.e. three coupons from each line every third month for a total of 12 months. The coupons are extracted from the top of the coupon rack without disturbing the remaining coupons.

### 3.4.3 Measurements

#### Metal loss and corrosion product accumulation

The extracted coupons are then dried (at 105 °C for 2 hrs) and weighted (called W<sub>2</sub>). Thereafter, the corrosion products are removed by using a stirred concentrated HCl-bath for 2 – 5 minutes, followed by brushing of the coupon, and another 2 – 5 minutes in the stirred concentrated HCl-bath. After the HCl-bath the coupon is thoroughly flushed and rinsed (in distilled water) before it is dried (at 105 °C for 2 hrs) and weighted (called W<sub>3</sub>). The exposure time is recorded and the weight loss of metal and the weight of accumulated corrosion products on the coupon are determined by:

$$\text{Weight loss of metal:} \quad W_1 - W_3$$

$$\text{Weight accumulation of corrosion products:} \quad W_2 - W_3$$

The same procedure is conducted for all the extracted coupons throughout the experimental period.

#### Corrosion products characterization

The corrosion products are characterized by Scanning Electron Microscopy (SEM) and X-ray using Electron Probe Micro Analyzer (EPMA). The samples are prepared from separate coupons in a two stage moulding using epoxy. A cross-section of the moulded coupon are then cut and polished for analysis.

The SEM is analyzed in back scatter mode for information about the nature of the corrosion attack, the thickness and structure of the corrosion product layer, etc, in addition to distinguishing between the different corrosion product phases. The results are documented by photos. The same cross-section area is analyzed by EPMA in order for semi-quantification of the elements occurring in various parts of the corrosion product layer. The results are documented by colour-photos using dot-mapping. Based on the results, point analyses for quantitative determination of the elements are conducted on selected areas.

The surface analyses data is used to determine the different compounds occurring in the corrosion product layer, evaluate their protective properties, etc. In addition, chemical wet analysis of the corrosion products may also be conducted to supplement the measurements.

Additional light microscopy may be conducted on the metal surface after the corrosion products are removed from the coupons in order to better characterize the corrosion attack, i.e. determine the area corroded, the number of corrosion pits and their shape and depth, etc.

### 3.4.4 Data evaluation

Iron pipes which have been in service for many years have developed corrosion product layers which have reach some kind of a steady state. A challenge when conducting iron corrosion experiments like coupon tests with a limited experimental duration time (even at after an experimental period of

6 to 12 months) is to reach a similar steady state conditions during the experimental conditions as occurring in the network. This can be illustrated by dividing the corrosion progress into three phases: 1) the initial phase, 2) the build-up phase, and 3) the steady state phase. During the initial phase the metal surface is relatively unprotected and consequently the corrosion rate is high. The duration of the initial phase is normally short. In the build-up phase, the corrosion rate declines with time as the corrosion product layer develop. The duration of this period will depend on the corrosion conditions (water quality and flow conditions), but it can be relatively long. Often corrosion rig experiments are conducted entirely within this phase. The steady state phase represents the relative constant long term corrosion rate occurring in the network. Normally it is this corrosion rate which is of main interest when conducting corrosion experiments.

In order to improve the evaluation of weight loss data (since all or most of the experimental data are from the build-up phase, while we primary are interested in the steady state phase), it is suggested to describe the build-up phase as follows [Broo *et al.*, 2006]:

It is assumed that the corrosion occurs at a rate,  $r$ , at the available non-blocked surface area,  $a$ :

$$r = k_1 \cdot a$$

where  $k_1$  is the rate constant. As the corrosion proceeds, and corrosion products develops, the remaining surface area available for corrosion is reduced with time. The time dependant rate of corrosion can then be written as:

$$r = k_1 \cdot a_0 \cdot (1 - k_2)^t$$

where  $a_0$  is the total surface area,  $t$  is the exposure time, and  $k_2$  is a constant describing the surface coverage. When this is integrated from time 0 to  $t$ , the weight loss,  $\Delta m$ , during the exposure time can be written as:

$$\Delta m = k_1 \cdot a_0 \cdot \left( \frac{(1 - k_2)^t - 1}{\ln(1 - k_2)} \right)$$

$k_1$  (rate of corrosion of available non-blocked surface) and  $k_2$  (coverage of blocked surface) can then be determined from the experimental data. A low  $k_1$  and a high  $k_2$  would give a low long term corrosion rate. These values would obviously depend on the water quality and the flow conditions. This model for data interpretation could probably be improved by also taking into account factors like pitting/non-uniform corrosion, the degree of blockage, etc

The corrosion data from the test rig, in combination with the data evaluation described above, and with additional characterization of the corrosion products as described, can then be used to estimate the potential for particles formation due to corrosion under various water qualities and flow conditions. It will, however, not consider under which conditions the particles (corrosion products) will be released.

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# 4 Microbiological problems associated with sediment composition and behaviour

## 4.1 Origins of microorganisms in distribution networks

Bacteria (both attached to particles or as planctonic cells) can be introduced into the distribution network through the water treatment plant or as a result of intrusion through leaks in pipes water distribution network and vents on storage reservoirs. Transient pressure events occur in distribution systems; that these events can result in negative pressures; that negative pressures provide a potential portal for entry of non-potable water into potable water distribution pipelines; and that fecal indicators and cultivable human viruses are present in the soil and groundwater exterior to the distribution system. Along with harmless microbes such as *Pseudomonas fluorescens* and *Aquabacterium commune* the potentially pathogenic (e.g. *Legionella* spp., *Mycobacterium* sp.), true pathogenic bacteria (e.g. *Shigella* spp, *Salmonella* sp., *E.coli*), protozoa (*Cryptosporidium* spp., *Giardia* spp.), viruses and small animals may enter distribution pipes.

Once organisms become established in the pipe environment, they will eventually form a biofilm (“microbial cells, attached to a substratum, and immobilised in an organic polymer matrix of microbial origin”). A biofilm is a collection of organic and inorganic, living and dead material collected on surfaces of pipes, suspended particles and deposits. It may be a complete film or, more commonly in water systems, small patches on pipe surfaces.

## 4.2 Influence of water quality on formation of biological deposits

Whilst it is suggested that many parameters govern the survival and growth of microorganisms in water systems (Figure 4.1), the most important variables to consider are particularly the water hydraulics, flux of cells, nutrients, characteristics of substratum, temperature and disinfectants (Percival *et al.*, 2000).

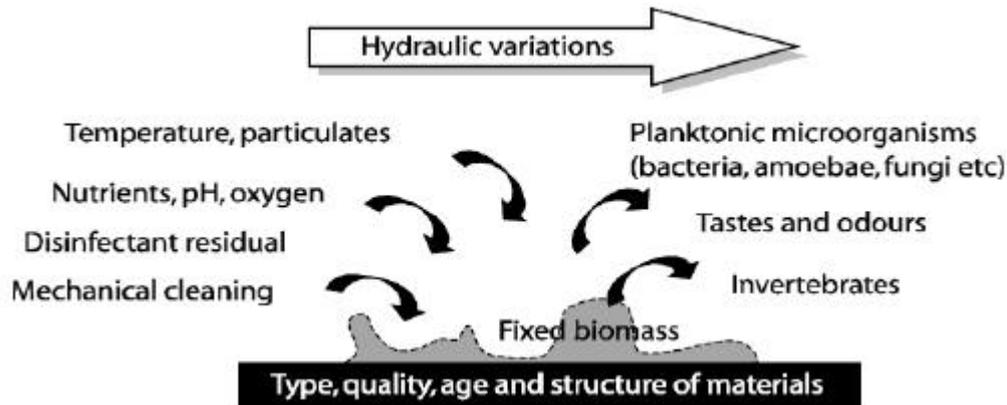


Fig. 4.1 The factors influencing microbial changes in water distribution networks (Ainsworth, 2004)

#### 4.2.1 Water hydraulics

Water flow rate effects accumulation of loose deposits in the system. In most distribution networks the water flow is slow enough to allow particles settle which promotes formation of loose deposits.

Effect of water velocity on biofilm is not straight forward. In general it is considered that higher water velocities cause a greater flux of nutrients to a pipe surface. Since mass transfer of nutrients is in major role in the growth of biofilms (Lehtola *et al.*, 2006) higher rate may promote growth of bacteria. On other hand it results in greater shearing of biofilm. Rapid changes in water flow rate resuspend biofilms and sediments which increased the concentrations of bacteria in water. The disturbance caused by the changing water flow also increases the particle counts and water turbidity recorded with online instrumentation. It was also shown that water velocities had also effect on nature and physiological activity of the some on biofilm. The manganese oxidising biofilm in distribution networks biofilm developed at a velocity of 0.5 m/s actively oxidized and deposited manganese, but at 0.01 m/s) no manganese was deposited (Sly *et al.*, 1988).

#### 4.2.2 Nutrient availability

The availability of nutrients is a necessity if bacteria are to grow in potable water. The general requirements for heterotrophic bacteria include carbon, nitrogen, and phosphorus in a ration of about 100:10:1. This ratio is necessary for the maintenance of effective growth. Trace elements and cofactors, necessary chemicals for enzymes to function, are also required as well as these principal nutrients. The phosphorus limitation has been observed in several countries (e.g. Miettinen *et al.* 1997). It is often the case with drinking water produced from humic water using chemical coagulation methods. It is because phosphorus is much more effectively removed with in coagulation step than carbon, rendering the former to be in deficit for bacterial nutrient requirements. In most of the water supply systems examined, organic carbon is limiting factor for bacterial growth.

Only fractions of dissolved organic carbon (DOC) in water can be used by bacteria. Biodegradable dissolved organic carbon (BDOC) measures the DOC concentration before and after incubation of a water sample with indigenous microorganisms. The analysis can be performed using microorganisms suspended in water, using sand containing microorganisms acclimatised to the water sample investigated or in biofilm reactors (Servais *et al.*, 1987). BDOC is calculated as the difference between the initial concentration and the minimum concentration when DOC has reached a plateau during the incubation. The sensitivity of the BDOC analysis is dependent on the TOC analyser for detection of differences in DOC concentrations that are very similar, and has been suggested to be approximately 0.1 mg/l, although lower concentrations have been reliably measured.

Assimilable organic carbon (AOC) utilises the degradative properties of certain strains of bacteria for organic carbon substrates typically found in drinking water (van der Kooij 1992). The original method uses the growth of *Pseudomonas fluorescens* strain P17, which has a high metabolic versatility. To also include organic molecules formed during ozonation, e.g. formate and oxalate, in the analysis, the *Spirillum strain* NOX could be additionally used. The water samples are pasteurized, the test strains are added and their growth are followed by spread plates on nutrient poor agar. AOC is calculated from the maximum bacterial concentrations of the test strains, and is converted to acetate carbon equivalents by obtained bacterial yield factors. The sensitivity of the AOC analysis is considerably higher than the BDOC analysis, and could measure concentrations below 5 µg acetate-C/l.

The methods of AOC and BDOC are not directly comparable. AOC has been shown to be of low molecular weight, less than 1000 g/mol, and would represent the easily biodegradable fractions of NOM, which would include monomeric molecules such as free amino acids, sugars and fatty acids. BDOC on the other hand measures all organic carbon that could be biologically oxidised. The majority of BOM is polymeric and requires exoenzymatic hydrolysis to monomers for bacterial utilization. BDOC has consequently been defined as the part of DOC susceptible for exoenzymatic hydrolysis and may also include humic substances. In carbon concentrations, AOC values are considerably lower than BDOC values and rather poor correlations between AOC and BDOC concentrations have been observed when comparing different water sources and the effects of different treatment steps.

### **4.2.3 Pipe material**

Most of the pipes used in drinking water distribution systems are made of plastic (PVC, PE, etc.) or metal (copper, cast iron) which can become highly corroded (Batte *et al.*, 2004). Whatever they are made of, pipes become extensively colonized by microorganisms. However, the nature of the pipe material plays a major role in the selection of biomass and its organisation. Through their roughness, wettability, adhesive properties, etc., materials affect the adhesion efficiency of pioneers and may act as a source of nutrients

or growth factors. For example, analysis based on the detection of cells by the oligonucleotide has shown that biofilm can be made up of 4 or 26%  $\alpha$ -proteobacteria, depending on whether it was formed on glass or on a polyethylene material. In another study, polyethylene, PVC, steel and copper materials were found to be covered with almost equal amounts of biomass, but there was much less bacterial activity on copper due to the toxicity of the released ions. The concern of using soft plastic materials for pipes is related to releases of organics which can be used by bacteria and accelerate biofilm formation. Corrosion of iron pipes leads to accumulation of a mixture of iron oxides which can effectively absorb bacteria on their surfaces. As shown in Figure 4.2, biomass production is very considerable in highly corroded pipes compared to slightly corroded ones. How iron oxy-hydroxides favour bacterial growth is not well known. It has been shown that they increase the growth of aerobic gram-negative bacteria like *Pseudomonas* sp., probably because they were used as a nutrient. In addition, the use of iron as a terminal electron acceptor in a respiratory chain (anaerobic respiration) is also well-known in several bacterial strains.

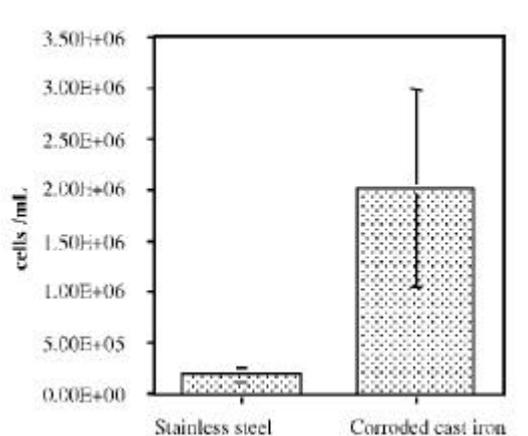


Fig. 4.2. Bacterial production in number of cells per milliliter of water (difference between entry into and exit from the reactor) in a Propella™ reactor fed continuously with drinking water (Hydraulic residence time of 24 h, temperature of 25 °C) (Batte *et al.*, 2004)

Iron corrosion products have also been reported to support the growth of *E. coli*. Complementary assays showed that the culturability of *E. coli* (CFU) in drinking water can be maintained for 550 h in the presence of iron oxy-hydroxide, compared to the results of an assay conducted in the absence of iron oxy-hydroxide, in anaerobic conditions. Thus, it is clear that the control of corrosion by decreasing water aggressivity or by anti-corrosion treatments is needed to limit the excessive growth of biomass in the network and to alleviate the coliform occurrence in water.

#### 4.2.4 Effect of chlorine

Chlorine, chloramine, chlorine dioxide, and chlorine compounds generally, have been used successfully for many years to disinfect drinking water.

Chlorine is able to limit bacterial regrowth both by injuring bacteria, thereby preventing their growth, and by limiting the production of bacteria in the system, either in the planktonic phase or in the biofilm (inter alia Batte *et al.*, 2004). However, the chlorine residual resulting from such a treatment, which is usually less than 1 mg Cl<sub>2</sub>/l, is insufficient to kill and remove all the attached biomass. Even large doses and drastic treatments are quite unable to eradicate biofilms. Several mechanisms may explain such a resistance (see Batte *et al.*, 2004), but the end result is that most of the biofilm remains alive. Another limitation is that chlorination of unchlorinated network determines a release of adsorbed organic matter (10 µg DOC/cm<sup>2</sup>) to which the bacterial cells contribute about 1%. It takes 1–2 months of continuous chlorination to release all the desorbable biofilm-entrapped organic matter (with a chlorination dosage of approximately 3.5 mg Cl<sub>2</sub>/l and a resulting chlorine residual of approximately 0.1 mg Cl<sub>2</sub>/l).

### 4.3 Biological composition of deposits

Loose deposits are found in many water distribution systems both in pipes and reservoirs. A study about loose deposits in water distribution systems of France (Gauthier *et al.*, 1999) showed that during one year deposit thickness may reach few millimeters. Loose deposits consist of mineral particles, organic matter and dead or alive biomass (Fig. 4.3). Organic matter contains about 2-10% of deposits dry weight, while microbial biomass represent 1-12% of organic matter.

A wide range of cultivable bacteria densities have been found in loose deposits, most of the recorded measurements being in the  $2 \cdot 10^6$  to  $2 \cdot 10^9$  CFU/g range. Total cell counts (epifluorescence microscopy after acridine orange staining) were found to be in the range of  $10^{10}$ – $10^{11}$  cells/g of deposit. Carriere *et al.*, (2002) (inter alia Batte *et al.*, 2004) measured deposit densities varying from 0.01 to 40 g/m<sup>2</sup> in three Canadian distribution systems, and found that it was not unusual for the results obtained from two pipe sections in the same distribution system to differ by a factor of 10. When computing the total number of deposit-associated cultivable bacteria from such quantities and comparing this number to the total number of biofilm bacteria (Gauthier 1998), it appears that the majority of cultivable bacteria may be located in the deposits rather than in the biofilm for deposit quantities >1 g/m<sup>2</sup>.

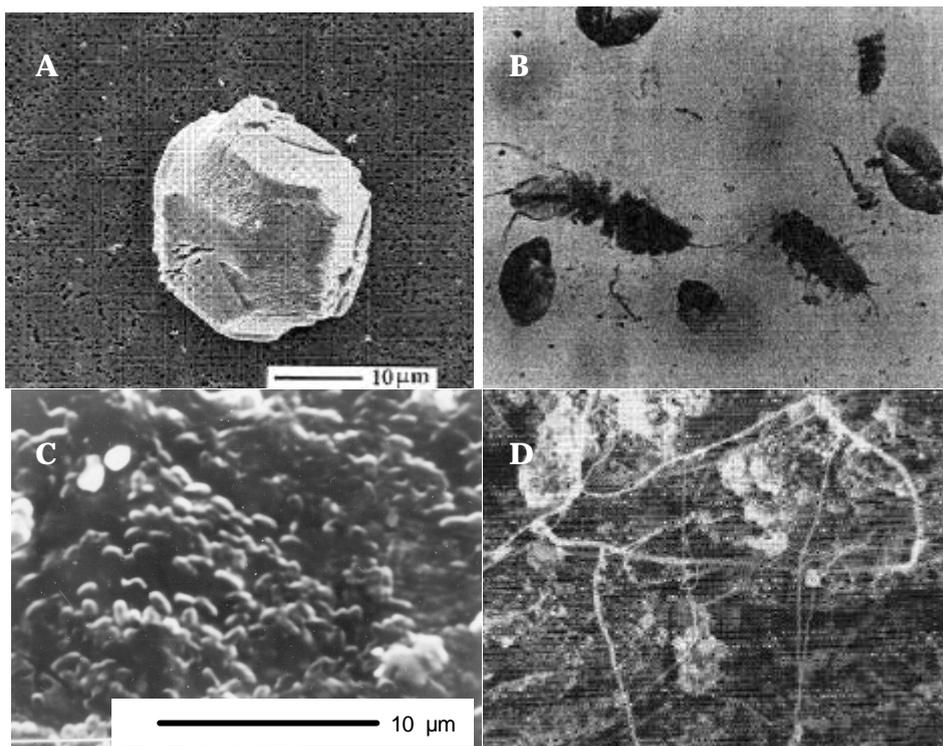


Fig. 4.3 Examples of particulate matter in drinking water networks (A) undissolved  $\text{CaCO}_3$  particle after 5 hours of water travel time in distribution (Alere and Hanaeus, 1997) (B) macroorganisms such as Asellus and Gasteropoda in biomass from reservoir (Gauthier *et al.*, 1999), (C) bacterial biofilm colonizing surface of PVC pipe, (D) actinomycetes colonizing pipe sediment (Geldreich, 1996).

When more than  $10 \text{ g/m}^2$  of deposits have accumulated, the bacteria associated to these deposits represent  $>80\%$  of the total bacteria in the pipe (Figure 4.4). Such deposit concentrations are not uncommon, especially in distribution systems which are rarely cleaned by flushing or mechanical means, or which are prone to deposit accumulation.

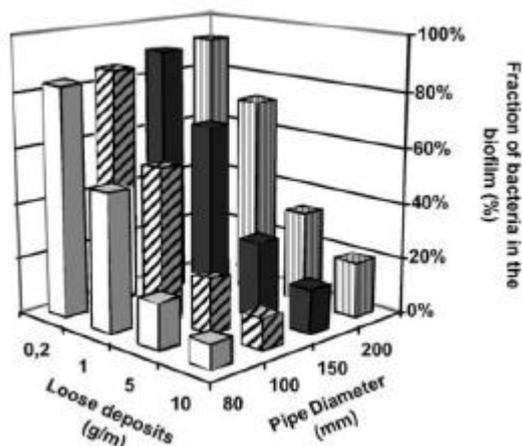


Figure 4.4. Biofilm fraction of cultivable bacteria in a 1-m pipe section – compared to the total number (biofilm + deposits) – as a function of deposit amount and pipe diameter (hypotheses: average concentration of cultivable bacteria in biofilms:  $10^5 \text{ CFU/cm}^2$ ; average concentration of cultivable bacteria in deposits:  $2.5 \cdot 10^8 \text{ CFU/g}$  (Gauthier *et al.*, 1996)).

Within bulk of loose deposits the environment is favorable for survival of pathogens and survival of opportunistic pathogens such as mycobacteria and legionella. Invertebrates also may also survive (and some may even reproduce) within the supply networks by deriving their food from microorganisms and organic matter present in deposits on pipe and tank surfaces (see Figure 4.1).

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## 5 Influence of mobilisation on microbiological water quality

Deterioration of water quality in distribution networks has become one of the major challenges in developed countries (Ford, 2006). The water quality changes during water transport in the pipes may produce esthetic (e.g. “red water” events, changes in taste, odor and color of the water, staining of laundry and appliances and presence of invertebrates in drinking water) and health related problems.

Historically, the provision of piped water directly to the household has been associated with improved hygiene and reduction in disease. However, as standards of living risen water infrastructure has aged, there has been growing recognition that water distribution systems are vulnerable to intrusion and contamination and may contribute to endemic and epidemic waterborne disease (Moe and Rheingans, 2006). Due to improved water treatment practices the proportion of waterborne diseases outbreaks associated with distribution systems is increasing (Fig. 5.1) .

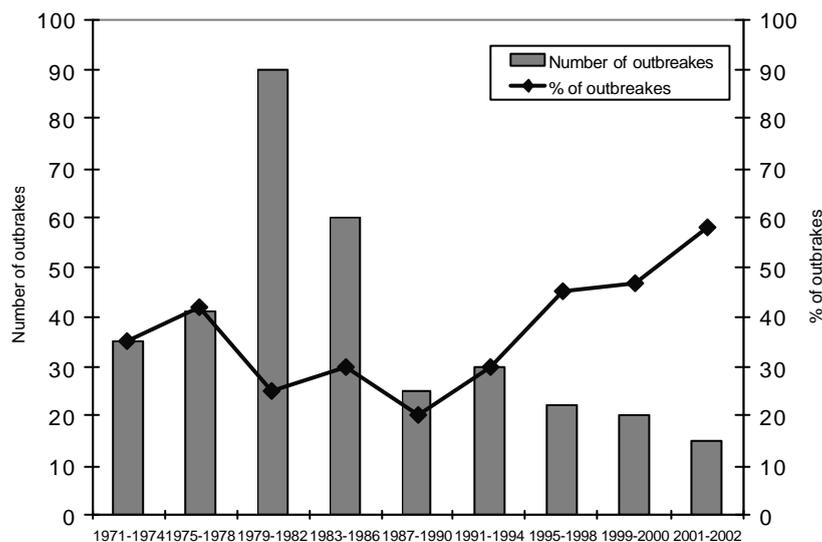


Figure 5.1. Reported waterborne disease outbreaks in the United States in community associated with distribution system deficiency (Moe and Rheingans, 2006)

Entrance of suspended particles and bacterial growth in distribution networks increase the risk of outbreaks. Suspended particles can carry bacteria fixed on their surface, which prevent them from disinfection and to the formation of loose deposits in reservoirs and pipes. Within the deposits bacteria may grow while pathogens which enter water distribution networks in small amounts, are concentrated. The loose deposits may be readily transferred into tap water if re-suspended during hydraulic changes in the distribution system. Such a re-suspension may affect routine coliform

monitoring results, heterotrophic counts, and increase spreading of pathogens.

## **5.1 Effect of biofilm growth**

Biofilms growing on surfaces of pipes or in loose deposits can be responsible for loss of distribution system disinfectant residuals, increased bacterial levels, reduction of dissolved oxygen, taste and odor changes, red or black water problems due to iron or sulfate-reducing bacteria, microbial-influenced corrosion, hydraulic roughness, and reduced materials life.

### **5.1.1 Increase of organism number**

#### *5.1.1.1 Coliforms*

A primary reason that many water utilities become concerned with biofilms in drinking water systems is due to growth of coliform bacteria in the pipe network. In 1993 in the United States alone, nearly 4,400 water systems affecting 21 million people violated drinking water standards for total coliform bacteria. Similar trends were noted for 1994 and 1995, with over 12,000 systems exceeding accepted coliform levels. Of concern are the nearly 2,000 systems every quarter that are significant noncompliers and repeatedly detect coliform bacteria in finished drinking water. Although some of these systems experience coliform occurrences due to cross connections and other operational defects, a large proportion of the systems can trace their problems to regrowth of the bacteria in distribution system biofilms. Occurrence of pathogens is linked to biostability and biofilm

#### *5.1.1.2 Heterotrophic plate counts (HPC)*

Heterotrophic microorganisms are indigenous to water (and biofilm) and always present in greater concentration than coliform bacteria in distribution and storage systems (Payment and Roberson, 2004). An increase in HPC indicates treatment breakthrough, post-treatment contamination, growth within the water conveyed by the distribution system or presence of deposits and biofilm in the system. A sudden increase in HPCs above historical baseline value should trigger actions to investigate and, if necessary to, remediate the situation. There is no evidence that heterotrophic microorganisms in distribution systems are responsible for public health effects in general population through ingestion of drinking water.

Some countries use a nonmandatory maximum HPC of 500 cfu/ml at 35°C because concentration greater than this interferes with the recovery of coliform bacteria by membrane filtration techniques based on lactose fermentation. However, newer coliform detection methods based on the metabolism of chromogenic substances are not prone to this interference.

### 5.1.1.3 Opportunistic bacteria growth

Most microorganisms developing within the distribution network are harmless (Ainsworth, 2004). Exceptions include *Legionella* and *Mycobacterium avium* complex, which are discussed below. There are no reports of public health problems arising from ingestion of opportunistic pathogens (e.g. *Aeromonas* and *Pseudomonas*) found in drinking-water biofilms. *Pseudomonas* and *Aeromonas* strains present in water usually do not have the same genetic pattern as those found in clinical cases during gastrointestinal infections. Although these organisms have not been implicated in waterborne outbreaks, *Pseudomonas* has been identified as the cause of several skin infections associated with swimming pools, hot tubs and other spa facilities.

*Legionella* and the *M. avium* complex merit special attention. *Legionella* in a piped distribution system can grow to significant numbers in warm waters and can colonise water heaters, hot tubs, hot-water lines and shower heads. The organism is also associated with cooling towers or evaporative condensers.

Special precautions need to be taken to prevent or control *Legionella* in environments such as hospitals and health care facilities, because aerosols generated by showers or spas can be a route of infection, and contamination with *Legionella* can be a significant source of nosocomial (hospital-acquired) infections.

The *M. avium* complex is a group of bacteria that are opportunistic pathogens in man, producing symptoms similar to *M. tuberculosis*. They are ubiquitous in soil, food and water, have been found in biofilms and are quite resistant to disinfection. Strains of these microorganisms that are found in the environment have been shown to cause disease in immunocompromised patients.

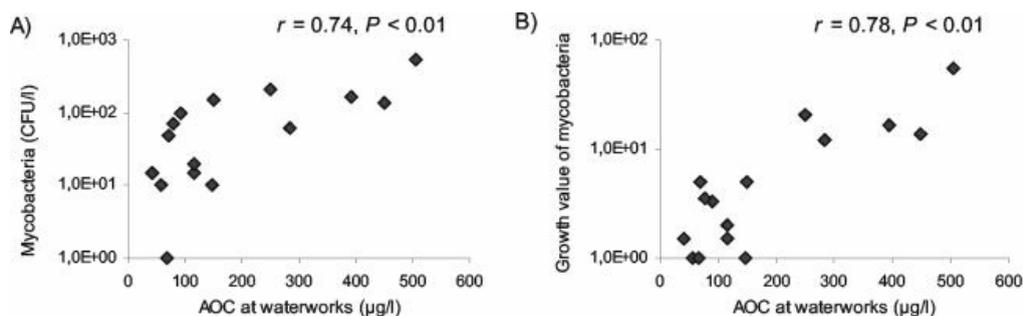


Fig. 5.2. Association between number (A) and growth value (B) of mycobacteria and AOC concentration in water.

Recent study in Finland (Torvinen *et al.*, 2006) found high numbers of mycobacteria in loose deposits of water distribution systems ( $1.8 \times 10^5$  to  $3.9 \times 10^5$  CFU/g) and has shown that the number and growth of mycobacteria in water correlate strongly with the concentration of AOC in water (Fig. 5.2). Thus, these bacteria are growing in water distribution networks.

## 5.2 Survival of pathogens in deposits

Biofilms, sediments and corrosion products may harbour pathogenic microorganisms introduced through inefficient treatment or breaches of the integrity of the distribution system. Buried in the sediments or embedded in the biofilm, pathogens could be released during repairs and cleaning operations, or by erosion caused by sudden changes in flow patterns. Survival depends on their nature and the microbial activity in the biofilm. Only a few pathogenic bacterial species may multiply if favourable conditions, such as warm water and appropriate nutrients, are present (Ainsworth, 2004).

Viruses and protozoan parasites are obligate parasites and they need a human or animal host to multiply. If they enter the pipe network, they can only survive for a limited period; the infective dose for human hosts is likely to be reached only if large accumulations occur within system deposits. Such accumulations may occur as a result of cross-connections, backflow or contamination. Although there are currently no reports of health effects directly attributed to the long-term survival of pathogens within a distribution system, such organisms have been shown to persist within biofilms, thereby presenting a potential underlying health concern to consumers. Biofilms contain many sorption sites that can bind and accumulate organic and inorganic contaminants, as well as particulate and colloidal matter. Within biofilms, microbial pathogens can be protected from biological, physical, chemical and environmental stresses, including predation, desiccation and changes or fluxes in the environment. Bacterial pathogens such as *Helicobacter pylori*, enterotoxigenic *E. coli*, *Salmonella typhimurium*, and *Campylobacter* species can persist within biofilms formed in experimental laboratory systems. The potential therefore exists for such pathogens to accumulate and persist within a municipal distribution system, although so far they have not been isolated directly from such systems. Model enteric viruses (B40-8 and MS2 bacteriophages) have also been shown to accumulate and persist within biofilms formed in the laboratory, although again these organisms have not been isolated directly from a municipal water distribution system. The interaction of viruses with pipe biofilms has been neglected or ignored in the past; however, recent research has demonstrated its potential significance. Problems may therefore arise in distribution pipe systems when clusters of biofilm-associated pathogens become detached from either substrata or biofilm matrices by physical, chemical or biological processes. Detached biomass could compromise the microbiological quality of distribution waters by providing a continual contamination of the bulk water through the release of sorbed pathogens and indicators. These mobilized pathogens, which may exist at concentrations greater than an infective dose, have the potential to reach consumers through the ingestion of contaminated water or food contaminated with such water, inhalation of aerosols or breaks in the skin. For example, in a risk model that has been developed for the distribution of recycled water there is evidence to suggest that, even during normal operating conditions (1 virus per 100 l of recycled water), enteric viruses may accumulate within distribution pipe biofilms in

sufficient numbers to present a risk to consumers should a biofilm slough off from the pipe. These studies support the view that preventing the accumulation of deposits and biofilms in a distribution system should be an important component of a water safety plan.

### **5.3 Presence of animals**

As a result of biofilm development the food chain is extending leading to growth of small animals in drinking water distribution networks. The presence of animals has largely been regarded by water suppliers in temperate countries as an “aesthetic” problem. The larger animals may be visible to the consumer and may be objectionable if they appear at the tap.

However, some animals such as free living amoebae e.g. *Acanthamoeba* and *Naegleria* can also harbour bacterial pathogens such as *Legionella* and mycobacteria, and may play a role in the survival of these organisms in drinking-water environments and in their pathogenesis.

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# 6 Methods used for the estimation of sediment formation and risk assessment

## 6.1 The Resuspension Potential Method

The Resuspension Potential Method (RPM) as developed in the Netherlands is based on measuring the mobility of the material in a network (Vreeburg *et al.*, 2004).

The RPM consists of a controlled and reproducible increase in the velocity within a pipe. An increase of 0.35m/s, in addition to the actual velocity, was determined empirically (Vreeburg *et al.*, 2004). The hydraulic shear stress as a result of the increased velocity causes particles to mobilise, affecting the turbidity of the water. The method is mainly applied in 100-150 mm pipes, hence the absolute difference in shear stress caused by the uniform velocity increase is not very large. The turbidity effect is monitored and translated into a ranking of the discolouration risk. The method is applied as follows:

- Isolate the pipe for which the discolouration risk is to be assessed, as for uni-directional flushing (Antoun *et al.*, 1999). The isolated length should be at least 315 meters long to be sure that only this single pipe is affected.
- Open a fire hydrant such that the velocity in the pipe is increased by the additional 0.35 m/s above the normal velocity and maintain that rate for fifteen minutes; after this reduce the flow to normal (total length affected is thus 315 m).
- Monitor turbidity in the pipe throughout the fifteen minutes of extra velocity and beyond that until turbidity returns to the initial level.

Schematically this procedure is schematically illustrated in Fig 6-1.

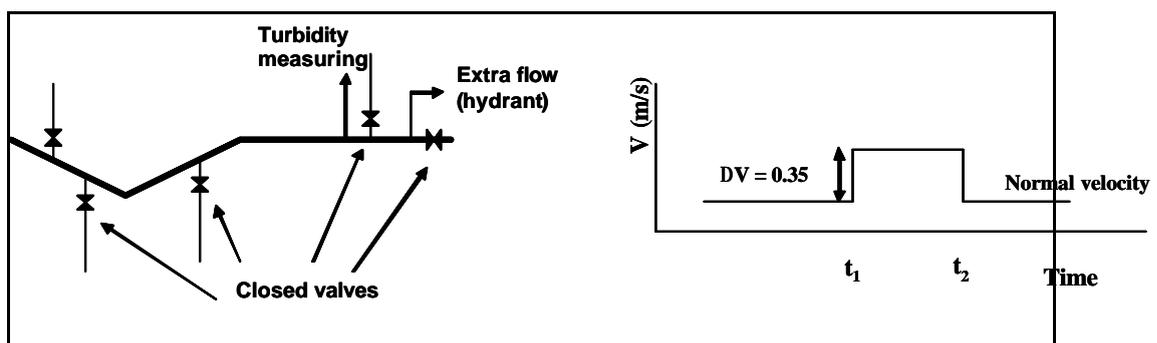


Fig 6-1 Principle Resuspension Potential Method: Increase velocity in isolated pipe

The result obtained from an RPM test is the turbidity response of a pipe. A typical example is shown in Fig 6-2 highlighting the four regions of the trace that are utilised to rank discolouration risk:

- Base turbidity level
- Initial increase in turbidity at the start of the hydraulic disturbance
- Development of turbidity during the hydraulic disturbance
- Resettling time and pattern to base (initial) turbidity level

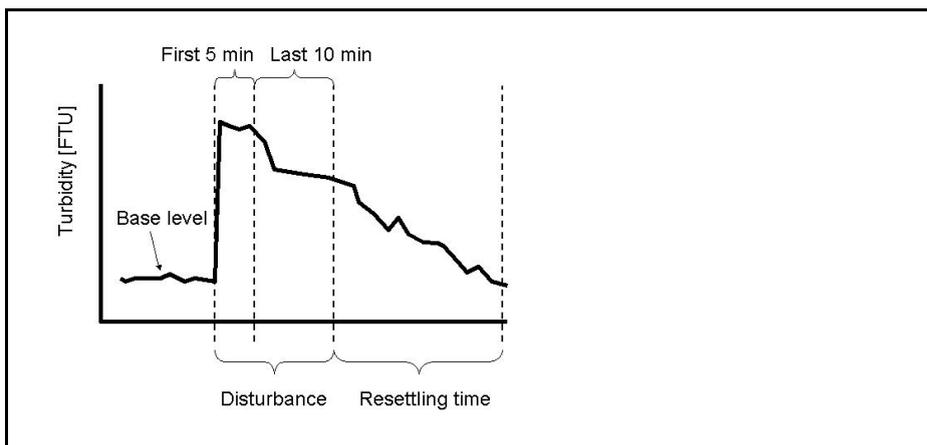
#### *Base turbidity level*

The base turbidity level is the level preceding the hydraulic disturbance. Base line turbidity can be linked to the turbidity of the source water and can give some insight into the source of the sediment. Baseline turbidity is also needed to judge the time required for the turbidity to resettle after the increased velocity has stopped.

#### *Initial increase in turbidity*

Following the actual disturbance the turbidity will rise immediately to a certain level. This initial increase indicates the instantaneous mobility of the sediment, resulting in peak turbidity. A loose layer that is immediately available causes the initial turbidity.

The initial increase is an indication of the maximum turbidity that can be caused by a hydraulic incident. High initial turbidity increases the chance that discolouration will be noticed.



*Fig 6-2 Typical RPM turbidity trace resulting from an RPM test, showing the four regions used to rate the discolouration risk.*

#### *Development of turbidity during the hydraulic disturbance*

The hydraulic disturbance is kept going for 15 minutes, allowing the turbidity to develop to a stable level. If the turbidity stays near the first five-minute level, then the amount of sediment is considerable and the composition of the sediment is homogeneous. A quickly reached level that remains fairly stable during 15 minutes indicates a high discolouration risk.

In many cases the turbidity drops during the 15-minute disturbance time.

Three phenomena can explain this:

- A relatively small amount of heavy sediment is present in the pipe. The extra forces in accelerating the flow also promote the initial whirling up of this heavy sediment. The significance of this sediment

is limited because it is settling even during the deviating hydraulic circumstances, making the discolouration risk smaller. The chance that the initial discoloured water can actually reach a tapping point in which it can be visually identified is small.

- A too-short length of isolated pipe. If the length is less than 315 meters, water is drawn from pipes upstream of the isolated pipe. This water originates from the looped network or from pipes with larger diameters and is less disturbed than the 0,35 m/s.
- A non-homogeneous deposit over the length of the tested pipe, for instance in a hilly area with concentrations of sediment in the depression of a pipe.

In all cases however the level of turbidity following the first peak determines the continued discolouration risk. This level will be present over a longer time, allowing customers more time to see it.

#### *Resettling to base level*

After closing the hydrant, it takes a certain amount of time for the turbidity to resettle again to base level. The time needed is important for the discolouration risk or, actually, the complaint risk. If the turbidity stays high during a longer period, the risk of noticing the turbidity in an application, such as filling a white basin (bathtub, washing bin, bathroom sink, etc) is greater.

#### **6.1.1 RPM and discolouration risk**

The RPM is interpreted on five aspect items that are rated equally:

- Absolute maximum value of turbidity during first five minutes of disturbance (20%)
- Average value of turbidity during first five minutes of disturbance (20%)
- Absolute maximum value of turbidity during last ten minutes of disturbance (20%)
- Average value of turbidity during last ten minutes of disturbance (20%)
- Time needed to resettle again to initial turbidity level. (20%)

For each aspect validation on a 0 to 3-point scale is made: 0 is the lowest or best rating and 3 the highest or worst rating. The lowest value equivalent with 'no discolouration risk' is thus 0 (zero) and the highest value or 'maximum discolouration risk' is 15.

For the rating per aspect a scale must be made that is calibrated to the turbidity equipment being used. Also, site-specific elements can be taken into account. If, for instance, the intuitive feeling of a network is that the discolouration risk is moderate, then the rating scale can be adjusted to this level. Changes in the discolouration risk that may occur when, for example, the treatment is improved or a cleaning program is started can be related to the objectified level. The discolouration risk established in this way is a relative figure that can be company-specific or even area specific. The effects of changed operation of the network can be assessed specifically and instantaneously. For every situation and type of measuring equipment a

ranking table can be made, depending on the type of turbidity-measuring equipment being used, local circumstances but also the goal of the measurements.

Table 6-1 and Table 6-2 give the values for discolouration risk for the Sigrist KT65 turbidimeter at a dedicated measuring point and the Dr Lange Ultraturb equipment at the flushing point, respectively.

*Table 6-1 Example of ranking RPM for discolouration risk using the Sigrist KT65 equipment at a dedicated measuring point*

	0	1	2	3
Absolute max first 5 min	<0,3 FTU	0,3-1,0 FTU	1,0-2,4 FTU	>2,4 FTU
Average first 5 min	<0,3 FTU	0,3-1,0 FTU	1,0-2,4 FTU	>2,4 FTU
Absolute max last 10 min	<0,3 FTU	0,3-1,0 FTU	1,0-2,4 FTU	>2,4 FTU
Average max last 10 min	<0,3 FTU	0,3-1,0 FTU	1,0-2,4 FTU	>2,4 FTU
Time to clear	< 5 min.	5-15 min	15-60 min	>60 min

*Table 6-2 Example of ranking RPM for discolouration using the Dr Lange Ultraturb equipment at the flushing point*

	0	1	2	3
Absolute max first 5 min	<3 FTU	3 – 10 FTU	10-40 FTU	>40 FTU
Average first 5 min	<3 FTU	3 – 10 FTU	10-40 FTU	>40 FTU
Absolute max last 10 min	<3 FTU	3 – 10 FTU	10-40 FTU	>40 FTU
Average max last 10 min	<3 FTU	3 – 10 FTU	10-40 FTU	>40 FTU
Time to clear	< 5 min.	5-15 min	15-60 min	>60 min

The ranking tables have a variety in the borders of the ranges that shows the flexibility of the methodology. Firstly, it allows for different turbidimeters and for different locations of measuring being used. The Sigrist KT65 measures with a white-light lamp and the Dr Lange Ultraturb with an infrared lamp resulting in different values for the same event. The first table uses a dedicated measuring point at the researched pipe itself, as is shown in Fig 6-1. The second table is meant for measuring at the flushing point, which is more practical, but gives higher values for turbidity (Slaats, 2002). Both measurement strategies can be tuned to each other in a calibration experiment using both methods simultaneously,

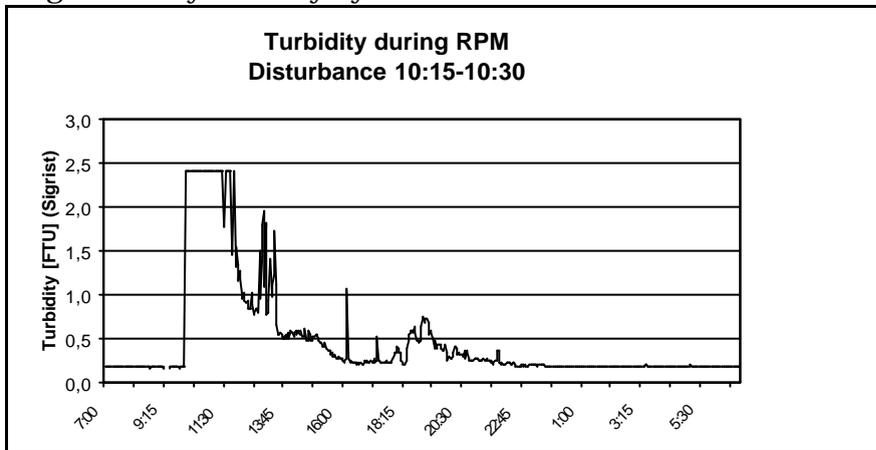
Secondly, the ranking can be tailored to the actual application. The first table is used to distinguish subtle differences in resuspension to see what the effects are of different cleaning methods. For that application it is important that the measurement be sensitive. The second table is used to prioritise the need for cleaning in a whole area that has a certain level of complaints. In that case the sensitivity should be more in the higher ranges to rank the areas that most urgently need cleaning.

With the ranking the sensitivity of the measurement can be tuned to the actual needs and equipment. This allows for a change in measuring

equipment or measuring locations without older data being lost. The ranking can also be adjusted to for instance changing standards or company policies. In the practical application of the RPM, the measuring of the resettling time is often limited to a maximum, for instance 30 minutes, or is ignored. If it is ignored the maximum ranking is set to 12 instead of 15.

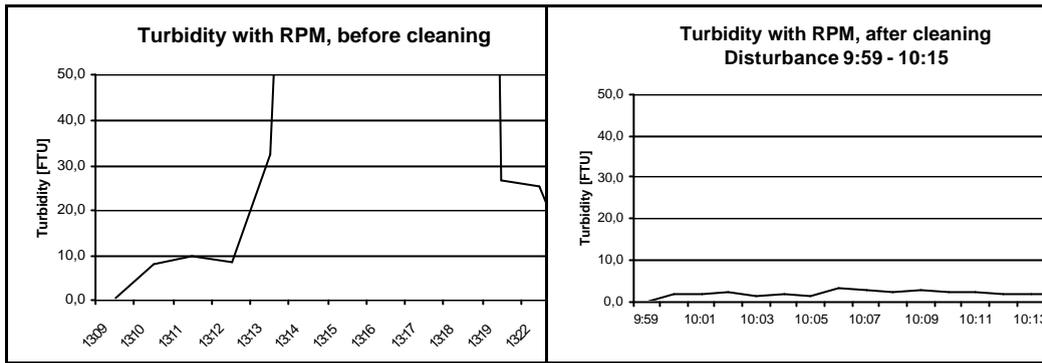
### 6.1.2 Typical RPM-curve

Fig 6-3 shows an example of an RPM with a high discolouration risk. The measurement has been made with Sigris-t-equipment, so Table 6-1 is applicable. The base turbidity is rather constant and low (0,22 FTU). The initial turbidity during the first five minutes as well as during the following 10 minutes is high and above 2,4 FTU, which is the maximum turbidity. The time to clear is several hours, meaning that on all the items the maximum score is reached, resulting in a maximum score of 15. Despite the fact that the base line turbidity is very constant and low, this location will experience a high turbidity with any hydraulic event.



*Fig 6-3 Example of RPM, measured with Sigris-t-equipment, indicating a high discolouration risk with a maximum score of 15.*

An example of the application of the RPM as an evaluating tool for a cleaning action is given in Fig 6-4.



*Fig 6-4 Results of RPM method applied pre- and post-cleaning to evaluate the effectiveness of the operations. The pre-cleaning RPM is 11 according to Table 6-2 and post-cleaning is 1; resettling time is ignored*

### 6.1.3 Flexibility of the RPM and alternatives

A full evaluation of the RPM requires a measuring time of at least a few hours, mostly consumed by determining the initial base level and pattern and the resettling time. It also requires turbidity-measuring equipment that can be costly and which requires skill to operate. Implementation of the principle of the RPM during the last decade has led to some alternatives in the analysis methods. The first adjustment is to limit the measuring time of the base level turbidity to some minutes prior to the disturbance or the time needed to isolate the pipe. Limiting the time used to measure the resettling time to a specific time, for instance 30 minutes is another adjustment. The examples shown in Fig 6-4 use this abbreviated version of the RPM. As seen the resettling time is not clear in the first graph. However the overall impression is that the discolouration risk is high (>12) and action is required. Limitation of the measuring time allows for more measurements in a working day. It is possible to do four measurements an 8-hour working day. Replacing the continuous monitoring by taking 5 to 8 grab samples during and after the disturbance limits the total measuring time even further. All the adjustments, however, cause the loss of information, specifically on the base level and the resettling time. For the assessment of the trigger level towards the discolouration risk, this is less important as the examples in Fig 6-4 show.

## 6.2 Cohesive transport model

As stated earlier (Boxall *et al.*, 2001) carried out theoretical analysis of the interaction of particles of the size found to predominate in discoloured water samples with respect to the hydraulic forces generated within distribution networks, concluding that forces and mechanisms above and beyond gravity settling forces must be in effect to inhibit particle movement. Rather than trying to identify or quantify specific contributing processes, they suggested a semi-empirical model that could be used to account for the effects of any such processes. The model they proposed was based on theory developed to

describe the erosion of estuarine mud (Parchure *et al.*, 1985) and as applied to in-sewer deposits by (Skipworth *et al.*, 1999).

The model is based on the concept that discolouration material is held in stable cohesive layers attached to the pipe walls of distribution systems and that these layers are conditioned by the usual daily hydraulic regime within the system. Within the model the material layers are described by a profile of discolouration potential versus layer strength, with an increase in potential corresponding to a decrease in strength. This strength, and hence layer state, is dictated by the shear stresses imposed by hydraulic conditions. Hence areas with low daily maximum hydraulic forces, such as dead-end pipes, redundant loops, over sized pipes, zone boundaries, extremities of loops etc. will have low strength characteristics and high discolouration potential, as has been noted in practice. The occurrence of disequilibria hydraulic conditions (burst, re-zoning, increased demand etc.) may expose the layers to shear stress in excess of their conditioned cohesive strength and lead to a mobilisation of the cohesive layers, resulting in a discolouration event.

It should be noted that neither the source of material nor the mechanisms and processes leading to accumulation and binding of particles are considered explicitly within the modelling approach. However through calibration of the empirical parameters describing layer strength characteristics, mobilisation and accumulation mechanisms a range of processes and materials may be simulated.

The model is used to predict turbidity as a result of hydraulic disturbance, and has been termed PODDS (Prediction Of Discolouration events in Distribution Systems). PODDS has been coded into EPANET (Rossman, 2000) and runs as a water quality element that utilises the EPANET hydraulic solution, substance tracking and transport algorithms. The incorporation of such a modelling approach into a calibrated hydraulic model allows the simulation of the discolouration risk (potential and impact) posed by different network areas and hydraulic scenarios. Once calibrated the model may be used to plan pro-active management strategies such as the flushing of systems to reduce the risk of discolouration events.

The model has been validated for data collected from flushing operations in the UK (Boxall *et al.*, 2005), and for data collected in Australia (Boxall *et al.*, 2006).

### **6.3 Other risk estimation tools and techniques**

The authors are aware of two other projects currently in progress for the development of practicable tools and techniques to aid the water supply industry with the identification and estimation of discolouration risk. The Discolouration Risk Management (**DRM**) tool which is being developed and rolled out to UK industry by Ewan Group plc and Yorkshire Water Services (Dewis *et al.*, 2005) DRM is a risk based assessment tool incorporating

likelihood of pipe failure, discolouration and consequence and is based on 'expert panel' risk trees. As such DRM is a pragmatic asset management and investment tool, but is limited by the discolouration knowledge and understanding of the required expert panels. The Particles Sediment Model (PSM) is being developed and trialled in Australia by the Cooperative Research Centre (CRC). However, as far as the authors are aware, little or no material is currently in the public domain relating to this.

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# 7 Water quality modelling

## 7.1 Overview

Network analysis models are an essential tool for knowledge gain and sensitivity analysis over system behavior, diagnosis of specific situations and general decision-making in planning, design, operation and maintenance of water supply systems. In general terms, the models allow the user to find out the values of flow rate, flow velocity, headloss, travel time, chemical concentration; and pressure or hydraulic grade, travel time, chemical concentration and source tracing at the nodes. Several other state variables are calculated, such as tank water levels, pump velocity and status, and valve status.

The water quality models that are incorporated in virtually all modern network simulators were developed in order to simulate the movement, mixing and transformation of chemical or biochemical water quality parameters that may be considered to be completely diluted in the water carried by the system (Coelho *et al.*, 2006; Lansey and Boulos, 2005; Walski *et al.*, 2003; Rossman, 2000; Rossman and Boulos, 1996). In addition, such models are capable of determining travel time between any two points (often referred to as residence time or water age), as well as tracing the proportion of water passing through any node that originates in any other given node.

There are three main processes being modeled by the above techniques. Two of those are purely a result of physics and are totally governed by pressure hydraulics laws: transport along the pipes, or advection, and mixture at junction nodes. The third process is in many ways more complex and almost always less known, and concerns the transformation undergone by the water quality parameter(s) at stake, as water travels in space and time. These transformations affect non-conservative substances, by definition, and may occur due to chemical reactions within the water itself – with itself and other components of the water – as well as reactions with the materials that water is in contact with, namely the inside surfaces of pipes, pumps, fittings and tanks, including deposits and sediments, and exchanges with the atmosphere at storage devices. Transformation processes may result in growth or decay, with partial or complete change into other products.

Besides inheriting all the assumptions of the hydraulics models upon which they are based<sup>1</sup>, the basic assumptions present in this type of models usually

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<sup>1</sup> Essentially, that the system is represented as a set of nodes connected by links, which are the physical components, such as pipes, valves and pumps; and nodes are arbitrary conventional elements, used for representing junctions, changes in pipe diameter or internal roughness, supply or demand points, sector boundaries or any other remarkable points (certain nodes with special properties represent tanks, reservoirs, boreholes). Demand and supply flows are concentrated at the nodes. The network representation may be simplified or *skeletonised*, although it is widely accepted that for water quality modeling purposes, models should include all the pipes in the network (all-mains models).

include complete instantaneous mixing at junction nodes and the absence of longitudinal mixing effects in pipes (Grayman *et al.*, 1988; Liou and Kroon, 1987). However, certain formulations also propose longitudinal dispersion (both molecular diffusion and eddy dispersion) as a quantifiable effect, especially for low velocity hydraulic regimes (Zhang *et al.*, 2004; Lansey and Boulos, 2005).

## 7.2 Formulation

The advection or transport process is fundamentally modeled by the equations of hydraulic equilibrium (Walski *et al.*, 2003, or Cesario, 1995, among many others, may be looked up for a detailed description), and specifically by the values of flow or velocity in each pipe and at each moment in time (Figure 7.1).



Figure 7.1 – Advection or transport with flow.

Modelling the mixing process that takes place at any junction node with more than one contributing pipe, including external supply, assumes instantaneous complete mixing and is ruled by the law of mass conservation (Figure 7.2).

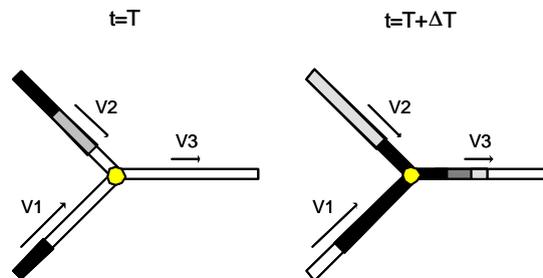


Figure 7.2 – Complete mixing at junction nodes.

At each moment in time, given any two or more pipes that contribute to a particular node and that carry different concentrations of the same parameter, the concentration of that parameter in all pipes leaving the node is homogeneous and corresponds to the complete mix of the contributing concentrations in the proportion of the respective flows.

For the generic node  $i$ , with  $U^i$  upstream nodes and  $D^i$  downstream nodes:

$$C_{ij} = \frac{\sum_{k=0}^{U^i} q_{ki} C_{ki}}{\sum_{k=0}^{U^i} q_{ki}}; \forall j \in D^i \quad (1)$$

where  $q_{ij}$  is the flow carried by the pipe linking  $i$  and  $j$ ;  $C_{ij}$  is the concentration of the modeled parameter in the flow from  $i$  to  $j$ ;  $q_{0i}$  is an external supply flow at the node, if any;  $C_{0i}$  is the concentration of the modeled parameter in that external supply flow.

The transformation process, resulting in changes in the concentration of the modeled parameter, is of the nature explored in previous sections of this text. In terms of a diluted substance, it may be described by the following generic equation, which is essentially an one-dimensional mass conservation differential equation. For a given link connecting node  $i$  to node  $j$ :

$$\frac{\partial C_{ij}}{\partial t} = V_{ij} \frac{\partial C_{ij}}{\partial x_{ij}} + RF(C_{ij}) \quad (2)$$

where  $t$  is time;  $x_{ij}$  is distance along the pipe, measured from the source node  $i$ ;  $V_{ij}$  is flow velocity;  $RF(C_{ij})$  is the reaction function being modeled.

In reality,  $C_{ij}$  in the equation above is a function both of time and distance:

$$C_{ij} = C_{ij}(x_{ij}, t) \quad (3)$$

In a simplified view, the problem thus consists of sequentially solving eq. (2) at each link (for all links), for which the initial condition at  $t=0$  is known, subject to a boundary condition  $x_{ij}=0$  given by eq. (1), that may be re-written thus (for all nodes):

$$C_{ij}(0, t) = \frac{\sum_k q_{ki} C_{ki}(L_{ki}, t) + q_{0i} C_{0i}}{\sum_k q_{ki} + q_{0i}}, \forall k \in U, \forall j \in D \quad (4)$$

where  $L_{ki}$  is the pipe length from node  $k$  to node  $i$ .

A particular treatment must be given to nodes that contain storage, such as tanks and reservoirs, as the stored bodies of water undergo changes in quality along time and by mixing with incoming water. As a base solution, it can be assumed that the contents of such storage facilities are completely mixed.

This is a reasonable assumption for many tanks operating under fill-and-draw conditions providing that sufficient momentum flux is imparted to the inflow (Rossman and Grayman, 1999). Under completely mixed conditions the concentration throughout the tank is a blend of the current contents and that of any entering water. At the same time, the internal concentration could be changing due to reactions. The following equation expresses these phenomena:

$$\frac{\partial(V_s C_s)}{\partial t} = \sum_{i \in I_s} Q_i C_{i|x=L_i} - \sum_{j \in O_s} Q_j C_s + r(C_s) \quad (5)$$

where  $V_s$  = volume in storage at time  $t$ ,  $C_s$  = concentration within the storage facility,  $I_s$  = set of links providing flow into the facility, and  $O_s$  = set of links withdrawing flow from the facility (Rossman, 2000).

More sophisticated models for a body of water can be extended from the above formulation, such as assuming there will be some stratification, leading up to considering, e.g., 2 separate volumes within the tank. Each of the volumes will have an uniform concentration and will be governed by an equation of the type of eq. (5), with a given rate of transfer assumed between the two bodies. Solutions of this kind are considered in many of the currently available modeling packages, and are well documented, e.g., in Lansey and Boulos (2005).

### 7.3 Model solutions

The above described models aggregate a large number of non-linear equations of eq. (2) and (3) type, for all links, plus eq.(4) for all nodes, at each moment in time of an extended period simulation. The set of equations is solved through numerical methods, of which a variety have been proposed since the late 1980's. Solutions have been grouped into Eulerian methods:

- Finite-difference method (FDM)
- Discrete-volume method (DVM)

and Lagrangian methods:

- Time-driven method (TDM)
- Event-driven method (EDM)

Other methods that have been proposed include control-based, backtracking methods and the Split-Operator (SOM) / Alternating Split-Operator (ASO) methods.

Lansey and Boulos (2005) and Rossman and Boulos (1996) contain detail descriptions and thorough discussions of the relative merits and shortcomings. Rossman (2000) provides a particularly well-supported and detailed description of the theoretical basis for the solution used in the widespread Epanet 2.0 modelling software and programmable library (dll).

#### 7.4 Model assumptions and limitations

As previously mentioned above, the models developed in the above manner are based on important assumptions, such as:

- complete mixing at junction nodes;
- cross-sectional homogeneity and absence of radial dispersion;
- Lagrangian (event-driven) formulations – absence of longitudinal dispersion at the plug’s head and tail boundaries;
- modelled parameters behave as if completely diluted;
- assumptions and simplifications inherent to the transformation functions deployed; and
- the assumptions of the underlying hydraulic model, particularly as regards the way demands are specified and estimated (at the nodes).

On the other hand, these models suffer from limitations induced by their deterministic nature. They attempt to describe in great detail chemical and microbiological processes which are not, as yet, fully understood or characterised. While trying to account for all possible interactions in a deterministic way, such formulations are also often hampered by over-parameterisation, leading to models that are realistically impossible to verify and calibrate.

Such water quality models fundamentally depend on sound hydraulic modeling to provide the basis for transport and mixing of water components. Even the best existing hydraulic models are not reliable at the level of detail needed by deterministic “all mains” water quality models. Their representation of consumption is based on average values at the nodes. These values are only really verified at the lumped level (i.e., at the upstream mains where demands are aggregated), and have high variance on network areas that represent smaller portions of the total demand. Such variance levels may significantly impact the inferences (advection and mixing) that are performed when modeling water quality. Those effects must be clearly quantified, establishing the limits to which the hydraulic models may be used as a basis for inference in water quality related modeling.

Hydraulic modelling assumptions limit the credibility of detail consumption and flow estimates. This usually has little effect on pressure estimates, especially in systems with low headloss (typical of many distribution sectors, especially in peripheral areas or network ends), therefore it is often perfectly acceptable for hydraulic analysis objectives.

Water quality models increase the demands on the hydraulic model because concentration and travel time calculations are based on time and space integration, hence any errors on flow and velocity are accumulated many times over.

Therefore, water quality models place increased calibration requirements, particularly on the demand allocation front, given the trend towards “all

mains” models. The complexity of process models adds to the questions raised regarding their effective usability.

## **7.5 Network modelling software**

### **7.5.1 Introduction**

This section reviews the features and capabilities of some of the most important modelling software currently available. The review begins with Epanet 2.0 and its upcoming evolution MSX Epanet, given the impact that this software has on the modelling community and on several of the other models available. Some commercially available models are subsequently introduced, such as Advantica/Stoner Synergee Water, DHI MikeUrban, Haestad/Bentley WaterCAD, MWHSoft H<sub>2</sub>O Net, Safège Piccollo and Wallingford Software InfoWorks WS <sup>(2)</sup>. The review ends with reference to other non-commercially available models and research-oriented models.

### **7.5.2 Epanet**

Epanet 2.0 (Rossman, 2000) is a public domain water supply network analysis modelling package developed and published by the U.S. Environmental Protection Agency (USEPA, or EPA). Epanet is essentially a stand-alone program for performing extended period simulation of hydraulic and water quality behavior within drinking water distribution systems.

The current version was introduced in 2000, building up and considerably expanding on the capabilities of the first version, published in 1993. This concerned several aspects, namely as regards the modeling environment, with full MS Windows support; the water quality modeling capabilities – where it introduced a very efficient EDM model that has become a reference and a standard in the industry; and a complete programmers toolkit that makes available all the modeling capabilities of Epanet, through a DLL library, to programs written in C/C#/C++, Pascal, Visual Basic and other languages, both in MS Windows and, more recently, in Unix.

Due to a policy of transparency on the part of EPA, who makes all materials freely available to the public domain, including the source code, and in no lesser degree to the intrinsic quality of the software – particularly of its hydraulic and water quality simulation engine – Epanet 2.0 has become a standard around the world. Large numbers of users and very active internet user communities virtually warrant a level of testing and debugging quite beyond the reach of any other similar software, which makes up for the absence of a formal maintenance and customer support structure. Given its widespread dissemination, and the sheer number of existing network models that have been formulated in the Epanet text format (.inp), many of the commercial packages are r/w compatible with that format.

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<sup>2</sup> All brand names mentioned in this section are trademarks and / or registered marks of their respective owner companies.

Epanet has a graphical user interface that allows for the correct formulation of models, and as such as been adopted by many professional users from the water industry. However, it has a simple project management, no power tools for user and data management, and no native support for GIS (although several 3rd-party solutions are available to allow export from the most common GIS software, or even to allow for Epanet to run from within those packages). Epanet is also relatively unsophisticated in terms of describing commercially available system components, such as some valve types and variable-speed pumps.

The Epanet water quality model uses a Lagrangian time-based approach to track the fate of discrete parcels of water as they move along pipes and mix together at junctions between fixed-length time steps. It is incorporated into the hydraulic solver and is both efficient from the computing viewpoint, and accurate in terms of the calculations performed to integrate the equations previously introduced in 7.2. In fact, the water quality solver (and often, the hydraulic solver) is used by many of the commercially available models mentioned below, through the Epanet toolkit. For that reason, it is described here in some detail.

Epanet 2.0 calculates either travel time, source trace (the % of water originating from any one node that flows through all pipes and nodes) or a concentration of a given parameter. It is capable of calculating bulk flow kinetics and wall effect kinetics. Several types of bulk flow kinetics can be accommodated (as described in Rossman, 2000):

- *Simple First-Order Decay* ( $C_L = 0, K_b < 0, n = 1$ )

$$R = K_b C \quad (6)$$

The decay of many substances, such as chlorine, can be modeled adequately as a simple first-order reaction.

- *First-Order Saturation Growth* ( $C_L > 0, K_b > 0, n = 1$ ):

$$R = K_b (C_L - C) \quad (7)$$

This model can be applied to the growth of disinfection by-products, such as trihalomethanes, where the ultimate formation of by-product (CL) is limited by the amount of reactive precursor present.

- *Two-Component, Second Order Decay* ( $C_L > 0, K_b < 0, n = 2$ ):

$$R = K_b C (C - C_L) \quad (8)$$

This model assumes that substance A reacts with substance B in some unknown ratio to produce a product P. The rate of disappearance of A is proportional to the product of A and B remaining. CL can be either

positive or negative, depending on whether either component A or B is in excess, respectively. Clark (1998) has had success in applying this model to chlorine decay data that did not conform to the simple first-order model.

- *Michaelis-Menten Decay Kinetics* ( $CL > 0, Kb < 0, n < 0$ ):

$$R = \frac{K_b C}{C_L - C} \quad (9)$$

As a special case, when a negative reaction order  $n$  is specified, EPANET will utilize the Michaelis-Menten rate equation, shown above for a decay reaction. (For growth reactions the denominator becomes  $CL + C$ .) This rate equation is often used to describe enzyme-catalyzed reactions and microbial growth. It produces first-order behavior at low concentrations and zero-order behavior at higher concentrations. Note that for decay reactions,  $CL$  must be set higher than the initial concentration present.

Koechling (1998) has applied Michaelis-Menten kinetics to model chlorine decay in a number of different waters and found that both  $Kb$  and  $CL$  could be related to the water's organic content and its ultraviolet absorbance as follows:

$$K_b = -0.32UVA^{1.365} \frac{(100UVA)}{DOC} \quad (10)$$

$$C_L = 4.98UVA - 1.91DOC \quad (11)$$

where  $UVA$  = ultraviolet absorbance at 254 nm (1/cm) and  $DOC$  = dissolved organic carbon concentration (mg/L). These expressions apply only for values of  $Kb$  and  $CL$  used with Michaelis-Menten kinetics.

- *Zero-Order growth* ( $CL = 0, Kb = 1, n = 0$ ),  $R = 1.0$   
This special case can be used to model water age, where with each unit of time the "concentration" (i.e., age) increases by one unit.

While flowing through pipes, dissolved substances can be transported to the pipe wall and react with material such as corrosion products or biofilm that are on or close to the wall. Pipe wall reactions are modeled assuming first-order kinetics, using the following expression (Rossman, 2000):

$$r = \frac{2k_w k_f C}{R(k_w + k_f)} \quad (12)$$

where  $k_w$  = wall reaction rate constant (length/time),  $k_f$  = mass transfer coefficient (length/time), and  $R$  = pipe radius. For zero-order kinetics the reaction rate cannot be any higher than the rate of mass transfer, which means:

$$r = \text{MIN}(k_w, k_f C)(2 / R) \quad (13)$$

where  $k_w$  now has units of mass/area/time.

The amount of wall area available for reaction and the rate of mass transfer between the bulk fluid and the wall will influence the overall rate of this reaction. The first is influenced by the surface area per unit volume (for a circular section, 2 divided by the radius), while the latter can be represented by a mass transfer coefficient whose value depends on the molecular diffusivity of the reactive species and on the Reynolds number of the flow (Rossman et. al, 1994).

Mass transfer coefficients are usually expressed in terms of a dimensionless Sherwood number ( $Sh$ ):

$$k_f = Sh \frac{D}{d} \quad (14)$$

in which  $D$  = the molecular diffusivity of the species being transported (length<sup>2</sup>/time) and  $d$  = pipe diameter. In fully developed laminar flow, the average Sherwood number along the length of a pipe can be expressed as

$$Sh = 3.65 + \frac{0.0668(d / L) Re Sc}{1 + 0.04[(d / L) Re Sc]^{2/3}} \quad (15)$$

in which  $Re$  = Reynolds number and  $Sc$  = Schmidt number (kinematic viscosity of water divided by the diffusivity of the chemical) (Edwards et.al, 1976). For turbulent flow the empirical correlation of Notter and Sleicher (1971) is used:

$$Sh = 0.0149 Re^{0.88} Sc^{1/3} \quad (16)$$

### 7.5.3 Multi-Species Extended Epanet (MSX Epanet)

MSX Epanet (Multi-species Extended Epanet) is an upcoming extension to the Epanet 2.0 programmable toolkit. It is essentially a new set of water quality modeling routines that extends on those previously available, namely allowing for multi-species, that is, the consideration of an array of concentrations, instead of the concentration of a single parameter (or travel time or source contribution). It keeps all the existing capabilities of Epanet, namely for extended-period hydraulic and water quality simulation.

Instead of suggesting a certain number of fixed kinetic models, such as presented above for the current Epanet version or offered by the majority of the best software available, it is open to any model that the user will care to specify. It uses its own very simple script language, and a parser to translate the formulas specified by the user into the calculation engine. It accepts any number of parameters and formulas for process models, in parallel. The formulas may be cross-dependent among them, or independent from one another – in other words, several processes may coexist and influence each other, if so specified. The formulas and parameters may be modified at will before each batch run, and the user has access to all the hydraulic and water quality variables under analysis as part of the set of parameters for calculation of each specified process.

It is expected that MSX Epanet will allow for more effective and flexible testing of process models, as it will become easier to implement individual or combined kinetics at network model level, and test the results.

MSX Epanet , currently in beta testing, will be launched solely as a toolkit extension, i.e., as a programmable library. At release time, it will not be available under the main (MS Windows – based) Epanet 2.0 software.

#### **7.5.4 Commercially available models**

The engineering software market offers a large variety of commercial software for modeling water supply networks. The most complete of these packages tend nowadays to offer GIS integration (several are already GIS-native), billing/customer system integration, database-oriented model data management, powerful model building environments, workgroup user management with version control, scenario managers and a wealth of extra capabilities to complement the standard hydraulic and water quality modeling.

These can include demand allocation and distribution, calibration aids, fire flow analysis, pump scheduling, network skeletonisation, pressure zone management, pipe sizing and design, transient analysis, booster chlorination siting, and sensor or sampling station optimisation, to name a few. Those with more advanced model data management offer the ability to connect to the SCADA/ telemetry databases, as well as to the maintenance/ component management data systems (often via the GIS) , the laboratory information management systems (LIMS) and other information systems in the utility.

Examples of such packages are:

- **Infowater** and **H<sub>2</sub>O<sub>N</sub>et** (MWHSoft) [<http://www.mwhsoft.com>]
- **InfoWorks WS** (Wallingford Software) [<http://www.wallingfordsoftware.com>]
- **MikeUrban** (DHI Software) [<http://www.dhi.dk/www,-d-,dhigroup,-d-,com/Software/Urban/MIKEURBAN.aspx>]

- **Piccolo** (Safège)  
[<http://www.safege.fr/french/dom/logiciel/reseaux/piccolo/present.htm>]
- **Synergiee Water** (Advantica/Stoner)  
[<http://www.advantica.biz/Default.aspx?page=321>]
- **WaterGems** and **WaterCAD** (Haestad/Bentley) [<http://www.haestad.com>]

Such packages are targeted at the water utilities and the consultants that directly support them in technical network management tasks, with particular benefits for larger-scale systems.

The added calculation and analysis capabilities may, or may not, be of importance depending on the specific network, scenario and operational/strategic objectives; many are undoubtedly useful and sophisticated. Although these models may sometimes be faster, especially for larger networks, the basic hydraulic and water quality simulation are, in general terms, not significantly more accurate than Epanet or many of the more frugal models (as mentioned before, the Epanet solver is often used by the larger packages as the base calculation engine).

What really sets these solutions apart are the capabilities for model building, version control, scenario- and user- management, and model data management. A crucial example is the option to concentrate on the GIS as the reference description of the system, and thus save on maintenance time and increase model accuracy and reliability (only the GIS is updated). Not only the models are generated much quicker, but the answers to the questions posed by performing hydraulic simulations feedback into the GIS for improved accuracy of the database, resulting in more reliable sets of data. Whenever the GIS and the customer/billing system can be bridged or integrated, the automatic model demand allocation is a great improvement, in terms of model update capability, and potentially in terms of the accuracy of the demand distribution. This is an important feature in terms of the accuracy of the hydraulic model, but most importantly, of the water quality model.

In terms of water quality calculations, and although both the model building and the result processing may be more advanced (including calibration aids, for example), the packages mentioned above seem, in general terms, to offer the same type of capabilities as those described for Epanet 2.0 (see 7.5.2). Several (such as **InfoWorks WS** or **MikeUrban**) use the Epanet water quality solver, as mentioned before.

**InfoWorks WS**, in addition to the Epanet WQ kinetic models, offers the capability to predict the sedimentation behaviour of a given type of material, through the **WatSed** module. The model is based on the Ackers-White general formulation (Ackers and White, 1973) that uses a set of 5 input parameters to define the characteristics of the solid and its suspension and deposition limits. The results are calculated throughout the hydraulic simulation process to predict the pipe carrying capacities of each pipe in the

network and whether suspension, deposition or variable conditions can take place.

### 7.5.5 *Research and non-commercially available models*

A great number of models have been proposed for network analysis, and specifically for simulation of a variety of water quality parameters. Of those that escape the mainstream logic described in the previous sections, perhaps the most controversial remain the attempts to generate mechanistic or deterministic models of microbiological processes.

One such proposal was the **Sancho** model (Servais *et al.*, 1994; Bois *et al.*, 1997), which attempted to describe the growth of biofilm or attached bacteria, the growth of suspended bacteria in the bulk water, the loss of substrate, the inhibition of growth by chlorine and the loss of chlorine by oxidation reactions. This model did not, however, take into consideration the hydraulics of the system, and therefore the transport and mixing processes were essentially not replicated. Instead, the results of a steady-state hydraulic model were used to determine travel times throughout the distribution system, and the were taken into consideration in predicting bacterial regrowth in a reactor-based logic. Sancho contains a large number of system parameters (19) that must be specified. Cigana *et al.* (1997) showed that only five of those – bacterial mortality, maximum capacity for bacterial fixation, bacterial yield, initial substrate concentration, and maximum uptake rate of the substrate – strongly influence the predictions of free bacteria in the bulk water. Sancho was not very flexible in terms of parameterization (the model was essentially fixed) and was a proprietary product; at the date of release of the present report, it does not appear to be available on the market.

The **PiccoBio** module (Dukan *et al.*, 1996) was proposed as part of the Piccollo package (see 7.5.4), as another, more advanced deterministic model of microbiological processes. It takes into consideration parameters such as biodegradable dissolved organic carbon (BDOC), temperature, residual chlorine, pH and the hydraulics in each pipe (steady-state). Although it has that advantage over Sancho, it also uses a higher number of parameters. A review by Gagnon *et al.* (1997) of this type of models points to over-parameterisation and complexity as barriers to their useful deployment.

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