The treatment of hospital wastewater: an appraisal

B. Pauwels and W. Verstraete

ABSTRACT

Hospitals discharge considerable amounts of chemicals and microbial agents in their wastewaters. Problem chemicals present in hospital wastewater belong to different groups, such as antibiotics, X-ray contrast agents, disinfectants and pharmaceuticals. Many of these chemical compounds resist normal wastewater treatment. They end up in surface waters where they can influence the aquatic ecosystem and interfere with the food chain. Humans are particularly exposed by the drinking water, produced from surface water. Microbial agents of special concern are multiresistant microbial strains. The latter are suspected to contribute to the spread of antibiotic resistance. In this paper, we will discuss the different approaches towards hospital wastewater treatment. The principle of uncoupling hospitals from public sewers warrants in-depth evaluation by technologists and ecotoxicologists as well as public health specialists.

Key words | estrogens, hospital wastewater, multiple antibiotic resistance, wastewater treatment, X-ray contrast media

B. Pauwels
W. Verstraete (corresponding author)
Laboratory of Microbial Ecology and Technology (LabMET),
Ghent University,
Coupure Links 653, B-9000 Ghent,
Belgium
Tel.:+32 9264 5976
Fax: +32 9264 6248
E-mail: *Willy.Verstraete@ugent.be*

INTRODUCTION

Hospital wastewater constitutes a major discharge of chemicals, but it is not unique in this respect. Residues of pharmaceuticals can be found in all wastewater treatment plant (WWTP) effluents, due to their inefficient removal in the conventional systems (Kümmerer 2001; Kolpin *et al.* 2002; Petrovic *et al.* 2003; Snyder *et al.* 2003; Carballa *et al.* 2004). It is difficult to distinguish between pharmaceuticals which originate from hospitals connected to the sewer and from household users. For substances such as iodinated X-ray contrast media, which are used for X-ray imaging of soft tissues, the hospital source is obvious. Non-prescription drugs are mainly used in hospitals (Kolpin *et al.*, 2002), but in households as well.

Besides recalcitrant and potent chemicals, hospitals discharge plenty of undesired potentially pathogenic propagules, e.g. antibiotic resistant bacteria, viruses and maybe even prions, etc. There may arise situations where a total exclusion of emission from the hospital is required, for instance in the case of multiple antibiotic-resistant strains (MARS).

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In this review, we pose the question "Can public policy continue to allow co-treatment of hospital wastewater with domestic sewage?" To evaluate this topic, an array of chemicals is scrutinized (Table 1) and potentially pathogenic propagules discharged by hospitals are reviewed. In the second section, emission abatement scenarios and their respective costs are examined.

EMISSION OF CHEMICALS

Chemicals

The presence of chemicals in wastewaters, surface waters, drinking waters and groundwaters has been reviewed extensively (Daughton & Ternes 1999; Jones *et al.* 2001; Sacher *et al.* 2001; Kolpin *et al.* 2002; Andreozzi *et al.* 2003; Petrović *et al.* 2003; Richardson 2003; Snyder *et al.* 2003; Anderson *et al.* 2004; Dębska *et al.* 2004. Most important chemicals in hospital wastewater are antibiotics (cf. above), cytostatic agents, anaesthetics, disinfectants (due to their

Table 1 | Estimates of the levels of different hospital related pollutants in hospital (resp. domestic) wastewater

Pollutant	Hospital wastewater	Domestic wastewater
Total antibiotics load (μg/L)	–	50[a]
Individual antibiotic concentration (μg/L)	2–83 measured; 5–50 estimated[a]	< LOD – 0.6[b,c], – 1.7[d], – 6[e], – 51[e]
Antibiotic resistant propagules (N/L)	–[1]	–[1]
Individual therapeutics concentration (μg/L)	5 – 50 estimated[a]	< LOD – 5.7[b]
Iodinated contrast media (μg/L)	–	< LOD – 6.6[b]
Estrogens (ng E2-eq/L)	>100[2]	20–100

a: Kümmerer (2001); b: Carballa *et al.* (2004); c: Yang & Carlson (2004); d: Ternes (1998); e: Ohlsen *et al.* (2003).
LOD: limit of detection; 1: no data on total amount of antibiotic resistant propagules available; 2: estrogen concentration is dependent on the number of pregnant women present in the hospitals maternity department. –: data not available.

major use in hospital practice), platinum, mercury (in preservatives in diagnostic agents and as active ingredients of disinfectants), rare earth elements (gadolinium, indium, osmium) and iodinated X-ray contrast media (Kümmerer 2001). Other pharmaceuticals which have been detected in WWTP effluents include lipid regulators, analgesics, antibiotics (cf. above), antidepressants, antiepileptics, antineoplastics, antipyretics, antiphlogistics, antirheumatics, β-blockers, broncholytics, β2-sympathomimetics, estrogens (cf. below), secretolytics, vasodilators and X-ray contrast media (cf. below) (Sacher *et al.* 2001; Ternes 2001).

Antibiotics

About 10 000 tonnes of antibiotics are consumed annually in Europe, of which roughly half are used in human medicine; the other half is used for veterinary purposes as a therapy or as a growth promoter. Of the antibiotics used for human purposes, 26% are used in hospitals (Kümmerer 2001). Antibiotics and their metabolites end up in the WWTP, since they are excreted with urine and faeces in wastewater.

A survey in a German university hospital revealed that antibiotic use increased by 16% for the surgical services, and by 20% for the medical services, in the period 1998–2000. This antibiotic use is expressed as defined daily doses per 100 patient days (DDD/100), (World Health Organization, WHO). Comparing the worldwide hospital antibiotic use, values of ~60 DDD/100 for surgical services and ~80 DDD/100 in medical services are consistent (de With *et al.* 2004). Cizman *et al.* (2004) report 16–78 DDD/100 for six Central Eastern European countries. Ohlsen *et al.* (2003) detected fluoroquinolones, erythromycin, gentamycin, trimethoprim, sulfamethoxazole, roxithromycin and clarithromycin in hospital wastewater at individual concentrations below 0.01 mg/L. In comparison with municipal wastewater, the total antibiotic load was reported to be relatively high, albeit not further specified.

Kümmerer (2001) estimated the total antibiotic load of municipal wastewater (which contains the contribution of hospitals) at 50 μg/L. This concentration takes into account outdated medicaments or remainders which are disposed of in household drains. These account for up to 20–40% of the total antibiotics. Giger *et al.* (2003) revealed that ciprofloxacin (a fluoroquinolone) concentrations in hospital wastewater were present above the predicted no-effect concentration (PNEC) of 3–10 μg/L. The authors add, however, that risk characterizations based on one compound are of limited value. Fluoroquinolones are very much related, so the total fluoroquinolone concentration should be considered in risk characterization. Andreozzi *et al.* (2003) detected some previously undetected fluoroquinolones in European WWTP effluents. These more recent fluoroquinolones add to the effect of the earlier generation of fluoroquinolone antibiotics. Hirsch *et al.* (1999) also consider agricultural run-off from manured land (manure

contains antibiotics which are used as growth promoters in veterinary medicine), aquaculture and landfills with pharmaceutical waste and pharmaceutical wastewater as important sources for antibiotics in the environment.

A lot of researchers focussed on the presence of antibiotics in surface waters and/or its implications for drinking water production technology (Sacher *et al.* 2001; Snyder *et al.* 2003). Webb *et al.* (2003) evaluated the risk of indirect exposure via drinking water of pharmaceuticals. For the most part of pharmaceuticals and *in casu* for antibiotics, at the present levels in drinking water prepared from surface waters there appears no risk in consuming 2 L of water daily during a lifetime of 70 years.

Jolibois *et al.* (2003) used the Ames test and the SOS chromotest to evaluate the overall toxicity of hospital wastewater. The authors contribute the genotoxic effect of 55% of the samples to anticancer drugs (e.g. ifosfamide, cisplatin) and antibiotics (e.g. ciprofloxacin). This genotoxic effect correlates with the findings of Kümmerer *et al.* (2000) in the Closed Bottle Test (CBT). The authors could not detect biodegradation of ciprofloxacin, ofloxacin and metronidazole in the CBT. The genotoxicity of these chemicals was not eliminated in this test.

These findings show that efforts are made to evaluate the risk of antibiotics. The risk cannot be estimated correctly due to the lack of concentration level data in hospital wastewater.

Iodinated contrast media (ICMs)

Iodinated contrast media (ICMs) are used for X-ray imaging of soft tissues. This industry has a turnover of $684 million worldwide in 2001 (Versweyveld 2004). Engels-Matena (1996, cited in Kalsch 1999) assumed a worldwide ICM consumption of 3460 tonnes in 1993. For one medical treatment, about 100 g of X-ray contrast media is used. This represents about 30 g of absorbable organic iodinated compounds (AOI) (Doll & Frimmel 2003). The AOI are biologically inert and stable towards metabolism during their passage through the body. They are excreted almost completely within a day after administration, ending up in the WWTP, where they are poorly removed (0–85% removal) (Steger-Hartmann *et al.* 2002). Due to their high hydrophilicity ICMs persist in the environment in the aqueous phase, rather than sorb onto organic material or accumulate in organisms (Kalsch 1999). Since not much is known about their fate and long term effects, there is a risk connected to their spread in the environment. They could end up in groundwater. More research is needed on this topic and precautionary measures should be taken.

Estrogens

Estrogens are of particular interest, since researchers in the UK observed the feminization of male caged fish at discharge sites of WWTPs (Purdom *et al.* 1994). The natural estrogens estradiol (E2), estrone (E1) and estriol (E3) together with the synthetic estrogen ethinylestradiol (EE2) are seen as the most important sources of estrogenic activity in environmental samples (Thomas *et al.* 2001). Estrogens are excreted in urine by both male and female mammals as sulfate or glucuronide bound complexes (Daughton & Ternes 1999). On a daily basis, women excrete on average approximately 32, 14 and 106 μg of conjugated E1, E2 and E3, respectively. Pregnant women excrete about 100 times this amount (D'Ascenzo *et al.* 2003). It can be assumed that in hospital wastewater elevated concentrations of E1, E2 and E3 can be expected. However, to the best of our knowledge, hospital wastewater estrogen levels have not been reported.

Other chemicals

The antimanic/antiepileptic drug carbamazepine has a chronic toxicity down to 25 μg/L on ceriodaphnids (toxicity test organisms). Risk quotients calculated for French and German WWTP effluents are greater than unity, thus meaning that these effluents pose a threat for aquatic life. This finding is corroborated by the fact that there is a continuous input of drugs in the environment, hence yielding chronic toxicity effects (Ferrari *et al.* 2003). Stackelberg *et al.* (2004) detected carbamazepine in the final drinking water of a US water production plant, even though granular activated carbon (GAC) filtration was used in this plant. Other drugs and antibiotics (erythromycin metabolite, sulfamethoxazole, acetaminophen, codeine, trimethoprim) were detected in the surface water used for drinking water production, but not in the final drinking

water. This finding confirms that component specific analysis is needed for risk analysis in drinking water production.

A mixture of 4 lipid regulators and 4 β-blockers, of which only 2 are considered as harmful based on toxicity categories regulated by European Directive 93/67/EEC and on EC_{50} values, was shown to be very toxic while each individual drug was only present at a low concentration of 2 μg/L (Hernando *et al.* 2004).

Cytostatic agents represent a danger because of their proven carcinogenicity, mutagenicity and embryotoxic properties. The largest emission of platinum stems from its use as a cytostatic agent. In this respect, the major source is excretion by patients (ng/L to μg/L levels in urine). Gadolinium (Gd) and recently indium (In) complexes are used in MRI (magnetic resonance imaging). These Gd and In complexes are non-biodegradable.

EMISSION OF PROPAGULES

Antibiotic resistant propagules

Bacteria have different mechanisms to become resistant to a specific antibiotic. Genes encoding for this resistance can be transferred vertically (i.e. to the bacteria's offspring) or horizontally (i.e. among bacteria of different taxonomic affiliation) (Schwartz *et al.* 2003). (Resistance) gene transfer is optimal at high cell densities and under high selective pressure (i.e. high antibiotic concentrations). However, under heterogeneous environmental conditions, this gene transfer can still occur at a significant level (van Elsas *et al.* 2000).

The emergence and spread of methicillin-resistant *Staphylococcus aureus* (MRSA) is of special concern. MRSA strains acquire multiresistance by means of additional resistant factors, such as conjugative gentamycin resistance plasmids (Ohlsen *et al.* 2003). Gentamycin resistance in *S. aureus* was transferred as efficiently in hospital sewage agar plates as on rich media, although the number of donor and recipient cells was decreased by about 1000-fold in sewage. Transfer of resistance genes was detectable in plain sewage at a frequency of $<5.0 \times 10^{-8}$ to 2.0×10^{-6} (Ohlsen *et al.* 2003).

Ruiz *et al.* (2004) reported higher antibiotic susceptibility of environmental *Pseudomonas aeruginosa*, which was collected from the hospital tap water and in the garden, relatively to clinical isolates from the same hospital. Clinical isolates were up to 32% resistant against several antibiotics, whereas environmental isolates were susceptible to these antibiotics and 5% were resistant to ofloxacin. Resistance to antipseudomonal antibiotics increased by >20% (34–37% for imipenem) in a US hospital over the five-year period 1998–2002. In 1998, 78% of isolates were susceptible to all four examined antibiotics, whereas in 2002 this number decreased to 27%. At the same time, the number of isolates that were resistant to all four antibiotics increased from none to 32% (Jung *et al.* 2004).

Blanch *et al.* (2003) studied the geographic differences in enterococcal populations in hospital wastewaters. Urban WWTPs receiving hospital wastewater in Sweden, Spain and the United Kingdom (UK) were similar in the total counts of enterococci and with regard to removal during treatment (about 2–2.5 log units decrease). The most common enterococci in wastewaters were *Enterococcus faecium* and *Enterococcus faecalis*, reflecting the contribution of animal and human faeces. These were also the most common vancomycin resistant enterococci. Resistance to the macrolide antibiotic erythromycin was common in the investigated areas. The authors conclude that the selective pressure from the environment could cause the high prevalence of resistant strains. Schwartz *et al.* (2003) could amplify the *van*A vancomycin-resistance gene from enterococci, the *mec*A methicillin-resistance gene from staphylococci and the *amp*C β-lactam-resistance gene from Enterobacteriaceae from hospital wastewater biofilms. *van*A genes and *amp*C genes were also detected in other wastewater and environmental biofilms. The same was observed on tetracycline resistant strains by Guillaume *et al.* (2000). The highest percentage of tetracycline resistant strains were found in hospital wastewater treatment facilities, where up to 12% of these isolates carried the *tet* A and *tet* C resistance genes.

These studies clearly demonstrate that hospital wastewaters are a source of bacteria with acquired resistance against antibiotics and this with at least a factor of 2–10 higher than domestic wastewater.

EMISSION ABATEMENT OF CHEMICALS

Removal efficiencies of different wastewater treatment techniques are commented on in the following paragraphs. A summary can be found in Table 2. For hospitals having their own on-site wastewater treatment plant, i.e. they do not discharge their raw wastewater into the sewer, data for chemical concentrations and/or chemical removal are not available in the literature.

Antibiotics

There are several options for antibiotic removal out of (hospital) wastewater. Carballa *et al.* (2004) detected a 65% removal of sulfamethoxazole during the biological step in a conventional WWTP. Andreozzi *et al.* (2003) examined the solar photodegradation of antibiotics. The half-life times for antibiotics were of the order of 5–10 d. Nitrate or humic acids can act as photosensitizers for antibiotics in river water and hence accelerate the breakdown with a factor of 2–5. Balcioğlu & Ötker (2003) reported in wastewater an increased biodegradability of cephalosporine, penicillin or quinolone after ozonation or O_3/H_2O_2 (3 g O_3/L h) treatment. Aksu & Tunç (2004) compared the removal of penicillin G (= benzylpenicillin) by biosorption with low-cost, natural and abundant sorbents (i.e. dried *Rhizopus arrhizus* biomass and dried activated sludge) with the removal by activated carbon. The dried activated sludge and *R. arrhizus* represented a valuable alternative to activated carbon, but the research was performed at environmentally irrelevant concentrations (1 g penicillin G/L).

In a series of articles (Giger *et al.* 2003; McArdell *et al.* 2003; Göbel *et al.* 2004) researchers presented results on the fate of macrolide and sulfonamide antibiotics in a catchment basin. The main outcome of this research was that WWTPs eliminated the macrolide antibiotics only for 20% (no statistically significant removal) and hence contributed to the antibiotics present in the rivers.

Iodinated contrast media (ICMs)

Activated sludge treatment is inefficient in removing ICMs. Carballa *et al.* (2004) detected no removal of iopromide in a

Table 2 | Removal characteristics of different hospital related pollutants by various processes

	Natural attenuation	Activated sludge	PAC/GAC	Ozonation	Ultraviolet photolysis	Reverse osmosis
Antibiotics	Poor	None[f]–poor (67%[a])	50–99%[g]	>95%[g]	50–80%[g]	>90%[g,1]
Antibiotic resistant propagules	Poor	<1[i], 2[b,j], >3[i] log units[b]	n.a.	–	–	n.a.
Therapeutics	None–poor[d, h]	7–90%[a,c,2]	90–99%[d,3]	Poor to >95%[e,4]	$t_{\frac{1}{2}}$ = 2.4–100 d[q]	50 to >90%[k,5]
Iodinated contrast media	None[6]	None[m]–85%[l,m,6]	–	Poor–14 to >80%[p]	$t_{\frac{1}{2}}$ = 5–10 h for metabolite[m]	–
Estrogens	Poor	65–99.9%[a,o]	>99.8%[e]	>80%[p]	–	95–99%[n]

a: Carballa *et al.* (2004); b: Reinthaler *et al.* (2003); c: Petrović *et al.* (2003); d: Ternes *et al.* (2002); e: de Rudder *et al.* (2004); e: Huber *et al.* (2003); f: Kümmerer *et al.* (2000); g: Adams *et al.* (2002); h: Doll & Frimmel (2003); i: Chitnis *et al.* (2004); j: Vilanova *et al.* (2004); k: Kimura *et al.* (2003); l: Kalsch (1999); m: Steger-Hartmann *et al.* (2002); n: Schäfer & Nghiem (2003); o: Ternes *et al.* (1999); p: Ternes *et al.* (2003); q: Andreozzi *et al.* (2003).
1: 99 and 99.9% reduction can be reached with two stage (resp. three stage) reverse osmosis units.
2: Low removal efficiencies for more polar compounds.
3: Specific throughput of > 70 m^3/kg sorbent, except for diclofenac (15–20 m^3/kg sorbent).
4: Not much information available on by-product formation.
5: Lowest removal obtained at low feed concentrations (100 ng/L) and for non-charged compounds.
6: Stable transformation products accumulate; experiments were conducted at environmentally irrelevant concentrations.
PAC: powdered activated carbon; GAC: granular activated carbon; n.a.: not applicable; –: not available.

good working activated sludge plant in Spain. However, Kalsch (1999) observed primary biodegradation of iopromide and diatrizoate. Stable transformation products were formed. Mineralization was not observed. Since these substances are highly hydrophilic, they do not absorb on activated sludge solids and end up in the WWTP effluent. Ozonation of ICM containing wastewater resulted in removal efficiencies higher than 80% for non-ionic ICM (iopamidol, iopromide and iomeprol), whereas ionic ICM triazoate exhibited only 14% removal (Ternes *et al.* 2003). Costs of the required application of 10 g O_3/m^3 wastewater were estimated to be lower than 0.04 €/m^3. Larsen *et al.* (2004) state that ozonation of wastewater is not expensive, but rather energy-consuming (40–50% increase in energy demand of a normal treatment plant). It is remarkable that this increase in energy demand is not reflected in the treatment costs, of which energy costs make up a big share.

Estrogens

Estrogens are excreted mainly in urine as glucuronide or sulfate conjugates. They reach a WWTP via the sewer, and are discharged mostly in surface water with the WWTP effluent. D'Ascenzo *et al.* (2003) observed significantly higher free estrogen concentrations in septic tank fluid of a condominium than in the urine of flat inhabitants. Deconjugation is attributed to high concentrations of *E. coli* and other fecal bacteria which produce β-glucuronidase enzymes (Ternes *et al.* 1999). WWTPs do not always succeed in removing estrogens adequately, yielding ng/L estrogenic activities in WWTP effluents expressed as E2-equivalents. Since estrogens are biologically active at 0.1–20 ng E2-eq/L, aquatic ecosystems are at risk (Purdom *et al.* 1994; Baronti *et al.* 2000; Witters *et al.* 2001). Some WWTPs remove estrogens very well (>90%), whereas others do not remove estrogens (EE2 in particular) at all. Estrogen removal seems to be positively correlated with the presence of nutrient removal in the WWTP, higher sludge age (sludge retention time, SRT 12–15 d) and the use of membrane bioreactors (MBRs) (Holbrook *et al.* 2002; Andersen *et al.* 2003; Joss *et al.* 2004). Since E1 is a biodegradation product of E2 (Onda *et al.* 2003), a prolonged hydraulic residence time (HRT) is needed to remove both the E1 present in the influent and the E1 resulting from E2 biodegradation. In the case of EE2, however, the ethinyl group hampers biodegradation. This was observed by Clara *et al.* (2004) who did not measure significant differences in EE2 removal efficiency between conventional activated sludge and MBR reactors. Both the conventional activated sludge plant and the MBR removed EE2 with 60–70% removal efficiency.

Activated carbon adsorption has been proposed as a promising technique with removal efficiencies greater than 99% to remove estrogens in drinking water production (Yoon *et al.* 2004). This removal efficiency is, however, only valid at initial concentrations of 500 ng/L and higher. These levels have not been detected in wastewater, where default concentrations of EE2 and E2 vary from below the detection limit up to 5 ng/L and from below the detection limit up to 150 ng/L respectively (Baronti *et al.* 2000; Ferguson *et al.* 2001). The adsorption capacity of GAC is only 1000 ng/g at environmentally more relevant concentrations of 20 ng/L (De Rudder *et al.* 2004). These authors evaluated the sorption of EE2 onto a MnO_2 upflow bioreactor. The MnO_2 reactor was not saturated at the predicted time, but kept on removing EE2. The authors suggest that the EE2 removal is due to a microbial regeneration of the MnO_2 sorbent.

Photolysis of E1 and E2 has been demonstrated with an UV (ultraviolet) disinfection lamp and a high pressure mercury lamp. The breakdown mechanism includes an oxidation of benzene rings to produce compounds containing a carbonyl group (Liu & Liu 2004).

Due to the oxidizing effect of chlorine, chlorination reduces the estrogenic potency of E2-containing solutions. However, disinfection byproducts were formed (Lee *et al.* 2004). Hu *et al.* (2003) identified these byproducts as chlorinated E2 derivates. After 10 min reaction time, almost all E2 had reacted to form mono- and dichlorinated E2 derivatives. 2,4-dichloro-E2 elicited an estrogenic response which represents about 40% of that of E2 in a yeast two-hybrid assay. 4-chloro-E2 had about the same estrogenic response as E2. 4-chloro-EE2 and 2,4-dichloro-EE2 have been described as the main chlorination products of EE2 (Moriyama *et al.* 2004). 4-chloro-EE2 had about the same estrogenic activity as the parent compound EE2, whereas the bichlorinated compound 2,4-dichloro-EE2 elicited an estrogenic response which was about ten times lower than EE2. In conclusion, one should reconsider the use of

chlorination for the removal of estrogens. Chlorination can be effective in decreasing the estrogenic activity, but on the other hand more recalcitrant chlorinated derivates can be formed which persist in the environment.

Other chemicals

Biodegradation of four relevant pharmaceuticals (the antiphlogistic diclofenac, the antiepileptic carbamazepine and the lipid regulators clofibric acid and bezafibrate) is assumed to be relatively low (Ternes *et al.* 2002). However, other factors should be considered since Clara *et al.* (2004) found >95% removal of bezafibrate and the analgesic ibuprofen in both AS and MBR systems. Carbamazepine was not removed at all in these systems. Diclofenac showed a more differentiated behavior in the treatment plants. In the conventional AS plant 40–60% removal was obtained, whereas in the MBR system, removal efficiency was dependent on the sludge residence time (SRT). With a SRT >10 d, similar results were obtained as in the AS plant. Flocculation with iron (III) chloride does not remove any of these four pharmaceuticals (diclofenac, carbamazepine, clofibric acid and bezafibrate) significantly, neither does slow sand filtration. GAC, in contrast, succeeded very well in removing the selected compounds (Ternes *et al.* 2002).

Ozonation was very effective in removing carbamazepine and diclofenac, decreased bezafibrate and primidone concentration levels considerably, but removed clofibric acid only to a limited extent (Ternes *et al.* 2003). In addition, Huber *et al.* (2003) evaluated advanced oxidation processes (AOPs) for the removal of selected pharmaceuticals. The latter were selected on the basis of their consumption rate and environmental relevance. The antibiotics sulfamethoxazole and roxithromycin, together with the synthetic estrogen ethinylestradiol, the antiphlogistic diclofenac and the antiepileptic/analgesic carbamazepine, had half-lives of 0.01–0.1 s in environmental conditions. It could be generalized that β-blockers, fluoroquinolones, macrolides, sulfonamides, tetracyclines and steroid hormones have similar high ozone rate constants, i.e. of the order of 10^4–10^6 1/M s.

Cyr *et al.* (2002) report the use of activated carbon for the removal of mercury from hospital wastewater. Thimerosal, an organic compound containing mercury, is present in hospital wastewaters. GAC removed 99.8% of the mercury in the wastewater. At the same time 90% of the copper was removed. Based on pilot scale experiments, a cost of 24 €/m^3 was calculated, which is about 50 times as expensive as compared to normal treatment costs of about 0.5 €/m^3.

EMISSION ABATEMENT OF PROPAGULES

Antibiotic resistant propagules

Reinthaler *et al.* (2003) examined the resistance towards 24 antibiotics of *E. coli* present in three different WWTPs, of which one was treating hospital wastewater. All WWTPs showed a 2.3 log decrease for total *E. coli*. The total amount of *E. coli* that was released into the environment by the WWTP was greater than 10^2 colony forming units (CFU) *E. coli*/ml. In this way, wastewater treatment contributes to the spread of resistant bacteria in the environment. Resistant strains towards 16 of the 24 tested antibiotics were observed. Of these 16, 14 had the highest resistance prevalence (percentage resistant strains of total strains) in the effluent of the WWTP treating (amongst other sources) hospital wastewater. In addition to this, this was the only plant with quinolone resistant *E. coli*. Quinolones are more recently discovered antibiotics.

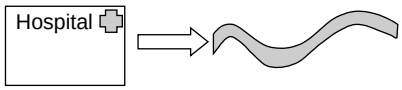
Vilanova *et al.* (2004) observed significant removal of bacterial populations in sewage treatment. They noticed, however, the persistence of vancomycin and erythromycin resistant Enterococci in the same proportions. This observation suggested that there was no selective elimination of bacterial populations during wastewater treatment processes. Chitnis *et al.* (2004) described the case of an Indian hospital, of which the WWTP effluent and the waste sludge were to be used (after disinfection) as irrigation/sanitary cleaning water and fertilizer, respectively, to make the treatment cost effective. The authors showed that the hospital was the only contributor to the multiple antibiotic resistance. Chlorination was necessary; especially to inactivate the 7.5×10^3 CFU of multiple antibiotic resistant strains (MARS) per mL of wastewater. After chlorination no MARS could be detected. Chlorination with sodium hypochlorite has been linked to AOX (adsorbable organic halogens) by Emmanuel *et al.* (2004). The authors found a positive correlation between these AOX (adsorbable organic halogens) in hospital wastewater and toxicity on *Daphnia magna*.

CRITICAL ISSUES

Hospital wastewaters urgently merit to be addressed as critical discharges to the environment in both developing and industrialized countries. In view of the above mentioned features, it is clear that hospital wastewater is a complex matrix which warrants treatment before discharge to the environment.

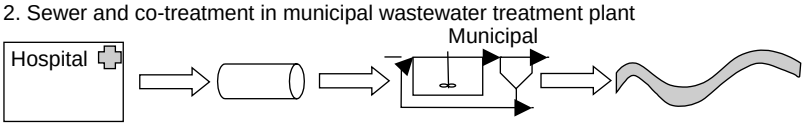
Four scenarios for hospital wastewater treatment can be envisioned: (1) direct discharge to the environment, (2) co-treatment in a municipal WWTP, (3) on-site wastewater treatment and subsequent discharge of the effluent to the environment, and finally (4) first on-site and subsequently municipal wastewater treatment. These scenarios are schematized in Figure 1. To the best of our knowledge, data on the occurrence of these different types of treatment

1. Direct discharge



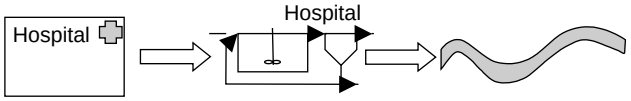
Advantages	Disadvantages
No investment, maintenance costs and process control	Very major danger of dissemination of the propagules and activation of the virulence due to putative short term cycling surface water → drinking water → human body. In case of epidemic, the whole of the raw sewage has to be chlorinated which may cause a lot of environmental damages.

2. Sewer and co-treatment in municipal wastewater treatment plant



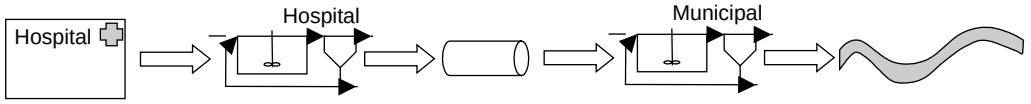
Advantages	Disadvantages
No direct discharge to the environment	Stormwater overflow creates dilution which hampers biodegradative processes at the WWTP

3. On-site wastewater treatment plant



Advantages	Disadvantages
Generally 90% decrease of load achieved	Very strict monitoring and process control necessary by both the process plant operators and in addition by the public authorities

4. On-site and subsequent municipal wastewater treatment plant



Advantages	Disadvantages
Double treatment and maximal safety	Expensive and complex

Figure 1 | Possible scenarios in hospital wastewater treatment and disposal.

are not available, neither for Europe nor the USA. Scenario 4 implies the highest costs, but at the same time the highest hazard reduction. Control procedures are necessary for all 4 abatement scenarios. A thorough evaluation of scenarios 3 and 4 is needed. The question of the dedicated treatment at the hospital site technically is necessary and effective; in other words if there is added value in submitting hospital WWTP effluent to a municipal WWTP is largely unanswered. Risk assessment for the hazard posed by the hospital wastewater should be done for all 4 scenarios. Scenario 3 (on-site treatment) could possibly give the highest efficiency and environmental benefits, since an expensive, highly effective technology on a small scale eventually can be more eco- and cost effective than a relatively cheaper technology on a large scale with smaller effects on the diluted hospital emissions.

Membrane bioreactors (MBRs)

Membrane bioreactors have been proposed as a promising alternative for conventional activated sludge treatment. Complete retention of the biosolids by the membrane (the so-called "bacteria behind membranes" concept) enables high mixed liquor suspended solids (MLSS) concentrations (12–15 g MLSS/L: some membrane manufacturers mention up to 50 g MLSS/L), yielding long sludge retention time (SRT) (20 d–∞) and low sludge loading rates (0.1 g COD/g VSS d). A low sludge production (Y = 0.23–0.32 kg MLSS/kg COD vs. Y = 0.4–0.5 kg MLSS/kg COD in AS) can be achieved resulting in overall lower sludge treatment costs (Yoon *et al.* 2004). An inquiry to 4 representative MBR selling companies revealed that there is only one MBR application for hospital wastewater treatment (Kamps, Kerkman, Futselaar & Fujimoto, all personal communications 2004). This MBR at the Kinki University Nara Hospital, Japan, has treated a hospital wastewater flow of 480 m^3/d since 1999 at 8–10 g MLSS/L and ensures a 7 log reduction of pathogens. The MBR effluent is discharged to the sewer, so this case represents scenario 4.

Interestingly, the United States (US) Commercial Service (2004) (www.buyusa.gov/print/china/en/sars15.html) has appealed to US companies to transfer MBR technology to Chinese hospitals after the recent SARS (Severe Acute Respiratory Syndrome) outbreak. SARS, however, is caused by a virus which is removed no less than 2 log units in MBRs, compared to a greater than 5 log removal of bacteria. US Commercial Service believes that MBR technology can play a key role in hospital wastewater treatment because of the high removal of bacteria. In this way, the spread of MARS can be limited as well.

Post-treatment technologies

As can be derived from Table 2, several so-called post-treatment technologies such as activated carbon, ozonation and UV photolysis remove hospital related pollutants quite well. Reverse osmosis (RO) is practically not possible and advisable, because of the required pre-treatment of WWTP effluent prior to using this technique and because of the generation of concentrated sidestreams. However, if effective further treatment of concentrate flows is possible, RO treatment of MBR effluent could be feasible.

Ozonation is a relatively cheap technique, but by-products are poorly characterized. Therefore, by-product prevalence in ozonated WWTP effluent (or more general effluent which has undergone AOP treatment) should be investigated, since most of the research is focussed on single compounds or compound groups. From the ecotoxicological viewpoint the same holds true.

Source separation

A proposal, which has been put forward by Webb *et al.* (2003), is the source separation of urine of patients which have undergone X-ray imaging. This urine, containing ICMs, can be processed as chemical waste. The same urine source separation could be applied to the urine of pregnant women in the hospital maternity department. This urine can be treated in a small scale WWTP which has been enriched with estrogen degrading organisms. However, the economical and also societal feasibility of such an approach has to be demonstrated. Larsen *et al.* (2004) highlight the 100–500 times higher concentrations of micropollutants in urine which allow more efficient conditions for removal by all types of wastewater treatment technologies. Hence, the WWTP involved in the treatment of pregnancy urine would have a small footprint with higher efficiencies and lower costs.

CONCLUSIONS

There is a remarkable paucity on data concerning the possible impacts of hospital discharges, direct or indirect, to the environment. The authorities responsible for hospital management and environmental health should address these aspects urgently and proper legislative actions are warranted.

There is a need to develop a matrix of treatment scenarios for hospital wastewaters, both with respect to attainable efficiency and costs per m^3 of water treated. Technologists and economists should be encouraged to develop and calibrate different operational configurations, thus generating the potential for practitioners to be informed on financial aspects and overall risks associated with putative treatments of hospital wastewaters.

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