Groundwater Information Sheet

The Impact of Agriculture

This is one of a series of information sheets prepared in relation to specific human activities which are of significant concern for the management of groundwater resources and protection of groundwater quality. The sheets aim to summarise the characteristics of each activity, describe the risk of each one impacting on groundwater, the possible approaches to their investigation and potential methods of control, mitigation or restoration. The purpose of these information sheets is to raise awareness of these issues amongst WaterAid Country Office staff, to provide guidance on taking the potential impacts of these activities into account in programme planning and implementation and on targeting monitoring and assessment efforts accordingly, and to encourage further thinking in the organisation on water quality and water management issues. The three sheets in this series (agriculture, industry and urbanisation) complement previous information sheets on specific groundwater quality parameters and for target WaterAid countries, and should be read in conjunction with these. The sheet on nitrate is particularly relevant, and material is not repeated here.

Agriculture as an environmental pressure

Cultivation of crops for food production and for economic benefit, and husbandry of livestock have for long been dominant activities in human communities. Population growth and increasing living standards and the associated demand for food production are the driving forces for expansion and intensification of agriculture. Of the world's total land area of 13 billion hectares, 12 per cent is cultivated and about 27 per cent is used for pasture (WWAP, 2006). The scope for meeting the growing demand for food by increasing the amount of agricultural land at the expense of forests and natural savannah is limited – most suitable land is already used. Demand is therefore largely met by increasing the productivity of existing cultivated land. Irrigation must continue to play its part in this by enabling new crops to be grown where they could not previously and by extending the growing season and cropping intensity on already cultivated land.

The 1.5 billion hectares of cultivated land include 277 million hectares of irrigated land, ie about 18 per cent of the total cropland. Irrigation represents between 70 and 80 per cent of total freshwater use, rising to more than 90 per cent in some countries. Much of this water comes from surface water storage in reservoirs or direct river diversions and the associated distribution systems. However, in favourable hydrogeological conditions highly productive aquifers provide substantial amounts of groundwater to support irrigated agriculture. The total irrigated area supplied wholly or partially from groundwater is officially reported at 69 million hectares, but could be as high as 100 million (Molden ed, 2007). Abstraction for irrigation is, therefore, a major environmental pressure factor, is by far the dominant use of groundwater globally and has grown dramatically in the last 50 years (Llamas and Custodio, 2003).

Impacts of agricultural abstraction

Most aquifers experience seasonal fluctuations in groundwater level or show declines at times of drought even when they are not exploited or are modestly used (Figure 1). During dry seasons or drought periods, river and spring flows and discharge to wetlands are maintained by the release of water from storage and groundwater levels decline. Water levels are able to recover during subsequent periods of recharge. If groundwater is exploited consistently at rates above the average annual net recharge (Figure 1), levels will decline, with the possible consequences shown in Table 1.

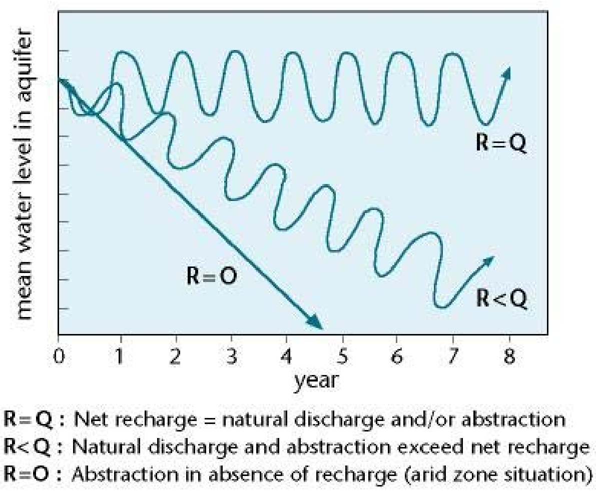

Figure 1. Patterns of groundwater level decline (Morris et al, 2003)

Table 1. Consequences of excessive groundwater abstraction (modified from Foster et al, 2000)

Consequences of excessive abstraction		Factors affecting susceptibility
Reversible interference	Pumping lifts/costs increase	Aquifer response characteristic
	Borehole yield reduction	Drawdown below productive horizon
	Springflow/river baseflow reduction	Aquifer storage characteristic
Reversible/ irreversible	Phreatophytic vegetation stress (natural and agricultural)	Depth to groundwater table
	Ingress of polluted water (from perched aquifer or river)	Proximity of polluted water
Irreversible deterioration	Saline water intrusion	Proximity of saline water
	Aquifer compaction/transmissivity reduction	Aquifer compressibility
	Land subsidence and related impacts	Vertical compressibility of overlying/ inter-bedded aquitards

These negative consequences can have severe socio-economic impacts which are slow to develop, not always immediately apparent and difficult to reverse (Table 1). Moreover, the impacts may be felt most severely by communities other than those who reap the benefits of the groundwater abstraction – drying up of shallow domestic water supply wells by large scale abstraction from deeper boreholes, for example, or subsidence damage to roads and houses.

Local hydrogeological conditions determine the susceptibility to the consequences listed in Table 1. In particular, consolidated rocks such as limestones and sandstones are unlikely to be affected by subsidence, whereas alluvial and especially lake sediments are susceptible (Foster et al, 2000). Coastal aquifers, whether consolidated or not, are clearly the most susceptible to saline intrusion.

Irrigation, waterlogging and salinity

The dramatic growth in irrigated agriculture and consequent demand for water is also met by diverting and/or storing surface water. However, withdrawing water from rivers to spread on land accelerates the accumulation of salts thorough evaporation. Moreover, the excess infiltration from the irrigated land causes the underlying groundwater levels to rise, sometimes by many metres, producing waterlogging in poorly drained soils where the groundwater level comes close to the ground surface. When the groundwater is within a metre or two of the land surface, capillary action allows water to rise further and evaporate from the land surface, resulting in salinisation of soils and water.

Overall assessment of the extent and severity of the impacts of salinisation is not easy. Smedana and Shiati (2002) suggest it seriously affects 20 to 30 million ha worldwide, ie about 10 per cent of the total area under irrigation and 25 per cent of that in the more arid regions, and may be extending at 0.25 to 0.5 million ha per year (WWAP, 2006). Much of this huge area is within the canal command areas of large irrigation schemes such as the Lower Indus in Pakistan and India, the Tigris and Euphrates, the Nile and the Aral Sea basin and in China and the United States. Severely waterlogged and saline soils become unusable, with major economic losses and destruction of livelihoods for the affected communities. Provision of drinking water supplies can become problematic in such areas; surface water in canals is likely to be unsuitable bacteriologically and groundwater too saline.

Agricultural intensification, agrochemical use and groundwater quality

Together with irrigation, the large increases in food production referred to above have been sustained only by the application of ever-increasing amounts of inorganic fertilisers and a wide spectrum of synthetic pesticides. While the rate of increase of fertiliser use is levelling off in the industrialised countries, rates of nitrogen fertiliser use have tripled in developing countries since 1975. In Asia a quarter of the growth in rice production has been attributed to increased fertiliser use.

Nitrate

Fertilisers are used to increase the availability to plants of the nutrients nitrogen, phosphorus and potassium. Of these, potassium and phosphate are rarely leached below the soil and seldom found at elevated concentrations in groundwater because of they are often the lesser components in fertiliser mixes and because phosphate is normally retained by adsorption onto clays in the soil. Nitrate is the principal nutrient leached to groundwater and is highly soluble, mobile and not readily degraded under aerobic conditions. Where KCl is an important component of the fertiliser, elevated concentrations of chloride may also occur.

The risk of groundwater pollution by nitrate depends on the interaction of the nitrogen loading and the vulnerability of the aquifer. The term 'vulnerability' can be defined as "the intrinsic properties of the strata separating a saturated aquifer from the land surface which determine the sensitivity of that aquifer to being adversely affected by pollution loads applied at the surface" (Schmoll et al, 2006).

To illustrate this rather difficult concept, a useful approach to thinking about groundwater pollution is shown in Figure 2. The vulnerability of the groundwater and receptor (which could be a well, borehole, spring, river or wetland) depends on the properties of the soil, the unsaturated zone and the saturated zone in the pathway. These properties determine the ability of water and pollutants to move from the surface to the receptor through the pore spaces and/or the fractures in the aquifer. Moreover, the properties of the materials along the pathway also determine the potential for attenuation of pollutants by processes such as sorption, degradation, ion exchange, filtration and precipitation (Morris et al, 2003). These processes are most active and effective in the soil where biological activity, organic matter and clay particles are usually much more abundant than at greater depths. The soil layer is therefore a key first line of defence against agricultural pollution, whereas it (and sometimes part of the unsaturated zone too) is often removed in urban, industrial and mining pollution scenarios (Foster et al, 2000).

The concept of groundwater vulnerability has gained broad acceptance amongst hydrogeologists, who use it in practice to describe in map form the degree of vulnerability as a function of hydrogeological conditions. Various approaches to defining and depicting vulnerability have been devised (Vrba and

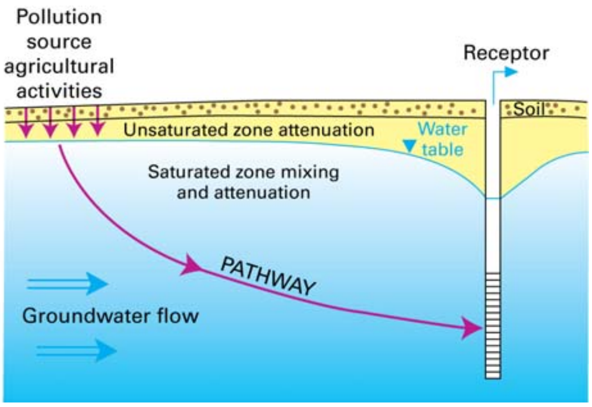

Figure 2. Source, pathway receptor model of groundwater pollution

Zaporozec 1994; Schmoll et al 2006), and it is worth gaining an appreciation from the literature of which types of aquifers and which hydrogeological settings are likely to be vulnerable (Morris et al, 2003).

Thus, areas underlain by thin permeable, aerobic soils and a permeable aquifer with a shallow water table will be especially vulnerable – fractured coastal limestones and coarse-grained alluvial aquifers in valleys and coastal plains. Because groundwater is readily available and the flat, low-lying land is suitable for cultivation, coincidence of the pressures of substantial and growing populations, intensive agriculture and vulnerable aquifers is common, as in India, Sri Lanka, China, Mexico, Egypt, and many other places. In Sri Lanka (Figure 3), irrigated triple cropping of onions on coastal sand dunes almost devoid of soil is sustained by applications of up to 500 kg/ha of nitrogen each year. Leaching of nitrate and re-cycling of chloride produces elevated concentrations of these parameters compared to neighbouring uncultivated land or less intensive, mixed farming (Figure 3).

The pollutant loading is defined principally by the types and quantities of fertiliser used. A useful summary of the chemical compositions and nutrient contents of various organic and inorganic fertilisers is provided by Schmoll et al, (2006). The type of cropping and irrigation regime also influences the risk of pollution. Nitrogen applications to relatively short duration crops, e.g. vegetables or wheat are likely to produce greater leaching losses than continuous crop cover e.g. sugar cane, citrus groves or coffee plantations. Even where high fertiliser applications are made, losses beneath wet rice cultivation are likely to be low as a result of volatile losses and denitrification in the waterlogged, anaerobic soil (Zhu et al, 2003). The nitrogen loading will be greatest where cultivation is intensive and double or triple cropping is practised. Especially high nitrogen leaching can occur from soils where irrigation is excessive and not carefully controlled.

The consequence of the frequent conjunction of intensive cultivation and vulnerable aquifers is that

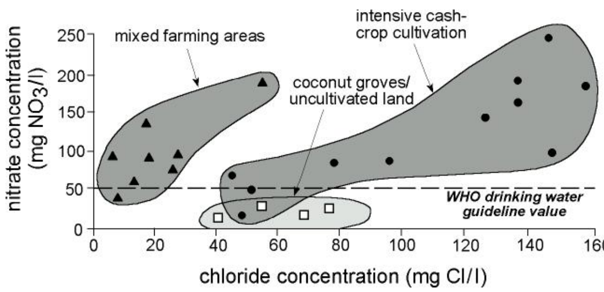

Figure 3. Quality of shallow groundwater in a coastal sand aquifer, Sri Lanka (Mubarak et al, 1992)

very high nitrate concentrations ranging up to several times the WHO Drinking Water Guideline value of 50 mg/l are widely observed in groundwater, as reported in the companion information sheet on nitrate.

Other sources of nitrate associated with farming

There are other sources of nitrate associated with farms and farming. Discharge of effluent from intensive livestock units and leachate from manure stores and leaking slurry pits, and slurry or manure spreading on land as organic fertiliser can all be sources of groundwater pollution (Cho et al. 2000). Use of partially treated or untreated wastewater in irrigation can also cause deterioration in the quality of the underlying groundwater (Al-Kharabsheh, 1999). The scale of operations is broad, ranging from local, peri-urban, often informal irrigation of small gardens with collected but untreated domestic wastewater, through large canal-commanded irrigation schemes using untreated or partially treated wastewater, to highly sophisticated, heavily controlled and managed soil aquifer treatment schemes in which the re-abstracted groundwater is of much improved quality.

Pesticides

All pesticides have the potential to pose a health hazard because they are chemically designed to be toxic and persistent enough to control weed, insect or fungal pests. Prior to the early 1980s, little thought was given to the possibility of groundwater pollution by pesticides, since agricultural scientists believed they would be attenuated either by sorption or degradation in the soil or by volatilisation. However, growing awareness of the processes of nitrate leaching, and the rapid increase in pesticide usage led to greater concern that pesticide residues could reach groundwater at problematic concentrations (Foster et al, 1991).

Although developing countries as yet use only a small proportion of the global total of pesticides applied, usage has increased rapidly, especially in the most quickly growing economies. Herbicide usage dominates in the temperate climates of Europe and North America, but insecticides are more widely used elsewhere. Pesticide usage is concentrated on a small number of crops; wheat, maize and soya bean and the plantation crops cotton, sugarcane, coffee, cocoa, pineapple, bananas and oil palm. Application rates are usually in the range 0.2 to 10 kg/ha of active ingredient (Morris et al, 2003). Usage on vegetables in particular is becoming more widespread, often with the highest application rates.

As for nitrate, the risk of pollution of groundwater depends on the interaction between pollutant load and vulnerability – the capacity to attenuate pesticides along the pathway through the soil and underlying aquifer (Figure 2). Again, as for nitrate, the mode of application is important. Soil-applied pesticides are more likely to be leached than leaf-acting compounds sprayed onto the plants.

The most important attenuation processes for pesticides – sorption, volatilisation and degradation are most active in the soil, with its high content of clay and organic matter and active microbial populations (Morris et al, 2003). Once below the soil, the very small proportion of the applied pesticide that does leach is likely to be more mobile and persistent; the attenuation processes, although still present, being much less active. In the saturated zone, dilution will be the main attenuation mechanism which will help to limit the concentrations of pesticides in groundwater arriving at wells or boreholes.

In general the same aquifers and conditions – thin permeable soils, permeable aquifers and shallow water tables are likely to be vulnerable for pesticides as for nitrate. Assessing the risk of pesticide pollution has, however, one crucial difference – the large numbers of pesticide compounds in common usage all have their own specific physico-chemical properties. The most important of these are solubility, partition coefficients and half-lives, and groundwater pollution risk assessment methods and pesticide leaching models using published information for these have been developed (Vanclooster et al, 2000; Dubus et al, 2003). Application of such leaching risk assessments is, however, made less certain for tropical environments because the published data refer almost entirely to standard, fertile, clay-rich soils in temperate climates, and there may be little equivalent data for more permeable soils and tropical conditions.

Study of pesticides in groundwater has been hampered in the past by the wide range of compounds in common use, the care required in sampling to avoid contamination or volatile losses and the low threshold values at which concentrations need to be determined. The EC Drinking Water Directive sets a very stringent maximum admissible concentration of 0.1 μg/l for any compound, whereas the WHO Drinking Water Guidelines (WHO, 2004) and US EPA (2009) set concentrations which are toxicity-based and hence vary from compound to compound. With greater routine sampling, and improved levels of analytical detection, more and more pesticides and their metabolites are being observed in groundwaters (Kolpin et al, 1996) and, as laboratory capabilities become stronger, in developing countries too.

On the Kalpitiya Peninsula in Sri Lanka, carbofuran was applied at 6 kg active ingredient /ha to

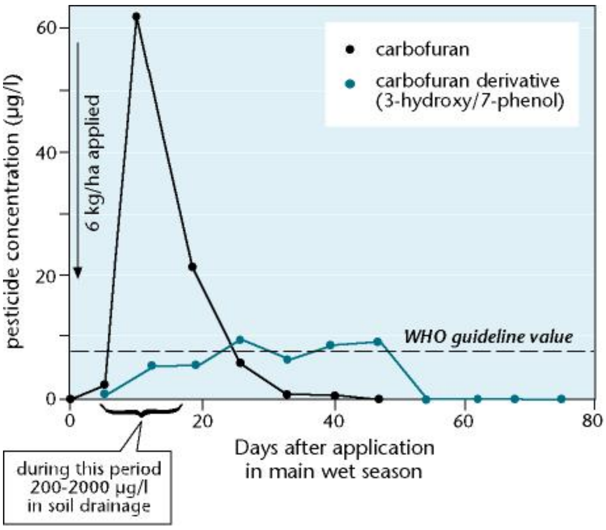

Figure 4. Carbofuran and one of its derivatives in shallow groundwater, Sri Lanka (Mubarak et al, 1992)

horticultural crops. This pesticide is highly mobile, and rapidly leached from the soil at concentrations of 200 to 2000 µg/l in soil drainage measured in lysimeters. Peak concentrations in excess of 50 µg/l were seen in the underlying shallow groundwater within 20 days of application (Figure 4). The carbofuran was, however, subject to rapid degradation leaving its more persistent metabolite carbofuran-phenol, which remained in groundwater for more than 50 days (Figure 4).

In general, measured concentrations in groundwater are in the same range as standards or guideline values (0.1-10 µg/l). Higher concentrations are associated with double or triple cropping and frequent heavy applications of the same compound (Morris et al, 2003), or indicative of point source pollution from mixing, storage, spillage or disposal rather than conventional field applications. Particular problems arise in developing countries where stockpiles of obsolete but highly toxic and persistent organ-chlorine pesticides present major difficulties of safe storage and disposal (Jovanovic, 2006).

Investigating and monitoring agricultural impacts

Impacts on groundwater resources

Investigating the potential impacts of agricultural water usage requires a sound conceptual model of the groundwater flow system, including sources and locations of natural and induced or imposed recharge and discharge. The former could include surface water in rivers, canals, irrigated land and drainage channels. Quantifying recharge from all sources is notoriously difficult and needs detailed investigation (Simmers, 1997). Determining the total abstraction for all uses is also difficult, but this needs to be done so that the balance of resources between replenishment and abstraction can be estimated.

Monitoring of the impacts of agricultural abstraction and the effectiveness of measures to control abstraction, alleviate the impacts or restore conditions requires the regular measurement of groundwater levels. Long term declines in levels are the best indication that heavy groundwater usage is affecting groundwater resources, and that the situation should be further investigated.

Impacts on groundwater quality

Evaluating the actual or potential impacts of agrochemical use on groundwater quality requires knowledge of the pollution load and the nature of the pollutant pathways. A first impression of the pollution potential can usually be obtained from observing land use and agricultural activities; if these are intensive further work is probably needed. This may require detailed surveys of fertiliser and pesticide use to determine the source term and pollutant load, using prepared survey forms (Chilton et al, 1991; Zaporozec, 2002) adapted to the local situation, or simple checklists (Schmoll et al, 2006). This information can then be compared with the aquifer vulnerability, ease and speed of water movement along the pathway and degree to which the pollutants might be attenuated to establish the risk of groundwater pollution.

A detailed description of groundwater quality monitoring is beyond the scope of this short note and reference should be made to suitable guidance material on the technical aspects of sampling (USGS, 2006) and the more strategic aspects of establishing suitable programmes (EC, 2007). Both of these are available from their respective websites.

Nitrate is relatively simple to measure and elevated concentrations are the most common indication of agricultural impact on groundwater quality. However, it is common to find agriculture, especially small scale but intensive urban and peri-urban horticulture intermingling with communities using unsewered sanitation and keeping livestock. In order to correctly target control measures, it may be essential to distinguish the sources of nitrate by chloride/nitrate ratios, comparison with other indicators, or nitrogen isotopes, as outlined in the nitrate sheet. To select pesticides for monitoring, the survey is essential to identify which are most heavily used and which, from their physico-chemical properties, are most likely to be leached to groundwater (Chilton et al, 1991).

Monitoring of agricultural impacts on groundwater quality needs to be focused on the upper part of the pathway in Figure 2 ie at or close to the water table beneath the intensively cultivated land. Existing domestic or irrigation wells or boreholes can be used, or if equipment is available and funds permit, new shallow monitoring boreholes can be constructed and sampled. Comparison of the results with those from suitable monitoring points in adjacent uncultivated areas (Figure 3) but with the same hydrogeological conditions can indicate the severity of the impact of the intensive agriculture.

Monitoring is also needed to observe the effectiveness of the control measures mentioned in the nitrate note. In developing countries, options for control which cut losses by means of restrictions on the timing and amounts of fertiliser application, using improved crop strains, using cultivation practices and irrigation methods which promote more effective use of nutrients (Shrestha and Ladha, 2002) may have serious economic consequences for the farmers involved, and may be difficult to monitor and enforce.

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British Geological Survey 2009