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A preliminary study on the identification of the sources of nitrate contamination in groundwater in Malta. Conclusions and policy implications

Groundwater Resources Programme

Commissioned Report CR/08/160

BRITISH GEOLOGICAL SURVEY

GROUNDWATER RESOURCES PROGRAMME

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Keywords

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Front cover

View of typical Maltese landuse with small agricultural plots near Bahrija Spring

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Foreword

This report is the final product of a study by the British Geological Survey (BGS) into the sources of nitrate in the groundwater of the Islands of Malta. It summarises the main findings of the project and sets the management of groundwater nitrate in Malta in the European regulatory context.

The study was funded by the Technical Assistance Programme under the Rural Development Programme for Malta 2004-2006. The start-up meeting was in December 2007 and the project was completed in March 2009.

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1 Introduction

1.1 BACKGROUND

The island of Malta has already been comprehensively studied over a long period from the perspective of its limited water resources (Tricker, 1977, Mangion and Sapiano, 2005). The scale of high groundwater nitrate concentrations is already well defined, and Malta has been described by the European Environment Agency as having the most widespread groundwater nitrate problem among the EU member states (EEA, 2008).

Generally, it is difficult to relate nitrate found in groundwater directly to any one of the possible sources of nitrate - agriculture, industry, sewered or unsewered sanitation. Experience suggests that this is even harder to do in island states where the pressures associated with the high population density and limitations on land availability produce a complex land-use scenario in which sources of nitrate are invariably mixed together.

Stable isotope techniques have been a valuable tool in hydrochemical research for over 30 years, and have been widely used in studies of the source, fate and behaviour of anthropogenically-derived contaminants, of which nitrate is probably the most important. $^{15}N/^{14}N$ measurements, in partnership with other chemical data, can provide information on both the sources of nitrate contamination, and the processes involved in transformations (notably denitrification). Other stable isotopes can also be applied to the identification of sources of pollution.

1.2 PROJECT OBJECTIVES

The overall objective of the project is to provide the Malta Resources Authority with the scientific basis to underpin policies and action programmes to address nitrate pollution of groundwater. Within this context, the main aim of the study is to identify the activities which are responsible for the high concentrations of nitrate currently found in the groundwaters of Malta and to evaluate the relative contribution of each activity. The dominant role of these activities may be due either to their widespread distribution in Malta as diffuse pollution sources or to high concentrations of nitrogen present in the local recharge associated with such activities as point sources.

Successful identification of these polluting activities would allow the regulatory agencies of Malta to design targeted Programmes of Measures to control the most important activities. This would have the aim of improving the chemical status of groundwater with respect to nitrate and enable Malta to meet the requirements of the Water Framework Directive, the Groundwater Directive and the Nitrates Directive.

1.3 SCOPE OF REPORT

This report comprises the technical output from Workpackage 3 of the project (Appendix 1). It brings together existing information on nitrate in groundwater in Malta as summarised in Workpackage 1 (Stuart et al., 2008a) and the interpretation of new data collected in this project as part of Workpackage 2 (Stuart et al., 2008b). The results are set in the context of the objectives and requirements of relevant EU legislation.

1.4 HYDROGEOLOGICAL SETTING OF MALTA

The Maltese Islands are composed of two coralline limestone aquifers (upper and lower) of Tertiary age separated by the impermeable Blue Clay (Figure 1). The Lower Coralline Limestone is overlain in part by the poorly permeable Globigerina Limestone. On both Malta and Gozo, the lower aquifer is in direct contact with sea water with a piezometric head controlled by abstraction from the public supply network. On Malta the upper perched aquifer consists of two larger and a series of discontinuous small bodies mainly used for agriculture and secondary purposes. On Gozo the Blue Clay and Globigerina Limestone provide extensive cover to the sea level aquifer but the perched aquifers are very limited in area. On Malta, north of the Pwales Fault (Figure 2), the Blue Clay dips below sea level and the perched groundwater bodies are in contact with the sea at the coast.

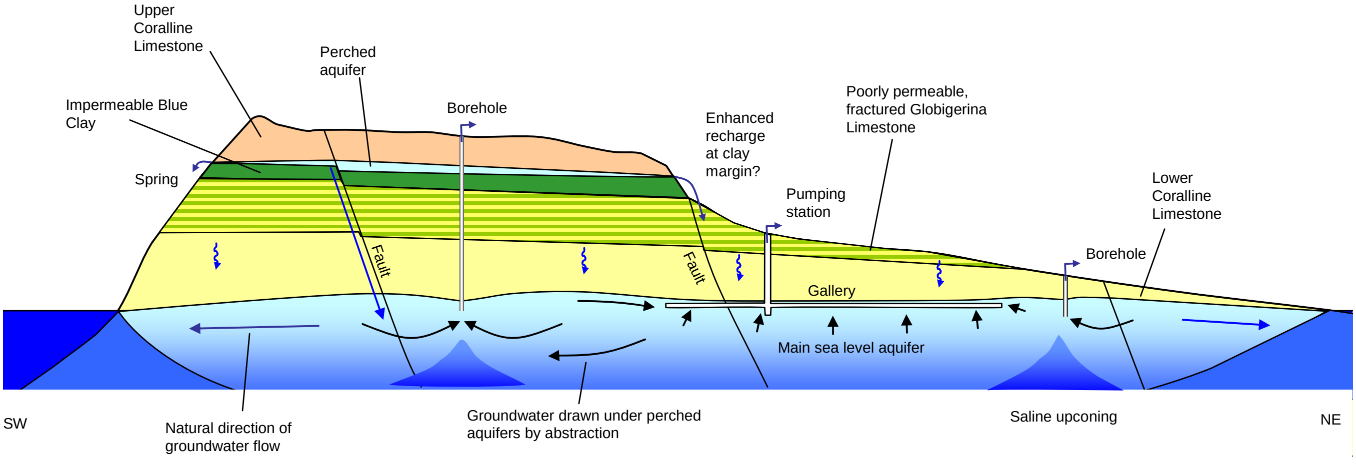


Figure 1 Hydrogeological cross-section of Malta

The climate is typically semi-arid Mediterranean with hot dry summers and mild, wet winters. There is considerable variation in both inter-annual and intra-annual rainfall with a frequent occurrence of low rainfall years. The potential evapotranspiration is high and estimates of effective rainfall are low varying from 95 mm/year to 200 mm/year. The annual replenishment from recharge is low compared to the groundwater storage resulting in likely residence times of decades in the sea level aquifers. Groundwater in the perched upper aquifers leaks to the underlying sea level aquifers.

Both slow infiltration through the pores of the rock matrix and faster movement through fractures and fissures are possible in such limestone aquifers and can result in a range of travel times from the surface to the water table. Travel to the perched aquifers is considered to be relatively rapid as these aquifers respond quickly to rainfall events whereas the Lower Globigerina and Lower Coralline Limestones appear to be massive and any fractures present to be blocked, suggesting infiltration is likely to be predominantly through the porous matrix.

2 Results of study

2.1 SUMMARY OF SAMPLING AND ANALYSIS

Fifty groundwater samples were collected representing the main sea level (MSL) aquifers on Malta and Gozo and the more important of the perched aquifers on Malta. Sample locations were selected to represent the distinct landuse types (urban, agricultural and areas irrigated with treated sewage effluent (TSE)) and proximity to point sources (sewer galleries, cesspits, cattle and pig farms). Landuse classifications were provided by the MRA as part of the sampling strategy (Stuart et al., 2008a).

Samples were analysed for nitrogen species and a wide range of indicators and potential co-contaminants: major and minor ions; trace elements; total organic carbon; stable isotopes of nitrate, carbonate, sulphate and water; fluorescence; faecal coliforms and residence time tracers.

2.2 WATER QUALITY SETTING

Water quality in the MSL aquifers is controlled by water-rock reactions with the limestone matrix and by saline intrusion, as well as by pollution from the surface. The major ion chemistry of groundwater suggests the incongruent dissolution of the limestone matrix with increasing residence time, with enhanced concentrations of Mg and Sr and an increase in the δ13C signature in the MSL aquifers on both Malta and Gozo relative to the perched aquifers. Groundwaters from areas of the aquifer confined by the Middle Member of the Globigerina Limestone Formation also show enhanced trace elements, such as Li and Mo. Saline intrusion into groundwater due to abstraction occurs in the MSL aquifers and affects the major ion composition and also a range of trace elements. Many of these solutes are also present in sewage and animal waste and they would otherwise be useful indicators. Sulphate $\delta^{34}S$ signatures are consistent with an increasing percentage of marine-derived sulphate in the MSL aquifers.

2.3 PRESENT NITRATE CONCENTRATIONS IN GROUNDWATER AND TRENDS

Nitrate concentrations throughout this report are expressed in mg/l as nitrate to be consistent with concentrations used in EU water quality directives.

In the perched aquifers the nitrate concentration was confirmed as being high with a median of 164 mg/l with a wide range of values: 41 mg/l to 411 mg/l (Figure 2). The majority of the sources are no longer used due to poor water quality. In the MSL aquifers concentrations are more-moderate and more consistent ranging from 10 mg/l to 159 mg/l with a median value of 62 mg/l in the Malta aquifer and from 24 mg/l to 106 mg/l with a median value of 44 mg/l in Gozo.

Nitrate concentrations have risen significantly in the perched aquifers over the last 10-20 years. Groundwater management measures have lowered the concentrations of

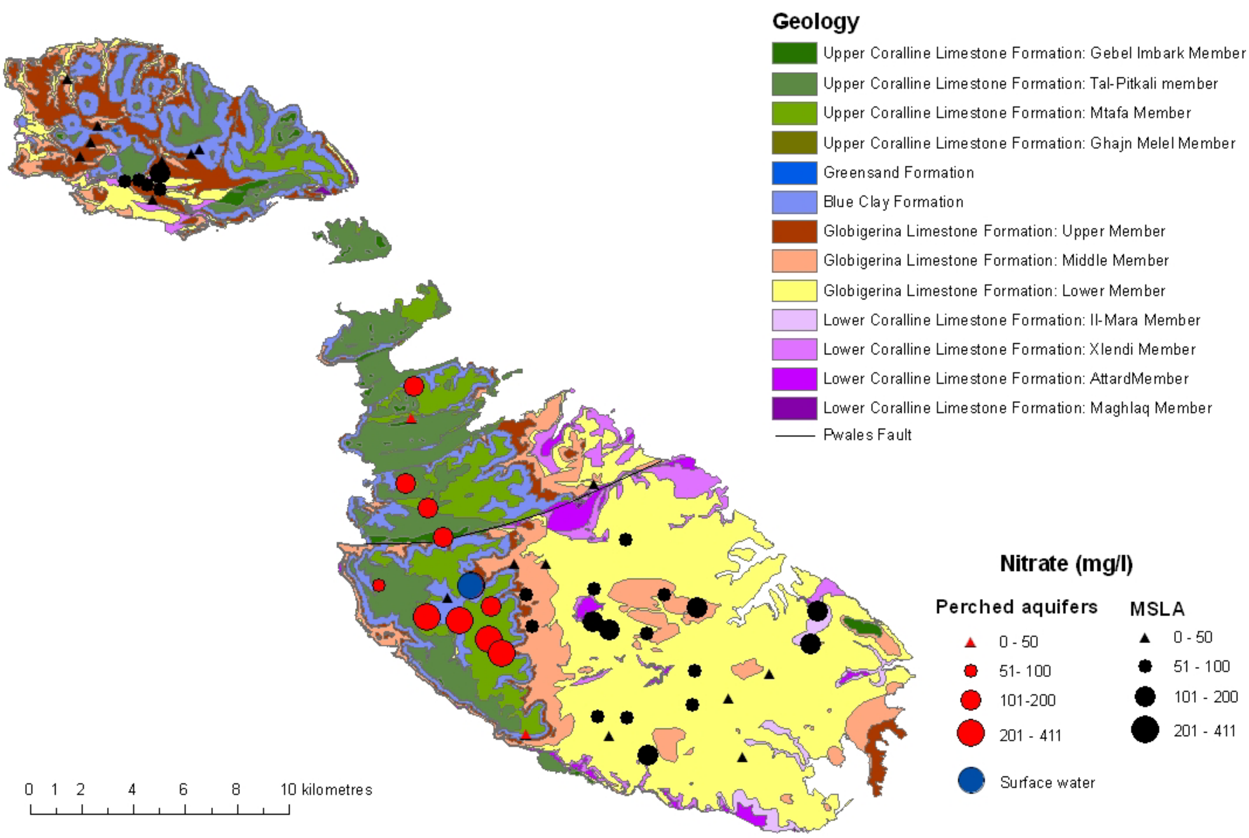


Figure 2 Distribution of nitrate concentrations in the perched and MSL aquifers

chloride in the MSL aquifers but data from two sites included in both the 1991 BRGM and the present studies indicates that groundwater nitrate has been stable over the last 30-40 years. This suggests that nitrate has reached equilibrium in the MSL aquifers. The area of agriculture has contracted over the last 40-50 years as the urban area has increased and this change of landuse will have affected the source and concentration of nitrate inputs to groundwater.

2.4 NITRATE ISOTOPE STUDY

Stable isotope ratios of nitrate ($^{15}N/^{14}N$ reported as $\delta^{15}N$ values, and $^{18}O/^{16}O$ reported as $\delta^{18}O$ values) are invaluable tools for helping to understand the geochemical origin and evolution of nitrate. The $\delta^{15}N$ and $\delta^{18}O$ values reflect the original sources of nitrogen (e.g. fertilizer, animal waste, etc.) and oxygen (water, O_2), as modified by any subsequent chemical transformation (e.g. denitrification).

All but two of the groundwater samples (i.e. 96% of the groundwater analysed) had nitrate $\delta^{15}N$ values in the range +7.2 to +13.2‰, and $\delta^{18}O$ values in the range +2.8 to +6.4‰. The two exceptions, both in the Malta MSL aquifer, displayed higher values in which the increases in $\delta^{15}N$ and $\delta^{18}O$ were in the proportion 2 to 1 – a characteristic of denitrification. The data therefore point to a very limited occurrence of denitrification, with no evidence for it being responsible for the lower concentrations of nitrate in the Gozo MSL aquifer.

The $\delta^{15}N$ and $\delta^{18}O$ values of the groundwaters, excluding the two denitrified samples, are shown in Figure 3 together with the values for potential sources of nitrate: fertilizers, sewage, animal wastes, and soil. Groundwaters from all three aquifer types – the perched and MSL aquifers on Malta, and the MSL aquifer on Gozo – displayed very similar ranges of $\delta^{15}N$ and $\delta^{18}O$ values, and are therefore not distinguished from one another in Figure 3. Whilst these similarities could be coincidental, they suggest that the source/s of nitrate is the same in all three aquifers, and that differences in nitrate concentrations relate to different hydrology.

The $\delta^{15}N$ and $\delta^{18}O$ values of potential nitrate sources in Figure 3 are based on measured values of nitrate in fertilizers, and on calculated values for nitrate derived from nitrification of ammonium in fertilizer, nitrification of ammonium in sewage, nitrification of ammonium in animal waste, and nitrification of organic nitrogen in soils. These calculated values assume that the nitrate formed by nitrification has: 1) $\delta^{15}N$ values the same as the $\delta^{15}N$ values measured for the ammonium or organic N; and 2) one of the three NO_3 oxygens derived from atmospheric O_2, with $\delta^{18}O$ = +23‰, and two of the oxygens from water whose $\delta^{18}O$ value was measured.

If the application of fertilizer is immediately followed by heavy rainfall, the fertilizer can comprise a direct supply of nitrate to groundwater - either directly as nitrate, or following nitrification of ammonium. The expected $\delta^{15}N$ and $\delta^{18}O$ values of this nitrate, based on analyses of Maltese inorganic fertilizers, are shown in Figure 3 and display a similar range to fertilizers globally. Both low $\delta^{15}N$ and high $\delta^{18}O$ values rule out fertilizer nitrate as a direct source, and the very low $\delta^{15}N$ value of fertilizer ammonium also makes it an unlikely source of nitrate in the groundwater.

Solid and liquid animal waste was collected from eleven sources, and had very high concentrations of ammonium with $\delta^{15}N$ values of +2.1 to +6.4‰ (mean +4.3‰, n = 9) for

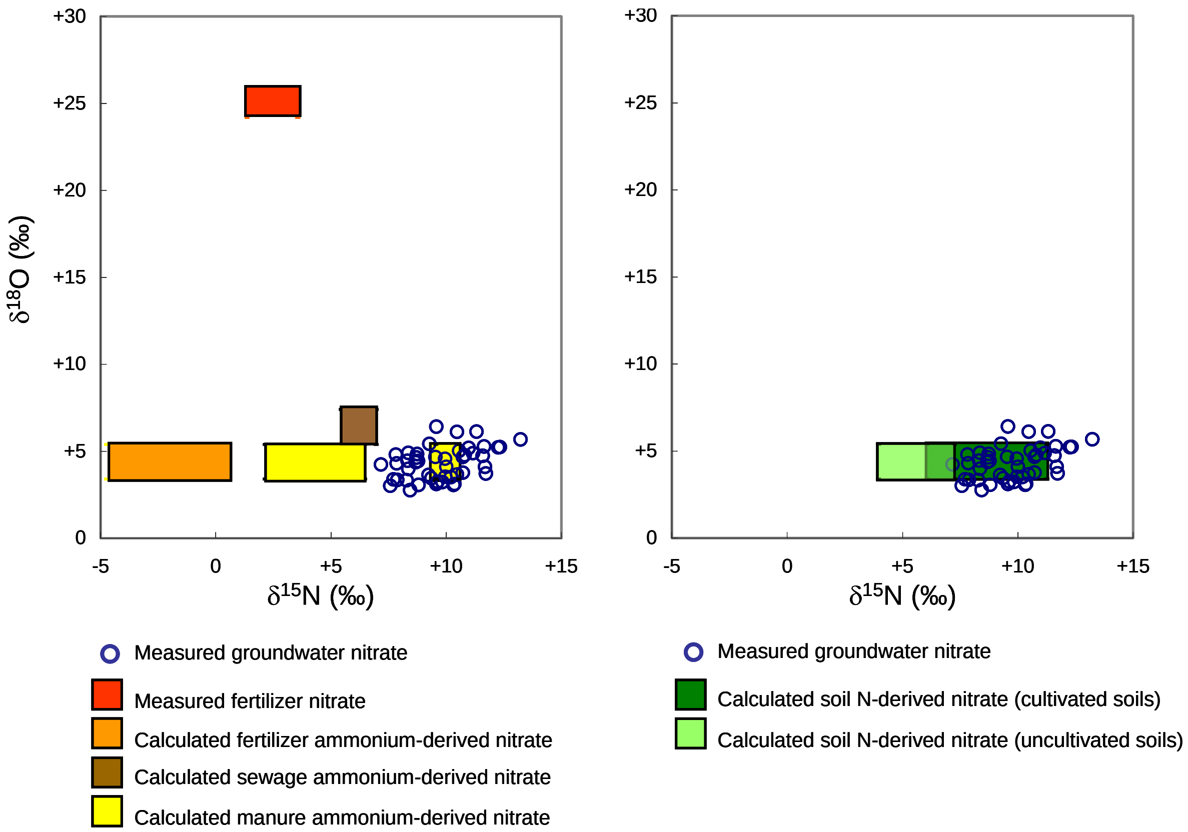


Figure 3 Summary of $\delta^{15}N$ and $\delta^{18}O$ in nitrate in groundwater and various potential nitrate sources

all five slurry and four solid wastes, and +9.8‰ and +10.1‰ for two solid wastes. The relatively low $\delta^{15}N$ values of most of the samples compared to groundwater suggests that most animal waste would not be a source of nitrate in the groundwater, whilst the two samples with higher $\delta^{15}N$ values could constitute such a source (Figure 3). This wide range of values exemplifies the difficulty of uniquely characterising animal waste, whose $\delta^{15}N$ values are greatly affected by loss of gaseous nitrogen, chiefly by ammonia volatilisation, during decomposition during storage. The process leads to an increase in the $\delta^{15}N$ value of the residual nitrogen converted to nitrate. Thus the animal wastes constituting most of the samples ($\delta^{15}N$ = +2.1 to +6.4‰) could, if subject to further ammonia loss during storage, produce nitrate with a $\delta^{15}N$ value in the range of that for groundwater.

Sewage samples (which includes samples from sewers and cesspits) contained very little nitrate, but had high ammonium concentrations which could be oxidised to nitrate if sewage leaked into an aerobic environment (the unsaturated zone or O_2-rich groundwater). The $\delta^{15}N$ and $\delta^{18}O$ data, however, do not support a sewage source for the nitrate in the groundwater (Figure 3). Unlike animal wastes on the surface, sewage constrained in sub-surface environments (sewers and cesspits) has limited opportunity to lose ammonia by volatilisation, and thereby increase its $\delta^{15}N$ value. It is therefore difficult to envisage a mechanism whereby sewage ammonium with a narrow range of $\delta^{15}N$ values between +5.4 to +6.9‰ could produce nitrate with $\delta^{15}N$ values in the range of those for groundwater (Figure 3). The presence of desalinated seawater in modern sewage waters (and TSE), moreover, will raise the $\delta^{18}O$ value of the H_2O contributing to two of the oxygens in the NO_3 molecule. The calculated $\delta^{18}O$ value of sewage-derived nitrate is therefore somewhat higher than the $\delta^{18}O$ value of groundwater nitrate (Figure 3), which also tends to mitigate against a (modern) sewage source for the nitrate. The $\delta^{15}N$ values of the soil organic nitrogen (+3.9 to +11.2‰, average +8.5‰) are at the upper end of the normal range for soils globally. The three soil samples from non-agricultural or abandoned sites tended to have lowest $\delta^{15}N$ values, and it may be that the generally high values for Maltese soils results from their extended cultivation. Considering organic nitrogen in the cultivated soils only (+6.0 to +11.2‰, average +9.1‰), and assuming that soil nitrification produces nitrate with similar $\delta^{15}N$ values, these values coincide very closely to those of nitrate in the groundwater (+7.2 to +13.2‰, average +9.7‰).

Overall, the $\delta^{15}N$ and $\delta^{18}O$ data favour a process whereby groundwater nitrate has been derived by leaching of nitrate formed by nitrification in soils; though derivation from stored animal wastes cannot be discounted. In the case of a soil nitrate source it must be emphasised that the isotope data do not rule out inorganic fertilizers and/or animal wastes as the *original* source of the nitrogen. The data are compatible with a process whereby nitrogen from inorganic fertilizers and/or animal wastes is assimilated into the soil organic nitrogen pool, and takes on the isotopic composition of this pool during the cycling of nitrogen attendant on cultivation, before nitrification and leaching to the underlying groundwater.

2.5 DENITRIFICATION

Isotopic evidence for the occurrence of denitrification in groundwater was limited to an area with urban landuse in the southeast of Malta where the main aquifer is confined beneath a perched aquifer. No current areas of reducing groundwater were detected although CFC data suggest that conditions may be, or have been, slightly reducing at some sites and a few show significant concentrations of nitrite. It is considered unlikely that nitrate in Gozo groundwater has been removed by denitrification in the confined aquifer and the lower concentrations found in groundwater must be due to protection of the aquifer by the overlying confining layers.

2.6 CO-CONTAMINANTS

The nitrate co-contaminant data are difficult to interpret. Animal wastes, and to a more limited extent, sewage do contain elevated concentrations of trace elements but these were not found to be diagnostic. All landuses appeared to be associated with increased trace element concentrations relative to the background Annunzjata spring. Co-contaminant data from the perched aquifers suggest the derivation of nitrate from animal farming and urban areas rather than agriculture or cesspits, in contrast to the nitrate isotopic data which suggest leaching from cultivated soils and probably animal wastes. Trace elements are also affected by saline intrusion and by residence time in the MSL aquifers. A number of trace elements are associated with groundwater from beneath the perched aquifers, resulting possibly from leakage of water from the overlying Blue Clay.

In the perched aquifers it is clear that urban areas and animal farming, particularly cattle, are having an impact on the dissolved organic carbon in the aquifer as measured by fluorescence. For the sea level aquifers the pattern is less distinguishable, but the impact of cattle farms on the Malta aquifer can be seen with an elevated protein-derived content. On Gozo, the pattern is confused with high protein-type fluorescence from under the perched aquifer and at agricultural sites. The ratio of protein type to soil-derived fluorescence can perhaps more securely indicate the presence of animal derived organic carbon. The highest ratios were measured in urban springs and boreholes. The fluorescence index, which gives an indication of the microbially derived organic content, also provides a mixed picture with impact from urban areas and pig farms in the perched aquifer, urban areas in Malta and most sites on Gozo.

E. coli were found in the perched aquifers in all but one sample and all landuses gave high results. In the main aquifer *E. coli* were detected in 6 out of the 24 borehole samples, generally with agricultural, urban/sewer landuses or TSE irrigation. On Gozo, *E. coli* were detected in 3 out of 14 sites with cattle, pig and agricultural uses. It is likely that the long travel times in the main aquifers limit the use of microbiological indicators for identifying potential sources of contamination.

2.7 TIMESCALES OF GROUNDWATER FLOW

The different flow regimes in the perched and MSL aquifers, particularly travel time, need to be taken into account in the interpretation of other indicators. The perched aquifers have very limited unsaturated and saturated thickness and presumably short residence times. The primary porosity of the Upper Coralline Limestone is high so there is likely to be recharge by slow matrix flow as well as rapid recharge from fissures and fractures. Other than a contribution to recharge by desalinated seawater, these aquifers are unaffected by saline water.

In the Malta MSL aquifer the unsaturated thickness is large as water levels are close to sea level. Parts of the aquifer are capped by the perched aquifers and more extensively by the relatively impermeable Globigerina Limestone. The limited detection of coliforms suggests little rapid recharge from the surface. Transmissivity is low and tritium and CFC data suggest that saturated zone travel times are in the range 15-40 years. Major ion chemistry shows the ingress of saline water due to abstraction. On Gozo the aquifer is similar but is more-extensively capped by impermeable Blue Clay. CFC data show the saturated travel time is from 25 years to possibly more than 60 years. Seawater intrusion is also more widespread. In both aquifers it is likely that nitrate and other solutes are retained in porewater and are moving slowly through the matrix, providing a long-term source.

There are significant differences in nitrate concentrations between the various aquifers with the perched aquifers having the highest concentrations and the Gozo MSL aquifer the lowest. This may be due to:

- difference in recharge concentration due to a long-term reduction in the area of agricultural land or small-scale landuse patterns;
- the dilution of modern recharge by older, low-nitrate water;
- differences in recharge distribution due to less permeable strata present at the surface impeding or delaying recharge and possibly enhanced recharge at the edge of such areas from run-off;
- differences in unsaturated zone thickness and therefore speed of arrival of modern concentrations at the water table.

There is little evidence for rising concentrations of nitrate in groundwater in the Malta MSL over the last 30 to 40 years and this would suggest that the system has reached at least a temporary equilibrium between surface sources and abstracted concentrations. The travel time may be very long in some parts of the MSL aquifers and there may still be high nitrate water yet to arrive at the pumping stations and boreholes, eventually bringing nitrate concentrations to the levels currently seen in the perched aquifers.

The recharge areas on Gozo look to be quite limited and it may be that the MSL aquifer receives a significant proportion of recharge from slow, and therefore old, infiltration through the overlying less-permeable strata.

2.8 SUMMARY OF GROUNDWATER NITRATE SOURCES

2.8.1 Nitrogen isotopes

Nitrogen isotopes showed that direct inputs of fertilizer or sewage derived nitrate were not major contributors to groundwater nitrate (Table 1). Leaching of nitrate from cultivated soils was likely to be the most important source, though derivation from animal wastes could not be discounted (Figure 3). There was little evidence of denitrification so the lower concentrations found in the Gozo aquifer must be due to the protection of the groundwater by the extensive Blue Clay cover on the island.

Table 1 Summary of likely importance of potential sources of groundwater nitrate

Sources	
Cultivated soils	●
Animal wastes	●
Bacterial nitrification	●
TSE	•
Sewage direct leaching	•
Fertilizers direct leaching	•

2.8.2 Nitrate and co-contaminants

The study has shown that in the perched aquifers the lowest nitrate concentration is about 40 mg/l at a site with natural vegetation. This gives the best available estimate of the background quality, that unaffected by anthropogenic activity, although even this is likely to reflect the long history of human activity in Malta. The background concentration is impacted by agricultural and urban landuses with cesspits having the least effect and cattle farming the most. This is corroborated by measurements of major and trace elements, dissolved organic carbon and by groundwater fluorescence (Table 2).

In the Malta MSL aquifer the pattern is different. The sites abstracting from below the perched aquifer have similar average nitrate values to the background (median concentration 42 mg/l). Concentrations overall are more moderate than in the perched aquifers with the highest concentrations found in the TSE irrigation area (140 mg/l) and possibly those associated with pig farms (102 mg/l). The results from fluorescence suggest the impact of cattle farms, urban areas and TSE irrigation areas. Trace metals were elevated in boreholes in urban areas not regularly used at the time of sampling.

On Gozo average concentrations are much lower with the highest values related to urban areas (20 mg/l). Again the lowest concentration was observed from beneath a perched aquifer. Low concentrations of nitrite and ammonia are also detected. Fluorescence measurements suggest a non-soil origin for the organic content of groundwater.

2.9 IMPLICATIONS FOR NITRATE MANAGEMENT

The results are not consistent with the tentative estimate of nitrogen leaching made by the MRA from agricultural activities alone using records and stock numbers (Stuart et al., 2008a). This suggested that in the order of 60% of leached nitrogen may come from animal waste stored on the ground, with only a further 20% from manure applied to crops and about 10% from synthetic fertilizers. The present study does not allow the contribution from sources other than cultivated soils to be estimated.

Another key finding of the study has been the confirmation of the long saturated zone residence times in

Table 2 Summary of co-contaminant and nitrate concentration ranges for different landuses

Aquifer	Landuse classification*	Metals	Organic carbon	Fluores-cence	E coli	Co-contam summary	Nitrate
Perched	Background	·	·	·	·	·	·
	Cesspits	·	·	·	•	·	·
	Urban	●	•	●	●	●	•
	Cattle	●	●	●	●	●	●
	Pig	•	●	●	●	●	●
	Agriculture	•	•	·	●	•	•
Malta MSL	Under perched	•	·	·	·	·	·
	Sewers	●	·	·	·	•	•
	Urban	●	●	·	•	•	•
	TSE	•	●	·	•	•	●
	Cattle	·	●	•	·	•	•
	Pig	·	·	·	·	·	●
	Agriculture	·	•	·	·	·	·
Gozo MSL	Under perched	•	●	•	·	•	·
	Urban	•	•	·	·	•	●
	Cattle	•	·	•	·	•	•
	Pig	●	·	•	·	•	·
	Agriculture	•	•	•	·	•	•

* primary landuse within 100 m of wellhead ● = high, • = medium, · = low or undetected

the MSL aquifers. This has important implications for any relationship between present-day activities and groundwater concentration and it would appear unrealistic that a clear pattern could be anticipated. The lack of widespread rapid pathways from the surface to the water table as deduced from microbiological evidence suggest that a major part of infiltration may occur by relatively slow flow through the aquifer matrix. The travel time for nitrate from the surface to an abstraction point could be several decades at some sites.

The nitrate stored in aquifer porewaters will act as a secondary source for a long period even if surface applications were to cease completely. If disposal and management of solid animal wastes were to be targeted as the most important source of nitrate contamination it is unlikely that significant improvements in groundwater quality in the MSL aquifers will be seen for a considerable period.

3 European groundwater quality directives

3.1 RELEVANT EU DIRECTIVES

3.1.1 Nitrates Directive

In response to the concern in Europe about the widespread occurrence of high nitrate concentrations in groundwater, and reflecting the growing evidence of elevated concentrations coming from Drinking Water Directive reporting, in 1991 Europe adopted the Nitrates Directive (91/676/EEC). This is an environmental measure designed to reduce pollution of water by agriculture and to prevent such pollution occurring in the future (Article 1). The Directive requires Member States to:

- establish as polluted all groundwater which already has, or trends indicate could reach, a concentration of 50 mg/l nitrate if no action is taken to reduce pollution (Article 3);
- designate as Nitrate Vulnerable Zones (NVZs) all land draining to waters that are affected by nitrate pollution (Article 3);
- establish a voluntary code of good agricultural practice to be followed by all farmers throughout the Member State (Article 4);
- establish information and training programmes for farmers (Article 4);
- regulate the capacity and construction of storage vessels for livestock manure (Annex II);
- establish an Action Programme of measures for the purposes of tackling nitrate loss from agriculture (Article 5). The Action Programme should be applied either within NVZs or throughout the whole country;
- review the extent of their NVZs and the effectiveness of their Action Programmes at least every four years and make amendments if necessary (Article 6).

The Action Programme referred to above must contain at least the measures prescribed in Annex III of the Directive related to fertilizer and manure spreading and manure storage (Table 3).

3.1.2 Water Framework Directive

The Water Framework Directive (2000/60/EEC) is intended to provide a comprehensive strategy for managing the water environment. Since it came into force in 2000, it's obligations for Member States and the clear timeframe for achieving and reporting on them already indeed sets the broad policy framework for managing the water environment. As stated in the ToR, the overarching policy objective is for Member States to achieve good quantitative and qualitative status for water by 2015 (Figure 4).

As a basis for managing and reporting on the water environment, Article 5 of the WFD requires Member States to analyse the characteristics of each River Basin District and to review the impacts of human activities on the status of surface waters and on groundwater. Annex II sets out the requirements for characterisation of surface water bodies and the initial and further characterisation of groundwater bodies and the review of the impacts of human activities. Initial characterisation assesses the uses of groundwater bodies and the degree to which they are at risk of failing to meet the objectives for each groundwater body under Article 4. The purpose of further characterisation is to refine the assessment of risk, to review the impact of human activities and to identify measures to be taken under Article 11 (the programme of measures).

For groundwater, Article 4(1) (b) sets out the environmental objectives to be achieved:

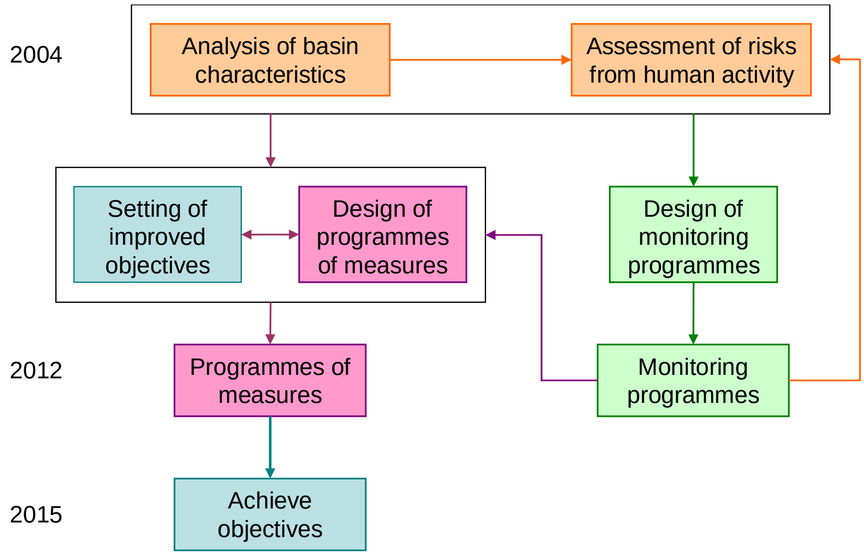
- to implement measures to prevent or limit the input of pollutants into groundwater and to prevent the deterioration of the status of the groundwater body;
- to protect, enhance and restore all bodies of groundwater, and ensure a balance between abstraction and recharge of groundwater, with the aim of achieving good groundwater status by 2015;
- to reverse any significant and sustained upward trend in the concentration of any pollutant that results from the impact of human activity in order to progressively reduce pollution of groundwater.

To determine compliance with Article 4 objectives, a risk assessment as part of the characterisation process must be undertaken for each groundwater body (or group of bodies) which must include the following:

Table 3 Nitrates Directive Action Programme measures

Activity	Control measure
Fertilizer spreading	Prohibition of use or certain types for high-risk periods. Limitation of land application of fertilizers based on soil conditions, type and slope, climatic conditions, land use and practices, and calculated crop requirement.
Manure storage	Capacity of vessels for livestock manure must exceed that required for the longest period where land application is prohibited.
Manure spreading	Maximum allowable animal manure application rate of 170 kg/ha following calculation of the amount of nitrogen in the manure unless an exception is justified.

Figure 4 Steps to achieving objectives under the Water Framework Directive (after EA, 2002)



- implement measures to prevent or limit input of pollutants into groundwater and to prevent the deterioration of groundwater body status. Thus characterisation needs to assess whether this is likely to be feasible. If suitable measures cannot be undertaken, it must be determined whether derogations can be applied;
- assess whether the groundwater body is at risk of failing to meet good status in 2015;
- assess whether measures to reverse significant trends can be implemented and made effective by 2015;
- assess whether protected area objectives are likely to be achieved by 2015.

If any one of these objectives is unlikely to be achieved, the body is characterised as 'at risk' of failing to meet Article 4 objectives.

Annex V of the WFD sets out the definitions of good status. For groundwater this includes both quantitative and chemical status assessment. Good chemical status requires no saline intrusion, the meeting of standards set out in Article 17 and no specified detrimental effects on associated surface waters or directly dependent terrestrial ecosystems.

Although the WFD aimed to provide for integrated management of surface waters and groundwaters, its provisions for groundwater were recognised as incomplete. Thus, Article 17 of the WFD foresaw additional measures to prevent and limit groundwater pollution.

3.1.3 Groundwater Directive

The Groundwater Directive (GD; 2006/118/EC) establishes criteria for the assessment of good groundwater status and for the identification and reversal of significant and sustained upwards trends and for the definition of starting points for the trend reversals required by the WFD.

Annex I sets out the quality standards for assessing chemical status for nitrate and pesticides. These standards are the same as those used for drinking water, namely 50 mg/l for nitrate, 0.1 µg/l for individual pesticides and 0.5 µg/l for total pesticides. Threshold values for other pollutants are to be established by individual Member States using criteria set out in Annex II.

Monitoring frequencies and locations will be sufficient to:

- distinguish trends from natural fluctuations;
- identify upwards trends;
- take into account the characteristics of the groundwater body.

Methods of monitoring and analysis should conform to international quality control principles and the assessment of trend should be made using a statistical method.

The starting point for implementing measures will be when the concentration of the pollutant reaches 75% of the quality standard set out in Annexes I and II.

3.2 IMPLEMENTATION OF THE DIRECTIVES

Implementation of these directives has proved to be a major task for Member States, demanding large technical resources for the characterisations and risk assessments outlined above, and for developing and operating the associated monitoring programmes. Moreover, the estimated cost of implementing the required programmes of measures targeted on the main polluting activities runs into hundreds of millions of Euros.

To assist in the implementation of these directives, a Common Implementation Strategy (CIS) has been set up by the Member States, Norway and the European Commission. Within the CIS, Working Groups are established to prepare guidance documents on specific aspects of implementation, which are published and openly available on the EU's CIRCA website. The discussion of implementation in the remainder of this section is focussed on groundwater and nitrate.

3.2.1 NVZ Designation

National approaches to the designation of NVZs have varied considerably. Some Member States took the decision right from initial implementation in 1993 to apply Article 3.5 of the Nitrates Directive, designating their whole territory and applying action programmes throughout. At the time of the 2000 EC synthesis report (EC, 2002), six of the then 15 Member States, Austria, Denmark, Finland, Germany, Luxembourg and the

Netherlands, had designated their whole territory, and these were joined in 2003 by Ireland (EC, 2007a).

Other Member States initially took a much less comprehensive approach, designating relatively small areas as NVZs, either because of a scarcity of evidence of high nitrate concentrations in groundwater or for national political reasons and opposition from the farming sector. This did, however, require the development of a robust methodology for designation, which would be publicly defensible when challenged by farmers. Thus, the UK Government designated only 8% of England initially in the first reporting, and this rose to 55% in 2002 and almost 70% in 2007. Significant increases between first and second rounds of designation were also reported for Italy, Sweden Belgium and Spain (EC, 2007a).

Three of the ten new Member States – Malta, Slovenia and Lithuania – have taken a 'whole territory approach' and the other seven have designated as NVZs percentages of their territories varying from 2.5% in Poland to 48% in Hungary (EC, 2007a).

3.2.2 NVZ Action Programmes

Actions by Member States to develop and apply the programmes of measures specified in the Directive were also often undertaken somewhat belatedly. The 2000 synthesis (EC, 2002) reported that nearly 200 Action Programmes had been published. However, a summary assessment of them for the report suggested that most countries were (at that time) failing to comply with measures on restricted periods for mineral fertiliser application even if they were complying for organic manures. Measures to ensure adequate storage of such manures to cover periods when application is prohibited were also not mandatory or were insufficient.

By the end of 2003, all fifteen Member States, with the exception of Ireland, had established Action Programmes, and Ireland established one in 2006 (EC, 2007a). In spite of some improvement, many of them still showed areas of non-conformity with the provisions in Annex III of the Directive (Table 3). These again included restrictions on application and storage capacity for livestock manures, and approaches to limiting total applications of organic and inorganic nitrogen. All of the new Member States have established Action Programmes (EC, 2007a).

3.2.3 WFD delineation and characterisation

The delineation of groundwater bodies by Member States has broadly followed the principles set out in CIS guidance Document 2 (EC, 2003a). Thus groundwater bodies should be delineated by geological and/or hydraulic boundaries, and should fall within a river basin or sub-basin. In detail, however, methodologies have been somewhat varied. Taken together with the huge variability of geological conditions and rock types across Europe, the result has been a range in national groundwater body numbers from a few tens or less, through hundreds to several thousands. The former is typical of the smaller, geologically simple countries (including Malta) and the latter applies to the vast number of small, disconnected (but often hydrogeologically similar) glacial and fluvio-glacial sedimentary aquifers of Norway, Finland and Sweden.

Approaches to the characterisation required under Article 5 of the WFD have also varied. Some Member States have been willing at the stage of initial characterisation to designate groundwater bodies to be 'at risk' based purely on the assessment of risks, and have taken a broad, precautionary 'one-out-all-out' view with regard to human activities causing pressures within the groundwater body. Others have not been prepared at that stage to report groundwater bodies 'at risk' unless there was at least some confirmatory groundwater quality information. The former could lead to a much higher proportion 'at risk' than the latter, and as a consequence, the proportion of groundwater bodies reported by Member States to be at risk following initial characterisation varies greatly.

3.2.4 Development of Monitoring Programmes

The Nitrates Directive requires all Member States to monitor their groundwaters and surface waters, and draft guidance on such monitoring has been produced (EC, 1999; EC, 2003b). According to this guidance, three type of monitoring can be distinguished:

- to identify areas for designation. This has usually been provided by existing networks, and is not required where whole territories have been designated;
- in 'whole territory' Member States, of important water bodies and intensively cropped areas. Again this has usually been provided by existing networks;
- to assess the effectiveness of action programmes. This clearly has to be concentrated in those areas where action programmes are being applied, and may require existing monitoring to be augmented.

In practice and in detail, Member States have applied their own interpretations of how this monitoring should be carried out (Fraters et al., 2003) and, as a consequence, periodic, Europe-wide summaries of the status and trends of nitrate in groundwater have suffered from major gaps and lack of compatibility (EC, 2002). Moreover, groundwaters in general respond more slowly than surface waters, and the hydrogeological variability across Europe means that this time lag is sometimes measured in decades. The delay between human actions at the ground surface and observed responses at groundwater monitoring points applies both to the impacts of increasing nitrogen loading and to the impacts of action programmes. This makes timely monitoring and reporting of the beneficial impacts on groundwater quality difficult. Some Member States have, therefore, considered other direct monitoring options, including sampling soil moisture, field drainage and shallow groundwater beneath agricultural land (Fraters et al., 2003), or indirect options such as farm statistics.

The Water Framework Directive is somewhat more specific about the requirements for monitoring than the Nitrates Directive. WFD monitoring is to be focussed primarily on the groundwater body and is intended to provide information to support the overall management of the river basin district. Article 8 specifies the establishment of programmes to monitor groundwater, the requirements for which are set out in Annexes II and V. Thus, in addition to a network for quantitative status assessment:

- a surveillance network is required to supplement and validate Article 5 characterisation and risk assessment for chemical status and to provide information for the assessment of long-term trends;

- an operational network to establish the status of all groundwater bodies which have been determined to be at risk and to establish significant and sustained upward trends.

Under the EC Common Implementation Strategy referred to above, guidance has been prepared to assist Member States to develop monitoring programmes to meet these objectives (EC, 2007b). This stresses the need to take proper account of the aquifer types and groundwater flow regimes – establishing a conceptual hydrogeological model – in designing networks. Specific guidance is provided on selection of sites, parameters and frequency for each network (EC, 2007b).

In practice, the information requirements of the Water Framework and Nitrates Directives overlap, and a combination of limitations on monitoring resources and budgets and practical constraints on the availability of sampling points mean that, in many countries, much of the groundwater quality monitoring output contributes to both.

3.2.5 Results of nitrate monitoring

As specified in the Nitrates Directive, nitrate monitoring results should be provided every four years to the European Commission, to be summarised in synthesis reports (EC, 2002; EC, 2007a). Member States should also report their monitoring results, or selected subsets of the results, periodically to the European Environment Agency (EEA).

In the most recent EEA report (EEA, 2004), the 50 mg/l limit was exceeded in at least one sampling site in a third of the 442 groundwater bodies reported. The most recent summaries available from both the EC (2007) and EEA (2004) suggest that there is no overall decreasing or increasing trend for nitrate in groundwater for Europe as a whole. Of the 142 groundwater bodies for which trends were reported, 31% showed statistically significant downward trends or a trend reversal, 49% showed no measurable improvement and 20% showed an upward trend (EEA, 2004).

Groundwater nitrate data show marked variation across Europe, many of the worst affected countries and regions being in the south and the less affected in the north (Figure 5). This general north-south gradient is mainly a reflection of climatic conditions, but also the underlying geological setting and the combined influence of these two on population distribution and agricultural activities (Table 4). Thus the Mediterranean regime has a warmer climate which allows more intensive double or triple cropping on large parts of the cultivated land, but provides less recharge to dilute the nitrogen loading. In contrast, the northern countries of Scandinavia have cold winters, heavier rainfall and less evapotranspiration, low population densities and underlying crystalline basement geological conditions that are generally not suitable for intensive cultivation, and low overall levels of nitrate pollution

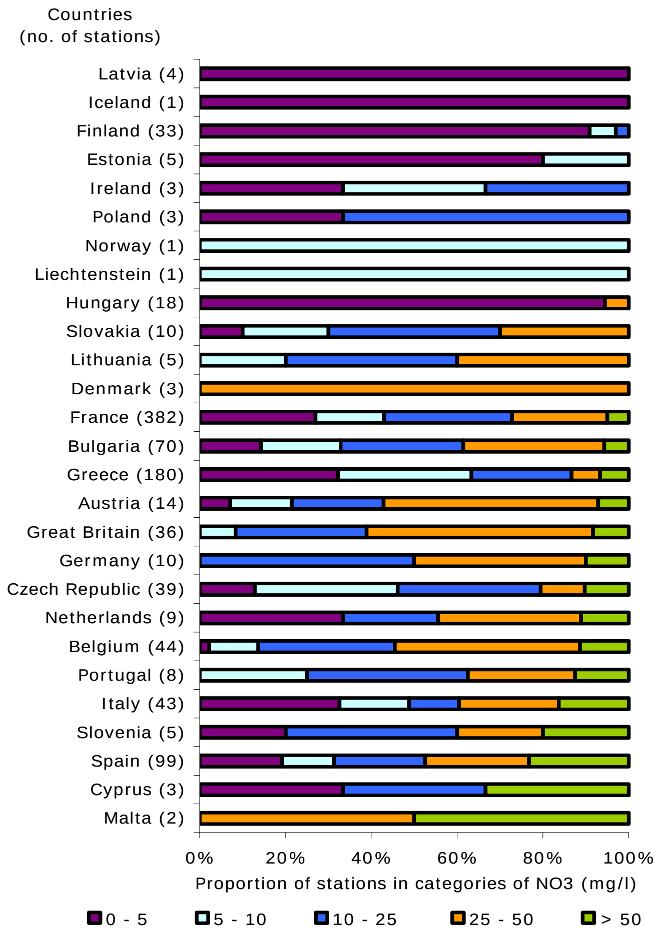


Figure 5 Nitrate concentrations in groundwater bodies in EU member states (EEA, 2008)

(Figure 5**)**. In between are several Member States (Netherlands, Belgium, and Denmark) with more moderate climates and suitable terrain in which intensively cultivation and livestock husbandry combined with high populations has resulted in significant nitrate problems in groundwater.

Even this broad subdivision is too simplified, as a number of the larger Member States have significant geological, climatic, land use and population variability within them, particularly from north to south (France, UK, Germany, and Italy). Many have much more complex geological settings than that implied by the simple description in Table 4. Moreover, the national figures in Table 4 for recharge, fertiliser use and population and livestock densities mask considerable differences between intensively cultivated regions (sometimes supported by irrigation) and much less well-developed, often hilly, mountainous or moorland areas. As the geological, climatic and economic variability across Europe increases with enlargement, it becomes more difficult for Member States to comply with the 'one size fits all' approach of EC Directives.

Table 4 Hydrogeological settings of EU member states

Country (groundwater dependence) ‡	Principal aquifers✧	Groundwater recharge (mm/yr)*	Mean fertilizer use (kg/ha)†	Population density in 2002 (person/km^2)§	Livestock density (head/km^2)#
Finland (10%), Norway (20%), Sweden (20%)	Small, thin, shallow aquifers in glacio-fluvial sands and gravels overlying crystalline bedrock in which local fracturing provides small aquifers	128 215 142	9 6 6	15 14 20	7 5 7
Denmark (25%)	Some Chalk (equivalent to UK) and recent sands and gravels, mostly shallow with thin unsaturated zone	363	69	125	330
Lithuania (6%), Latvia (45%), Estonia (19%	Sedimentary sequences and karstic limestones	153 159 172	30 8 6	53 36 30	29 13 13
Germany (13%)	Thick alluvial plains in the north, consolidated sediments in the centre and south	201	73	231	110
Poland (16%)	Sedimentary sequences and karst limestone (2.5% of land mass), some volcanics	144	48	123	79
France (16%)	Some Chalk in the north as in UK, thick alluvial plains, limestones in the centre and south	200	72	109	62
Netherlands (13%), Belgium (9%)	Thick alluvial sequences with water table very close to surface – thin unsaturated zone, some limestones	355 275	94 94	450 335	420 293
United Kingdom (19%)	Chalk in the south and east, sandstones and limestones	339	74	243	62
Ireland (19%)	Small, thin aquifers in older basement rocks, karstic limestones in the west, small, shallow alluvial aquifers	373	84	56	122
Hungary (16%), Czech Rep (18%), Slovakia (41%)	Large alluvial basins of the Danube and its tributaries and karstic limestones	73 92 116	54 47 25	109 130 110	49 54 33
Romania (12%), Bulgaria (31%)	Large alluvial basins of the Danube and limestones	93 77	14 15	92 71	40 14
Austria (34%), Slovenia (22)	Karstic limestones and some alluvial basins and river plains, some older fractured rocks in Alpine regions	163	25	97	61
Spain (9%), Greece (26%), Portugal (42%), Italy (23%)	Karstic limestones, coastal alluvial plains and some large alluvial basins (Po, Tagus, Guadalquivir,)	70 91 142 147	43 31 23 28	82 84 113 193	62 12 41 51
Malta (45%)	Limestone	142	22	1248	294
Cyprus (30%)	Fractured igneous rocks and limestones, and deltaic, fluvial and aeolian sands and gravels	30–70	17	87	55

‡ EEA, 1999; EEA, 2000; ICID, 1996
✧ Burdon, 1954; CGMW, 2005
* Döll and Fiedler, 2008; ICID, 1996; Zagana et al., 2007
† National use from UNEP, 2008 divided by area
§ UNEP, 2008
# Earthtrends, 2008-cattle and pigs

4 Managing groundwater nitrate pollution in Malta

4.1 LEGAL AND INSTITUTIONAL CONTEXT

The Nitrates Directive was transposed into Maltese legislation as Subsidiary Legislation 435.40. Annex VI states that all of Malta shall be one entire Nitrate Vulnerable Zone, and Malta has therefore taken the 'Whole Territory Approach' referred to in the previous chapter. The associated Maltese Code of Good Agricultural Practice was produced in 2003.

The Water Framework Directive was transposed into Maltese legislation as Legal Notice 194 in 2004 (Water Policy Framework Regulations, 2004). These regulations define the Malta Resources Authority as the Competent Authority, with the exception that inland surface waters were placed under the Malta Environment and Planning Authority.

4.2 GROUNDWATER BODIES

To meet the obligations of the WFD, sixteen groundwater bodies have been delineated based on geological characteristics. The two largest comprise the Main Sea Level aquifers in Malta and Gozo and the smaller ones are the various hydraulically-distinct parts of the perched aquifer (Stuart et al., 2008a). Malta's Article 5 assessment report to the EC indicated that all bodies, apart from Comino, could be considered to be at risk of failing to meet the objectives of the WFD (MRA, 2005).

4.3 ESTIMATING NITRATE LEACHING

The hydrogeological setting of Malta described in the main project report (Stuart et al., 2008b) can be summarised as two distinctive limestone aquifers separated by less permeable material, and experiencing a dry, Mediterranean climate. It is this setting which helps to establish the observed groundwater nitrate concentrations and places Malta into the situation it occupies in Figure 5. It also provides the physical environment against which the likely beneficial impact of any measures to control nitrate pollution must be evaluated. Some simple scoping calculations help to illustrate the scale of the problem.

The overall groundwater recharge for Malta has been estimated as 40-45 × 10^6 m^3/year (Sapiano et al., 2006) over a total area of 316 km^2. Although some of the recharge mechanisms are clearly localised, if this volume were taken as uniformly distributed, it would suggest an average infiltration to groundwater of 135 mm/year. Nitrate concentrations in groundwater from the perched aquifers measured during the project have a median value of 164 mg/l. If it is assumed (again unrealistically) that all the infiltrating water contains this level of nitrate, a simple volumetric conversion suggests an average equivalent nitrogen leaching rate of about 50 kg/ha/year over the whole aquifer. The comparable calculation for the MSL aquifer would suggest that the mean concentration of 62 mg/l nitrate would be equivalent to an average leaching rate of 19 kg N/ha/yr.

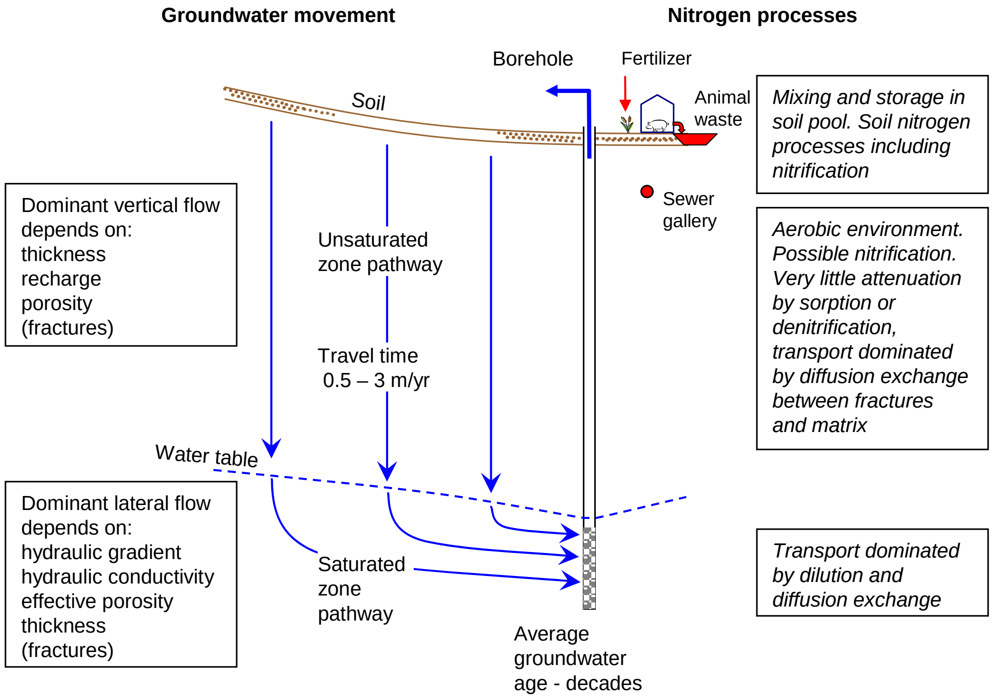
The island of Jersey is geologically different (Stuart et al., 2008a) to Malta, but comparable in terms of its population density and intensive mix of arable and livestock farming. Moreover, the groundwater recharge rate is very similar at 130 mm/yr. Using the same simple estimate, leaching losses of 25-50 kg N/ha/yr from applications of 100-200 kg N/ha/yr produce nitrate concentrations in groundwater such that 70% of them exceed 50 mg/l (Robins and Smedley, 1998). In SW Spain, in probably a similar physical setting (Stuart et al., 2008a), 187 kg/ha/yr of nitrogen fertiliser applied to potatoes and cotton on a limestone aquifer resulted in an average concentration of 160 mg/l nitrate in groundwater.

By sampling groundwater directly beneath the cultivated land and knowing the amount of nitrogen applied, the proportion leached can be estimated by the simple method used here. These general estimates have proved most useful where land use, farming activities and hence nitrogen applications are relatively uniform and, moreover, where there is information about the proportion of the nitrogen load that is leached to groundwater. Research has been able to establish this for major agricultural crop types and inorganic nitrogen fertilisers, and the results are available in the literature.

For organic nitrogen, however, the nitrogen cycle processes of volatilisation and nitrification, which are likely to affect the leaching behaviour, are more variable and uncertain. Application rates are also much more variable (and usually less well recorded) and the nitrogen content is less certain. As a consequence, nitrogen leaching losses are much more difficult to estimate. As the results of this study point more towards organic animal waste than agricultural fertilisers as the source of nitrate in groundwater, this remains an important uncertainty.

The results of the present study suggest that groundwater nitrate concentrations are significantly higher in the perched aquifers and still rising, but more modest and largely stable in the MSL aquifers of both Malta and Gozo. The possible reasons for this are set out in the main project report (Stuart et al 2008b) and the processes governing the movement of water and behaviour of nitrogen are shown conceptually in Figure 6. If this difference not caused by changes in land use, pollution sources and loading factors, then the unsaturated and saturated zone processes (Figure 6) in the two aquifers must be different. The young ages and high and rising nitrate concentrations in the perched aquifers suggest recharge is rapid and dominantly through fractures. If recharge is slower and more distributed in the MSL aquifers at rates more akin to those in Figure 6, then it is possible that there is time for diffusion exchange and delay to nitrate transport. Additional nitrate may be yet to arrive at the water table, and concentrations may rise in the future. However, if this were the case, it would be expected that observed nitrate concentrations would be rising at locations where the travel times were shortest. Perhaps the time series information is not comprehensive enough to provide an answer. Transport in the saturated zone (Figure 6) of the MSL aquifer may also be slower than

Figure 6 Groundwater movement and nitrogen processes in Malta aquifers



typical of limestones because the transmissivity is lower and because of the very low hydraulic gradient.

The alternative view is that the MSL has already reached some degree of equilibrium between relatively stable nitrate inputs and observed concentrations in the abstracted water. In either case, the saturated zone residence times in the MSL aquifer are confirmed by the results of this study to be long, and the beneficial impacts of any control measures introduced are likely to be more slowly observed in the MSL than in the perched aquifer.

4.4 MANAGING NITRATE POLLUTION

An approach already adopted in Malta has been for the Water Authority to withdraw springs and boreholes with very high nitrate concentrations from the public distribution system. As a result, no groundwater is now drawn from the Malta perched aquifers for public water supply. The shortfall has been made up from desalinated seawater, which in 2003/04 comprised over 50% of Malta's total public supply (Sapiano et al., 2006). Even the simple estimations above make clear the need for substantial reduction in nitrate leaching to groundwater to meet a target of 50 mg/l nitrate. Malta's low annual groundwater recharge cannot dilute the leached loading.

Malta has developed a comprehensive Code of Good Agricultural Practice to help manage the impact of farming. This incorporates the Action Programme developed for Malta under the Nitrates Directive, as well as addressing other important environmental concerns. Within the code, obligatory Action Programme provisions related to the storage and application of manures include a 'close' season of five winter months from mid October to mid March. This has implications for the capacity of storage that must be made available by livestock farmers.

The provision under the Nitrates Directive for land applications of total nitrogen in livestock manure to be initially limited to 210 kg/ha/yr for the first four years of the Action Programme and to 170 kg/ha/yr from Year 5 onwards is incorporated into the code. The simple estimation above for the perched aquifer indicates that nitrogen applications would probably need to be lower than this in order to achieve a meaningful reduction in groundwater nitrate concentration. As agricultural land use occupies between 40% and 60% of the area of the perched aquifers, a simple conclusion would be that nitrate leaching from this land needs to be reduced by almost 75%.

In practice, and particularly so in Malta, the land surface of a groundwater body or the capture zone of an individual borehole or spring is made up of a complex patchwork of different land uses. The activities on some of these patches contribute recharge with high nitrate leaching losses, whilst from others the infiltrating water helps to dilute the overall nitrate concentration. For managing quality at the groundwater body scale, therefore, the key factor is the relative proportions of 'high' and 'low' leaching land uses, and the number and size of 'point' sources of nitrogen.

This leads to possible pollution control options of either:

- relatively modest reductions in nitrate loading over the whole catchment or groundwater body to increase the proportion of 'low' leaching land;
- stronger reductions, complete prohibition or removal of heavily-leaching point sources;

or a combination of both. The latter could be targeted on the land closest to abstraction sources with the most rapid groundwater flow paths for quickest benefit. This applies more in the case of well-defined capture zones or protection areas around large public supplies, than at the groundwater body scale. The approach has been tested in Denmark and Germany by small municipal water utilities that already own or buy the land adjacent to their groundwater supply sources and lease this land for prescribed, non-polluting, often recreational activities (Folmand et al., 2006).

The results of the isotopic study indicate that the 'source' nitrogen has passed through the soils. This implies that both agricultural fertilizers and organic manures applied to the land need to be controlled. The Code of Practice contains measures to control the amounts and timing of fertilizer applications. Thus, for the livestock source in Malta, the Code of Practice prescribes the maximum organic loading of 170 kg N/ha/yr, but does not refer to the proportion of the land area to which this is applied. The most recent edition of the NVZ Action Programme for England (Defra, 2008) specifies that this limit applies to the whole farm, but allows up to 250 kg N/ha/yr on any one field. To achieve this, farmers often transfer their excess manure to neighbouring farms, and the Action Programme requires both the sender and the recipient to record the type and quantity and the origin and destination (Defra, 2008). The Maltese Code of Practice also requires such information.

Controlling groundwater pollution from organic manure clearly depends on managing the relationship between storage capacity and land application. Other possible management options for reducing nitrate pollution from farming include:

- an overall reduction in livestock numbers;
- alternative use of livestock manure, for example for biogas production;
- reducing the availability of land for manure application and better techniques of application at field level to reduce leaching;
- introducing agricultural techniques to improve nitrogen uptake by choice of crop and cultivation of cover crops.

The Code of Good Agricultural Practice contains provisions related to irrigation which can also be used to help manage the impact of irrigation with treated sewage effluent, below which elevated concentrations of nitrate in groundwater are observed (Stuart et al., 2008b).

One approach to the management of water quantity which could have benefits for groundwater quality would be the use of artificial recharge. This could potentially be used to help improve quality from both a nitrate and salinity point of view. Artificial recharge needs a reliable source of water and suitable hydrogeological conditions for infiltration into the underlying aquifers. In Malta, source water could be provided by rainwater captured by surface structures or by treated wastewater. The most important constraint in such a crowded island is likely to be the availability of suitable land for recharge structures. There are only a few valleys and gullies which might provide sites for such structures, and the MSL aquifer itself may not be sufficiently permeable to allow rapid infiltration. Where land is scarce, recharge boreholes have been employed, but again the hydraulic conductivity could be a constraint on infiltration rates. However, the artificial recharge option needs a scoping study of its feasibility in Malta.

While the Nitrates Directive Action Programme is targeted towards farming activities, the potential for polluting leakage from large sewers needs to be kept under review and lining improved where ‘hot spot’s are identified, especially in the upper reaches of the collection system where the sewers are likely to be well above the water table.

4.5 STAKEHOLDER ACCEPTANCE

In targeting farming activities in its Action Programme, the Nitrates Directive accords with the ‘Polluter Pays’

Table 5 Acceptability of possible mitigation measures for reducing nitrogen leaching to groundwater

Mitigation measures	Disincentives	Possible incentives	Nitrates Directive
Improved manure storage facilities (clamps)	Cost	Conditional on financial assistance being provided Dry product with easier handling	✓
Controls on field storage of manure	Inconvenience		✓
Controls on location of livestock units	Cost		
Reduction in livestock numbers	Reduced profitability		
No sacrificial disposal	Alternative method not available	Suitable facilities developed	✓
Limit on fertilizer applications	Possible lower crop yield	Cost saving	✓
Stringent limit on fertilizer application	Possible limits on crop type Reduced profitability		
Controls on timing of applications	Inconvenience		✓
Soil incorporation	Inconvenience, greater effort		
Splitting of fertilizer applications	Greater effort	Improved efficacy	✓
Controls on crop type and introduction of cover crops	Reduced profitability		
Reduction of agricultural area	Reduced profitability		
Alternative use of manure, e.g. biogas	Capital cost	Reduced cost of energy	
Lining of sewer galleries	Capital cost	Cost to state rather than private individuals	

principle. While this is likely to meet broad acceptance amongst most stakeholders, some control measures, such as building storage capacity for the period of the closed spreading season, may necessitate significant direct expenditure by the farmers themselves (Table 5). Grants are needed to ensure that these provisions are complied with by the construction of adequate and correctly specified storage facilities, and the Government of Malta has established a programme to provide such grants. Other measures (limiting nitrogen applications, limiting livestock numbers, cover crops) may result in actual or perceived loss of crop productivity and income (Table 5) which can be difficult to quantify. Some measures may require extra work and greater costs (soil incorporation, improved application techniques), or may limit the farmers' scope for spreading (avoiding sloping land, proximity to surface water and the coast).

Replacing high nitrate groundwater with desalinated sea water for public supply is essentially an 'engineering' rather than 'environmental' solution. While this may meet Drinking Water Directive obligations for water in supply, it would not meet Malta's more environmental obligations under the Water Framework and Groundwater Directives, which require water within groundwater bodies to meet good chemical status. Moreover, there is a high cost of alternative supply from desalination, and the whole economy and community pays for poor groundwater quality rather than the polluter. Good chemical status is not going to be easy or quick to achieve by any combination of control measures. For Malta, as for all Member States, controlling nitrate pollution from diffuse and point sources is a complex technical, socioeconomic and institutional issue.

4.6 TECHNICAL RECOMMENDATIONS

The following activities would contribute to an increased understanding of nitrate transport and storage in the groundwater system, which would inform the design of appropriate programmes of measures and contribute to the prediction of the timescale required for improvements to groundwater quality to be effected.

4.6.1 Determine nitrate leaching from cultivated soils

Collect leaching data from the soil root zone in arable and horticultural areas. This would confirm that leaching from agricultural soils is a major source of nitrate in groundwater and inform the calculation of the nitrogen application reductions which would need to be made to bring leaching down to the level required for groundwater to meet the regulatory obligations.

This is difficult to do in this type of aquifer as typical samplers such as porous pots (vacuum lysimeters) are problematic to install in fractured media and structured clay soils. It may be possible to employ wick samplers or zero tension lysimeters (interception trays) which may be better at intercepting macropore flow (Holder et al., 1991; Zhu et al., 2002).

4.6.2 Determine nitrate storage in the unsaturated zone

Investigate the unsaturated zone porewater concentrations by cored drilling and extraction of porewater. This has been successfully used for limestones in the UK to identify the amount of nitrate held in store in the unsaturated zone (Smith-Carington et al., 1983). This would inform whether nitrate stored in the unsaturated zone will lead to future increases in groundwater nitrate concentration. This can be a costly technique both for cored drilling, particularly for the depths which would be required here and for processing and analysing the samples. The core can be used to provide other information, for example aquifer properties data, although this should be available for the Globigerina limestone from quarried material.

There has been some speculation on the feasibility of reducing the cost by lateral coring from the shaft of one of the big pumping stations. We are unable to comment on this from our own experience but, in principle, it would appear reasonable. It is not known what the zone of influence of the shaft would be and therefore what penetration would be required.

4.6.3 Quantify sewer leakage

Attempt to quantify leakage from the remaining unlined sections of sewer galleries. Most work in this area has concentrated on the use of artificial tracers added to the sewage. (Vollertsen and Hvitved-Jacobsen, 2003; Rieckermann et al., 2007). An approach using flow gauging at the input and exits of galleries without input connections, analogous to that used for estimating canal leakage, may be helpful.

4.6.4 Improve characterisation and flow modelling of aquifers

Numerical modelling of groundwater flow and transport would enable a link to be made between the nitrate concentrations at the base of the soil zone and concentrations at various points in the aquifer. This would require the collection of new data from the above activities to quantify the various elements of the model, namely:

- nitrate leached from the base of the soil zone;
- nitrate transport through the unsaturated zone by both rapid and matrix routes;
- transport of nitrate from the unsaturated to the saturated zone;
- nitrate transport through the saturated zone;
- capture by abstraction.

The model could then be used to predict the response time of the aquifers to changes in surface activities as a result of programmes of measures designed to improve groundwater quality.

4.6.5 Improved nitrate monitoring for trend detection

Improved collection of groundwater quality data to provide time series for nitrate concentrations under the Nitrates Directive and for other determinands under the Water Framework and Groundwater Directives. This is required for trend quantification and assessment of the efficiency of the Programme of Measures. Existing water quality information is not adequate for the precise determination of trends.

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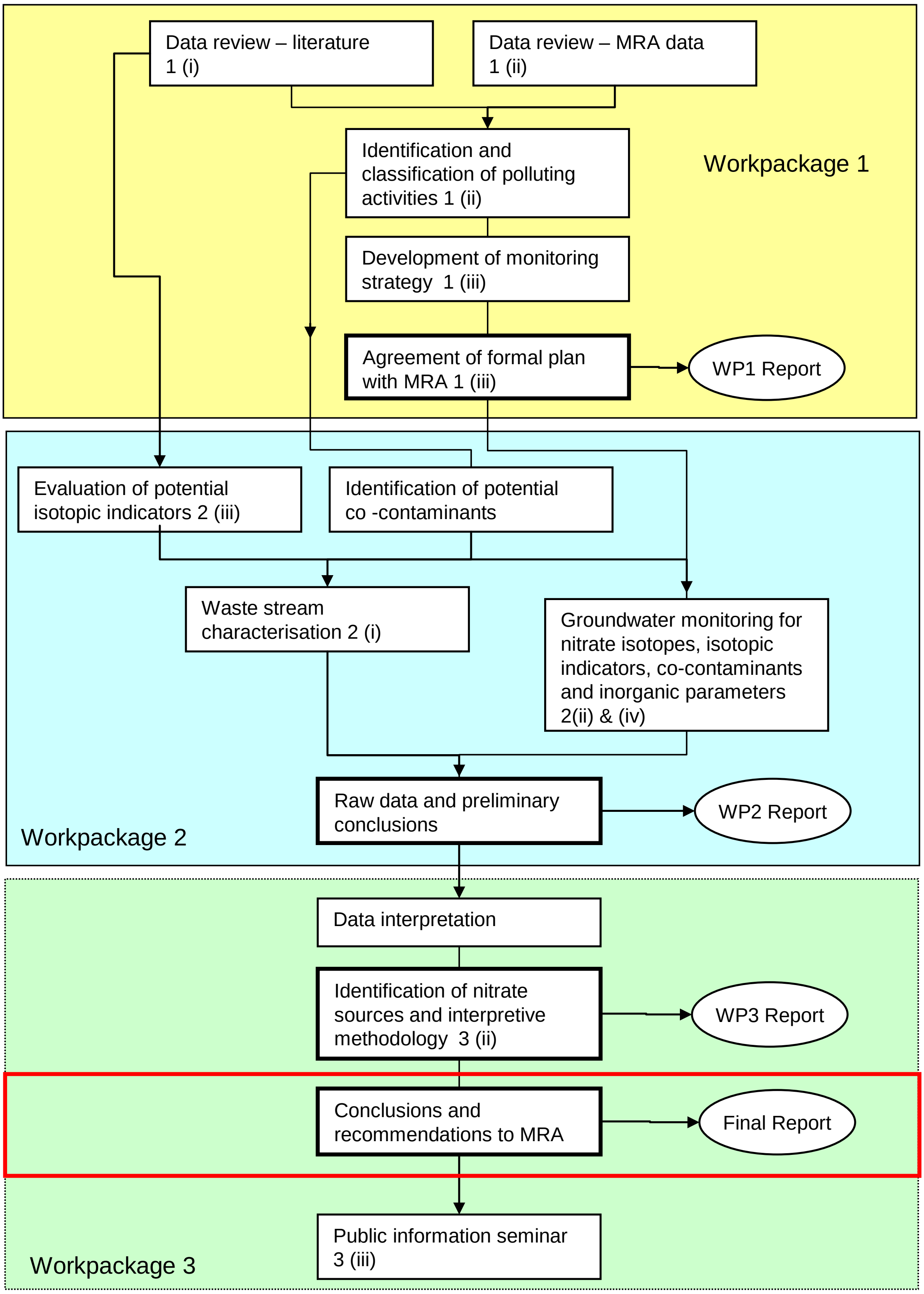
Appendix 1 Flowchart of project activities



Numbered activities refer to tasks in Annex 2, Section 4 of the project TOR

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