Investigating the effectiveness of NVZ Action Programme measures:

Development of a strategy for England

Report for Defra Project No. NIT18

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EXECUTIVE SUMMARY

The Nitrates Directive specifies a 4-yearly review for the European Commission of progress in implementing the Directive. A part of this review must include an evaluation of the impact of the Action Programme measures, which should feed into predictions of future water quality benefits and subsequent revisions to the Action Programme measures as necessary. The aim of this project was to provide this assessment of Action Programme measures.

This project based its approach around the following actions:
- Field measurements on commercial farms, to provide evidence of the size of nitrate losses and the effects of mitigation methods under ‘real world’ farming (rather than experimental plots).
- Use of field-scale and catchment-scale models (validated against the field data) to quantify the likely effectiveness of combinations of measures as applied to the NVZs.
- Use of other data and information to supplement our own assessments.

This multi-strand approach follows the recommendations in the EC guidelines for monitoring the effectiveness of the NVZ Action Programme (AP). The project started in spring 2004, allowing three winters for measurement (2004/05, 2005/06 and 2006/07), and ended with final reporting in May 2007. Figure 1 summarises the overall approach of the project, a combination of field measurement and modelling.

The project produced a large amount of measurement and modelling data, as well as several summary reports, all of which contribute to the overall assessment of Defra’s NVZ Action Programme (as detailed in the project objectives). In order to keep this main document brief, we have presented the detail as Appendices. The remainder of this main report focused on interpreting these results, placing in context with other data and developing the policy implications for Defra.

Field measurements

A network of sites was established on both surface and groundwater catchments. The main activities were:
- Establishment of groundwater and surface water sites, and monitoring during winters 2004/05, 2005/06 and 2006/07

Figure 1. Schematic representation of the project structure.
Measurements of nitrate leaching and water flux at each site
Measurement of late autumn Soil Mineral Nitrogen (SMN), as an indicator of N at risk of leaching, and soil properties at each site
Annual collection of field-level site and management data to aid interpretation of the field measurements and as input data for the models to be validated against the field measurements

A total of 203 fields were monitored on groundwater sites in 8 locations and 125 fields on surface water sites in 8 micro-catchments, ensuring measurements across a range of cropping systems and geoclimatic regions. The farms were all subject to the current NVZ Action Programme. The British Geological Survey (BGS) provided supporting information on the choice of groundwater aquifers used in the project. The approach taken in this project, of scientific data and models supported by field-specific measurements, is specifically recommended by EC in their guidance for implementing the Nitrates Directive.

The measurement data showed that nitrate concentrations in leachate from typical agricultural land within NVZs often exceed 50 mg/l, especially under arable cropping or intensive dairying. Nitrate losses were elevated where manure was used; where fertiliser N inputs were high relative to crop N uptake; and in some intensively stocked grassland systems. Proper adjustment of fertiliser inputs to take account of the crop-available N supplied by manure is effective in reducing the additional losses. However, the data confirm that losses remain greater where manure is used in the rotation than where no manure is applied.

There was some evidence from the field measurements that further reductions in nitrate leaching from slurry (and, by inference, poultry manure) applications could be achieved by extending the Closed Period to cover all soil types, not just sand/shallow soils.

In the surface water micro-catchments (drained clay soils), there was sometimes evidence of increased nitrate concentration in drainage water in the spring (normally, nitrate concentrations were decreasing at this time). This suggests new sources of nitrate were being leached, such as recently applied manure or fertiliser.

Where additional fertiliser N is applied to wheat to boost grain protein, there is an indication that this increases N loss, compared with fertilising for yield only.

Nitrate losses from land were elevated where crop cover was nil or small for a long period prior to the winter – for example, after early harvested peas, and some rotational set-aside. In such situations, the only effective way to reduce nitrate leaching risk is to ensure green cover is established early enough in autumn to take up the high N supply – whether a commercial crop such as oilseed rape, or a purpose-sown cover crop.

Modelling

Measurements of nitrate leaching can demonstrate the impact of management on nitrate emissions. In order to generalise this understanding to quantify impacts at the catchment scale, we need to be able to extrapolate the effects to other cropping, management, soil or climatic conditions. To this end, a field-scale model (NIPPER) was developed and tested within this project. A system for integrating the model with catchment-level spatial statistics on land use, livestock numbers, weather and soils has been developed so that the impact of measures can be assessed for particular catchments.

In order to assess the impact of the NVZ AP, the baseline conditions (representing practice prior to implementation of measures) were taken from survey data. The British Survey of Fertiliser Practice has recently been adapted to provide improved, field-by-field information to allow estimation of change in management since implementation of the NVZ Action Programme, including quantity and date of application of manures, and the
degree to which this affects fertiliser inputs. Scenarios representing prior and post-implementation management, for the same catchment and land use, were developed and their effects modelled. The likely effect of the AP measures depends on the prior farmer practices within the catchments (e.g. what proportion of farms made allowance for manure N inputs when planning fertiliser inputs), on which there is some uncertainty.

The impact of the current NVZ AP was modelled in two contrasting, catchments (the mainly arable, relatively dry, groundwater catchment of the Meden and the dominantly grassland, wetter, surface-water catchment of the Taw) to illustrate the likely range of effects on nitrate leaching. The impact of further, more severe, changes in management were also explored to indicate the scale of response which might be expected under current land use (Tables 1 and 2). As indicated, responses differed between catchments, but also between sub-catchments within a catchment depending on factors including cropping, soil type, and livestock numbers and type.

Table 1. Modelled results for the Meden, showing N leached per hectare of agricultural land and nitrate concentrations in drainage from agricultural land, along with the average effect of a mitigation measure across the whole catchment and the range of effect in the various sub-catchments.

<table>
<thead>
<tr>
<th></th>
<th>N Leached (kg/ha)</th>
<th>Nitrate Concentration (mg/l)</th>
<th>Average Reduction (%)</th>
<th>Range of Reduction (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>58.4</td>
<td>147</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Current AP measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Do not exceed crop N requirement</td>
<td>54.6</td>
<td>138</td>
<td>6.5</td>
<td>1.7 – 20</td>
</tr>
<tr>
<td>Closed period PLUS do not exceed crop N requirement</td>
<td>49.6</td>
<td>125</td>
<td>15.0</td>
<td>1.9 – 23</td>
</tr>
<tr>
<td><strong>Additional measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10% Fertiliser Reduction</td>
<td>54.1</td>
<td>137</td>
<td>7.4</td>
<td>4.0 – 11.7</td>
</tr>
<tr>
<td>Remove all Manures</td>
<td>43.3</td>
<td>109</td>
<td>25.9</td>
<td>5.8 – 37.4</td>
</tr>
</tbody>
</table>

These Tables illustrate some of the lessons learnt from the NIT18 project:

- **Arable losses often greater than grassland losses** (for the same rainfall) because grassland provides all year cover.
- **The relatively large contribution of manures to N losses in many catchments** (depending, obviously, on their composition in terms of livestock number and type. Here, the Meden is intensively stocked compared with some arable catchments. Removing all manure reduces nitrate losses by 25% and 9%, respectively, for the Meden and Taw.
- **Good fertiliser practice reduces nitrate loss** – the size of effect varies, but in the two example catchments ranges from 2-7%, on average.
- **The closed period on sandy/shallow soils decreases nitrate loss** – by 15%, on average in the Meden catchment, by about 7% on a more typical catchment with lower stocking densities. This measure was not applicable in the Taw because of the absence of sandy and shallow soils.
- **Reducing N fertiliser applications below optimum decreases nitrate loss** - with a greater reduction (c. 7%) on arable land compared with c. 3% on grassland.

Table 2. Modelled results for the Taw, showing N leached per hectare of agricultural land and nitrate concentrations in drainage from agricultural land, along with the average effect
of a mitigation measure across the whole catchment and the range of effect in the various sub-catchments.

<table>
<thead>
<tr>
<th></th>
<th>N Leached (kg/ha)</th>
<th>Nitrate Concentration (mg/l)</th>
<th>Average Reduction (%)</th>
<th>Range of Reduction (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>23.0</td>
<td>29.3</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Current AP measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Do not exceed crop N requirement</td>
<td>22.7</td>
<td>28.8</td>
<td>1.5</td>
<td>0.1 – 5.0</td>
</tr>
<tr>
<td><strong>Additional measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10% Fertiliser Reduction</td>
<td>22.4</td>
<td>28.5</td>
<td>2.7</td>
<td>0.5 – 5.4</td>
</tr>
<tr>
<td>Remove all Manures</td>
<td>21.0</td>
<td>26.7</td>
<td>9.0</td>
<td>2.7 – 27.5</td>
</tr>
</tbody>
</table>

**Assessment of the time-scale of response**

The EC requires some assessment of time-scale of response in terms of when change in farm practice will be expressed as change at the sampling point. Most management changes rapidly affect nitrate loss from the land. Changes are rapidly transmitted through clay soils to surface water catchments. However, it can take many years before effects are evident in groundwater abstractions, or in groundwater-fed rivers. Groundwater modelling is difficult, data-hungry and expensive. It was decided during project specification that it would not be practicable within the resources available to model response times for every groundwater NVZ. British Geological Survey were, therefore, charged with reviewing literature and modelling approaches for estimation of groundwater response times, and developing a classification of aquifers and associated modelled response times which could inform reporting to the EC.

The approach divided England and Wales, into a manageable number of type classes for modelling, on the basis of likely responses of the aquifers/groundwater systems. Table 3 shows the time (in years) for 50% of the change to register at the abstraction point and the percentage of the change that would be observed after 10 years. The most important aquifers in terms of total area are Chalk and Permo-Triassic Sandstone.

**Table 3. Generalised response times for the main aquifer types in England and rainfall categories.**

<table>
<thead>
<tr>
<th>Aquifer type</th>
<th>Area km²</th>
<th>% change after 10 years</th>
<th>Time to 50% (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>&lt;650 mm</td>
<td>&gt;850 mm</td>
<td>&lt;650 mm &gt;850 mm</td>
</tr>
<tr>
<td>Chalk</td>
<td>19,511</td>
<td>8</td>
<td>10</td>
</tr>
<tr>
<td>Sandstone</td>
<td>15,415</td>
<td>8</td>
<td>10</td>
</tr>
<tr>
<td>Limestone</td>
<td>5,829</td>
<td>28</td>
<td>30</td>
</tr>
<tr>
<td>Crag</td>
<td>3,095</td>
<td>&lt;5</td>
<td>&lt;5</td>
</tr>
<tr>
<td>Alluvium</td>
<td>8,289</td>
<td>48</td>
<td>58</td>
</tr>
<tr>
<td>Greensand</td>
<td>5,098</td>
<td>10</td>
<td>15</td>
</tr>
</tbody>
</table>

**Assessment of impacts on eutrophication**

Some NVZs are designated because they drain to waters identified as eutrophic, on the basis of elevated nitrate levels (albeit not necessarily above 50mg/l) and biological indicators demonstrating accelerated growth and undesirable disturbance. Therefore, an additional study was commissioned, via the Centre for Ecology and Hydrology (CEH) and Plymouth University, to assess the effect of any changes in nitrate loss arising under the NVZ Action Programme, on eutrophication. The case study focussed on the Taw estuary, which has been designated as an NVZ on grounds of eutrophication within the estuary.
Inputs of total inorganic N (TIN) to the Taw Estuary are dominated by rivers (90%), with the River Taw the largest contributor (65-70%). For the purpose of the study, a relatively large reduction in nitrate emissions was postulated, of c. 10% between September and May. The corresponding reduction from June to August was c. 2%. The conclusions from the study were:

- The Taw Estuary system has the potential to respond relatively quickly to reductions in nutrient inputs. The estuary is macro-tidal with short water residence times (2-3 days, depending on river flow). The inter-tidal sediments are largely sand and gravel, with relatively little mud, which implies that the sediments do not represent a significant reservoir of nutrients.
- Reduced inputs of nutrients from the catchment would, if sufficient, lead to an improved trophic status within the estuary.
- The NVZ AP measures would not appear to be sufficiently stringent to bring such a change about.
- Beneficial changes to the trophic status of the estuary are also likely to be impeded if other nutrient inputs, particularly from Ashford STW, but also possibly from the atmospheric deposition of nitrogen to the catchment, are not controlled.

**Overall conclusions**

The analysis within this project has clearly demonstrated that the most effective measures within the current NVZ Action Programme are focused on manure and fertiliser management:

- Minimise post harvest soil nitrate (i.e. at nitrate at risk of winter leaching) by using a fertiliser recommendation system to minimise the risk of over-fertilising the crop.
  - This includes making FULL allowance of manure applications when developing the fertiliser recommendation.
- Closed window of application for N fertiliser during periods when crop uptake rate is small and leaching risk is high.
- Closed window of application in the autumn on sandy/shallow soils for manure with a large proportion of readily available N.

Even so, field measurements within the project (from sites within the NVZs) show many instances of high nitrate concentrations. Modelling of the current NVZ AP measures also indicates that the impact of the AP, while locally important, is often relatively small when averaged across a catchment or the whole NVZ area:

- A potential reduction in nitrate leaching of c. 7% in a predominantly arable catchment with manure and a significant proportion of sandy/shallow soils (i.e. closed periods for manure application) – half of this benefit is lost if the soil type is such that closed periods do not apply.
- A potential reduction in nitrate leaching of c. 2% in a predominantly grassland catchment on clay soils. Impacts of fertiliser compliance measures are smaller in most grassland systems, and closed periods for manure applications do not apply on this soil type.

Reviewing current and potential mitigation methods, we can conclude that:

- Good manure and fertiliser management is vital to reducing nitrate loss.
- Under the current NVZ AP
  - Ensuring N fertiliser and manure inputs do not exceed crop requirement is effective, especially as regards improved adjustment for N supply from manure and other sources.
• The Closed Period for application of organic manure is effective, but the current formulation applies only to a small proportion of soils, which severely limits its impact overall.
• The Closed Period for applications of chemical fertiliser affects relatively few fields because good practice in this regard is already widespread.
• Modelling and experimental data confirm that additional measures could reduce nitrate leaching further, albeit at a cost:
  • Extending the Closed Period for application of organic manure to a wider range of soils would reduce nitrate losses further.
  • Recent data have demonstrated that clay soils are not always ‘retentive’ soils as previously assumed - nitrate and other manure-related pollutants may be lost following applications in autumn and especially during the period when drains flow. In contrast to sandy and loamy soils, applications of both manures and fertilisers in early spring can result in some transfer of nitrate and other manure-related pollutants to waters. Best practice requires careful balancing of the risks to all pollutants associated with different closed periods for application, taking account of the practical constraints of farming as well.
  • Establishment of crop cover by early autumn can reduce nitrate losses substantially, especially where soil N supply is high due, for example, to the use of manure in the rotation.
  • Reduction of nitrogen chemical fertiliser applications to a little below crop requirement would further reduce nitrate leaching, at a cost. The greater the reduction, the smaller the benefit per unit cost.
  • Other measures have been reviewed in previous research projects, but no additional evidence was developed in this project. These include for example reduction in the N content of livestock feed in order to reduce manure N content; and incineration of poultry litter.
• In many catchments, especially arable catchments in the drier parts of the country, nitrate concentrations may still remain above 50 mg/l even after application of all of the measures. Further reduction (if required) would imply major change in farming systems, including for example:
  • Reduction in livestock numbers (or removal of livestock manures). Nitrate losses were elevated in rotations where manure was used, even in years when the manure was not applied. Reduction in stock numbers would therefore reduce nitrate leaching, although the cost implications are serious.
  • Conversion of some arable land to extensive (unfertilised) grassland. Nitrate losses from such ‘arable reversion’ (and indeed from unfertilised long-term setaside) are typically less than 10% of the loss from arable systems. Again, costs are very high.
• Many other practices are considered ‘Good Agricultural Practice’ and are included in the current Defra Codes. Most relate to maintaining the soil in good condition and avoiding the risk of pollution via surface runoff. (Avoiding cultivation or stocking when wet, avoiding manure applications close to ditches and streams, for example). These measures are especially important for minimising the loss of other manure-derived pollutants such as P, ammonium, and pathogens, and for minimising erosion and sediment transfer.

Individual AP measures can have large effects at field scale, and in some catchments where they apply to a large proportion of the land (see Tables 1 and 2). However, taking account of land use, soil types and livestock numbers across the whole NVZ area, our assessment suggests that the overall effect of the current NVZ AP is small: somewhere in
the order of a 3% reduction in nitrate leaching loss. The change would be greater in catchments with large numbers of livestock and in groundwater catchments.

Assessment of the impact of change in loss from agricultural land on the observed status of waters in England indicated:

- Nitrate concentrations in surface water catchments fed by clay soils would respond quickly to any change in nitrate loss from agricultural land.
- The response time of groundwater aquifers (and groundwater-fed surface waters) is variable depending on rainfall (recharge) and aquifer type. The proportion of the effect of the NVZ AP that would be measurable at groundwater abstraction points within 10 years ranged from <5% to 50%.
- Impacts of the current AP measures on river and estuarine eutrophication are likely to be small.
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1 INTRODUCTION

1.1 Background

The Nitrates Directive requires Member States to take action against water pollution caused by nitrates from agricultural sources, and provides a framework for identifying such polluted waters. The European Commission acknowledges that the agricultural sources of nitrate pollution are diffuse (multiple discharges, difficult to locate), and that farms are highly sensitive to anything that affects the economic viability of their activity.

Member States are required to ensure farmers follow ‘Action Programme’ (AP) ‘measures’ (i.e. mitigation methods) to reduce nitrate pollution, either throughout the whole country or within specific Nitrate Vulnerable Zones (NVZs). In England, 66 NVZs were originally designated in 1996 for the protection of drinking water sources. The NVZs covered 8% of England’s land area, within which farmers have been required to follow an Action Programme of measures since 19 December 1998. Following a judgement by the European Court of Justice in December 2000 that the UK failed to implement the Directive fully for the protection of all waters, additional NVZs were designated in 2002 for the wider protection of all surface waters and groundwaters. These new NVZs covered approximately 47% of England and increased the total coverage to almost 55%. The Action Programme of measures entered into force within these NVZs on 19 December 2002.

The Directive specifies a 4-yearly review for the European Commission of progress in implementing the Directive. A part of this review must include an evaluation of the impact of the Action Programme measures, which should feed into predictions of future water quality benefits and subsequent revisions to the Action Programme measures as necessary.

The aim of this project was to provide this assessment of Action Programme measures.

1.1.1 The NVZ Action Programme

The Nitrates Directive states that measures in the Action Programme must include rules relating to:

- Periods when the land application of certain types of fertiliser and manure is prohibited.
- The capacity of storage vessels for livestock manure; this capacity must exceed that required for storage throughout the longest period during which land application in the vulnerable zone is prohibited.
- Limitation of the land application of fertilisers, consistent with good agricultural practice and taking into account factors such as soil type, climate, previous and current cropping and soil nitrogen (N) supply.
- Limits to the amount of manure that can be applied annually to a farm.
- Keeping of adequate farm records, including cropping, livestock numbers and the use of organic manure and N fertilisers.

Table 1.1 summarises the detail of Action Programme for the 1996 and 2002 designated NVZs.
Table 1.1. Summary of measures currently required under the NVZ Action Programme.

<table>
<thead>
<tr>
<th></th>
<th>Arable</th>
<th>Grassland</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Closed Periods</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Manufactured Nitrogen Fertiliser (on all soil types)</td>
<td>1 Sept – 1 Feb</td>
<td>15 Sept – 1 Feb</td>
</tr>
<tr>
<td>Organic Manures (with high available N on sandy and shallow soils only e.g. slurry and poultry manure)</td>
<td>1 Aug – 1 Nov (arable land with no autumn sown crop)</td>
<td>1 Sept – 1 Nov (including arable land with autumn sown crop)</td>
</tr>
<tr>
<td><strong>Nitrogen Limits</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fertiliser Nitrogen (on all soil types)</td>
<td>do not exceed crop requirement</td>
<td>do not exceed crop requirement</td>
</tr>
<tr>
<td>Organic Manures:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(i) whole farm within NVZ (including grazing deposition)</td>
<td>210 kg/ha total N(^1)</td>
<td>250 kg/ha total N</td>
</tr>
<tr>
<td>(ii) field limit(^2) (excluding grazing deposition)</td>
<td>250 kg/ha total N</td>
<td>250 kg/ha total N</td>
</tr>
<tr>
<td><strong>Spreading Controls</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Do not apply N fertiliser or organic manure when the soil is waterlogged; or flooded; or frozen hard; or snow covered.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Do not apply N fertiliser or organic manure to steeply sloping fields.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Spread N fertiliser and organic manure evenly and accurately.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Do not apply N fertiliser in a way that contaminates watercourses.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Do not apply organic manure within 10 metres of watercourses.</td>
<td></td>
</tr>
<tr>
<td><strong>Slurry Storage</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>There must be sufficient storage to meet the autumn closed period for spreading slurry.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>All new or substantially enlarged/reconstructed storage facilities must comply with the relevant regulations.</td>
<td></td>
</tr>
<tr>
<td><strong>Record Keeping</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Adequate farm records must be kept for at least five years covering cropping, livestock numbers, and the use of N fertilisers and organic manures.</td>
<td></td>
</tr>
</tbody>
</table>

\(^1\) reduced to 170 kg/ha after the first four years of Action Programme in NVZ; Farmers located within the existing NVZs designated in 1996 have been required to adhere to a lower limit of 170 kg/ha total N per year for spreading manure on arable land since 19 December 2002. From 19 December 2006, farmers located in the new NVZs have been required to adhere to this lower limit.

\(^2\) NB available N from organic manures must not exceed crop requirement

1.1.2 Review of the Action Programme

The EC requires an assessment of progress at a number of different levels:

- Current measured water quality (rivers and borehole)
- Designation of Nitrate Vulnerable Zones
- Development and implementation of Codes of Good Practice
- Development and implementation of Action Programme
- An evaluation of the effectiveness of the Action Programme
- Estimates of future water quality
This project contributes to the last two points. The EC has prepared (draft) monitoring guidelines to assist in the assessment. These guidelines recognise the difficulties of assessing the impacts and the effectiveness of measures solely from measurements of water quality at catchment-scale (i.e. at the scale monitored by the Environment Agency). These difficulties are due to:

- The difficulty in detecting a small change in N concentration against background variability (especially in surface waters).
- The long time-scale for response to changes in land management practices (decades to centuries in groundwaters or groundwater-fed surface waters).
- The difficulty in distinguishing effects from other factors (e.g. weather patterns influencing N loss or drivers of farm management change other than the NVZ AP, such as economics or epidemics such as Foot and Mouth disease).
- The difficulty in assessing the impacts of individual measures because the Action Programme requires combinations of measures to be applied on a farm.
- The contribution of point sources to the overall catchment load of nitrate (and phosphorus).

The guidelines, therefore, provide for Member States to integrate information from a number of sources to produce an estimate of the likely long-term effect of the Action Programme. The emphasis is on providing an evidence-base for assessing the effectiveness of individual measures.

Section 2 (‘Approaches’) details the methodology and the rationale for the approach in more detail. However, in summary, this project has based its approach around the following actions:

- Field measurements on commercial farms, to provide evidence of the size of nitrate losses and the effects of mitigation methods under ‘real world’ farming (rather than experimental plots).
- Use of field-scale and catchment-scale models (validated against the field data) to quantify the likely effectiveness of combinations of measures as applied to the NVZs.
- Use of other data and information to supplement our own assessments.

1.2 Project objectives

The overall objective was to provide robust scientific evidence, from modelling and monitoring data, of the impact of the NVZ Action Programme measures in NVZ areas that would:

- Be appropriate for feeding into the wider Nitrates Directive policy question of what further changes to the individual components of the Action Programme measures could be needed in the future, and by how much these changes would increase environmental benefit?
- Be suitable for feeding into any future wider, long-term agricultural monitoring strategies and reporting requirements (e.g. under the Water Framework Directive, OSPAR, the HARP process);
- Satisfy EC monitoring requirements under the Nitrates Directive (Article 7) in order to feed into the next review of NVZ designations and Action Programme measures in 2004 and beyond;
- Satisfy four-yearly reporting requirements to the European Commission under the Nitrates Directive (Article 10 report) and beyond (next report is due by June 2004 for years 2000 to 2003).

1.3 Structure of this report

The report is separated into the key elements of the project:
Approaches (Section 2) – summarises the approaches taken within the project and the rationale for the methodology, i.e. a combination of field measurements and modelling of nitrate losses.

Measurements (Section 3) – reports on the key pieces of evidence from the field measurements, in terms of factors affecting nitrate loss and mitigation methods for reducing losses. A key aspect of this project is that the measurements were taken on commercial farms, rather than experimental plots.

Approach to model development for assessment of impacts (Section 4) – describes the approach used for model development approaches and validation of these models against field data.

Modelling the impact of measures (Section 5) – describes the modelling undertaken in support of this project and the data sources.

Implications and discussion of impacts of the NVZ AP (Section 6) – takes the key results from measurements and applies modelling to provide an assessment of the effectiveness of the current AP.

Conclusions (Section 7) – includes an assessment of potential new mitigation methods that might be considered for future Action Programmes.

The project has produced a large amount of measurement and modelling data, as well as several summary reports, all of which contribute to the overall assessment of the NVZ Action Programme. To keep this main report brief, the detail is presented in Appendices (Table 1.2). The remainder of this main report focuses on interpreting these results, discussing them in context with other data and developing the main policy implications.

Table 1.2. Locations of NIT18 project outputs in report appendices.

<table>
<thead>
<tr>
<th>Title</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>I Selection and description of monitoring sites</td>
<td>Includes an overview of the monitored surface water and groundwater catchments, and provides contextual information.</td>
</tr>
<tr>
<td>II Monitoring results</td>
<td>Presents detailed data from the field sites monitored:</td>
</tr>
<tr>
<td></td>
<td>- Land use and environmental conditions</td>
</tr>
<tr>
<td></td>
<td>- Farming practice</td>
</tr>
<tr>
<td></td>
<td>- Measurements of SMN and nitrate leaching</td>
</tr>
<tr>
<td>III Model description</td>
<td>Description of model approaches developed under NIT18 to allow assessment of NVZ effects.</td>
</tr>
<tr>
<td>IV Model Evaluation and Results</td>
<td>Description of the testing and validation of the developed model using NIT18 and other data, and application to selected catchments.</td>
</tr>
<tr>
<td>V Review and categorisation of nitrate transport in groundwater systems</td>
<td>Report produced by BGS to address aquifer response times and place our nitrate leaching assessments to groundwater in context for a range of aquifer types.</td>
</tr>
<tr>
<td>Title</td>
<td>Comments</td>
</tr>
<tr>
<td>------------------------------------------------------------</td>
<td>--------------------------------------------------------------------------</td>
</tr>
<tr>
<td>VI Eutrophication in rivers: an ecological perspective</td>
<td>Report produced by CEH, assessing the implications of APs on the designation of eutrophic NVZs.</td>
</tr>
<tr>
<td>VII A Review of Estuarine Eutrophication, with specific reference to England and Wales</td>
<td>A review of the role of nitrogen (N) and phosphorus (P) in the eutrophication of estuaries within England and Wales</td>
</tr>
<tr>
<td>VIII The Taw River Catchment and Estuary: A case study for the effects of NVZ measures</td>
<td>A case study to assess the potential effects of NVZ measures on the eutrophic status of the freshwater streams and the loads of nutrient being delivered to the Estuary</td>
</tr>
<tr>
<td>X Other relevant projects</td>
<td>A summary of other (Defra-funded) projects that have direct relevance to the NIT18 project objectives, and which contribute information and understanding to synthesis of the project.</td>
</tr>
<tr>
<td>XI Model uncertainty</td>
<td>Discussion document addressing the issues</td>
</tr>
</tbody>
</table>
2 APPROACHES

2.1 Overview

This methodology provides an evidence-based approach to assessment of the effectiveness of the Action Programme on nitrate loss from agricultural land, including the effectiveness of individual measures or changes. The project complements other work and data sources (Appendix X).

The project started in spring 2004, allowing three winters for measurement (2004/05, 2005/06 and 2006/07), and ended with final reporting in May 2007. Figure 2.1 summarises the overall approach of the project.

Figure 2.1. Schematic representation of the project structure.

The whole project is based around the linkage between field measurements and modelling (A-E, Fig. 2.1) and is the prescribed approach presented in the EC’s draft reporting guidelines. It is important to stress the need for both approaches. Whereas field measurement data are essential to providing robust information on 'typical' nitrate losses under commercial agriculture, it is not feasible to monitor every field within the NVZs and modelling is required to bring in the understanding gained from the whole scientific knowledge base, and to scale up the results.

A network of measurement sites was set up (A, Fig. 2.1) and monitored over three winters (B, Fig. 2.1). Sites were carefully chosen to represent the range of land uses and livestock systems soil types and climate within NVZs. The field measurements collected within the project were able to provide evidence from real commercial farms on the factors that affect nitrate loss, and the scale of nitrate loss from agricultural land under a range of conditions. Past experience with other schemes (e.g. the Nitrate Sensitive Areas (NSAs) and the first round of NVZs) shows that such realistic evidence makes a valuable contribution to stakeholder discussion and facilitates change by providing a clear and readily understood evidence base. It extends the evidence provided by experimental

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1 Measurement data collected and archived. However, timing of the project deadline did not allow the final year's data to be incorporated into the project report.
data, and is demonstrably relevant to current farming conditions. Monitoring data also often provide evidence relating to issues not covered by experimental data (for example, there may be changes in management of certain crops such as set-aside; or new crops; or abnormal weather).

Extending these measurements to build up a quantitative assessment of measures and to scale up to catchment and national effects, however, requires the use of robust nitrate leaching models (Fig. 2.1). The models must be able to use farm management data (e.g. fertiliser application rates, cropping and manure applications) to provide an estimate of likely leaching losses at the field and catchment scale.

An important component of the project was, therefore, to develop these models (C, Fig. 2.1) and validate them against the field measurements collected in the project (D, Fig. 2.1), as well as against other experimental data. The measurement data were crucial for evaluating model performance, including responsiveness to management or other factors, (e.g. if weather conditions are unusual), and the model’s ability to deal with the full range of conditions on commercial farms.

The model system was then used to assess the effectiveness of the Action Programme measures by combining with farm practice, climate and soil data (E, Fig. 2.1).

2.2 Field measurements

In summary, sites were established on a range of commercial farms to measure nitrate loss, and soil nitrate at risk of leaching. Section 3 provides more detail on sites and their locations and the rationale for their choice. A network of sites was established on both surface and groundwater catchments. The main activities are summarised thus:

- Establishment of groundwater and surface water sites, and monitoring during winters 2004/05, 2005/06 and 2006/07
- Collection of permanent site details and weather data at each site
- Measurements of nitrate leaching and water flux at each site
- Measurement of late autumn Soil Mineral Nitrogen (SMN), as an indicator of N at risk of leaching, and soil properties at each site
- Annual collection of field-level site management data to aid interpretation of the field measurements and as input data for the models to be validated against the field measurements

A total of 203 fields were monitored on groundwater sites in 8 locations and 125 fields on surface water sites in 8 micro-catchments, ensuring measurements across a range of cropping systems and geoclimatic regions. The British Geological Survey (BGS) provided supporting information on the choice of groundwater aquifers used in the project.

Soil Mineral Nitrogen (SMN) was measured in late autumn on all fields as an indicator of risk of nitrate loss arising from the management of that particular field. These measurements provided some insurance against problems with measurement of actual nitrate leaching (such as very dry winters). They also facilitated attribution of N loss in surface water catchments, where monitored drains sometimes represented flow from several fields.

On groundwater sites, nitrate leaching was also measured on about 100 fields each winter, using porous ceramic cups. Porous ceramic cups (Webster et al., 1993) have been shown to be appropriate for measurement of nitrate loss from the free draining soils (Fig. 2.2) on the groundwater catchments (with the exception of the chalk sites: see Williams and Lord, 1997). The technique gives results for individual fields, which facilitates linking field management data to nitrate loss.
Porous cups are not suitable for quantifying nitrate loss from clay soils, which are the dominant soils in many surface water catchments. Instead it was necessary to identify hydrological ‘micro-catchments’, and monitor individual drains or groups of fields within the micro-catchment using flow monitoring and automated water samplers (Fig. 2.3). The 23 water sampling points were chosen such that activity in a field or collection of fields could be linked to the measured nitrate loss, and such that there was no significant point source (sewage) contribution. Where sampling stations represented more than one field, the SMN measurements and field management data were used to differentiate the likely contributions of individual fields to loss at that site.

Detailed land use, cropping & land management data have been collected for 2002-2006 harvest years. A database was developed to manage and check this large dataset. The farm management, soil and climate data provided the input data for estimating nitrate leaching by models.

2.3 Model development and evaluation

Measurements of nitrate leaching can demonstrate the impact of management on nitrate emissions. In order to generalise this understanding to quantify impacts at the catchment scale, we need to be able to extrapolate the effects to other cropping, management, soil or climatic conditions. To this end, a field-scale model (NIPPER) was developed and
tested within this project. Because of growing concern regarding patterns of nitrate losses from clay soils, and their implications for mitigation strategies, particular attention was paid to hydrological and associated nitrate transport modelling for clay soils. For this reason also, a daily time step was chosen, to allow the model to reflect the rapid variations in nitrate concentration in drainage from clay soils.

A system for integrating the model with catchment-level spatial statistics on land use, livestock numbers, weather and soils has been developed so that the impact of measures can be assessed for particular catchments.

Model results were evaluated initially against a wide range of data from controlled experiments, and against the behaviour of other proven models such as MANNER (Chambers et al., 1999). The response to implementation of change was tested in the same way. The model was then evaluated against the monitoring data from the current project, to check that it could reproduce nitrate losses from land under a wide range of current conditions on commercial farms.

### 2.4 Assessment of effectiveness

The purpose of the project is to provide a toolkit for assessment of the impact of measures. This assessment may be at field scale or catchment scale. Field scale is the most appropriate scale for proper validation and evaluation of the results, because of the detailed, site-specific input data and measurements available. It is at this scale that the model's ability to reproduce the measured data from the monitoring project is tested.

At catchment scale, the same approach is used, but with statistically representative management scenarios. The results can be compared to catchment nitrate concentrations, but this test is not robust, because input data are rarely known with certainty, but rather based on survey statistics; the single measured value represents results from hundreds or thousands of fields; and many factors may intervene to modify or delay the effect at the sampling point of change in emissions from the land.

In order to assess the impact of the NVZ Action Programme, the baseline conditions (representing practice prior to implementation of measures) were taken from survey data. The British Survey of Fertiliser Practice has recently been adapted to provide improved, field-by-field information to allow estimation of change in management since implementation of the NVZ Action Programme, including quantity and data of application of manures, and the degree to which this affects fertiliser inputs. Scenarios representing prior and post-implementation management, for the same catchment and land use, were developed and their effects modelled.

### 2.5 Additional studies within the project

Although most of the project focused on measurement and modelling of nitrate loss from the ‘agricultural zone’ of the soil, an assessment of effectiveness needs to consider effects beyond edge of field. Two complementary studies were, therefore, undertaken to assess consequent impacts on water status. These studies related to (a) time-scale of response in groundwater aquifers and (b) impact of the Action Programme on eutrophication of rivers and estuaries.

#### 2.5.1 Time-scale of response within groundwater aquifers

The EC requires some assessment of the time-scale of response of water quality to the measures applied to land. While most management changes rapidly affect nitrate loss from agricultural land, and these effects are rapidly transmitted via drains and ditches to surface water systems, it can take many years before these effects are evident in
groundwater abstractions. This delay also affects groundwater-fed river systems. Detailed modelling of such responses is both expensive and uncertain for any individual borehole. However, there are broad principles relating rock type, drainage volume and aquifer depth to typical response rates.

The British Geological Survey (BGS), therefore, undertook to investigate the responsiveness of groundwater aquifers in order to provide a sound evidence base for Defra's response to the EC. The work comprised three components:

- Review existing available modelling techniques.
- Subdivide national hydrogeology into a set of workable categories for modelling. The outcome has been a scientifically sound broad basis for classifying groundwater bodies in terms of their typical response times to change at the soil surface.
- Undertaking some simple modelling of response times for a range of representative aquifers. The effects of these delays on the response time at the groundwater abstraction point was modelled in more detail for a case study catchment in sandstone, to illustrate the approach.

Appendix V reports the outputs from this work and it is summarised in Sections 4.4, 5.4 and 6.4.

2.5.2 Eutrophication

Eutrophication is defined in the Nitrates Directive as:

"the enrichment of water by nitrogen compounds, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned".

If waters are found to be eutrophic, or in the near future may become eutrophic, and a significant amount of the nitrate present in the waters comes from agricultural sources, then the Member State must designate the land draining into the affected waters as NVZs and put in place Action Programmes or adopt Action Programmes across the total territory.

Additional studies were undertaken by the Centre for Ecology and Hydrology (CEH) to assess the effect of any changes in nitrate loss arising under the NVZ Action Programme, on eutrophication in rivers and estuaries.

The approach comprised of three parts:

- A review of riverine eutrophication (Appendix VI) - The purpose was to review the role of N and P in the eutrophication of rivers within England and Wales. The report reviewed nutrient inputs to rivers; the role of nutrients in the eutrophication of rivers; the response of biota to nutrient enrichment; monitoring of changes due to eutrophication and the management of riverine eutrophication, including the potential impact of the NVZ Action Programme measures on riverine eutrophication in England and Wales.
- A review of estuarine eutrophication (Appendix VII) - The purpose was to review the role of N and P in the eutrophication of estuaries within England and Wales.
- A case study for the impacts of NVZ measures on eutrophication (Appendices VIII and IX) - The aim of this work was to examine how the current trophic status of the Taw Estuary may change in response to the implementation of NVZ Action Programme
Measures in the catchment of its main freshwater source, the River Taw. Details of the methodology and results are summarised Sections 4.5 and 5.5 and in Appendices VIII and IX.

2.6 Activity in other Member States

The Nitrates Directive requires that Member States set up monitoring programmes to test the effectiveness of the NVZ Action Programmes, and the EC accepts that information from a wide range of sources must be integrated to provide this evidence, as described earlier. The methodology adopted in this project fits very well with the approaches prescribed by the EC in their draft guidelines. Other Member States have adopted variations on this theme and some examples are provided below.

2.6.1 Denmark

The NVZ Directive was incorporated into existing Danish Water Protection Legislation. The latest targets (introduced in 2004) were:

- 13% reduction of N-leaching in 2015 compared to 2003
- P-surplus in Danish agriculture to be halved by 2015.

In terms of monitoring, a yearly interview is undertaken with farmers in 5 small catchments (green circles in Fig. 2.4) to determine trends in agricultural practice and to collect data for modelling nutrient leaching. In these catchments, there is also a measuring programme for soil water, drainage, groundwater and streams, which documents trends in nutrients in waters and is used for modelling hydrological pathways.

![Figure 2.4](image)

*Figure 2.4. A summary of some of the Danish monitoring activity in support of the Nitrates Directive.*

This is a very similar approach to that taken in England under this project (NIT18), where there is a detailed monitoring programme in surface water and groundwater catchments. Like Denmark, detailed information on agricultural practices is collected from the farmers in these catchments. Together with the monitoring results this data is used to model trends at a national scale. A major difference is that the Danish approach also focuses on P.
2.6.2 Belgium (Wallonia)

In Belgium (Wallonia), a research partnership has been set up (November 2000) to monitor farming activity and water quality. Called GRENeRA (Group de Recherche ENvironnement et Ressources Azotées) the partners are NitraWal (overall management, communication and reports to the EC), FWA (popularisation of rules and communication with farmers), AquaWal (water quality monitoring and communication with the general public), UCL (validation/modification of the Action Programme and monitoring) and FUSAGx (validation/modification of Action Programme and monitoring).

The aim is to validate the Action Programme, propose modifications and carry out agricultural monitoring (including setting the standard each year for soil N levels). Thus, there are some similarities with activities within this project, namely monitoring and assessment of the Action Programme.

Soil mineral N sampling is an essential component in Belgium. Soil N measurement is carried out five times a year on representative farms: in spring (to assist with N fertiliser recommendations) at harvest, and monthly during the autumn (October, November and December, see Fig. 2.5).

In 2006, 213 samples were taken. There is a website where farmers can compare their own data against the standards for different crop types.

These measurements correspond with the SMN analysis carried out as part of Defra’s English monitoring programme and as in Belgium the data have been used to provide farmers with advice about spring N requirements both as individual letters to participating farmers and in the national farming press and via email to PLANET users.
In Belgium relationships (presence and type) between cover crops (CIPAN) and manure use are studied on 8-10 reference farms, this type of study also forms part of the monitoring/modelling described in this report.

In summary, all Member States need to provide evidence for the effectiveness of the NVZ Action Programmes in their Territories. Although the precise details differ the main approaches are to combine information from a wide range of sources at different scales. This includes national level monitoring (such as the EA water quality monitoring programme), but also finer scale measurements, combined with information on farmer practices, and with modelling, i.e. the approaches developed in this project, NIT18.

2.7 Other information sources

NIT18 was designed to provide improved data and improved tools for answering policy questions relating to NVZ Action Programmes. However, answering such policy questions requires information from a wide range of sources, many of which fall outside the scope of this project, and are separately funded. These include agricultural census data, spatially interpolated (within the MAGPIE database maintained for Defra by ADAS); survey data (the British Survey of Fertiliser Practice has been adapted to give improved data relating to the effects of NVZ measures); and the Farm practice Survey, which provides data on manure management); together with soil and climate data mapped within the MAGPIE database. The modelling within the project also draws on many previous projects.

Appendix X summarises the most relevant projects that contribute to the assessment of NVZ Action Programme measures, and other projects that have been commissioned by Defra to provide information on impacts of measures.
3 FIELD MEASUREMENTS

3.1 Field sites and site selection

3.1.1 Overview

Monitoring sites were selected to represent the range of cropping, land management, soil and climate conditions within Nitrate Vulnerable Zones. Previous monitoring of nitrate schemes (Nitrate Sensitive Areas, groundwater NVZs) has concentrated on permeable soils over aquifers. Most data on the effects of land management on nitrate leaching also derives from such situations, because it is easier to set up controlled experiments and measure leaching on replicated trials on such soils. However, the majority of the land area within NVZs is in surface water catchments, and a substantial proportion of this is on clay soils. Therefore, this project included monitoring of sites on both groundwater and surface water clay catchments. The method of sampling differs because of the different flow paths of water draining from the land.

The general following criteria were taken into consideration during site selection:

- Representative spatial distribution across NVZ areas
- Representative range of land use (arable, grass; with and without manures; high and low N input systems)
- Farmer co-operation
- Farm enterprises should be viable long term (as far as possible)
- Rainfall range to represent main contrasts in NVZ areas

The NVZs are located mainly in the southern and eastern areas of England (Fig. 3.1) and therefore have relatively low rainfall, and a large proportion of arable cropping.

Figure 3.1. Location of Nitrate Vulnerable Zones, and of groundwater and surface water monitoring areas within this project.
3.1.2 Surface water catchments

The majority of the currently defined NVZ area covers surface water catchments, because any failure point on a river results in designation of the whole of the land upstream that contributes to the flow at that point. Many surface water catchments are fed by clay soils. Pollutants, especially those on the soil surface, can flow rapidly from clay soils to rivers. In contrast, once pollutants have penetrated into the clay matrix, they are somewhat protected, and leach less rapidly. These characteristics mean that the relationship between date of manure or fertiliser application, and risk of associated pollution, is different for clay soils than for permeable soils feeding groundwaters.

Within this study, a network of ten surface water micro-catchments were established involving around 25 individual farms within designated NVZs, covering over 100 fields across England. The land use included arable, grassland (intensive and extensive, dairy/beef), with some land receiving pig and poultry manures.

In addition to the general criteria, special criteria for these catchments were:

- Catchment headwaters, representing a small group of fields, with little/no sewage inputs or other non-agricultural sources
- Clay soils, such that the majority of flow would be captured by the monitoring of drains or headwater streams
- ‘Eutrophic’ NVZ designation criteria to include at least one catchment

Table 3.1. Overview of surface water catchments.

<table>
<thead>
<tr>
<th>Site number</th>
<th>Catchment name</th>
<th>Location</th>
<th>Land use overview</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>Whittle Dene</td>
<td>Northumberland</td>
<td>Mixed arable crops and grassland with beef cattle</td>
</tr>
<tr>
<td>11</td>
<td>Colworth</td>
<td>North Bedfordshire</td>
<td>Arable mainly cereal crops with no manures on chalky boulder clay soil</td>
</tr>
<tr>
<td>12</td>
<td>Blackwater</td>
<td>Essex</td>
<td>Mainly arable crops with straw-based pig manure and slurry on boulder clay soils.</td>
</tr>
<tr>
<td>13</td>
<td>Weaver</td>
<td>South Cheshire</td>
<td>Arable crops and grassland with dairy cattle on Crewe and Cottam series soils</td>
</tr>
<tr>
<td>14</td>
<td>Waveney</td>
<td>Norfolk</td>
<td>Mixed arable farms with pig and poultry manure applications on Beccles soils</td>
</tr>
<tr>
<td>15*</td>
<td>Holderness</td>
<td>East Yorkshire.</td>
<td>Mixed grassland and arable crops with imported organic manure on clay loam soil</td>
</tr>
<tr>
<td>16</td>
<td>Inkberrow</td>
<td>Worcestershire</td>
<td>Mixed grassland and arable crops with dairy slurry and straw-based pig manure on soils predominantly of the Evesham and Denchworth Associations</td>
</tr>
<tr>
<td>17</td>
<td>Ditchford</td>
<td>Worcestershire</td>
<td>Mixed grassland and arable crops with dairy slurry and straw-based pig manure mostly on Evesham and Denchworth soil associations</td>
</tr>
<tr>
<td>18</td>
<td>Pugsley</td>
<td>North Devon</td>
<td>Mainly grassland with dairy cows on Denbigh, Neath and Hallsworth soil series</td>
</tr>
<tr>
<td>19</td>
<td>Edgeworthy</td>
<td>North Devon</td>
<td>Mainly grassland with dairy cows on Denbigh, Neath and Hallsworth soil series</td>
</tr>
<tr>
<td>20*</td>
<td>Catwick</td>
<td>East Yorkshire</td>
<td>Mainly arable crops, cereals and peas with imported organic manure on clay loam soils.</td>
</tr>
</tbody>
</table>

* Site 15 monitored 2004/05 only; replaced by catchment 20 winter 2005/06 onwards
The ten surface water micro-catchments, including two existing sites, are outlined in Table 3.1 and were monitored using a combination of stream, ditch or open main drain and field drain monitoring, plus autumn soil mineral nitrogen (SMN) sampling. The objective was to monitor the smallest practicable spatial unit, so that the water quality results could be related back to land use. Water flux was monitored continuously, and flow proportional samples were taken for nitrate analysis after approximately every 2 mm of flow, giving a detailed picture of the pattern of nitrate loss. The measurement of SMN on all fields within the micro-catchment provided a more detailed indication of how the nitrate at risk of leaching related to the land management of the individual fields.

Further details on the sites and monitoring data are contained in Appendix I (sites) and Appendix II (results).

3.1.3 Groundwater clusters

More than 200 fields were monitored within Groundwater NVZ catchments (Table 3.2). Some of these areas had previously been monitored under the earlier Nitrate Sensitive Areas scheme (Lord et al., 1999; Silgram et al., 2003; Silgram, 2005) which ended in summer 2003. Fields in groundwater NVZs (over sandy, chalk or limestone rock) were selected in 8 clusters. Soils in these sites are typically sands, and light to medium loams.

There were eight groundwater clusters involving a total of around 45 farms with approximately 200 individual fields in designated NVZs across England. The land use included arable (cereals, low and high nitrogen break crops), grassland (including both intensive dairy and extensive); and pig and poultry manure on some sites. During the site selection phase, the British Geological Survey assessed the suitability of the sites as representative of the main aquifer types within the broader NVZ area.

Soil mineral N was measured in late autumn on all fields, as an indication of nitrate at risk of leaching. The SMN measurements, because they are a good indicator of leaching potential, also provided a degree of insurance against the risk of very dry winters or other problems affecting the porous pot leaching measurements (i.e. if there was insufficient rain to wash nitrate out of the soil profile).

Nitrate leaching in drainage waters was measured at frequent intervals during the October to April period from a subset of about 80 fields using porous ceramic cups to sample nitrate concentrations at 90 cm depth (which was taken as the depth beyond which recovery on nitrate by the following crop was not likely). The technique gives an assessment of nitrate loss in individual fields, which enabled a direct link to be established between land use management and nitrate leaching losses. As it was not possible to measure soil drainage in a non-invasive manner on these soils, nitrate leaching fluxes were calculated using a combination of measured concentrations and modelled soil drainage from the root zone using the field scale water balance model, IRRIGUIDE (Bailey & Spackman, 1996). IRRIGUIDE is based on the UK Meteorological Office’s MORECS model version 2 (Hough et al., 1996). IRRIGUIDE estimates soil moisture deficit, evapotranspiration and drainage volumes given crop type, rooting depth, soil texture and daily agrometeorological data. The IRRIGUIDE water balance model is used to underpin a number of key Defra research and consultancy projects, and has recently been updated in Defra project NT2517.

Further details on the sites are contained in Appendix I.
### Table 3.2. Overview of groundwater clusters.

<table>
<thead>
<tr>
<th>Site number</th>
<th>Catchment name</th>
<th>Location</th>
<th>Land use overview</th>
<th>Geology</th>
</tr>
</thead>
<tbody>
<tr>
<td>21</td>
<td>Nottinghamshire Yorksop, North Notts</td>
<td>Arable crops and grassland with dairy and beef; pigs and poultry</td>
<td>Sand</td>
<td></td>
</tr>
<tr>
<td>22</td>
<td>Yorkshire Doncaster, South Yorkshire Newport</td>
<td>Arable crops with pigs and poultry</td>
<td>Sand</td>
<td></td>
</tr>
<tr>
<td>23</td>
<td>West Midlands Newport</td>
<td>Arable crops, grassland, beef and dairy, pigs and poultry</td>
<td>Sand</td>
<td></td>
</tr>
<tr>
<td>24</td>
<td>Devon Duckaller / Otter valley, near Mamhead Hampshire Avon</td>
<td>Mixed grassland with beef and dairy plus arable crops including maize</td>
<td>Sand</td>
<td></td>
</tr>
<tr>
<td>25</td>
<td>Hampshire Hampshire</td>
<td>Arable with no manures; some grass with dairy or beef</td>
<td>Chalk</td>
<td></td>
</tr>
<tr>
<td>26</td>
<td>Lincolnshire Wolds</td>
<td>Mainly combinable arable crops with pigs</td>
<td>Chalk</td>
<td></td>
</tr>
<tr>
<td>27</td>
<td>Sleaford Lincolnshire</td>
<td>Mainly combinable arable crops with pigs</td>
<td>Limestone</td>
<td></td>
</tr>
<tr>
<td>28</td>
<td>Old Chalford Oxfordshire</td>
<td>Mainly arable cereals with no manures; some extensive grassland</td>
<td>Limestone</td>
<td></td>
</tr>
</tbody>
</table>

### 3.2 Data collection

Within this project, a database was constructed including land use, land use management practices, and resulting nitrate loss which covers three winters of leaching data, and associated management information. Data from two winters are reported in this report: data relating to nitrate losses for the third winter (2006/7) has been completed but is not reported within the scope of the current project. Overall, about 300 fields were monitored each year (200 in groundwater areas and 100 in surface water areas). The land use management information collected included data at farm level (livestock numbers of different types and ages, and at field level (including land use, sowing date, harvest date, estimated yield, number of cuts of grass, primary cultivations, and the amounts, types and loadings of fertilisers and manures). Land use and land use management data were collected annually, and included additional information relating to the two years prior to the start of this project (to enable us to take into account the effect of previous land use on the risk of nitrate loss).

### 3.3 Site data

#### 3.3.1 Soil properties and total nitrogen content

The soil types within the monitoring sites could be broadly classified as:

- **Clays to depth.** These soils are characteristic of the surface water sites. Those in arable cropping generally have artificial drainage, while those under grass may depend more on ditches with occasional drains.

- **Medium soils.** Soils over chalk typically have topsoils of loamy or clayey texture, but are permeable to depth. Medium soils also occur within groundwater catchments over limestone, sandstone and clay, especially in valleys. They have moderate to good water holding capacity.

- **Light sandy or shallow soils.** These soils are typical of groundwater sites over sandstone; and soils over hard limestone such as in Lincolnshire. Many of these soils are subject to the NVZ AP closed period for manure applications.
Soil total N content is an indicator of the soil’s inherent capacity to supply nitrate through mineralisation. Nitrate supplied during the growing season may be used by crops, but that supplied in autumn, especially on arable soils, is at risk of leaching. Topsoil N content, measured on all sites at the start of the project, ranged from close to 0.15% on the most sandy sites, to 0.5% or more on long-term grassland on clay soils (e.g. Inkberrow and Weaver surface water catchments) (Table 3.3). SMN in late autumn, and hence leaching risk, would be expected to be greater on soils with higher total N content. However on grassland sites this is counterbalanced by a smaller mineralisation rate (due to lack of soil disturbance), and the presence of an established crop during the winter, which reduces the accumulation of nitrate.

Table 3.3. Total %N in topsoil (0-15 cm depth) summarised by catchment (n=358).

<table>
<thead>
<tr>
<th>No.</th>
<th>Catchment name</th>
<th>Total %N in topsoil</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>Whittle Dene, Northumberland</td>
<td>0.27</td>
</tr>
<tr>
<td>11</td>
<td>Colworth, Bedfordshire</td>
<td>0.23</td>
</tr>
<tr>
<td>12</td>
<td>Blackwater, Essex</td>
<td>0.28</td>
</tr>
<tr>
<td>13</td>
<td>Weaver, Cheshire</td>
<td>0.60</td>
</tr>
<tr>
<td>14</td>
<td>Waveney, Norfolk</td>
<td>0.21</td>
</tr>
<tr>
<td>16</td>
<td>Inkberrow, Worcester</td>
<td>0.63</td>
</tr>
<tr>
<td>17</td>
<td>Ditchford, Worcester</td>
<td>0.31</td>
</tr>
<tr>
<td>18</td>
<td>Pugsley, Devon</td>
<td>0.35</td>
</tr>
<tr>
<td>19</td>
<td>Edgeworthy, Devon</td>
<td>0.39</td>
</tr>
<tr>
<td>20</td>
<td>Catwick, Yorkshire</td>
<td>0.24</td>
</tr>
<tr>
<td>21</td>
<td>Nottinghamshire Sands</td>
<td>0.15</td>
</tr>
<tr>
<td>22</td>
<td>South Yorkshire Sands</td>
<td>0.15</td>
</tr>
<tr>
<td>23</td>
<td>West Midlands Sands</td>
<td>0.16</td>
</tr>
<tr>
<td>24</td>
<td>Devon Sands</td>
<td>0.14</td>
</tr>
<tr>
<td>25</td>
<td>Hampshire Chalks</td>
<td>0.40</td>
</tr>
<tr>
<td>26</td>
<td>Lincolnshire Chalks</td>
<td>0.19</td>
</tr>
<tr>
<td>27</td>
<td>Lincolnshire Limestone</td>
<td>0.23</td>
</tr>
<tr>
<td>28</td>
<td>Oxfordshire Limestone</td>
<td>0.37</td>
</tr>
</tbody>
</table>

3.3.2 Land use

The land use on the monitored sites reflected fairly closely that within the NVZ area as a whole (Fig. 3.2). The sites contained less grassland than England and Wales as a whole, because the NVZs are largely in eastern areas, which are predominantly arable. The grassland sites were selected with emphasis on intensive systems, although extensive systems were also represented. The lightest soils under arable cropping were in mixed winter and spring crop rotations, and included crops such as potatoes, sugar beet and peas. The clay soils (surface water catchments) were either under grass or predominantly winter cropping (winter cereals, oilseed rape).
3.3.3 Fertiliser and manure inputs

Fertiliser inputs were similar to national averages by crop, as recorded within the British Survey of Fertiliser Practice (BSFP) and the recommendations in Defra’s Reference Book 209 (MAFF, 2000). Fertiliser inputs were greatest to potato and winter oilseed rape (OSR) crops, closely followed by winter wheat (Fig. 3.3). As would be expected, N-fixing crops such as peas and clover received negligible fertiliser N inputs.

Management information revealed that overall around 25% of fields received manure applications. On these fields, loadings ranged from less than 100 to about 250 kg/ha N as manure (mean 170). The proportion of this estimated to be ‘readily-available N’ i.e. available to crops or at risk of leaching, varied from 10 to 50%, being greatest for slurry and poultry manures and least for old (stored) farmyard manure (FYM). The dates of manure applications broadly reflected those recorded by national surveys such as the
Farm Practice Survey (Table 3.4). Only a minority of the area was subject to an autumn closed period under current NVZ Action Programme rules.

Table 3.4. Manure application timing within the monitored NVZ areas.

<table>
<thead>
<tr>
<th>Month</th>
<th>Percentage of applications</th>
<th>Percentage of total manure N load</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jan</td>
<td>2</td>
<td>5</td>
</tr>
<tr>
<td>Feb</td>
<td>14</td>
<td>9</td>
</tr>
<tr>
<td>Mar</td>
<td>12</td>
<td>18</td>
</tr>
<tr>
<td>Apr</td>
<td>10</td>
<td>8</td>
</tr>
<tr>
<td>May</td>
<td>7</td>
<td>7</td>
</tr>
<tr>
<td>Jun</td>
<td>5</td>
<td>2</td>
</tr>
<tr>
<td>Jul</td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Aug</td>
<td>7</td>
<td>8</td>
</tr>
<tr>
<td>Sep</td>
<td>9</td>
<td>18</td>
</tr>
<tr>
<td>Oct</td>
<td>9</td>
<td>6</td>
</tr>
<tr>
<td>Nov</td>
<td>7</td>
<td>10</td>
</tr>
<tr>
<td>Dec</td>
<td>11</td>
<td>5</td>
</tr>
</tbody>
</table>

Figure 3.4 shows data for fertiliser and manure N applications to all groundwater fields with winter wheat in harvests 2004 and 2005, grouped into sites which received no manures, and those that did receive manures in the current and previous harvest years. These results show that fertiliser N inputs tend to be smaller on fields that received manures relative to those that did not under comparable management (Fig. 3.4). This is in line with the NVZ Action Programme requirements, and with the recommendations in RB209, and provides evidence that NVZ farmers are making allowance for the N made available from manures applied in the current and previous years. This is in contrast to the situation prior to NVZs as recorded in the British Survey of Fertiliser Practice in earlier years. The fertiliser and manure loadings across the monitored sites were broadly similar to those recorded nationally by the British Survey of Fertiliser Practice.

![Figure 3.4. Groundwater sites: fertiliser N application rates to winter wheat in harvests 2004 and 2005, in situations with and without manure applications in the current and previous years.](image)
3.3.4 Crop yields

Crop yields were broadly in line with expectation on good commercial farms (Table 3.5). Since inputs of fertiliser and manure were also typical, nitrate leaching from these sites is considered to be fairly representative of the situation in the NVZs as a whole.

Table 3.5. Arable yields in monitored fields, summarised by crop and harvest year.

<table>
<thead>
<tr>
<th>Harvest crop</th>
<th>Harvest year 2002</th>
<th>Harvest year 2003</th>
<th>Harvest year 2004</th>
<th>Harvest year 2005</th>
<th>Total reported area in 2005 (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wheat (winter)</td>
<td>8.3</td>
<td>7.5</td>
<td>8.1</td>
<td>7.6</td>
<td>959</td>
</tr>
<tr>
<td>Barley (winter)</td>
<td>6.3</td>
<td>6.8</td>
<td>6.1</td>
<td>6.0</td>
<td>400</td>
</tr>
<tr>
<td>Barley (spring)</td>
<td>5.7</td>
<td>5.6</td>
<td>5.3</td>
<td>5.2</td>
<td>406</td>
</tr>
<tr>
<td>Wheat (spring)</td>
<td>3.8</td>
<td>6.6</td>
<td>7.4</td>
<td>5.5</td>
<td>47</td>
</tr>
<tr>
<td>Oilseed rape (winter)</td>
<td>3.1</td>
<td>3.2</td>
<td>3.0</td>
<td>3.4</td>
<td>216</td>
</tr>
<tr>
<td>Peas (dried and vining)</td>
<td>4.1</td>
<td>4.4</td>
<td>3.8</td>
<td>3.6</td>
<td>145</td>
</tr>
<tr>
<td>Potatoes</td>
<td>33.4</td>
<td>47.5</td>
<td>49.7</td>
<td>48.5</td>
<td>46</td>
</tr>
<tr>
<td>Sugar beet</td>
<td>54.5</td>
<td>57.3</td>
<td>48.3</td>
<td>54.6</td>
<td>136</td>
</tr>
<tr>
<td>Forage maize</td>
<td>37.0</td>
<td>42.9</td>
<td>40.0</td>
<td>44.6</td>
<td>35</td>
</tr>
</tbody>
</table>

3.4 Autumn Soil Mineral Nitrogen (SMN)

Soil mineral N (SMN; nitrate plus ammonium content to 90 cm depth) in late autumn is used as an indication of N at risk of leaching in the subsequent winter. In contrast to topsoil total N content, SMN varies from month to month and from year to year depending on the balance of processes that consume and release mineral N.

Systematic differences in SMN were identified between catchments, with the largest values measured in the Taw catchment NVZ in Devon (Pugsley and Edgeworthy sites) in
predominantly grassland systems on clay soils, and the smallest values in mainly arable systems in the east (e.g. Notts sands, Lincs limestone, Colworth) (Fig. 3.5).

These differences reflect the combined effects of differences in topsoil total N content, and hence capacity to generate nitrate after crop growth ceases or slows; and the mix of land use and land use management. Late autumn SMN was consistently greater after break crops such as peas, potatoes and oilseed rape due to the relatively high N content in crop residues, and in the case of peas, oilseed rape and early potatoes due to the early harvest (Fig. 3.6). SMN was usually small after (or under) sugar beet, as its late harvesting (into November and December) allows an extended period of plant uptake. SMN measurements tended to be fairly high following maize, which may be associated with a greater proportion (87%) of maize fields receiving manures (46 out of 53 fields) compared to 21% across all arable land (179 out of 869 fields). This reflects typical farm practice, where land cropped to maize is often used as a disposal route for manure materials. SMN under grassland varied widely. SMN was high under intensive grassland systems, although actual nitrate leaching tended to be less than this might suggest (see later), due to the continued presence of an established crop over the winter drainage period.

![Figure 3.6. Late autumn SMN (0-90 cm depth) across all surface water and ground water sites in winters 2004/05 and 2005/06, summarised by previous crop (n=300 Autumn 2004; n=301 Autumn 2005). Standard errors are shown.](image)

The large amount of SMN under and following some set-aside crops was notable (Fig. 3.6), and has been reported elsewhere (e.g. Silgram, 2005). This was mainly due to the early crop destruction commonly practised. Manure may have also been applied. These management practices may be good on ecological or economic grounds, but the resultant nitrate leaching can be extremely large.

SMN is not a perfect predictor of nitrate leaching, because the pool of N may change due to inputs or crop uptake; and because the proportion of this N that actually leaches depends on soil type and prevailing rainfall. However, within a system, it can still serve as a meaningful indicator of relative risk. Collection of SMN data also gives some insurance against failure in measurements of loss of nitrate leaching (e.g. due to very dry winters limiting drainage, or due to accidental damage of field instrumentation).
3.5 Nitrate leaching: groundwater sites

The two winters monitored for nitrate leaching (winters 2004/5 and 2005/6) were unusually dry, with mean hydrologically effective rainfall (HER) on the groundwater sites typically less than 100 mm. Some sites yielded no drainage at all in one or other of the two winters, and on those sites where drainage did occur nitrate concentrations tended to be greater than normal (and loads smaller) due to this lack of dilution. Results are presented in Table 3.6 for winter 2005/6 only, and for both winters in Figures 3.7 and 3.8.

The relationship between late autumn SMN and nitrate leaching is illustrated by data from groundwater sites in 2005/6 (Table 3.6). Those land uses with elevated SMN were in general those which resulted in greatest nitrate concentrations in drainage waters. Nitrate leaching from arable land was smallest after sugar beet (due to a combination of late crop harvest and low drainage), and greatest after peas and potatoes (both of which leave large residues of N-rich material which mineralise rapidly). Losses under grassland were smaller than under arable, despite the relatively high SMN on some sites, partly due to the presence of a crop throughout the winter. Such data are invaluable to support the development and testing of field-scale nitrate leaching models, such as the NIPPER model developed under this project.

Table 3.6. Late autumn SMN and nitrate leaching from groundwater sites, winter 2005/6. Results for all sites (including surface water sites) are presented in Figures 3.7 and 3.8.

<table>
<thead>
<tr>
<th>Previous crop</th>
<th>Soil Mineral Nitrogen</th>
<th>Nitrate leaching</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Late autumn SMN (kg N/ha)</td>
<td>n</td>
</tr>
<tr>
<td>Wheat (winter)</td>
<td>89</td>
<td>48</td>
</tr>
<tr>
<td>Barley (winter)</td>
<td>68</td>
<td>26</td>
</tr>
<tr>
<td>Barley (spring)</td>
<td>85</td>
<td>33</td>
</tr>
<tr>
<td>Sugar beet</td>
<td>50</td>
<td>14</td>
</tr>
<tr>
<td>Peas</td>
<td>80</td>
<td>10</td>
</tr>
<tr>
<td>Potatoes</td>
<td>93</td>
<td>5</td>
</tr>
<tr>
<td>Set-aside</td>
<td>201</td>
<td>6</td>
</tr>
<tr>
<td>Grass ley &gt; 3 years</td>
<td>96</td>
<td>21</td>
</tr>
<tr>
<td>Grass / clover</td>
<td>97</td>
<td>5</td>
</tr>
<tr>
<td><strong>Grand Total</strong></td>
<td><strong>88</strong></td>
<td><strong>205</strong></td>
</tr>
</tbody>
</table>

* excludes field 22/2/2 following sugar beet receiving 150 kg N/ha as old cattle FYM in mid Nov 05 (mean nitrate concentration 234 mg NO₃/l, flux 14 kg N/ha)

Mean nitrate concentrations exceeded 50 mg/l nitrate following all crops except sugar beet, and under grass/clover. Both reported winters were unusually dry, and in a more typical winter nitrate concentrations would be expected to be somewhat lower than the values reported in Table 3.6 and Figure 3.8. However these results are consistent with findings from previous work, in that nitrate concentrations in leachate from arable cropping in eastern areas of England (i.e. the majority of the NVZ area) typically exceed 50 mg/l.

Nitrate concentrations following rotational set-aside were extremely variable, reflecting variable management practices. For example, in winter 2005/06 three set-aside fields were monitored using porous pots, with losses ranging from 13-78 kg N/ha (drainage varied from 39-240 mm). Such measurements demonstrate that losses can be extremely large, and this is especially true in situations where the crop is destroyed early and/or manures have been applied.
3.6 Groundwater sites: case studies

3.6.1 Grassland management

Results from the groundwater monitoring fields provide evidence of the contrasting concentrations of nitrate leaving the root zone under low and high input grassland. Figure 3.9 shows results from four groundwater fields, two of which were long term grass managed intensively (fields 24/3/1 and 24/3/2), and two of which were managed as low input extensive grass/clover swards (fields 24/2/1 and 24/2/2) in a manner more akin to the Premium Arable option under the NSA scheme (Lord et al., 1999; Silgram et al., 2003; Silgram, 2005). The grass/clover swards received no mineral fertiliser and only field 24/2/2 received a small loading (60 kg N/ha) of manure. In contrast, the intensively
managed grass received 225 and 232 kg N/ha as mineral fertiliser in addition to 150 and 39 kg total N/ha as manure to fields 24/3/1 and 24/3/2 respectively. Leaching results reveal that, even under NVZ rules concerning manure loadings and closed periods, concentrations of nitrate in drainage waters in winter 2005/06 were extremely high from more intensively managed grass sites, although this effect is partly a reflection of the very dry winter (drainage was less than 150 mm over the winter period from these four fields).

Figure 3.9. Grassland, winter 2005/6: contrasting nitrate concentrations in leachate from intensively managed fields (24/3/1 and 24/3/2) and low-input extensively managed fields (24/2/1 and 24/2/2).

3.6.2 Leaching of autumn applied slurry

Field 23/3/1 provides an example of the effect of a high loading of slurry to vulnerable soils in autumn. Following harvest of a potato crop which had received manure, pig slurry was applied on 10 October 2004, as the field’s soil texture was sandy loam and therefore not subject to the 2002 Action Programme’s closed period. The field was ploughed and drilled to wheat in November, with little growth before winter. Late winter SMN was 222 kg/ha, and nitrate loss was 182 kg/ha N, resulting in an exceptionally high mean concentration of 558 mg/l nitrate in winter 2004/5 (Fig. 3.10). The closed period on sandy and shallow soils is designed to reduce the risk of such very large N losses on the most vulnerable soils, but under the existing 2002 Action Programme this applies to under 10% of soils.

The following spring, the winter wheat crop received 174 kg N/ha as fertiliser. The field was left bare over winter 2005/6. Figure 3.10 illustrates that nitrate concentrations in winter 2005/6 were initially high but rapidly fell to a more typical level of about 100 mg/l nitrate (total winter loss was still high at 105 kg N/ha).
3.6.3 Losses following legumes

The large N losses that typically occur following legume crops may be illustrated by field 21/4/1. SMN was low (40 kg/ha N) following carrots in 2004, and mean nitrate concentration in leachate that winter was 83 mg/l (still well in excess of 50). The following year, after a pea crop, autumn SMN was 86 kg/ha and mean nitrate concentration in leachate was 265 mg/l.

As is the case for most monitored fields, field 21/4/1 did not receive any N fertiliser to the pea crop, which is consistent with the recommended practice for this crop (based on RB209) and with the requirements of the NVZ Action Programme. However the British Survey of Fertiliser Practice for 2004 indicates that 7% of peas and beans did receive...
organic manures and 4% of peas (for human consumption) and 11% of beans (for animal consumption) received manufactured N fertiliser. Applications of fertiliser and manure have been made to some pea fields in the monitoring dataset (see Figure 3.3 and Appendix 2), indicating that this aspect of the AP may not be well understood.

3.6.4 Losses following set-aside

Some very high nitrate concentrations have been reported following rotational set-aside (e.g. from NSA monitoring data reported in Silgram (2005)), although this is not always the case. Data from the NIT18 project presented here also reveal elevated, but highly variable, losses under set-aside land (Table 3.6; Fig. 3.8).

For example, field 23/3/7 was in set-aside in harvest 2005 with a low SMN of 36 kg N/ha measured in December 2004. The field received 50 t/ha of dilute pig slurry in March and July 2005, and was ploughed out and drilled to winter wheat in autumn 2005. Late autumn SMN was exceptionally high at 431 kg N/ha, reflecting a combination of manure inputs and early-killed crop cover.

Another site under set-aside in 2005 followed by winter wheat had late autumn SMN of 304 kg/ha, resulting in mean nitrate concentrations in leachate of 683 mg/l. However, not all sites had elevated leaching following rotational set-aside, indicating that issues including establishment, duration, destruction, and the extent of good ground cover can influence nitrate leaching risk. Optimum management may differ depending on the balance between biodiversity, the risk of nitrate leaching, and economic factors.

Losses under longer-term set-aside which does not receive large inputs of N as manure or fertiliser would be expected to be smaller than losses from rotational set-aside or arable crops, because of the presence of continuous established crop cover.

3.7 Surface water sites: results and case studies

An overall summary of the surface water data is presented in Table 3.7. Specific case studies drawn from this dataset are presented in Sections 3.7.1 to 3.7.5. The two winters monitored for nitrate leaching (winters 2004/5 and 2005/6) were unusually dry, with many sites yielding <50% of average drainage.
Table 3.7. Summary of monitoring results related to specific surface water gauges for winters 2004/5 (top) and 2005/6 (bottom). The first two digits of each gauge reference number denote the number of the catchment in which the gauge is located (see Tables 3.1 and 3.2 for key).

<table>
<thead>
<tr>
<th>Winter 2004/2005</th>
<th>Surface water gauge reference number</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>101</td>
</tr>
<tr>
<td>Total agricultural area (ha)</td>
<td>114</td>
</tr>
<tr>
<td>Total contributory area (ha)</td>
<td>128</td>
</tr>
<tr>
<td>Area weighted annual fertiliser loading (kg N/ha)</td>
<td>134</td>
</tr>
<tr>
<td>Area weighted annual manure loading (kg N/ha)</td>
<td>10</td>
</tr>
<tr>
<td>Total winter flow (mm)</td>
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</tr>
<tr>
<td>Flow weighted mean concentration (mg NO₃/l)</td>
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</tr>
<tr>
<td>Total load (kg N/ha)</td>
<td>10</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Winter 2005/2006</th>
<th>Surface water gauge reference number</th>
</tr>
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<tbody>
<tr>
<td></td>
<td>101</td>
</tr>
<tr>
<td>Total agricultural area (ha)</td>
<td>101</td>
</tr>
<tr>
<td>Total contributory area (ha)</td>
<td>128</td>
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<tr>
<td>Area weighted fertiliser loading (kg N/ha)</td>
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<tr>
<td>Area weighted manure loading (kg N/ha)</td>
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<tr>
<td>Total winter flow (mm)</td>
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</tr>
<tr>
<td>Flow weighted mean concentration (mg NO₃/l)</td>
<td>35</td>
</tr>
<tr>
<td>Total load (kg N/ha)</td>
<td>13</td>
</tr>
</tbody>
</table>
3.7.1 Losses following peas

At Colworth the results from adjacent gauges 112 and 114 show the contrasting effect of previous crop and associated management on nitrate concentrations in drainage waters. The field draining into gauge 112 was in peas during harvest 2005, with associated high autumn 2005 SMN and measured stream concentrations (see Fig. 3.12 below). In contrast, the adjacent field draining into gauge 114 was in winter wheat in harvest 2005, resulting in more modest autumn 2005 SMN and lower stream nitrate concentrations. In such cases good autumn establishment of the next crop, or a cover crop, could have helped mitigate against the potentially high nitrate loss. Total drainage from catchments 112 and 114 was 57 mm and 38 mm respectively, giving flow weighted nitrate concentrations of 137 mg/l and 72 mg/l.

![Figure 3.12](image)

*Figure 3.12. Contrasting nitrate concentrations following peas (gauge 112, top) and winter wheat (gauge 114, bottom).*

3.7.2 Late spring increases in nitrate concentrations

The 2005-2006 data from the Weaver catchment gauge 131 (draining an area in grass) show both increases and decreases in nitrate concentration during flow events (see Fig. 3.13). The fact that increases occur late in the season (as well as the usual autumn nitrate flush) suggests new sources of nitrate were becoming available. This may be a consequence of slurry spreading during drier spring periods, or spring fertiliser application.
3.7.3 Nitrate “spikes” in late spring 2006

The data from Ditchford catchment gauge 171 (Fig. 3.14) show that the management regime associated with forage maize cropping (i.e. high organic manure inputs) can cause high nitrate concentrations in drainage waters (e.g. in response to rainfall in May 2006). The fields contributing to this gauge received farmyard manure on the 10th of March 2006, which had an N loading of 222 kg/ha and in addition also received 43 kg/ha N as mineral fertiliser on the 12th of April. The observed high nitrate concentrations were enhanced by the low drainage volumes (i.e. there was little dilution).

Similarly, measurements at the Inkberrow catchment gauge 161 showed a large spike between 15th to 23rd May 2006. Approximately two thirds of this catchment received 50 kg/ha N as cattle FYM on the 17th of March 206 and the remainder received 42 kg/ha of N as fertiliser on the 16th of May 2006. Little flow occurred between March 17th and May 15th and again the concentrations were enhanced by the small flows.
Figure 3.15. Losses of nitrate from a lowland dairy farm in the Inkberrow catchment.

These results show the difficulty of ensuring that losses from systems where organic manures and fertilisers are applied during the period of risk of drain flow do not cause enhanced nitrate concentrations in streamwaters. In this instance, the unusually dry winter of 2005/06 was followed by a notably wet period in May 2006 which accounts for the high peak concentrations observed at this time at both of these catchment gauges.

3.7.4 Correlation between high SMN and high stream concentrations

It is notable that data from Catwick sub-catchment gauge 202 shows high nitrate concentrations in drainage, declining from 180 to 100 mg/l over-winter 2005/06, and yet the overall catchment outlet gauge 201 shows much lower nitrate concentrations within the range 20-80 mg/l (see Appendix II). Only field 20/2/1 (OSR Harvest 2005; high SMN 298 kg N/ha Autumn 2005) contributes to flow at Gauge 202, whereas a total of 10 fields with a mean SMN of only 93 kg N/ha contribute to the flow measured at the main catchment outlet at gauge 201 (i.e. Field 20/2/1 has by far the highest Autumn SMN of all fields in this catchment in Autumn 2005). This is a good example of the correlation between high SMN in autumn and high measured nitrate concentrations in drainage waters.

3.7.5 Contrasting stream nitrate responses from Devon sites

The nitrate concentrations in water draining from the two catchments of Pugsley (18) and Edgeworthy (19) were notably different given their relatively close proximity to one another and the broadly similar topsoil total N contents (Fig. 3.16; Table 3.3). This point is discussed further in Section 3.11.4. Data in Figure 3.16 cover both winters 2004/05 and 2005/06, and include flow expressed as mm equivalent. The Pugsley site has a greater proportion of arable land, and a significant proportion of fields that are ploughed out of grass each year, while Edgeworthy is predominantly under grass.
Figure 3.16. Flow and nitrate concentrations at (a) Pugsley winter 2004/05, (b) Pugsley winter 2005/06, (c) Edgeworthy winter 2004/05 and (d) Edgeworthy 2005/06.
Streamwater nitrate concentrations from Pugsley (catchment 18) were much higher (typically 40-70 mg/l over-winter) compared to Edgeworthy (catchment 19) (typically 10-20 mg/l, except for a “spike” in May 06 probably associated with spring fertiliser applications). This pronounced difference in streamwater nitrate concentrations is in contrast to the differences observed in autumn SMN. The elevated stream nitrate concentrations measured in Pugsley are at least in part due to the far greater proportion of arable cropping, including land ploughed out of grass leys with high N inputs. NIPPER modelling predicted nitrate concentrations in the Pugsley catchment of more than 3 times those in the Edgeworthy catchment, due chiefly to the greater proportion of arable in the latter. Despite elevated late autumn SMN measurements for grass, of a similar order to arable, nitrate loss from grass appears much smaller. Part of the reason is continued crop N uptake over winter. In many clay grassland catchments, including these and the Weaver, measured nitrate concentrations under grass were even smaller than modelled, and much smaller than from equivalent grassland on sandy soils. This may be due to denitrification especially where drainage is imperfect.

The elevated concentrations of nitrate at Edgeworthy in May 2005 followed applications of fertiliser in April and May (no manure applications were recorded at this time). Agronomically, grass needs N fertiliser by this date because it is making very fast growth. The data show that even so late, there are risks of loss to drains. However the actual quantity of drainage at this time was small, so that the loss itself was a small proportion of the annual total.

An additional influence at Pugsley is the potential for rapid transfer from the farmstead to the stream, and one particular sloping field which lies between the farm’s milking parlour and stream. Both of these factors would tend to increase emissions of P, BOD, FIOs and other pollutants associated with excreta and manures. These are important points in terms of mitigation, as it suggests targeting management on a few fields and on a key loss pathway could help reduce losses of these pollutants. Measurements of P and other pollutant emissions at these sites could provide valuable information for targeting integrated catchment management measures.

3.8 Prior levels of nitrate leaching from agricultural land in England

The present project measures nitrate leaching from land within designated NVZs. Other sources of data can be used to indicate the situation prior to NVZs being introduced.

Designation of surface water NVZs is based on whether concentrations of nitrate in waters exceed 50 mg/l nitrate on more than 1 in 20 occasions. The Environment Agency estimated that in 1996-2000, about 36% of river reaches failed this criterion. The proportion was greater in the dry, intensively farmed east (62% in Anglian region) and smaller in the wetter west. The area designated as NVZs is greater than this proportion would suggest, because all upstream land contributing to the catchment is included in the designation. A 10% reduction in the value of this upper 95th percentile would reduce the failure proportion to 26%; and a 20% reduction would bring it down to 19%. A monitoring point that fails the 95th percentile criterion will have a mean nitrate concentration below 50 mg/l, typically only 30-40 mg/l nitrate.

Nitrate concentrations in surface waters in England have stabilised in recent years, in line with stabilisation and decline in fertiliser inputs, and implementation of environmental measures including NVZs. In some areas, concentrations appear to be falling slightly.

Nitrate concentrations in groundwater abstractions are less variable from day to day, and represent ‘older’ water, with a damped and lagged nitrate signal that has been integrated over time and space. Because of the delay between water leaving the base of the soil root
zone and being abstracted at the groundwater borehole, groundwater nitrate concentrations are still rising in many areas, and in many cases will continue to do so for some time even if nitrate concentrations at the surface are reduced substantially immediately (e.g. Silgram et al., 2003). Measurements of nitrate concentrations in water leaving the soil presented in this report show average concentrations well in excess of 50 mg/l in most arable systems in NVZs, with a more variable picture evident for grassland systems.

The following sections synthesise results by considering in turn the factors which are relevant to an assessment of the impacts of farming practice on nitrate concentrations in waters, and how the results of this project inform that assessment.

3.9 Measured nitrate leaching from agricultural land in NVZs

Mean nitrate concentrations in leachate from groundwater sites in this project were 106 mg/l nitrate, or twice the 50 mg/l criterion (Appendix II). Soil SMN in late autumn averaged 88 kg N/ha in these fields. The sites were all subject to the 2002 NVZ Action Programme, so these data indicate that even under existing Action Programme rules, concentrations will remain well above 50 mg/l nitrate in many areas. The winters were drier than average (HER of less than 100 mm), but data collected since 1990 from groundwaters (NSAs and then NVZs) confirm that nitrate concentrations under arable cropping are on average well above 50 mg/l.

Concentrations of nitrate at catchment level will be smaller than concentrations in leachate from the fields monitored, because of land that is not in agricultural use – including woodland, amenity, road margins and rough grazing. Losses of nitrate from such land are usually smaller (e.g. Silgram et al., 2004).

Mean nitrate concentrations in leachate from the surface water sites also exceeded 50 mg/l in most locations (Appendix II). These sites drain to surface water catchments, where the criterion for designation is that no more than one measurement in 20 should exceed 50 mg/l nitrate at the monitoring point. To achieve this requires mean concentrations that are smaller than 30-40 mg/l nitrate. The majority of sites, except those in long-term low-input grassland, exceeded these concentrations.

Thus, the results indicate that nitrate concentrations in leachate from agricultural land within NVZs exceed – sometimes very markedly – the levels that would trigger designation, even under the current NVZ Action Programme. These results are considered in more detail below.

3.10 Nitrate leaching risk: Baseline issues

Nitrate loss is determined by two main factors:
- quantity of nitrate ‘at risk of leaching’ during late autumn and winter (‘source’)
- the effectiveness with which this is actually leached or lost to water (‘transport’)

The former can be changed by land use and management practice, as discussed earlier. Within England and Wales there are also major contrasts in soil and climate which affect the effectiveness with which nitrate in soil in autumn is leached quantity and concentration of nitrate reaching ground and surface waters.

3.10.1 Climate

In dry areas or dry winters, nitrate leaching may be incomplete (i.e. total loss may be small) but low drainage volumes result in high nitrate concentrations leaving the soil root zone. Under wetter conditions, nitrate concentrations are smaller but total loss (in kg N/ha) is greater. For this reason, NVZs are predominantly located in the drier parts of England. Earlier data (1990-2000) from a similar project shows that mean measured nitrate
concentrations in leachate after cereals (the dominant arable crop) within NVZs were well above 50 mg/l, and often above 100 mg/l, even in winters that were much wetter than the average for arable cropping in NVZs (Fig. 3.17).

![Figure 3.17. Effect of over-winter drainage volume on nitrate loss and mean nitrate concentration in leachate following cereal crops within groundwater NVZs, 1993-99.](image)

A similar, though slightly more variable, picture emerges when data for all crops and grass are combined; mean nitrate concentrations are slightly smaller but still remain above 50 mg/l through the main climatic range covered by Nitrate Vulnerable Zones (Fig. 3.18).

![Figure 3.18. Effect of drainage volume on nitrate loss and mean nitrate concentration in leachate following all crops and grass within groundwater NVZs, 1993-99.](image)

The two winters of monitoring reported from this project (2004/05 and 2005/06) were particularly dry. For example, mean over-winter drainage in the groundwater fields in the North and East group (which includes sites 21, 22, 26, and 27) was only 96 mm and 68 mm in winters 2004/05 and 2005/06 respectively, while even in the wetter South and West group (which includes sites 23, 24, 25, and 28) drainage was only 100 mm and 129 mm in winters 2004/05 and 2005/06 respectively (see Appendix II for details). In comparison, over-winter drainage from 32 groundwater sites monitored under the Nitrate Sensitive Areas scheme
ranged from 105-279 mm in the winters 1994/5 to 2003/4 inclusive (Silgram, 2005). The abnormally low drainage for two consecutive winters resulted in notably high nitrate concentrations at the base of the root zone, as there was little dilution of soil water nitrate in drainage waters.

3.10.2 Soil type
The other control on leaching is soil type (e.g. Fig. 3.19). Light sandy or shallow soils need relatively little water to leach the nitrate in the soil profile, while more retentive loamy, silty and clay soils need more and may not be fully purged of nitrate under the typical rainfall conditions within most NVZs.

![Figure 3.19. Effect of soil type and annual rainfall on nitrate leaching after winter wheat. The ‘loamy sand’ soil represents the lightest soils, while the ‘silty’ soil is more typical of permeable soils. Results using the NIPPER model (Section 4).](image)

Clay soils comprise two components of response: rapid transfer of any nitrate at the soil surface, and a slower leaching of nitrate that is within the clay peds. Concentrations of nitrate leaving clay soils therefore tend to fluctuate widely from day to day, and these fluctuations are rapidly transferred to the river.

While mean concentrations of nitrate in drainage from clay soils may be less than those from the permeable soils found over groundwaters, due to dilution during heavy rains, they often exceed 50 mg/l. Furthermore concentrations fluctuate, and the upper 95th percentile concentrations on which NVZ designations are based will exceed the mean, and in most sites monitored exceeds 50 mg/l under arable cropping. For example, a site with an average nitrate concentration of only 35 mg/l (due to low N inputs and 25% non-agricultural land) still recorded just over 5% of winter measurements in excess of 50 mg/l nitrate. If this were a river monitoring point, the catchment would require designation. Figure 3.20 illustrates this issue with the somewhat greater nitrate concentrations more typical of arable systems.

Climate and soil type also affect land use, with arable concentrated in the drier areas, and this, in addition to their direct impact, helps explain the systematic trend towards greater N concentrations in the drier E of England, as mapped by Lord & Anthony (2000).
3.11 Nitrate leaching risk: land use and management impacts

In this Section we briefly summarise the aspects of nitrate leaching risk that are susceptible to change through modified land management, with reference to the specific case studies described for ground water and surface water NVZs in Sections 3.6 and 3.7.

The ‘quantity of nitrate at risk of leaching’ identified in Section 3.10 is most strongly influenced by land use (crop type), crop management (e.g. drill and harvest date; irrigation); N additions during autumn and winter months; and crop N uptake during autumn which reduces the soil nitrate content. The NVZ Action Programme places controls on three main management issues that affect the quantity of nitrate at risk of leaching:

- the quantity of N fertiliser applied
- the date of application of manures
- the quantity of manure and excretal N applied

Grassland management and the presence of cover crops on land which would otherwise be left bare over-winter also affect the risk of nitrate leaching, although these are not directly addressed in the current AP. The NVZ Action Programme also contains a number of ‘good practice’ measures (largely aimed at preventing direct runoff of pollutants to surface waters) which have little impact on nitrate leaching. Land management issues are considered further below.

3.11.1 Land use: arable and grassland

Soil mineral N in late autumn is an indicator of the pool of N at risk of leaching. Arable crops are harvested in late summer, and nitrate mineralised from the soil during autumn is not taken up. This nitrate is, therefore, at risk of leaching. Results presented earlier in Section 3 (e.g. Fig. 3.6) and in other research (e.g. Lord et al., 1999; Silgram, 2005) have shown that SMN levels and associated nitrate leaching losses are strongly influenced by land use type and by land management, with generally greater values:

- for crops that are harvested early (e.g. peas, oilseed rape).
- for crops that leave high-N plant residues (e.g. potatoes, oilseed rape, peas)
- for crops that receive more N input than they can efficiently use (e.g. some potato and vegetable crops; wheat for bread making)
For example, Figure 3.12 showed an almost doubling in nitrate concentration drainage from soil as a consequence of the residues left after peas (flow weighted nitrate concentrations of 137 mg/l compared to 72 mg/l).

In addition, it is clear that the large SMN under some set-aside crops can also present a high nitrate leaching risk (Table 3.6; Fig. 3.8). Managing set-aside for economic or biodiversity objectives can result in increased nitrate leaching and nitrate leaching losses and mean over-winter concentrations in drainage waters have proved to be extremely variable following and under set-aside (Appendix II).

SMN under grassland varied widely (Section 3.6.1; Fig. 3.9; Appendix II), with the largest values under the more intensive grassland systems, although the presence of an established crop cover did go some way towards limiting the actual nitrate leaching under such situations. The interaction between percentage total N in topsoil, soil texture, and crop type also influences nitrate leaching risk. The effect of longer-term grassland in building up organic N reserves largely accounts for the high topsoil %N found on such sites (0.51% N in this project). The N built up within such longer-term grassland can produce a large flush of nitrate this grass is subsequently cultivated (see for example, Silgram & Shepherd, 1999; Silgram, 2005).

The dry winters monitored in this project resulted in higher nitrate concentrations and lower nitrate leaching losses than might be expected under more typical weather conditions. This in turn tended to enhance the residual soil mineral N reserves remaining in the soil and available to the crop the following spring. The implications of this in terms of a reduced requirement for fertiliser N were publicised as advice to farmers via several articles in the farming press in spring 2005 and spring 2006.

### 3.11.2 Autumn crop cover and cover crops

Autumn sown crops can take up N that would otherwise leach. Most cereal crops are drilled too late to be effective. However, winter oilseed rape is normally drilled during September. Late autumn SMN was, on average, 10 kg/ha less where WOSR was grown than under winter cereals, despite the fact that some of the WOSR crops received autumn fertiliser.

Cover crops were a requirement under the Nitrate Sensitive Areas scheme, which ran from 1990-2003, although they are not a requirement of the 2002 NVZ Action Programme. Cover crops proved cost-effective as a means of reducing nitrate leaching (Lord et al., 1999), reducing leaching by about 40% compared to no over-winter crop or to winter cereals under equivalent conditions. By taking up nitrate that would otherwise leach, cover crops proved to be capable of reducing nitrate leaching by over 20 kg N/ha under research conditions (e.g. Silgram & Harrison, 1998; Final report for Defra project NT1508).

Within the dataset from this project, less than 5% of fields monitored in groundwater NVZs had an intentional catch or cover crop, while the conditions suitable for use of cover crops occur in about 30-40% of fields in any one year on these soils (about 70-80% of spring crops are typically suitable. Cover crops are not practicable following late harvested crops such as sugar beet and maize, because insufficient growth can be made before winter). It can therefore be concluded that there is further scope for integrating cover crops into such rotations as a mitigation tool when ground would otherwise be left bare over winter (which might represent one year in four in an arable rotation). However, research has shown that to be effective cover crops must be established early – preferably by mid-September (Lord et al., 1999), and the ability of cover crops to reduce nitrate leaching depends on choice of species which can establish green cover rapidly (Harrison & Silgram, 1998). Volunteer
barley, weed regrowth, and stubble turnips among others were all found to be effective provided good even cover was established early.

3.11.3 Fertiliser N inputs to arable land

A main objective of the NVZ AP is to avoid excessive N inputs, which are inefficiently used by crops and, therefore, increase the risk of nitrate leaching. Most arable crops are given approximately the economic optimum quantity of fertiliser for yield. Over-fertilisation results in half or more of the additional N being at risk of leaching. Reductions in fertiliser inputs below the optimum results in a decrease in residual N, but this reduction is limited because N uptake efficiency is greater at smaller N inputs. The economic optimum cannot be predicted with perfect precision in advance, so even where the recommended quantity of N is applied, it is likely that a proportion of fields will be over-fertilised. Overall reductions in fertiliser input decrease the proportion of crops which cannot use all of the supplied N. Recent changes in the ratio between fertiliser price and product value have reduced the economic optima for many crops by 10-30 kg/ha, which should result in reduced inputs and reduced overall nitrate leaching in the near future.

‘Quality’ wheat destined for bread making receives more N than other wheat, in order to increase the protein in grain, and this results in an increase in nitrate residues post harvest. The effect for arable crops is illustrated by output from the NIPPER model compared with the results of replicated experiments on sandy soils (Fig. 3.21).

![Figure 3.21](image)

*Figure 3.21. NIPPER modelled (dashed, unfilled) and observed data (solid, filled) showing the effect of applied fertiliser on late autumn SMN on a sandy soil at Swindon, Shropshire following winter wheat (triangle, square) and winter barley (circle).*

3.11.4 Grassland N inputs and stocking density

N inputs to grassland vary greatly between farms. Intensively stocked systems receive greater N inputs and greater manure returns than extensive systems, resulting in greater nitrate leaching risk per unit of stock carried, as well as greater ammonia emissions and reduced efficiency of use of N inputs. This is illustrated by data from groundwater NVZ sites (including some from earlier years of monitoring to increase the number of site years), Figure 3.22.
Nitrate concentrations in leachate from grass on groundwater NVZs, as a function of nitrogen fertiliser input.

Figure 3.9 illustrates this for two grassland fields each receiving more than 225 kg/ha N as fertiliser in addition to manure, where nitrate concentrations in leachate were high throughout the winter, while those under extensively stocked grass/clover systems were much smaller, and below 50 mg/l on average (Fig. 3.9). This effect of stocking density and associated N inputs on nitrate loss from grassland systems is consistent with the Nitrate Directive requirement for limits on manure and excretal returns at farm level.

Although losses from some intensive sites were high, several sites received little or no N fertiliser, some of these relying on clover to supply N. The mean nitrate concentration in leachate from groundwater sites in winter 2004/5 and 2005/6 in this project was smaller from grassland than from arable, despite similar late autumn SMN values and a smaller water flux on the grassland sites (e.g. Figs 3.7 and 3.8),

Stocking densities across all the farms in the surface water catchments in this study ranged from 0.5 to around 3 LU/ha (where LU is livestock units), with the higher values related to the two catchments located in Devon: Edgeworthy and Pugsley.

These two surface water catchments also consistently have the greatest mean values of all monitored catchments for late autumn SMN (185 and 190 kg N/ha respectively in autumn 2004, and 275 and 180 kg N/ha respectively in autumn 2005), as well as relatively high values of 0.34-0.39% for total N in topsoil.

Nitrate concentrations tend to be smaller in wetter areas; and in drainage from grassland on clay soils, where substantial denitrification may occur. The national average nitrate concentrations will be smaller than those reported above for groundwaters because much grassland is relatively extensive, much is in wetter areas, and some is on clay soils.

3.11.5 Manure management

Nitrogen supplied to the crop from manure

Livestock recycle much of their N intake as excreta, most of which is applied to land either directly (at grazing) or as manure, collected from housed stock. The N in the excreta/manure is partly as complex organic matter, which breaks down only slowly, and partly in forms
(ammonium, uric acid) which rapidly release nitrate. The latter can substitute for fertiliser N. Proper adjustment for this source of N is complicated, and until recently this has often been ignored despite the fact that it could save farmers money. The NVZ Action Programme requires that farmers take full account of N supply from all sources; and the main area where change is required is in adjustment for N released by manures. The British Survey of Fertiliser Practice indicates improvements since the introduction of the NVZ AP, and such improvements have been facilitated by provision of calculation details in paper and user-friendly electronic form (RB209; MANNER; PLANET, etc).

Overall, around 25% of fields monitored in this project received manures. However within the dominantly grassland catchments of the Weaver, Ditchford, Pugsley and Edgeworthy, more than half of their agricultural land received manure each year, as is typical of grassland systems. Under the NVZ rules imposed on land in this project, there was evidence that fertiliser inputs were reduced on fields that received manure (Fig. 3.4).

Timing of manure applications

A major measure within the NVZ Action Programme is the closed period for manure applications. During autumn and winter, crop uptake is limited and nitrate leaching occurs because rainfall exceeds evapotranspiration. Nitrate released from manure applied during this time is at risk of loss by leaching. Other manure-derived pollutants (such as FIOs) may also be transferred to waters, usually through surface runoff, or via preferential flow through clay soils.

The closed period of the 2002 NVZ AP (August or September to October inclusive) currently applies only on 'sandy or shallow' soils, which comprise less than 10% of the current NVZ area. It does not apply to straw-based farmyard manures (from cattle or pigs) because these have a small content of available N and, therefore, make a relatively small contribution to leaching risk in the winter of application. For the fields in this project, the majority of manure was applied in the spring (February, March) and post-harvest (August, September) periods (Table 3.4). This distribution is similar to that recorded in national surveys, and demonstrates that the fields monitored in this project were broadly representative of the wider NVZ area.

Slurry applied in early autumn can release nitrate rapidly, and the effect on nitrate leaching has been illustrated in Figure 3.10, which showed very high nitrate concentrations (mean 558 mg/l) in the winter following the harvest of a potato crop. High nitrate concentrations continued into the second winter, before declining towards more typical levels. In this example, the pig slurry application was made on 10 October 2004, which was permitted as the field was a sandy loam soil and therefore not subject to the 2002 Action Programme closed period.

The 2002 Action Programme closed period restriction on sandy and shallow soils is designed to reduce the risk of such very large N losses on the most vulnerable soils. However, this example shows that further reductions in nitrate leaching could be achieved by extending the measure to cover all soil types, which would limit applications of manures such as that reported in this example, and thereby reduce nitrate leaching. In this particular case, use of an over-winter cover crop in winter 2005/6 would also have helped to reduce nitrate leaching losses – which even with the two dry winters was still high (185 kg N/ha in 2004/05 and 105 kg N/ha in 2005/06).
4 APPROACH TO MODEL DEVELOPMENT FOR ASSESSMENT OF IMPACTS

4.1 Overview

The approach to assessment of the impact of measures concentrated effort upon assessment of the effects of land management on emissions from agricultural land, since this is the system on which the NVZ Action Programme seeks to effect change. In addition, as described earlier, studies were commissioned to provide a framework for prediction, and case studies of, the time-scales of response in groundwater catchments (led by BGS), and implications for risk of eutrophication in surface water catchments (led by CEH).

Estimation of impacts of measures was carried out using a field scale model (NIPPER), which allowed evaluation against measured data of the responsiveness of the model to mitigation measures and other factors. The model development built on other well-tested models or modules as far as possible. The model was then linked to spatial data on land use and environmental factors, using management scenarios developed from survey data, to provide estimates of nitrate loss from agricultural land at catchment scale, where detailed field-by-field data are not available. Catchment-scale processes of mixing, delay and retention were simulated to allow comparison with measurements in large surface-water catchments.

4.2 Development of field scale model

The NIPPER model was developed within this project to enable assessment of the impacts of management change on nitrate losses. It takes into account weather and soil factors, allowing the implications of measures to be assessed for different locations and dates. The work builds on existing work including the field-scale NITCAT model (Lord, 1992 and subsequent revisions), MANNER (Chambers et al., 1999) for determining the fate of manure-derived N; algorithms developed within the SNSCAL project relating to mineralisation processes (King, 2004); aspects of the NCYCLE model behaviour for grassland (Scholefield et al., 1991) and improved treatment of clay soils based on recent experimental data.

This model is distinct from existing UK nitrate modelling tools in that within a single unit it deals with all agricultural land uses and all soils; uses relatively simple input data; is designed to be responsive to nitrate mitigation measures (such as manure timing); and has improved representation of risks of pollutant transfer from clay soils.

Particular attention was paid to modelling of clay soils. These are the dominant soils in surface water catchments, and their response to mitigation methods can differ from that of more permeable (e.g. sandy) soils because of the risk of rapid flow from field to ditch via cracks, drains or over the land surface. The model has a daily time-step in order to allow model output to be compared with measurements of the fluctuating flows and nitrate concentrations in water draining from clay soils.

4.2.1 Model Outline

The NIPPER model simulates the leaching of nitrate from a soil profile to ground and surface waters. This is achieved by modelling sources and sinks of Soil Mineral Nitrogen (SMN), the effects of land management on SMN, and the transport of N in soil water and runoff. The model is modular in structure, with sub-models predicting changes in SMN arising from a group of associated processes (such as crop growth and the associated uptake of N), and the transport of nitrate through the soil profile. The model predicts crop growth solely in

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order to estimate the associated uptake of N; it is not designed to provide accurate predictions of yield required for cost-benefit analyses.

The field scale model has been designed to have a relatively small data requirement based around farm management data that could be available by questionnaire, and which were collected for the monitoring sites within this project. The low data requirement meant that spatially interpolated data on crop areas, livestock numbers and manure management could be used in conjunction with expert agronomic knowledge and survey data to determine the inputs to drive the model at catchment scale.

A catchment is assumed to be a collection of fields that operate independently of their neighbours, so a catchment can be modelled using a collection of field scale runs, with the results additive to provide catchment scale effects. Mixing, transport delays and off-site retention effects may then be modelled at sub-catchment scale.

The model has been designed to provide the flexibility needed in order to investigate a range of methods designed to mitigate nitrate leaching, such as altering the timing and degree of incorporation of manure applications or altering N contents of manures, use of cover crops, alterations to stocking densities (and thus manure and fertiliser rates) and land use.

4.2.2 Field Scale Model Structure

The field scale model consists of a data input routine, and a series of submodels simulating the processes that affect the fate of N in agricultural systems. NIPPER is able to respond to crop and animal husbandry decisions as well as landscape characteristics. These husbandry decisions are implemented as a series of events and associated dates and are thus termed “diary events”. These events include:

- Crop drill and harvest dates (for arable fields)
- Number of cuts and/or end of grazing period (for grass fields)
- Stock type (for grazed grass fields)
- Cultivation (to a specified depth)
- Application of manure (of a specified type and rate)
- Application of fertiliser (at a specified rate)

On any time-step, the model implements any scheduled events, before calling the various sub-models which calculate crop growth and associated N uptake, changes to SMN from soil N turnover and the mixing and leaching of soil N (Fig. 4.1).

Although many of the processes simulated by NIPPER act relatively slowly, the model utilises a daily time-step to accurately represent the mixing and transport of N through the soil profile. The user may determine the geometry of the model soil profile, but to be compatible with this short model time-step, this will typically consist of layers 50 mm in thickness, to a total depth of three metres.

Input data include environmental conditions (soil properties; drainage system; daily weather; atmospheric N input) and management actions.

The crop growth model provides the basis for calculation of N uptake and evapotranspiration. It is driven by a combination of accumulated temperature and radiation, modified by parameters that vary with the crop and growth stage. The approach is similar to that in WOFOST (Supit et al., 1994 modified where necessary to fit UK data.

Crop N uptake is controlled by growth stage, dry matter and the typical N content of the crop at that stage, following Greenwood et al. (1990) and others. N uptake is progressively restricted when N is in short supply. Grass growth follows the same principles, with
parameters modified to reproduce typical UK results as summarised within the NCycle model (Scholefield et al., 1991).

Water balance (which drives the leaching process) is based upon the Penman-Monteith methodology as described in Allen et al. (2005). Crop growth is reduced progressively under water stress, using a method outlined by Bailey & Spackman (1996).

Mineralisation of N derived from soil and crop residues is based upon the SNSCAL (Soil Nitrogen Supply CALculator) model (King, 2004). Denitrification of soil N is calculated as a (negative) function of topsoil air content, with separate relationships for grass and arable (i.e.
denitrification occurs more readily in grassland soils), following data in Conen et al. (2000) and in NCycle (Scholefield et al., 1991).

Manure N transformations are based upon the MANNER model (Chambers et al., 1999). They take account of ammonia volatilisation as a function of manure type, conditions of application and delay before incorporation (cultivation) and of mineralisation of manure organic matter N. Manure composition data are also taken from MANNER. Nitrification of manure-derived ammonium N follows the relationship derived by Bradbury et al. (1993).

Soil drainage and leaching are calculated by the EDEN submodel (Gooday et al., 2007) developed within this project. The objective in the development of EDEN was to create an explicit and readily parameterised model of soil drainage and nitrate leaching, that simulates the effects of preferential and drain flow, and which is suitable for application across the whole of England and Wales. It was designed such that a single model structure deals with both impermeable (clay) soils (where much flow is via preferential pathways) and permeable soils (where flow is chiefly via the soil matrix), and can be parameterised from nationally available soils data. The model calculates vertical flow between, and lateral exchanges within, discrete soil layers. In order to capture the effects of processes controlling the behaviour of impermeable soils, the EDEN model operates on a daily time-step. Lateral water movement (and accompanying nitrate transport) only happens in soil above layers of restricted permeability. Natural lateral flow is assumed to occur as a result of slope, and is modelled as a function of hydrological pressure head and slope angle. Additional lateral flow can occur from field drains by application of Hooghoudt’s Formula (Hooghoudt, 1940). Nitrate dispersion and diffusion are controlled by empirical functions with rate parameters calibrated against field scale observations. Water (and associated nitrate) may leave the agricultural soil profile via surface runoff or lateral flow, which feed surface waters; and via deep percolation, which feeds groundwaters.

A detailed description of the model is given in Appendix III. Evaluation of model performance is reviewed in Appendix IV, with examples cited below.

4.3 Development of catchment-scale model system

At the scale of large catchments and regions, spatially interpolated data based on statistical sources must be used to estimate crop areas, livestock numbers and management of crops and manures. Mapped data on soils and climate are also required. The field scale model was designed with a low input data requirement so that it could easily be scaled up and used at catchment scale. A catchment is modelled as a collection of field scale runs.

This project used the MAGPIE database as the source of spatial data, at 1 km resolution (Lord & Anthony, 2000). MAGPIE was developed (under Defra and Environment Agency funding) as a policy tool to provide the data for catchment-scale modelling of water quality and agri-environmental issues. The land use mapping in MAGPIE derives from agricultural census, integrated with ancillary data sets such as urban boundaries, forest and amenity land. Agricultural census data on crops and livestock numbers are spatially interpolated from the Defra June Census. Soil data and parameters are provided by NSRI, augmented and gap-filled by ADAS. UKCIPS climate data or real weather data may be used as input.

Management data are taken from surveys where possible. The British Survey of Fertiliser Practice (BSFP) is the source of data on management change within NVZs (Goodlass & Allin, 2004), and the Farm Practice Survey is the source of manure management data, which are then automatically mapped according to livestock numbers and land use.
Calculations are made on sub-catchments, which are defined based upon topography, soil and the drainage network, with the aim being that each sub-catchment is relatively homogeneous.

Nitrate leaching depends not just on crop management for the current year, but on the previous cropping and management. The effect of break crops such as potatoes carries through as an increased soil N supply to the following cereal crop. Therefore, for each sub-catchment, typical crop rotations were generated, based on reported crop areas and soil types, using a crop rotation tool developed for the purpose. Crop management within each of these rotations is based upon survey data for the baseline scenario. All phases of each rotation are modelled, together with their antecedent 2 years of cropping, in order to simulate cumulative effects of management.

The results for all rotations are area-weighted to produce an aggregated estimate of water and nitrate emissions from agricultural land for the sub-catchment, via surface, lateral and deep seepage pathways.

Further processing for surface water catchments (e.g. adding point source inputs; denitrification; and attenuation) will generate the expected flow and nitrate concentration time series at the river sampling points. A drainage routing model based upon a one-dimensional form of TOPMODEL (Beven et al., 1994) simulates the river hydrograph and mixes rapid and slow soil drainage derived from different depths in the soil profile. The model is parameterised based on soil HOST class (Boorman et al., 1995). Nitrate losses from a river system by denitrification and plant uptake, (‘retention’) is calculated using empirical relationships between discharge and channel geometry to estimate the proportion of nitrate removed by bed processes in the river channel.

Transfers through aquifers are not modelled explicitly within this project, but would require processing via generic estimates of response times classified by aquifer type and climate, which have been developed by BGS, as described below.

4.4 Assessment of the time-scale of response

The EC requires some assessment of time-scale of response in terms of when change in farm practice will be expressed as change at the sampling point. Most management changes rapidly affect nitrate loss from the land. Changes are rapidly transmitted through clay soils to surface water catchments. However, it can take many years before effects are evident in groundwater abstractions, or in groundwater-fed rivers.

Groundwater modelling is difficult, data-hungry and expensive. It was decided during project specification that it would not be practicable within the resources available to model response times for every groundwater NVZ. BGS were therefore charged with reviewing literature and modelling approaches for estimation of groundwater response times, and developing a classification of aquifers and associated modelled response times which could inform reporting to the EC. Further details are provided in Appendix V.

4.4.1 Broad aquifer classification

The first step was to divide, on the basis of likely responses of the aquifers/groundwater systems of England and Wales into a manageable number of type classes for modelling, aiming in principle for the smallest convenient number (6-8) of groupings. The approach used to develop a matrix of aquifer types for modelling is briefly described below.

The main useable criteria selected to characterise areas that are representative of a typical range of responses were porosity/flow type and recharge. Other important criteria that contribute and could have been included are unsaturated zone and saturated zone
thickness, dip/scarp slope position and confined/unconfined. However, these are likely to vary on a scale that does not lend itself to definition of a small number of classes. Thus the geology was divided into four main hydrogeological types:
1. Intergranular
2. Dominantly intergranular
3. Fracture flow
4. Dominantly fracture

Rather than use recharge for characterisation of aquifer types, mean annual rainfall was used a simple surrogate for defining the categories (although recharge was used in the modelling). Three rainfall ranges were used:
1. 0 – 650 mm/a
2. 651 – 850 mm/a
3. >850 mm/a

This produces broadly north-east to south-west boundaries between the categories (Table 4.1). The next step was to verify the matrix, to establish the area of aquifer within each category (and by implication the volume of useable and protectable groundwater resource), and to ensure that the smaller aquifers falling within the likely area of interest were adequately represented.

**Table 4.1. Matrix of response scenarios for NVZ modelling.**

<table>
<thead>
<tr>
<th>Type of porosity and groundwater flow</th>
<th>Annual Rainfall (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>High (&gt;850)</td>
</tr>
<tr>
<td>Intergranular</td>
<td>Alluvium</td>
</tr>
<tr>
<td>Dominantly intergranular</td>
<td>Sherwood Sandstone●●</td>
</tr>
<tr>
<td>Dominantly fracture</td>
<td>Old Red Sandstone</td>
</tr>
<tr>
<td>Fracture</td>
<td>Chalk</td>
</tr>
<tr>
<td></td>
<td>Cotswold Limestone●</td>
</tr>
</tbody>
</table>

● Sites for ADAS evaluation by porous cup installations

The areas obtained in this way for the categories are shown in Table 4.2. A decision was made to include only areas greater than 1000 km² (shaded). This cut-off was taken to reduce the number of categories to be included in later modelling.

**Table 4.2. Areas of aquifer falling within each category (km²).**

<table>
<thead>
<tr>
<th>Geology</th>
<th>Category¹</th>
<th>Rainfall (mm/a)</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>0 - 650</td>
<td>651 - 850</td>
</tr>
<tr>
<td>Magnesian Limestone</td>
<td>F</td>
<td>613</td>
<td>845</td>
</tr>
<tr>
<td>Chalk</td>
<td>DF</td>
<td>7584</td>
<td>9837</td>
</tr>
<tr>
<td>Oolite</td>
<td>DF</td>
<td>2172</td>
<td>2716</td>
</tr>
<tr>
<td>Greensand</td>
<td>I</td>
<td>1590</td>
<td>1799</td>
</tr>
<tr>
<td>Crag</td>
<td>I</td>
<td>2951</td>
<td>145</td>
</tr>
<tr>
<td>Alluvium</td>
<td>I</td>
<td>2077</td>
<td>2798</td>
</tr>
<tr>
<td>Permian &amp; Triassic</td>
<td>DI</td>
<td>2916</td>
<td>3972</td>
</tr>
</tbody>
</table>

¹ Intergranular; Dominantly Intergranular; Dominantly Fracture; Fracture
Table 4.2 confirms that the most important aquifers (Chalk and Permo-Triassic Sandstone) occur in all three rainfall categories. The majority of the Chalk outcrop area is in the dryer south and east, whereas the Permo-Triassic Sandstone is more dominantly in the wetter central and western part of the England and Wales, but still with a substantial eastern outcrop in Nottinghamshire and South Yorkshire. Most of the NVZ areas receive less than 850 mm annual rainfall.

4.4.2 Modelled response times
Response times were then calculated using the MAP model, using the following steps.

1. Develop groundwater flow model using MODFLOW.
2. Undertake particle tracking to calculate travel times (MODPATH model).
3. Use MAP model to examine impact of step change of Nitrate input.
4. Produce time series for 200 years.
5. Examine results and draw conclusions.

Theoretically, the following factors impact the size and shape of a borehole catchment:
- Pumping rate
- Recharge rate and distribution
- Transmissivity
- Location within aquifer system (e.g. Scarp slope vs dip slope)

For the modelling, the Multiple Analytical Pathways (MAP) model was used in conjunction with a groundwater flow model (MODFLOW). These were used to simulate groundwater flow and nitrate transport for a wide range of aquifer scenarios chosen to give a representative coverage of the country, across three rainfall bands (high >850 mm, medium 650-850 mm and low <650 mm). Abstraction rate was also factored in to the scenarios. Table 4.3 summarises the aquifer conditions that were tested. The results of the assessment are reported in Section 5.4.
### Table 4.3. Aquifer conditions tested in BGS model simulations of nitrate response times.

<table>
<thead>
<tr>
<th>Aquifer type</th>
<th>Rainfall category</th>
<th>Abstraction rate</th>
<th>Other details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chalk</td>
<td>M</td>
<td>10 ML/d</td>
<td>Representation of stream flowing into larger river. Higher hydraulic conductivity in the valley than the rest of the system. The hydraulic conductivity decreases with depth below the water table.</td>
</tr>
<tr>
<td></td>
<td>M</td>
<td>5 ML/d</td>
<td></td>
</tr>
<tr>
<td></td>
<td>M</td>
<td>20 ML/d</td>
<td></td>
</tr>
<tr>
<td></td>
<td>L</td>
<td>5 ML/d</td>
<td></td>
</tr>
<tr>
<td></td>
<td>H</td>
<td>5 ML/d</td>
<td></td>
</tr>
<tr>
<td>Sandstone</td>
<td>H</td>
<td>20 ML/d</td>
<td>Groundwater flow occurring towards large river with the Sandstone partially covered by drift (hence 75% of recharge rate applied).</td>
</tr>
<tr>
<td></td>
<td>L</td>
<td>20 ML/d</td>
<td></td>
</tr>
<tr>
<td>Limestone</td>
<td>L</td>
<td>10 ML/d</td>
<td>Lincolnshire Limestone - Highly permeable, but thin aquifer with recharge occurring on the outcrop area only. The abstraction occurs in the confined part of the system.</td>
</tr>
<tr>
<td></td>
<td>L</td>
<td>5 ML/d</td>
<td></td>
</tr>
<tr>
<td></td>
<td>H</td>
<td>5 ML/d</td>
<td>Based on Lincolnshire Limestone, but modified to represent Oolitic Limestone in western England.</td>
</tr>
<tr>
<td>Crag</td>
<td>L</td>
<td>0.5 ML/d</td>
<td>Relatively low permeability aquifer, which is largely drift covered. The majority of the recharge is assumed to occur in river valleys where drift cover is absent.</td>
</tr>
<tr>
<td></td>
<td>L</td>
<td>2 ML/d</td>
<td></td>
</tr>
<tr>
<td>Alluvium</td>
<td>H</td>
<td>0.5 ML/d</td>
<td>Simulating narrow river valley aquifer with enhanced recharge at edges due to runoff.</td>
</tr>
<tr>
<td></td>
<td>L</td>
<td>0.5 ML/d</td>
<td></td>
</tr>
<tr>
<td>Greensand</td>
<td>H</td>
<td>0.5 ML/d</td>
<td>Thin aquifer (~20m thick) with medium hydraulic conductivity that results in a medium transmissivity. Limited width of unconfined outcrop receives recharge that flows into the confined. Abstraction can occur from the unconfined or confined part of aquifer. Well in confined aquifer.</td>
</tr>
<tr>
<td></td>
<td>L</td>
<td>0.5 ML/d</td>
<td>Well in confined aquifer.</td>
</tr>
<tr>
<td></td>
<td>H</td>
<td>0.5 ML/d</td>
<td>Well in unconfined aquifer.</td>
</tr>
<tr>
<td></td>
<td>L</td>
<td>0.5 ML/d</td>
<td>Well in unconfined aquifer.</td>
</tr>
</tbody>
</table>

### 4.5 Assessment of impacts on eutrophication

If waters are found to be eutrophic, or in the near future may become eutrophic, and a significant amount of the nitrate present in the waters comes from agricultural sources, then the Member State must designate the land draining into the affected waters as NVZs and put in place Action Programmes or adopt Action Programmes across the total territory. Eutrophication is defined in the Nitrates Directive as:

"the enrichment of water by nitrogen compounds, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned".

Therefore, an additional study was commissioned, via the centre for Ecology and Hydrology (CEH) and Plymouth University, to assess the effect of any changes in nitrate loss arising...
under the NVZ Action Programme, on eutrophication. Detailed reports are provided in Appendices VI to IX. In addition to reviews of the evidence, a case study focussed on impacts for the Taw estuary, which has been designated as an NVZ on grounds of eutrophication within the estuary.

The aim of this work was to examine how the current trophic status of the Taw Estuary may change in response to the implementation of NVZ Action Programme Measures in the catchment of its main freshwater source, the River Taw.

The assessment for the estuary was undertaken by interrogating two datasets;
1. Estimates of inputs of the nutrients nitrogen and phosphorus to the estuary from river inflow and direct discharges. For the purpose of this exercise, an estimate of the upper limit of potential impact of the current NVZ AP measures was provided, indicating decreases in inputs from the River Taw of ca 10 % between September and May, but only 1.9 % from June to August.
2. Estuarine water quality measurements undertaken from 1990-1005 by the Environment Agency (EA) within the Harmonised Monitoring Scheme (HMS).

These data were used for several analyses which were designed firstly to estimate the trophic status of the Estuary from 1990 – 2004 at six sites from Newbridge (freshwater) to Airy Point at the mouth of the estuary and, secondly, to assess the factors influencing the trophic status of the estuary. The analyses were:

- An assessment of the spatial and temporal (monthly) distribution of chlorophyll a concentrations across the six sites.
- An estimation of the annual inputs (expressed as t a⁻¹ and %) of nitrate, ammonium and phosphate to the Taw Estuary from the River Taw, other rivers and the Ashford sewage treatment works (STW).
- Comparison of the average annual concentrations (µmol/l) of ammonia, phosphate and nitrate in the effluent of Ashford STW.
- Calculation of the molar ratio of nitrogen to phosphorus for the complete temporal dataset at each of the six sites.
- An assessment of the effect of river flow (high, medium and low) on the chlorophyll a concentrations at each of the six sites.
- Principal component analysis of the data to investigate interactions between key variables (chlorophyll a, nitrate, ammonium, phosphate, N:P ratio, ammonium:nitrate ratio and salinity).

The results of the assessment are reported in Section 5.5.
5 MODELLING THE IMPACT OF MEASURES

5.1 Overview of impacts

The current NVZ AP would be expected to bring about the following changes:

- **Fertiliser: quantity**
  - Reduce N fertiliser inputs especially in areas where manures are used (due to better adjustment for the crop-available N contained in the manure)
  - Have only small effects on fertiliser use elsewhere, e.g. prohibit N fertiliser use on pea crops

- **Fertiliser: timing of application**
  - Prevent applications of chemical fertiliser at times when there is no crop requirement. The proportion of fertiliser currently applied during these closed periods is small.

- **Manure: timing of application**
  - Prevent applications of slurries and poultry manures during September and October (and reduce applications in August), on sandy and shallow soils (around 10% of agricultural land, very variable between catchments) and correspondingly increase applications at other times
  - Have no effect on manure timing on other soils, or for Farmyard Manures.

- **Manure: quantity**
  - Prevent applications of manures to crops with no fertiliser N requirement (e.g. peas and beans) and reduce applications to other crops which were previously in excess of crop requirement
  - Result in more even spatial distribution of manures from pig and poultry farms because of the annual field-scale limit on manure inputs
  - On the most intensively stocked dairy farms, result in some export of manures due to the farm-scale limit on excretal returns

Other measures within the current AP are expected to have only minor impact on the risk of nitrate leaching, although they have value in minimising the risk of direct loss of other manure and fertiliser-related pollutants to waters.

In calculating the impact of measures, it is necessary to consider the impacts on surrounding land. For example, reduction in manure inputs in one area will increase inputs in another area.

The model estimation of the impact of management on nitrate loss is evaluated below against field scale data. Estimation of nitrate leaching at catchment scale is then illustrated for two contrasting catchments, the Meden in Nottinghamshire, and the Taw in Devon.

5.2 Evaluation of model against field-scale measurements

The ability of the model to simulate impacts of management practice on nitrate loss was tested against data from a series of controlled experiments; and against measurements made from commercial farms within NVZs as part of the present project.

5.2.1 Fertiliser N input

NIPPER reproduces the form of response to fertiliser N in terms of crop N uptake, and residual SMN (representing N at risk of leaching) in the autumn following crop harvest (Fig. 5.1).
Figure 5.1. Comparison between modelled predictions (dashed, unfilled) and observed data (solid, filled) for yield, N offtake and autumn SMN for an N response experiment on a sandy soil at Swindon, Shropshire. Winter wheat (triangle, square) and winter barley (circle).

5.2.2 Manure management

Nitrate loss to water associated with manure applications is greatest where the manures are applied in autumn, and decreases for applications in late winter and spring. On sandy soils, there is little risk of loss from applications after early January, under annual rainfall of about 700 mm, typical of arable Eastern areas.

For clay soils, the risk of loss from autumn applications is smaller than for sandy soils; but some risk persists for applications made in winter and early spring, because of the
occurrence of rapid flow from the surface via cracks and preferential flow paths. This contrast is illustrated in the modelled results in Figure 5.2.

The model correctly simulated measured nitrate losses from a heavy clay soil (Defra project ES0106) to which slurry had been applied at different timings (Fig. 5.3). (Predictions for August applications on these arable sites were the same as for September applications, under the same cropping conditions).

![Figure 5.2. NIPPER simulated effects of date of manure application on nitrate loss from sandy and clay soil, annual rainfall 700 mm.](image)

![Figure 5.3. Comparison of observed and predicted amounts of leached nitrate at Brimstone (Defra project ES0106) over two winters for fields with different manure application timings. Standard errors shown for observations (n=3).](image)

The impact of date of application of chemical fertilisers on nitrate leaching follows similar patterns to those for manure.

The effect of the N supplied to the crop by manure applications was simulated for wheat crops on permeable soils, within this project, where nitrate leaching was measured with
porous cups. The model agreed with the finding that nitrate leaching was increased after crops to which manures were applied, even though fertiliser inputs were decreased (Fig. 5.4).

![Graph showing comparison between observed and modelled SMN following winter wheat crops depending upon the manure history of the field.](image)

**Figure 5.4.** Comparison between observed and modelled SMN following winter wheat crops depending upon the manure history of the field.

### 5.2.3 Crop type

NIPPER model simulations of nitrate leaching following crops monitored within this project on commercial farms are shown below. The risk of nitrate leaching after harvest varies with crop type, being greater for some break crops (e.g. potatoes and oilseed rape) which may receive large N inputs and which leave residues rich in N. It also tends to be greater after crops that are harvested early, unless the following crop is drilled early in autumn. The model reflected these differences, except for an under-estimation of mean N losses from grass crops, and an overestimation of losses from winter wheat (Fig. 5.5).

![Graph showing comparison of observed and modelled mean nitrate concentrations for all ground water sites with porous pots. Data are averaged across winters 2004/05 and 2005/06 and summarised by previous crop.](image)

**Figure 5.5.** Comparison of observed and modelled mean nitrate concentrations for all ground water sites with porous pots. Data are averaged across winters 2004/05 and 2005/06 and summarised by previous crop.
Detailed data are shown below to illustrate the effect on nitrate leaching in the following winter, as measured on sites within this project, of a pea crop and of ploughing out intensive grassland (Figures 5.6 and 5.7).

**Figure 5.6.** Comparison of observed and modelled nitrate concentrations showing the effects of peas grown on 2005 on nitrate concentrations the following winter. (Data: this project).

**Figure 5.7.** Comparison of observed and modelled nitrate concentrations showing the effects of ploughing out of grassland autumn 2005 (winter wheat crop sown). Data: this project.

### 5.2.4 Surface Water Sites

For the surface water sites, flow and nitrate concentration were monitored in streams, ditches or open main drains and the husbandry recorded for all fields within the micro-catchments that fed the watercourses.

The Whittle Dene catchment, in Northumberland, had both arable crops and grassland, with some beef cattle, although very little applications of managed manure (Fig. 5.8). The model simulates the 'flashy' pattern of flow, which is typical of clay catchments. Modelled concentrations are somewhat less than observed, particularly in the first winter. The model reproduces the observed dilution events during times of heavy flow, when a proportion of the...
water reaching the gauge has travelled by rapid flow pathways and has not equilibrated with nitrate in the soil.

![Graph of observed and modelled flow](image1)

![Graph of observed and modelled nitrate concentration](image2)

**Figure 5.8.** Comparison of observed and modelled flow and nitrate concentrations for Whittle Dene, a mixed arable and grassland farm in Northumberland.

The Pugsley and Edgeworthy catchments are both in the Taw river basin in Devon, which is an area of high rainfall. Edgeworthy is an almost exclusively grassland catchment, whereas Pugsley is only 60% grassland, with dairy cattle found on both sites. The catchments received large amounts of manure and fertiliser, with the manure loadings of 200 kg/ha N in Edgeworthy particularly high. Observed and modelled flow hydrographs are similar at Pugsley (Fig. 5.9), although the fact there is some observed flow even during the summer hints at the possibility of there being a base flow contribution which may be from outside the delineated catchment. The graphed modelled flow is smaller than measured because it does not include the deep percolation component.
Comparisons of the observed and modelled concentration time series at Pugsley are good, with remarkable similarity in the first winter (Fig. 5.9). As with the Whittle Dene data, there are dilution events at times of high flow in both the observed and modelled concentrations. The model correctly predicts smaller concentrations in the Edgeworthy catchment due to the smaller proportion of arable land (Fig. 5.10). The intensity of management of grassland is similar to Edgeworthy. Measured nitrate concentrations are smaller than predicted, and remarkably low for such an intensively stocked catchment. There is little evidence of underdrainage in the catchment, and the implication is that substantial denitrification (of the order of 50%) occurs on the way to the stream monitoring point, presumably within the saturated soil and subsoil horizons.

Figure 5.9. Comparison of observed and modelled flow and nitrate concentrations for Pugsley, a mixed arable and grassland catchment in Devon.
5.3 Modelled impacts of measures at field scale

The impact of measures within the NVZ Action Programme, and other mitigation approaches, was assessed at field scale by model runs for a range of conditions. The results indicated large potential nitrate savings on individual fields, by improvement of relevant practices.

5.3.1 Applying no more fertiliser than crop requirement

The NIPPER model and experimental data indicate that nitrate leaching increases steeply with inputs if more N is applied than the crop requires. Typically, a surplus of 20 kg/ha N would increase nitrate leaching following cereals by about 20%. This degree of 'over-fertilisation' is sometimes inevitable, due to our inability to predict crop growth perfectly. Under-fertilisation by the same amount, which also occurs, reduced nitrate leaching by 10-15%.

One of the most common causes of over-fertilisation is failure to reduce fertiliser inputs to account fully for N supplied by manure. On a field receiving poultry manure or slurry, at application rates of 170 kg/ha total N, and with fertiliser inputs not adjusted at all for this input, nitrate leaching was increased by 60-100% compared to a correctly fertilised crop.

This measure is very effective, and does not have any adverse impact on yield. About one in 6 arable fields receives manure in any one year, of which the majority is pig slurry or poultry manure (the remainder being FYM and a little dairy slurry). Some farmers already take account of manure N and other factors affecting crop N supply, and this needs to be taken into account in assessing the impact of this measure in practice.

5.3.2 Timing of manure applications

Autumn applications of slurries or poultry manure (at a typical rate of 170 kg/ha total N) were calculated to more than double nitrate leaching from arable fields relative to fields which did not receive manure. For a site with annual rainfall of 700 mm, typical of much of the NVZ area, August or September applications to loamy sand soils increased nitrate leaching by 150%. November applications increased leaching by only 45%, and January applications by
less than 10%. Actual N losses were smaller on grassland. They were also smaller in drier areas, although the associated concentrations were greater, and the proportional increase was little changed (i.e. the background N loss was also smaller in dry areas). In the driest areas, the date after which applications were unlikely to cause nitrate loss was earlier (sometimes as early as November, on silty or loamy soils) while in wetter areas some leaching could occur even after applications in February.

On a clay soil under equivalent conditions, autumn applications doubled nitrate loss relative to no manure. Applications in January increased losses by 30%. Thus the nitrate leaching risk from autumn applications was smaller on clay than sands. Later application reduced the risk on both soils, but by late winter the risk from sandy soils was negligible, while that on clay soils remained significant. This is due to the risk on clays that during heavy rain, some water can pick up pollutants from near the soil surface and carry them via rapid flow pathways – cracks, drains – to surface waters.

Survey data indicate that about half of the pig slurry and poultry manure applied to arable land is applied during the autumn (August to October) with a smaller quantity during the winter months. About one arable field in 7 or 8 receives slurry or poultry manure during any given year. The proportion of grassland receiving manure is greater (40-50%) but the proportion applied during autumn is smaller. A greater proportion of grassland manure (e.g. 80% of beef manure) is as FYM (rather than slurry), which because of its low proportion of readily-available N is not subject to the Closed Period.

5.3.3 Timing of chemical fertiliser applications

Chemical N fertilisers applied in England are mainly ammonium nitrate and urea. In ammonium nitrate, half the N is as nitrate and immediately at risk of leaching, while the other half is as ammonium and requires nitrification. Urea is rapidly converted to ammonium in the field, and then is nitrified to nitrate. Manures contain readily-available N as uric acid or ammonium, both of which require nitrification. This process takes days to weeks depending on the temperature. The nitrate leaching risk from applications of chemical fertiliser are therefore very similar to the risks from application of manure. A typical autumn fertiliser application on arable crops would be 30 kg/ha N. The increase in nitrate leaching due to such an application was 0 to 50% of the level where autumn fertiliser was not applied, depending on the N taken up by the crop. Experimental data indicate that oilseed rape can take up this quantity of fertiliser N in addition to soil N supply, but that this is not always the case.

In practice only 1% of fertiliser is applied in autumn or winter, and much of that is prior to crops such as winter oilseed rape where there is a crop requirement. Applications of autumn N fertiliser have been declining in recent decades. This measure is therefore effective, but affects only a small proportion of land.

5.3.4 Reducing chemical fertiliser inputs to less than crop requirement

As described above, reducing crop N supply reduces leaching risk. The saving in leaching risk diminishes as the N input falls below crop uptake capacity (which usually occurs at a point close to the recommended N input). A reduction of chemical N fertiliser applied to arable crops by 10% compared to the recommended quantity was calculated to reduce nitrate leaching by 8-12% compared to the baseline, depending on the crop and conditions.

Reductions in grassland systems were also variable depending on the current inputs assumed. The average reduction was estimated to be 5-10% if it was assumed that stock numbers were not changed. This latter calculation was considered to be subject to fair uncertainty, in relation to assumptions about the efficiency of grass utilisation that is attainable in practice.
5.3.5 Removal of all livestock manures

The model takes account of the effect of previous land management history on soil N supply and leaching risk. A large proportion of the N in manure is as organic matter, which is mineralised slowly, a little each year, and is therefore difficult to take into account when planning fertiliser applications. However this increase in soil N supply can increase leaching both because it is a cause of additional N supply to the crop, and because it increases nitrate accumulation on arable land in autumn.

The effect of removing all manures from a rotation was therefore investigated, to give an indication of the proportion of nitrate leaching which is attributable to livestock manures. Removal of all manures from an arable rotation receiving 170 kg/ha total N as livestock manure one year in 3 was estimated to reduce nitrate leaching by 20-30%, depending on manure type and cropping system.

5.4 Catchment case studies: modelled impacts on emissions from agricultural land

The NIPPER model was used to explore the impacts of mitigation practices in two case study catchments (Meden and Taw). The purpose of these studies was to explore how measures that are highly effective at the field scale would translate to a larger scale.

The impact of a measure at catchment scale is the product of its effectiveness when applied at field scale; and the degree to which it is actually applied at the catchment scale. The latter depends on opportunity – measures relating to manure management apply only in catchments where there is manure. Agricultural census data, spatially interpolated, together with mapped soil and climate data, allow assessment of opportunity for measures within each catchment. Impact of the measure also depends on the contrast between current practice, and the assumed practice in response to a measure. If all land is already fully compliant with a measure, making the measure law can have no additional impact. This issue can be addressed by using survey data collected prior to the implementation of the measure, and ideally afterwards as well. Failing this, an estimate must be made of the likely response. Unfortunately fully analysed data for the ‘post 2002 NVZ’ case were not available to us in time for this report.

As an example of the issues, the Closed Period applies only to certain soils, and to certain manure types. Furthermore, it affects only that proportion of manure that is currently applied during the proscribed period. And the fate of the manure that would otherwise have been applied during the closed period must be estimated. Use of spatially interpolated agricultural census and survey data on manure management allows an assessment to be made of the applicability of the measure in a given catchment, and the prior management practice. The climate and soil data for that catchment then allow the impact on actual nitrate loss to be assessed. Mapped data from the MAGPIE system were used together with survey information and the crop rotation tool to generate crop scenarios of sub-catchments within each case study.

The Meden and Taw catchments were chosen as being typical of contrasting elements of the NVZ area. The Meden is a groundwater catchment, partly on sandy soils to which the Closed Period for manure application applies, dominantly arable but with some intensive grass, and with a high livestock density. It is in the eastern half of the country, and in both climate and land use is representative of much of the NVZ area. The Taw catchment has much greater rainfall, and is a clay surface water catchment dominated by grassland. Closed periods for manure application do not apply here.
Two measures were modelled which were considered, on the basis of preliminary modelling and data on current practice, to represent the main impact of the current (2002) NVZ Action Programme. These were:

- **Apply no more N than the crop requires, taking account of all other sources of N**
- **Do not apply slurries or poultry manures during autumn on sandy or shallow soils**

Two other significant measures that were not explicitly modelled were:

- The current farm-level limit on N input as manure and excretal returns, set at 170 kg/ha N on arable land and 250 kg/ha N on grassland. This will result mainly in movement of manures. The movement of manure to avoid applications in excess of crop requirement is subsumed within the measure 1 below. Analysis of data by Defra indicates that few grassland farms would be affected by the 250 kg/ha N limit, and these would be able to respond by movement of manure or acquisition of land, rather than needing to reduce stock numbers.

- The closed period for chemical fertiliser application. This is a necessary rule, which could have significant impact in principle, just as the closed period for manures. However, the impact was considered to be very small because of the high level of prior compliance. Less than 1% of N fertiliser is applied in autumn, and part of this is applied to early-drilled crops such as winter oilseed rape, which take up a large quantity of N in autumn and therefore have a crop requirement for N fertiliser.

In order to explore the potential for minimising nitrate reduction at catchment scale, two further and far more severe measures were explored. It must be emphasised that these are not part of any Action Programme, but are purely indicative. The measures were:

- **A 10% reduction in chemical fertiliser inputs**
- **Removal of all managed manures from the catchment**

Assumptions used to implement actions within the model were:

1. **Apply no more N than the crop requires, taking account of all other sources of N**

   This measure represents the major effect of implementation of the requirement that N inputs should not exceed crop requirement. Manure applications provide an additional source of N to aid crop growth which farmers should take account of when determining levels of inorganic fertiliser to apply. It was assumed that the fertiliser rates taken from BSFP represent fertiliser rates where no account has been made for the crop available N in manure. Reduction of these fertiliser rates to account for the available N from manure thus represents a maximum effect that could be gained from this mitigation measure.

   It was assumed that MANNER (Chambers et al., 1999) would have been used to estimate how much of the available N from manure applications would have been leached given the timing, soil type and climate. The BSFP rates of inorganic fertiliser applications were reduced accordingly to account for the estimated remaining crop available N.

2. **Do not apply slurries or poultry manures during autumn on sandy or shallow soils**

   Slurries and poultry manures are not applied in the autumn (August to October) to soil series with a topsoil texture described as sand, loamy sand or sandy loam. Instead, these types of manure are stored and applied in the late autumn and winter (November, December and January). This is not an unreasonable assumption on these soils, which remain trafficable for most of the winter. On these light soils, some leaching would take place from a winter application, although this would be less than from applications in early autumn. It was assumed that this measure would be used in conjunction with
measure 1 (since if implemented on its own, some of the N saved from leaching would be surplus to crop requirement and would leach the following winter).

3. **A 10% reduction in chemical fertiliser inputs**

A uniform 10% reduction in fertiliser rates was applied to all arable land and grassland. It was assumed stocking densities on grass would remain the same even though grass production would fall. By implication, any deficit would be made up by imported feed.

4. **Removal of all managed manures from the catchment**

The removal of all managed manures from within a catchment is an unrealistic scenario for nearly all catchments. However, implementation of this via the model does provide an estimate of the maximum effect that any mitigation measure that aims to alter manure management could hope to achieve. It removes the medium-term impacts on soil N supply as well as the immediate effects of freshly-applied manures.

5.4.1 **Results: Meden, Nottinghamshire**

The Meden groundwater catchment in Nottinghamshire is fairly typical of the NVZ area in climate and land use, but it has a higher proportion of sandy soils, and almost double the average quantity of manure per hectare. The 26,400 ha catchment ranges in elevation from 30 m a.s.l. in the east to 180m a.s.l in the west, with a corresponding variation in average annual rainfall from 600 mm up to 730 mm, with a catchment average of 680 mm. Approximately 65% of the catchment is covered by sandy soils, with the remaining 35% covered by free-draining clay loams over permeable sandstone. The area is approximately 45% agricultural land, mainly arable. The remaining land use is split fairly evenly between rough grazing and woodland, although the town of Mansfield in the southwest contributes to the 6.4% urban land use (Table 5.1). The arable land is primarily in rotations of winter and spring cereals with sugar beet, potatoes, and oilseed rape (Table 5.1).

In the Meden catchment, annual manure loadings average 74 kg/ha N of agricultural land, which is about twice the national average. The highest average rates of manure application are found in the east (which is also where there is more agriculture). Poultry numbers within the catchment are large (Table 5.2), especially in the east, such that poultry manure alone provides 48 kg/ha N agricultural land. Since almost two thirds of poultry manure is applied in the autumn, an autumn closed period should have a significant effect in this area.

Table 5.1 shows the assumed rates of fertiliser and manure application by crop type averaged across the whole catchment.

Based on stocking densities, it was assumed that 74% of the grassland is used for farming beef and sheep, whilst the remainder is used for dairy cattle. Dairy cattle are typically stocked more densely than beef or sheep (based on survey data) and, so, calculated fertiliser rates to dairy land are almost twice those required to support beef and sheep. Mean application rates for all agricultural land were estimated as 138 kg/ha N for fertiliser and 74 kg/ha N for manure.
Table 5.1 Land use within the Meden and the assumed fertiliser rates and average manure application rates.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Area (ha)</th>
<th>Percentage Total Area (%)</th>
<th>Fertiliser † (kg/ha)</th>
<th>Manure Total N (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter Wheat</td>
<td>2703</td>
<td>10.2</td>
<td>192</td>
<td>89</td>
</tr>
<tr>
<td>Winter Barley</td>
<td>1905</td>
<td>7.2</td>
<td>149</td>
<td>103</td>
</tr>
<tr>
<td>Spring Barley</td>
<td>1067</td>
<td>4.0</td>
<td>107</td>
<td>57</td>
</tr>
<tr>
<td>Potatoes</td>
<td>587</td>
<td>2.2</td>
<td>166</td>
<td>120</td>
</tr>
<tr>
<td>Sugar Beet</td>
<td>1146</td>
<td>4.3</td>
<td>106</td>
<td>45</td>
</tr>
<tr>
<td>Beans</td>
<td>38</td>
<td>0.1</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Peas</td>
<td>151</td>
<td>0.6</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Maize</td>
<td>66</td>
<td>0.2</td>
<td>48</td>
<td>144</td>
</tr>
<tr>
<td>OSR</td>
<td>799</td>
<td>3.0</td>
<td>205</td>
<td>24</td>
</tr>
<tr>
<td>Linseed</td>
<td>251</td>
<td>0.9</td>
<td>59</td>
<td>4</td>
</tr>
<tr>
<td>Set-aside</td>
<td>939</td>
<td>3.5</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Other Crops</td>
<td>489</td>
<td>1.8</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Grass &lt; 5 yrs</td>
<td>528</td>
<td>2.0</td>
<td>105 / 212</td>
<td>125</td>
</tr>
<tr>
<td>Grass &gt; 5 yrs</td>
<td>1409</td>
<td>5.3</td>
<td>105 / 212</td>
<td>125</td>
</tr>
<tr>
<td>Rough Grazing</td>
<td>6905</td>
<td>26.0</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Woodland</td>
<td>5726</td>
<td>21.5</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Urban</td>
<td>1706</td>
<td>6.4</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

† taken from BSFP 2002 except for grassland which are calculated from stocking densities (values for Beef & Sheep / Dairy)

Table 5.2. Stocking densities within the Meden.

<table>
<thead>
<tr>
<th>Stock Type</th>
<th>Density †</th>
</tr>
</thead>
<tbody>
<tr>
<td>Adult Beef</td>
<td>0.29</td>
</tr>
<tr>
<td>Adult Dairy</td>
<td>0.47</td>
</tr>
<tr>
<td>Beef Followers</td>
<td>0.64</td>
</tr>
<tr>
<td>Dairy Followers</td>
<td>0.13</td>
</tr>
<tr>
<td>Young Cattle</td>
<td>0.69</td>
</tr>
<tr>
<td>Adult Sheep</td>
<td>0.87</td>
</tr>
<tr>
<td>Lambs</td>
<td>0.87</td>
</tr>
<tr>
<td>Layers</td>
<td>47.79</td>
</tr>
<tr>
<td>Broilers</td>
<td>19.32</td>
</tr>
<tr>
<td>Pullet</td>
<td>11.92</td>
</tr>
<tr>
<td>Other Poultry</td>
<td>0.30</td>
</tr>
<tr>
<td>Pigs</td>
<td>0.82</td>
</tr>
</tbody>
</table>

† Cattle per ha managed grass, Sheep per ha all grassland, other per ha agricultural land.

Baseline nitrate leaching: Meden

The average amount of nitrogen leached was 58 kg N per ha of agricultural land, with considerable spatial variation (Fig. 5.11).

The pattern of leaching across the catchment is a combination of the differences in soil type, rainfall, land use and manure applications. The greatest leaching occurs in the middle of the catchment - up to 90 kg/ha – this corresponds with the band of sandy soils, as sandy soils are less retentive than heavier soils. The sub-catchments in the east on loamy soils receive greater manure loadings than those on the loamy soils in the west. Because the drainage is lower, the amount of nitrate leached is comparable to that in the west; but the nitrate concentrations in the east are greater. The largest concentrations are found on the sandy
soils in the centre of the catchment, where manure loadings are large and soils are less retentive of nitrate.

![Figure 5.11. Nitrate concentrations in drainage from agricultural land.](Image)

Even in the western sub-catchments, concentrations of nitrate in leachate from agricultural land are above 50 mg/l, the level at which action is required under the Nitrates Directive. Concentrations in groundwaters, and rivers fed by this leachate, will be somewhat smaller due to the substantial area of non-agricultural land. These data are consistent with data from the Boughton Nitrate Sensitive Area (NSA) groundwater catchment, which partly overlaps the Meden catchment and has very similar land use. Here, mean nitrate-N loss prior to implementation of the NSA scheme was estimated as 59 kg/ha N of agricultural land, resulting in a mean concentration in leachate to groundwater of 70 mg/l nitrate, taking due account of the 46% of non-agricultural land in that NSA (Lord et al., 1999).

**Impact of Action Programme measures: Meden.**

The NVZ AP requires that fertiliser inputs should not exceed crop requirement. Fertiliser recommendation tables indicate crop requirement based upon the results of N response experiments. Data from the BSFP (Goodlass & Allin, 2004 and others) show that the main cause of non-compliance with recommendations is failure to reduce inputs to account for N supply from manures. Reducing inorganic fertiliser rates to account for the crop-available N from manure applications resulted in a decrease in the average rate of fertiliser application across the catchment from 138 kg/ha to 126 kg/ha. This change reduced modelled nitrate leaching in this catchment by 7% (Table 5.3). The actual impact of the measure may be less than this because farmers did already make some adjustment for manure N supply prior to the Action Programme, but there may be gains in improved adjustment for factors other than manure, including previous cropping and soil type. There is considerable variation within the catchment in the effectiveness of this measure (Fig. 5.12), which is due to the large variation in manure loadings within the catchment.

Introduction of a closed period, preventing autumn applications of slurry and poultry manure on sandy soils and adjusting inorganic fertiliser inputs to take account of the additional quantity of crop-available N (which reduced fertiliser rates to 118 kg/ha) reduced average nitrate concentrations in the catchment by a further 8% (Fig. 5.13, Table 5.3). This last estimate is subject to some uncertainty, since farmers have some freedom to circumvent the
closed period by moving manures onto different soil types, or applying them earlier in the
autumn before the closed period begins.

Table 5.3. Modelled results for the Meden, showing nitrogen leached per hectare of
agricultural land and nitrate concentrations in drainage from agricultural land, along with the
average effect of a mitigation measure across the whole catchment and the range of effect in
the various sub-catchments.

<table>
<thead>
<tr>
<th></th>
<th>N Leached (kg/ha)</th>
<th>Nitrate Concentration (mg/l)</th>
<th>Average Reduction (%)</th>
<th>Range of Reduction (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>58.4</td>
<td>147</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Do not exceed crop N requirement</td>
<td>54.6</td>
<td>138</td>
<td>6.5</td>
<td>1.7 – 20</td>
</tr>
<tr>
<td>Closed period PLUS do not exceed crop N requirement</td>
<td>49.6</td>
<td>125</td>
<td>15.0</td>
<td>1.9 – 23</td>
</tr>
<tr>
<td>10% Fertiliser Reduction</td>
<td>54.1</td>
<td>137</td>
<td>7.4</td>
<td>4.0 – 11.7</td>
</tr>
<tr>
<td>Remove all Manures</td>
<td>43.3</td>
<td>109</td>
<td>25.9</td>
<td>5.8 – 37.4</td>
</tr>
</tbody>
</table>

Figure 5.12. Effect of accounting for available manure N when applying inorganic fertilisers
on nitrate leaching from agricultural land.

The calculation assumed that manures that could not be applied in early autumn were
instead applied in winter. This is not an unreasonable assumption on these soils, which
remain trafficable for most of the winter. As the closed period only affects sandy soils, it only
alters the leaching across 65% of the catchment. On the sandy soils, the closed period
actually reduced leaching by 10%. The fact that the closed period has such a large net effect
for the catchment is a result of the fact that manure loadings from poultry are large and
approximately two thirds of the poultry manure was applied in the autumn before
implementation of the NVZ Action Programme, so the timing of its application is impacted by
the closed period.
The estimated total nitrate leaching reduction due to the current NVZ AP in this catchment is therefore about 15%, which still leaves nitrate concentrations in leachate from the agricultural land within the catchment significantly above 50 mg/l limit. The estimated reduction in nitrate loss due to the NVZ AP for arable sandy soils with quantities of manure N more typical of the NVZ area (which is just under half of the application rate found within the Meden) would thus be about 7%.

**Impact of additional measures: Meden.**

A uniform decrease in chemical fertiliser rates of 10% to all agricultural land (but without any change to stocking densities on grassland) reduces nitrate leaching by 7% from the baseline value (Fig. 5.14, Table 5.3). There is limited spatial variation in the reduction in leaching, which is consistent with the universal nature of this measure.
The removal of all manure from the Meden, which sets a theoretical upper limit to what could be achieved through altering manure practices, reduced nitrate leaching by 26% overall, and up to 40% within sub-catchments with highest manure loadings (Fig. 5.15). These catchments were of course the ones with the greatest nitrate loss under baseline conditions.

The results indicate that even under this measure, nitrate concentrations in leachate from agricultural land would be in excess of 100 mg/l (Table 5.3), and mean concentrations for the catchment as a whole even after allowing for the large proportion of agricultural land would be close to or just above 50 mg/l.

Figure 5.15. Effect of not applying any managed manure on the nitrate leaching from agricultural land.

5.4.2 Results: Taw, Devon

The Taw Catchment in Devon is a surface water catchment, dominantly grassland and with high rainfall which is fairly typical of the extreme western NVZ catchments. The 110,000 ha catchment ranges in elevation from sea level at the estuary in the north-west to around 500m a.s.l. in the southern and north-eastern edges, with a corresponding variation in average annual rainfall from 900 mm to over 2000 mm. Over 95% of the catchment is covered by clay and clay loam soils. Over 75% of the catchment is used for agriculture, (60% managed grassland, 15% arable) with woodland and rough grazing being the other significant land uses (Table 5.4). In the Taw catchment, manure loadings, mainly from grazing stock, were estimated as 41 kg/ha N of agricultural land, with a substantially larger quantity of N returned as excreta at grazing by cattle and sheep. Cattle are found throughout the catchment, but stocking densities are lowest on the north eastern and southern edges with some poultry near the centre of the catchment. Based on stock numbers, it was assumed that 77% of the grassland is used for farming beef and sheep, whilst the remainder is used for dairy cattle – dairy cattle are typically stocked more intensively than beef or sheep. Mean application rates for all agricultural land are 122 kg/ha N for fertiliser and 41 kg/ha N for manure, both of which are substantially smaller than in the Meden.

Mean nitrate concentrations measured in the river are about 15 mg/l, corresponding to loads of the order of 25 kg/ha N. The catchment is designated on grounds of eutrophication.
Table 5.4. Land use within the Taw and the assumed fertiliser rates and average manure application rates.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Area (ha)</th>
<th>Percentage Total Area</th>
<th>Fertiliser † (kg/ha)</th>
<th>Manure Total N (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter Wheat</td>
<td>3578</td>
<td>3.2</td>
<td>192</td>
<td>37</td>
</tr>
<tr>
<td>Winter Barley</td>
<td>3116</td>
<td>2.8</td>
<td>149</td>
<td>23</td>
</tr>
<tr>
<td>Spring Barley</td>
<td>3716</td>
<td>3.3</td>
<td>107</td>
<td>1</td>
</tr>
<tr>
<td>Potatoes</td>
<td>107</td>
<td>0.1</td>
<td>166</td>
<td>159</td>
</tr>
<tr>
<td>Sugar Beet</td>
<td>88</td>
<td>0.1</td>
<td>106</td>
<td>142</td>
</tr>
<tr>
<td>Beans</td>
<td>215</td>
<td>0.2</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Peas</td>
<td>58</td>
<td>0.1</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Maize</td>
<td>1305</td>
<td>1.2</td>
<td>48</td>
<td>55</td>
</tr>
<tr>
<td>OSR</td>
<td>563</td>
<td>0.5</td>
<td>205</td>
<td>1</td>
</tr>
<tr>
<td>Linseed</td>
<td>1352</td>
<td>1.2</td>
<td>59</td>
<td>0</td>
</tr>
<tr>
<td>Set-aside</td>
<td>1331</td>
<td>1.2</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Other</td>
<td>456</td>
<td>0.4</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Grass &lt; 5 yrs</td>
<td>11051</td>
<td>9.9</td>
<td>99 / 195</td>
<td>46</td>
</tr>
<tr>
<td>Grass &gt; 5 yrs</td>
<td>57024</td>
<td>51.1</td>
<td>99 / 195</td>
<td>46</td>
</tr>
<tr>
<td>Rough Grazing</td>
<td>9157</td>
<td>8.2</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Woodland</td>
<td>12799</td>
<td>11.5</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Urban</td>
<td>4732</td>
<td>4.2</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

† taken from BSFP 2002 except for grassland which are calculated from stocking densities (values for Beef & Sheep / Dairy)

Table 5.5. Stocking densities within the Taw.

<table>
<thead>
<tr>
<th>Stock Type</th>
<th>Density †</th>
</tr>
</thead>
<tbody>
<tr>
<td>Adult Beef</td>
<td>0.18</td>
</tr>
<tr>
<td>Adult Dairy</td>
<td>0.36</td>
</tr>
<tr>
<td>Beef Followers</td>
<td>0.41</td>
</tr>
<tr>
<td>Dairy Followers</td>
<td>0.16</td>
</tr>
<tr>
<td>Young Cattle</td>
<td>0.41</td>
</tr>
<tr>
<td>Adult Sheep</td>
<td>3.24</td>
</tr>
<tr>
<td>Lambs</td>
<td>3.28</td>
</tr>
<tr>
<td>Layers</td>
<td>0.95</td>
</tr>
<tr>
<td>Broilers</td>
<td>6.26</td>
</tr>
<tr>
<td>Pullet</td>
<td>0.47</td>
</tr>
<tr>
<td>Other Poultry</td>
<td>1.35</td>
</tr>
<tr>
<td>Pigs</td>
<td>0.14</td>
</tr>
</tbody>
</table>

†Cattle per ha managed grass, Sheep per ha all grassland, other per ha agricultural land

Baseline nitrate leaching: Taw.

The average amount of N leached across the whole catchment was 23 kg/ha, but there is considerable spatial variation (Fig. 5.16). Only in 2 of the sub catchments does the mean concentration of nitrate in drainage from agricultural land exceed the 50 mg/l criterion of the Nitrates Directive (Fig. 5.16). However the Taw is designated on the grounds of eutrophication risk in the estuary, and generally has nitrate concentrations that are lower than 50 mg/l but sufficiently elevated to be potentially contributing to eutrophication.

As soil type, land use and manure application rates are all relatively homogenous within the catchment, the nitrate concentrations are broadly in inverse relationship to the amount of drainage, with greatest concentrations in the middle of the catchment.
Impact of Action Programme measures: Taw

Reducing inorganic fertiliser rates to account for the crop-available N from manure applications resulted in a decrease in the average rate of fertiliser application across the catchment from 121 kg/ha down to 114 kg/ha. This change reduced modelled nitrate leaching in this catchment by only 2% (Table 5.6).

Table 5.6. Modelled results for the Taw, showing nitrogen leached per hectare of agricultural land and nitrate concentrations in drainage from agricultural land, along with the average effect of a mitigation measure across the whole catchment and the range of effect in the various sub-catchments.

<table>
<thead>
<tr>
<th>Measure</th>
<th>N Leached (kg/ha)</th>
<th>Nitrate Concentration (mg/l)</th>
<th>Average Reduction (%)</th>
<th>Range of Reduction (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>23.0</td>
<td>29.3</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Do not exceed crop N requirement</td>
<td>22.7</td>
<td>28.8</td>
<td>1.5</td>
<td>0.1 – 5.0</td>
</tr>
<tr>
<td>10% Fertiliser Reduction</td>
<td>22.4</td>
<td>28.5</td>
<td>2.7</td>
<td>0.5 – 5.4</td>
</tr>
<tr>
<td>Remove all Manures</td>
<td>21.0</td>
<td>26.7</td>
<td>9.0</td>
<td>2.7 – 27.5</td>
</tr>
</tbody>
</table>

The reason for the smaller change in the Taw compared with the Meden, despite a similar change in fertiliser rates, is that the increase in leaching per additional kg of fertiliser is generally less for grassland than for arable, at these levels of intensity of stocking. The actual impact of the measure may be less than this because farmers did already make some adjustment for manure N supply prior to the NVZ Action Programme, and fertiliser inputs to grassland have fallen substantially already in recent years. There is some variation within the catchment (Figure 5.17), due to the poultry farming in the centre of the catchment providing greater amounts of manure.
Over 95% of the catchment is on soils that are not required to comply with a closed period, and the sandiest areas have limited amounts of agriculture, so a closed period would have minimal effect in the Taw.

The one other measure that could be relevant within the Taw is that which limits farm-averaged N inputs as manure and grazing returns. Some dairy farms are stocked at more than the proposed 170 kg/ha N limit but very few, if any, are stocked at more than the 250 kg/ha limit currently in force. Under the NVZ Action Programme, such farms are required to export manure or reduce stock numbers. The net effect of such export on nitrate leaching was estimated under the 250 kg/ha N limit to be very small because the total quantity of manure applied, and the timing of this, will be unchanged. Reduction in stock numbers due to this measure is considered extremely unlikely.

Overall, therefore, the current NVZ AP is predicted to have reduced nitrate leaching in the Taw catchment by less than 5% (mean 1.5%). However, nitrate concentrations were already less than the 50 mg/l criterion of the Nitrates Directive (the Taw was designated an NVZ on the grounds of eutrophication).

These results illustrate the difficulty of achieving major impact on nitrate leaching in clay catchments, and in grassland areas which are not especially intensively stocked.

**Impact of additional measures: Taw**

A uniform decrease in fertiliser rates of 10% to all agricultural land (but without any change to stocking densities on grassland) reduces nitrate leaching by 3% from the baseline value (Table 5.6). There is limited spatial variation in the reduction in leaching (Fig. 5.18).

The removal of all manures from the Taw, which sets a theoretical upper limit to what could be achieved through altering manure practices, reduced nitrate leaching by 9% within the catchment as a whole, with greatest effects around the poultry farms (Fig. 5.19).
These two examples serve to illustrate some of the causes of variation between catchments in current nitrate pressures, and in the opportunities for change.

**5.5 Time-scale of response: Groundwater catchment (Meden)**

The time-scale over which any change in agricultural practice would be reflected in measured nitrate concentrations at an abstraction borehole was calculated for the Meden, using a special case of the methodology developed by BGS for classification of response times in aquifers.
As part of the numerical work looking at time-scales within aquifer systems, a model description for a 0.5 Ml/d abstraction in a small sandstone aquifer catchment had been developed. These are conditions similar to those found in the Meden Valley. This model was used to demonstrate how a change in nitrate leaching from the soil propagates to abstracted water. Figure 5.20 and Table 5.7 summarise the behaviour. They show that it takes a significant amount of time for a change at the surface to be manifest in the abstraction. Only 25% of the change will be registered 20 years after it has occurred at the surface. (See Appendix V for more detailed discussion).

![Figure 5.20. Percentage of change in N loss from land which is reflected in nitrate concentrations at the borehole, as a function of years since the change.](image)

Table 5.7. Percentage of change in N loss from land which is reflected as change in nitrate concentrations at the borehole, as a function of years since the change.

<table>
<thead>
<tr>
<th>Change in nitrate concentration (% of total)</th>
<th>Years since change</th>
</tr>
</thead>
<tbody>
<tr>
<td>25</td>
<td>20</td>
</tr>
<tr>
<td>50</td>
<td>35</td>
</tr>
<tr>
<td>75</td>
<td>55</td>
</tr>
</tbody>
</table>

5.6 Implications for risk of eutrophication: Surface water catchment (Taw)

The report from CEH and Plymouth University on implications of the NVZ AP for the Taw estuary eutrophication risk is summarised here and given in full in the Appendices VIII and IX.

The Taw was chosen for the case study because the estuary of the River Taw (North Devon) and its freshwater catchment have been designated as an NVZ on the basis that its estuary is eutrophic. The Taw Estuary is 23 km in length, extending from its tidal limit at Newbridge to its mouth at Crow Point (Fig. 5.21). The estuary is macro-tidal, with a flushing time of 2-3 days, depending on river flows. There are extensive inter-tidal sand and gravel beds (with some mud) at the mouth, and mud flats in the upper reaches. Above Barnstaple, the estuary is freshwater dominated, whilst the mid and lower reaches are dominated by coastal water. The largest tributary to the estuary is the River Taw. The type of soil and subsoil, and the
low groundwater storage in the catchment, means that the river responds quickly to rainfall, with rapid rises in river levels i.e. it is flashy in nature. Although groundwater reserves are generally low, they do contribute to river base flows during dry periods.

Figure 5.21. Location map of the Taw Estuary. Filled circles represent Environment Agency water quality monitoring sites. 1, Newbridge; 2, Little Pill; 3, Barnstaple; 4, Ashford STW; 5, RAF Chivenor; 6, Airy Point.

Inputs of total inorganic N (TIN) to the Taw Estuary are dominated by rivers (90%), with the River Taw the largest contributor (65-70%) – see Figures 5.22 and 5.23. Modelling studies under the current NVZ Action Programme measures predicted a decrease in inputs from the River Taw of c. 10% between September and May, but only c. 2% from June to August. The reason for this is that the NVZ measures impact on agricultural land, which contributes nitrate during the drainage period, i.e. mainly in winter, whereas sewage inputs persist throughout the year. Impacts on biological activity, including eutrophication, will depend largely on nutrient concentrations during the warmer months, when biological activity is greater. This greater growth can mean that N becomes the limiting nutrient in summer. For this reason, the seasonality of nutrient inputs to waters can be important in relation to ecological impacts.

Figure 5.22. Annual inputs (t a⁻¹) of nitrate and ammonium to the Taw Estuary from each source (no STW nitrate data for 1997; no riverine data for 2005).
Little is known about how small, well-mixed estuaries with short residence times, such as the Taw, respond to variations in nutrient loading. In particular, how do physical and biological processes control nutrient concentrations, and how do such systems respond to changes in catchment run-off and inputs?

The high (>100 μg/l) concentrations of chlorophyll a observed during June to August and the high TIN:P ratios in the River Taw and the estuary strongly suggest that reductions in N export from the catchment due to NVZ Action Programme measures will have little impact on the trophic status of the estuary. A high N:P ratio indicates that the N supply remains plentiful even in summer, so that the small reductions likely as a result of the NVZ AP are unlikely to reduce it to the point where N availability limits growth. Furthermore, under low river flow conditions, concentrations of chlorophyll a in the estuary are mainly governed by inputs of ammonium and phosphate from the Ashford STW.

Any reduction in nitrate loads is unlikely to have a significant impact on algal growth in the upper and middle estuary as nitrate concentrations in the water column are still high enough to allow maximum uptake rates by phytoplankton for most of the spring and summer period.

It is important to emphasise that any legislative drivers may require a significant period before having a positive, observable effect. For example, the Danish Action Plan measures, which went beyond the requirements of the Nitrates Directive, appear to show that the response time of Denmark’s estuarine and coastal waters is longer than 10 years and they may be experiencing the historical legacy of past nutrient inputs (stored in sediments).

Any assessment of estuarine eutrophication should take account of all potentially bioavailable nutrients, including organic forms of N and P. This study suggests that ammonia mostly overrides nitrate, particularly under low flow conditions and in the upper estuary. Nitrate appears to play only a certain role under intermediate flow conditions in the lower reaches of the estuary when ammonia concentrations drop due to dilution effects. However, the dataset did not contain information on DON, which leaves some uncertainties about the role of regenerated nitrogen within the studied system. For N, actions required to reduce fluxes from the catchment should also consider atmospheric deposition.

Figure 5.23. Annual inputs (t a-1) of phosphate to the Taw Estuary from each source (no STW data for 1990, 1991, 1992 and 1997; no riverine data for 2005).
The conclusions from the study were:

- The Taw Estuary system has the potential to respond relatively quickly to reductions in nutrient inputs. The estuary is macro-tidal with short water residence times (2-3 days, depending on river flow). The inter-tidal sediments are largely sand and gravel, with relatively little mud, which implies that the sediments do not represent a significant reservoir of nutrients.
- Reduced inputs of nutrients from the catchment would, if sufficient, lead to an improved trophic status within the estuary.
- The NVZ AP measures would not appear to be sufficiently stringent to bring such a change about.
- Beneficial changes to the trophic status of the estuary are also likely to be impeded if other nutrient inputs, particularly from Ashford STW, but also possibly from the atmospheric deposition of N to the catchment, are not controlled.
- The N:P ratios in the estuary indicate that algal growth is P limited. However, the concentrations of nutrients are sufficiently high to support the extensive algal growth observed. The estimated reductions in TIN inputs from the river to the estuary due to NVZ AP are small and will not impact significantly on the TIN concentration in the estuary.
- Recent improvements at Ashford STW resulting from implementation of the UWWTD over the period 1997 to 2004 may result in sufficient P reduction to reduce the incidence of summer algal growth within the estuary. It is therefore recommended that water quality parameters in the estuary are systematically monitored over the next two to five years to confirm this. However it should be noted that the response time of estuarine and coastal waters is likely to be longer than 10 years due to the historical legacy of past nutrient inputs stored in sediments.
6 IMPLICATIONS AND DISCUSSION OF IMPACTS OF THE NVZ AP

6.1 Implications of the NVZ AP for whole NVZ area

The Meden and Taw catchments were chosen to explore impacts in the arable and grassland sectors, respectively. The Meden catchment is fairly typical of the NVZ area in climate and land use. It has a higher proportion of sandy and shallow soils, larger quantities of manure, and substantial areas of non-agricultural land. The Taw catchment, being dominantly grassland and with high rainfall, is more typical of the extreme western outliers of the NVZ area.

The characteristics of the current NVZ area in England (prior to the 2006/7 review) are summarised in Table 6.1, for comparison with those of the case study catchments.

Table 6.1. Comparison of characteristics of agricultural land in the NVZ area and England with those of the Meden and Taw catchments

<table>
<thead>
<tr>
<th></th>
<th>Current NVZ area</th>
<th>England</th>
<th>Meden</th>
<th>Taw</th>
</tr>
</thead>
<tbody>
<tr>
<td>Annual rainfall arable (mm)</td>
<td>644</td>
<td>675</td>
<td>680</td>
<td>-</td>
</tr>
<tr>
<td>Annual rainfall grassland (mm)</td>
<td>783</td>
<td>872</td>
<td>-</td>
<td>900-2000</td>
</tr>
<tr>
<td>Arable, % of agricultural land</td>
<td>67</td>
<td>56</td>
<td>84</td>
<td>20</td>
</tr>
<tr>
<td>Total annual manure loading (kg/ha N)</td>
<td>33</td>
<td>39</td>
<td>74</td>
<td>41</td>
</tr>
<tr>
<td>% Sandy or shallow soils</td>
<td>8</td>
<td>6</td>
<td>65</td>
<td>0</td>
</tr>
</tbody>
</table>

The measure limiting N inputs to crop N requirement was estimated to reduce nitrate loss in the Meden by 7%, and in the Taw by 2%. Total N applied as manure in the Meden is about twice the national average, while the Taw is more typical of grassland areas within NVZs. Taking these factors into account, the total estimated impact of this measure was 2-4% for the current NVZ area, and a little less (2-3%) for the whole of England.

For the Closed Period, no impact was estimated in the Taw, and a 10% reduction in those areas of the Meden that were on sandy soils and subject to the Closed Period. The Meden contains a large proportion of poultry manure and, therefore, a larger proportion than average of manure applied in autumn. The quantity of slurry and poultry manure N that was estimated to be applied in the August to October period in the Meden was 5 to 6 times the NVZ mean. The area of sandy and shallow soils within NVZs is about 8%, and impacts of the Closed Period are smaller on grassland because of the N uptake by grass when manures are applied in early autumn. Taking these factors into account, the maximum impact of the current Closed Period was estimated to be a reduction of less than 0.1% in nitrate leaching, although, locally, much greater impacts are to be expected. It should be noted that the assumptions used here were relatively optimistic – that where the Closed Period operated on arable soils, all manures which were formerly applied in autumn would be moved to winter. In fact, some application is permitted in August, and applications made immediately upon the end of the Closed Period in November could result in significant nitrate loss.

For arable areas where the Closed Period applies, and with much of the manure applied being poultry manure, the estimated mean reduction in nitrate leaching was 5-10%. Without a closed period, the mean reduction was estimated as 3-4%. In dominantly grassland areas, the expected reduction in nitrate leaching was estimated as 0-2%, depending on the intensity of stocking.
The farm-level limit on excretal returns, currently operating at 250 kg N/ha on grassland, is considered (on the basis of analysis of farm statistics) to have a small impact overall. This is because the number of farms and cattle affected are small and because, in most cases, the response on affected farms will be to move some manure to neighbouring farms. The impact on pig and poultry farms is subsumed within the measure that requires that N inputs should not exceed crop requirement.

No impact was assigned for measures aimed at minimising the risk of surface runoff from manure and fertiliser (e.g. applying manure/fertiliser on frozen or wet ground, use of a buffer near to watercourses). These measures are important, but especially so for pollutants such as P, organic material (BOD, organic N); FIOs and ammonia, which travel chiefly by surface or preferential flow pathways (Jarvis, 2007). On average, it is expected they would have little impact on total annual nitrate emissions to water. This is confirmed in the DPI User Manual (Cuttle et al., 2006), which assigns a very small effect on nitrate loss to these measures.

The percentage impact on nitrate leaching will be only slightly affected by rainfall or soil type (other than where this affects the AP measures themselves). The impact in terms of change in concentration will be greater in drier areas (which tend to have greater concentrations), whilst the impact in terms of load will be greater in wetter areas, under similar land use.

Table 6.2. Comparison of modelled impacts of measures on nitrate loss from agricultural land in the NVZ area and England with those of the Meden and Taw catchments.

<table>
<thead>
<tr>
<th>Baseline nitrate loss, kg N/ha agricultural land</th>
<th>Current NVZ area</th>
<th>England</th>
<th>Meden</th>
<th>Taw</th>
</tr>
</thead>
<tbody>
<tr>
<td>% reduction due to:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>• Limit N inputs to no more than crop requirement</td>
<td>2 to 4</td>
<td>2 to 3</td>
<td>6.5</td>
<td>1.5</td>
</tr>
<tr>
<td>• Closed period for manure application on sandy and shallow soils</td>
<td>&lt;0.1</td>
<td>&lt;0.1</td>
<td>8.5</td>
<td>0</td>
</tr>
<tr>
<td>• Farm level limit on manure loading</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Total % reduction</td>
<td>1 to 4</td>
<td>1 to 3</td>
<td>15</td>
<td>1.5</td>
</tr>
</tbody>
</table>

The impact of the measures will in general be greater in areas of high livestock numbers, especially under arable, and these are the areas with the greatest nitrate concentrations at present.

These estimates are uncertain due partly to uncertainties in modelling, but chiefly to uncertainties as to exactly how manures and fertilisers are currently managed and how farmers will respond to the measures. Survey data can help, but carry their own uncertainties and are confounded by underlying changes in land management caused by other factors.

The conclusions are, however, clear – the existing measures have relatively small impacts, and these impacts will be greatest in arable areas with large numbers of livestock, and on sandy or shallow soils; these are also the areas which typically have the greatest nitrate concentrations at present.
In these areas, however, the measures will often be insufficient to reduce nitrate concentrations to less than 50 mg/l. This fact is borne out by direct measurements of nitrate loss from land within the current NVZs as part of this project (see earlier sections of this report); and by measurements and data analysis of N losses within the Nitrate Sensitive Areas Scheme, which implemented more severe measures on groundwater catchments now subsumed within the NVZ area (Lord et al., 1999).

6.2 Implications of other measures

The large predicted reductions in leaching caused by assuming all manure in a catchment could be removed resulted in much greater reductions in nitrate leaching (27% in the Meden, 9% in the Taw). This is an unfeasible option for all but small areas, but it does highlight the fact that manures increase nitrate leaching in ways which are not dealt with by the current AP measures. This is partly because closed periods cannot be made entirely comprehensive, for practical reasons, but chiefly because manures contain organic N. This N is released slowly, making it difficult to account for in fertiliser recommendations, and causing increases in the quantity of nitrate that accumulates in autumn and is at risk of leaching. Measures such as cover crops where land would otherwise be bare over winter could make an important contribution to minimising such nitrate losses.

A 10% reduction in fertiliser inputs was estimated to reduce nitrate leaching by 7% in the Meden and 3% in the Taw. This measure had a more evenly distributed impact than measures focussed on manure management. Implementation of this measure in legal terms would of course require a more detailed approach, but the work gives an indication of the scale of response that might be expected.

6.3 Time-scale of responses

The EC requires an assessment of the time-scale of responses. Most management changes rapidly affect nitrate loss from the land, and hence the signal in surface water catchments. However, it can take many years before these effects are evident in groundwater abstractions, or in groundwater-fed rivers. To aid Defra in assessing the likely range of response times for any given location, BGS have produced a detailed report embodying a methodology for arriving at a generalised assessment of aquifer response times (Hughes et al., 2006; see Appendix V).

Each aquifer will be unique in its response time to nitrate movement through the unsaturated zone to the saturated zone. The flow pathways to rivers and boreholes are summarised in Figure 6.1, showing that the abstracted or discharging water can be a mixture of water of different ages, originating at the land surface at different times. In addition to path length, the effects of processes such as mixing and diffusion must be taken into account. Overall, the response times are a combination of unsaturated and saturated zone residence times. Thus, rapid responses are to be expected in alluvial sands and gravels, which are generally thin and shallow; but decades are required for responses in deeper sandstone and chalk aquifers.

Aquifers in England were classified on the basis of geology, hydrology and climate as described in Section 4. Rainfall categories were <650, representing much of the arable area of England, >850, representing dominantly grassland areas, and the intermediate class. Table 4.2 above confirms that the most important aquifers (Chalk and Permo-Triassic Sandstone) occur in all three rainfall categories. The majority of the Chalk outcrop area is in the dryer south and east, whereas the Permo-Triassic Sandstone is more dominantly in the wetter central and western part of the England and Wales, but still with a substantial eastern outcrop in Nottinghamshire and South Yorkshire. The aquifer classification is mapped in Figure 6.2. Most of the NVZ area receives less than 850 mm annual rainfall.
By multiple modelling runs with the MAP groundwater model, typical response times were calculated for different aquifer types. These have been summarised in Figure 6.3, showing that rapid responses are to be expected in alluvial sands and gravels, which are shallow; but decades are required for responses in sandstone and especially chalk aquifers.

The MODFLOW and MAP codes were then used to simulate groundwater flow and nitrate transport for a wide range of aquifer scenarios chosen to give a representative coverage of the country.
Conceptual models representative of each aquifer type were developed using the combined experience of BGS hydrogeologists who have worked in these areas. For each type, aquifer thickness, hydraulic conductivity and abstraction rates were the key variables used in the models. The mathematical models produced from these conceptual models were designed to be representative of typical ranges of conditions for that particular aquifer type: they do not represent specific cases, and are not designed to illustrate extreme behaviour. To this end, the models have not been calibrated against measured data, rather hydrogeological expertise has been used to ensure that the models are behaving in a reasonable manner.

![Summary aquifer response times (from Hughes, 2006).](image)

The first modelling runs showed that, as could have been predicted, the results for the middle rainfall category fall between those for the high and low rainfall categories. Also there was no significant difference in the shape of the three curves. Therefore, it was decided to model only the high and low rainfall categories.

Figure 6.4 shows the response of the various aquifer types to a step change in nitrate loading at the surface, for the high and low rainfall scenarios. The Crag aquifer was modelled only for low rainfall as it only occurs in East Anglia. The response plots take into account travel of nitrate through the unsaturated zone, the part of the system between the land surface and the water table. The greater the thickness of the unsaturated zone, the longer the response time of the aquifer to changes in nitrate concentration. The results of the other model runs detailed in Table 4.3 are shown in Appendix V.

Table 6.3 shows what percentage of the change will be registered at the abstraction point 10 years after the step change in nitrate loading at the surface and also the time (in years) for 50% of the change to register at the abstraction point. Detailed response curves are given in Figure 6.4.
Table 6.3. Modelled response times for the main aquifer types.

<table>
<thead>
<tr>
<th>Aquifer type</th>
<th>% change after 10 years</th>
<th>Time to 50% (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low rainfall</td>
<td>High rainfall</td>
</tr>
<tr>
<td>Chalk</td>
<td>8</td>
<td>10</td>
</tr>
<tr>
<td>Sandstone</td>
<td>8</td>
<td>10</td>
</tr>
<tr>
<td>Limestone</td>
<td>28</td>
<td>30</td>
</tr>
<tr>
<td>Crag</td>
<td>&lt;5</td>
<td>175</td>
</tr>
<tr>
<td>Alluvium</td>
<td>48</td>
<td>58</td>
</tr>
<tr>
<td>Greensand</td>
<td>10</td>
<td>15</td>
</tr>
</tbody>
</table>

The details of the model runs performed are summarised in Table 4.3. Various different abstraction rates were modelled. However, the magnitude of the abstraction (within the expected range) did not have a significant effect on the response of the system to the change in nitrate concentration. The reason that the response is not sensitive to the abstraction rate is because the larger capture zone of a large abstraction also contains higher water table gradients. Thus, even though the water has a longer distance to travel, it is moving faster, resulting in similar travel times.

Rainfall >850 mm/year:

Rainfall <650 mm/year:

Figure 6.4. Modelled response time of aquifers following a change in nitrate inputs at the surface

6.3.1 Summary

This work confirms the very long time-scales over which the transport of nitrate to groundwater abstraction or discharge points can occur, and highlights the difficulty of being able to measure the impact of NVZ measures on groundwater quality. However, differences exist between the response times for each aquifer type. The extremes are the most rapid response for the Alluvial aquifers to the slowest response observed in the Crag. The former, the Alluvium, are typically, small, shallow aquifers with short groundwater travel pathways. The latter, the Crag, was modelled with recharge at the margins of the aquifer only and long travel pathways to the abstraction boreholes.

The results for the Chalk and Sandstone aquifers show very similar timings; 8–10% change after 10 years and 61 and 64 years to 50% change for low rainfall and 41 to 46 years for 50% change for high rainfall.

At first sight, this is surprising since nitrate movement in Chalk and Sandstone aquifers occurs in different ways. In Chalk aquifers, saturated-zone flow is predominately through
fractures and nitrate can move by diffusion between the flowing fractures and the matrix (blocks of Chalk). Sandstone aquifers have a high porosity (water filled parts of the rock) and groundwater movement is slow. Historically, however, the groundwater nitrate problem was recognised at similar times in both aquifers and this indicates that overall average time of travel in both aquifers is similar, although the transport processes in the unsaturated and saturated flowpaths are distinctive.

6.4 Impacts on eutrophication

Some NVZs are designated because they drain to waters identified as eutrophic, on the basis of elevated nitrate levels (albeit not necessarily above 50 mg/l) and biological indicators demonstrating accelerated growth and undesirable disturbance. Therefore, as described in Section 2.5.2, the Centre for Ecology and Hydrology (CEH), undertook an assessment of the likely impacts of the NVZ Action Programme on eutrophication, comprising:

- A review of the factors affecting riverine eutrophication
- A review of the factors affecting estuarine eutrophication
- A case study of the Taw estuary to assess impacts of the Action Programme.

Details of the approaches and results are in Appendices VI - IX and the key points are summarised below.

6.4.1 Eutrophication in rivers

From Newman (2006):

There are clear ecological effects of raised nutrient concentrations in both lakes and rivers, including:

- Increases in planktonic algae in lakes and slow-moving rivers
- Increases in filamentous algae in streams
- Changes in macrophyte communities to fast-growing, competitive (weedy) species particular impact on species or habitats of high conservation value
- Impacts on fisheries, particularly economically important salmon rivers

The nutrients of primary concern for eutrophication are N and P compounds. However, in rivers, plant communities respond to flow, sediment type, and underlying geology more than any transient changes in dissolved nutrient status derived from external inputs. Flushing in flowing systems tends to reduce exposure times to enhanced nutrient loads, thereby reducing the scale of any change.

Increases in both N and P cause changes in plant communities similar to those observed for P-enrichment only. It can be deduced that P is the main driver for change in freshwater aquatic plant communities, rather than N. This is not to say that N does not have a role to play in some circumstances (especially shallow lakes).

Enrichment by N tends to be associated with dissolved nutrients in the water column, whereas enrichment by P is associated with both sediment-bound and water column nutrients. It is, therefore, theoretically possible to reduce the effects of N-enrichment relatively easily over a relatively short timeframe if inputs are controlled, while the effects of P will be less easily resolved over short time-scales. Assuming that the major observable effects are P-driven, and exacerbated by N enrichment, then the observable effects of a reduction in N may not be detectable until P is also reduced.

There are large regional differences in the degree to which rivers deviate from a 'natural/background' level, with less impacted sites predominating in less agriculturally-intensive (and less populated) landscapes, particularly Cumbria and the Scottish Highlands.
and Islands. Lowland regions in Central/Southern England, Northern Ireland, Wales and Central Scotland are generally impacted much more severely.

The quantitative ecological response of individual waters to raised nutrient concentrations is, however, more uncertain and is partly mediated by their geoclimatic setting and land–use history as well as sensitivity factors such as flushing/flow regime. Much greater understanding of these sensitivity factors is required to enable more reliable predictions of likely ecological responses to and increase or decrease in nutrient loadings from the catchment.

The general regional pattern in ecological response does, however, reflect a broad regional split between present day Total P concentrations typically observed in upland and lowland waters. The greatest ecological change is observed in lowland regions in Northern Ireland and England where present-day nutrient concentrations are often well in excess of background or reference concentrations. The study by Bennion et al. (2004) supports this conclusion, indicating that 36% (79) of their study lakes showed low floristic change, with the majority of these located in the uplands of Scotland (with a few sites in upland Wales and the English Lake District).

Eutrophication of rivers is best managed by reducing inputs to the river system, rather than any in situ remedial action. However, in these severely impacted regions, the extent of nutrient changes may be so great, that large reductions in nutrient loadings from catchments are likely to be needed before any ecological response is observed.

Eutrophication can be spatially restricted in river systems, for example downstream of point source inputs. Dilution tends to restrict both the magnitude and extent of these impacts. Nutrient sources are not necessarily all agricultural, with significant point sources in some catchments.

### 6.4.2 Eutrophication in estuaries

From Nimmo-Smith et al. (2006):

The enrichment of estuaries with nutrients is the most common factor that initiates the eutrophication process.

Estuarine eutrophication has increased on a global scale in recent decades but the nature of the problem faced in England and Wales is not straightforward. Many of our estuaries experience hypernutrification (an excess of nutrients without adverse effects occurring), rather than eutrophication (an excess of nutrients and giving rise to an undesirable disturbance of the ecosystem). As a consequence, nutrients may be flushed from estuaries to adjacent coastal waters, where they may cause undesirable disturbance.

Phosphorus is the generally considered the most common limiting nutrient in UK mainland estuaries. However, this is not always so: NVZs are designated around eutrophic estuaries because it is considered that N is limiting. In these circumstances, therefore, taking action to reduce losses of N from agriculture is likely to have an impact on estuarine eutrophication if the measures are sufficient to cause a significant reduction in N loss. Similarly, removal of N from sewage treatment works discharges into eutrophic estuaries is required under the Urban Waste Water Treatment Directive.

Nutrient fluxes to estuaries are dominated by riverine inputs, which have experienced increased N and P concentrations over time. The average ammonium and nitrate riverine load to UK estuaries of $1.4 \times 10^3$ kg N km$^{-2}$ a$^{-1}$ is almost the same as the value of the average total N load from catchments around the North Sea. In contrast, the average total dissolved
inorganic phosphorus (DIP) riverine load to UK estuaries of 152 kg P km\(^{-2}\) a\(^{-1}\) is above the average DIP load of 117 kg P km\(^{-2}\) a\(^{-1}\) for catchments around the North Sea. Estuary area-normalised riverine loads are greatest on the east and south coasts of England. Furthermore, these loads may be under-estimated as only a limited number of N and P species are measured. Estuaries also potentially receive nutrients from submarine groundwater discharges (SGD) and atmospheric inputs, often poorly estimated, which could be of significance on a local scale.

However, the quantification of riverine inputs requires revision in order to include dissolved organic and particle associated N and P fractions that have historically been ignored. This has been due to a focus on the effect of inorganic N runoff from fertiliser application on water quality and a lack of suitable analytical methodologies for determining organic N and P species. More regular chemical monitoring, development of methods to measure biological effects, and speciation of N and P are required. Furthermore, estuaries also receive other inputs, such as those from the atmosphere and SGDs, which are not accurately quantified at present. Hotspots for nutrient enrichment are where estuary area-normalised loads are high i.e. along the east and south coasts of England. However, complete reliance on nutrient load data to predict the potential for estuarine eutrophication is inadvisable, as complex nutrient cycling and removal processes occur within estuaries.

Nutrient limitation plays a key role in estuaries and may switch from P in the spring to N in the summer. Biogeochemical processes transform nutrients within estuaries. N is permanently removed from the estuary to the atmosphere via the biological process of denitrification, although estuaries with long residence times and high nitrate concentrations experience greater losses. This loss of N may contribute towards N limitation in estuaries during the summer.

In contrast, P is temporarily buried in estuarine sediments and also rapidly exchanged between aqueous and particulate phases. The wide range of observed behaviour of P complicates attempts to calculate how river inputs affect estuarine eutrophication. A seasonal pattern of P cycling is apparent, where burial occurs in winter and spring, and P is released during summer. This cycling may contribute towards P limitation in estuaries during the spring. There is considerable uncertainty surrounding the potential of estuaries to permanently remove P via burial.

Understanding the role of nutrient limitation in estuaries is essential when considering management of the eutrophication process. However, estuaries probably are not limited by one nutrient, as lakes and open coastal waters tend to be, but may experience a switch from P limitation in the spring to N limitation in the summer.

There is conflicting evidence from studies in the EU and USA on the ability of nutrient load control measures to reduce nutrient enrichment and undesirable disturbance. Denmark’s multi-dimensional (e.g. nutrient load control, wetland re-establishment, afforestation) Action Plans for the Aquatic Environment appear to have the most potential, although the historical legacy of past and present nutrient inputs will impact on the response rate of estuarine and coastal waters in the near future. Interestingly, studies from the USA (Boesch et al., 2001; Whitall et al., 2004) go a step further and suggest that a multi-sectoral approach is required to effectively manage nutrient pollution.

In relation to specific EU policy measures, consideration of NVZ Action Programme measures revealed that there is some potential to reduce nutrient enrichment in estuaries. However, as a result of uncertainty surrounding the complex biogeochemical cycling of nutrients within estuaries and their catchments, it is difficult to generalise.
6.4.3 Summary

- Eutrophication is a complex process involving many factors, including not just nutrients but flow rates and climate.
- In rivers, eutrophication is limited by $P > Si > N$. Since $N$ is rarely limiting, reduction in $N$ loss is unlikely to change river status. If the NVZ AP reduces the risk of surface runoff of $P$, this could reduce the risk of eutrophication. Estimation of this effect is very difficult because it comprises changes in risk management, for which no prior or post survey data are available.
- In estuaries, there may be seasonal changes, with $P$ limiting algal growth in spring, and $N$ in summer. Any changes that increase $P$ inputs in spring are therefore likely to be detrimental.
- A too-severe closed period in future, preventing manure applications in autumn could increase inputs in late winter and spring, which could increase $P$ emissions dramatically especially from clay soils. The changes in $N$ emissions attributable to the current NVZ AP are unlikely to be large enough to have major beneficial impact. These potentially conflicting factors need to be taken into account in determining Closed Periods for manure application.

A detailed case study to illustrate the issues using the Taw catchment (a mainly grassland catchment with moderate to high rainfall, in Devon) suggests that it is unlikely that AP measures will probably contribute to a reduction in estuarine nutrient loadings, and ecological response in the estuary may result, although other nutrients have a part to play and the loadings from the sewage treatment works may be more significant. Impacts on existing plant and diatom communities present in the river Taw are not likely, as the communities probably do not indicate eutrophic conditions at present.

6.5 Review of nitrate mitigation measures

Nitrogen cycling, nitrate loss and control under UK conditions are well researched topics, with a huge base of scientific knowledge underpinning the choice of measures in the current NVZ Action Programme (e.g. see SUM, 2000; Shepherd & Chambers, 2007). Generally, all of these measures seek to reduce or prevent the accumulation of high levels of nitrate in the soil at times when leaching or surface run-off is likely to occur. Soil nitrate that remains after harvest can be lost during autumn and winter drainage, while nitrate that is mineralised during this period is also susceptible to leaching. The majority of practical methods for decreasing nitrate leaching, therefore, seek to minimise the amount of soil nitrate in autumn.

However, controlling the source of nitrate is not the only potential solution to reducing nitrate loss. Mitigation measures that also focus on mobilisation or transport of $N$ may also have a part to play. The source-mobilisation-transport concept is a common framework for assessing how mitigation methods function.

In reviewing the current Action Programme, we need to consider if there are other potential mitigation measures that would be effective (or, indeed, if any of the current mitigation methods are ineffective). Haygarth et al. (2006) identified a list of over 40 potential mitigation methods for controlling on-farm losses of nitrate, phosphorus, sediment and pathogens (Table 6.4). These can be categorised in terms of management of soil, crop, fertiliser, manure or animal, as well as infrastructure changes, indicating that there are many points in a farm system where mitigation methods could be applied. It should be noted that this list is not exhaustive: Cuttle et al. (2004) included a wider range of measures. Furthermore, new measures are continually developing, e.g. development of a nitrification inhibitor as a bolus to feed to cattle (S. Ledgard, Pers. Comm.). However, the list does cover the majority of the current main types of potential mitigation methods.
An analysis of the 44 mitigation methods identified by Haygarth et al. (2006) shows (Table 6.4) that:

1. A significant proportion of the methods is likely to have little (or uncertain) effect on nitrate loss and they are more appropriate for controlling P/sediment or pathogen losses.
2. Of those methods that are effective for nitrate, many (focusing on manure and fertiliser management) are included in the current Action Programme in some form, although there may be scope for amending their formulation. Others, such as the use of cover crops and reducing N fertiliser applications below the economic optimum, are likely to be effective but have not been included.
3. There may be scope to include other mitigation methods or broaden the scope of some current AP measures in a revised Action Programme, but at increased cost.
4. Some mitigation methods that show potential are practically difficult to incorporate into a farming system and/or carry a high cost. Translation of some measures into enforceable legal requirements is also problematic.

We now review these categories. The analysis is not meant to be a complete critical appraisal of each mitigation method; this has been undertaken in other reports and papers. Rather, this is a summary assessment to place possible measures in the context of the current AP.

It is important to note that the overall effectiveness of a measure to control leaching will also be determined by how widely it can be applied. For example, the effectiveness at a regional scale will be much reduced if the measure only achieves significant reductions in leaching on a particular soil type, which does not occur extensively, or if the measure is directed at management practices that are undertaken by only a few farmers. Equally, the degree of current compliance with a measure affects the improvement that its enforcement will bring. This is particularly true of measures classed as ‘good practice’.

In selecting mitigation measures, it is also important to take account of impacts of measures on other forms of N or other pollutants – whether beneficial or adverse. (see Section 6.7).
Table 6.4. An assessment of the mitigation methods for diffuse pollution control listed by Haygarth et al. (2006), categorised by likelihood of a significant effect on nitrate loss (\(\checkmark\checkmark = \text{‘yes’}, X = \text{‘no’}, ? = \text{‘uncertain’}) and inclusion in the current NVZ Action Programme.

<table>
<thead>
<tr>
<th>No.</th>
<th>Method</th>
<th>Sig. effect on NO(_3)?</th>
<th>Current AP?</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>1a</td>
<td>Convert arable land to extensive grassland (no livestock)</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>Highly effective as, demonstrated in the pilot Nitrate Sensitive Areas scheme</td>
</tr>
<tr>
<td>1b</td>
<td>Convert arable land to extensive grassland (include livestock)</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Establish cover crops in the autumn</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>Effective, but restricted opportunities for application due to rotations and soil type</td>
</tr>
<tr>
<td>3</td>
<td>Cultivate land for crop establishment in spring, not autumn</td>
<td>?</td>
<td>X</td>
<td>Not practical for all soils, uncertain effects</td>
</tr>
<tr>
<td>4</td>
<td>Adopt minimal cultivation systems</td>
<td>?</td>
<td>X</td>
<td>Not practical on all soils, uncertain effects</td>
</tr>
<tr>
<td>5</td>
<td>Cultivate compacted tillage soils</td>
<td>?</td>
<td>X</td>
<td>Of indirect benefit by improving N uptake by crops</td>
</tr>
<tr>
<td>6</td>
<td>Cultivate and drill across the slope</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>Leave autumn seedbeds rough</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>8</td>
<td>Avoid tramlines over winter</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>9</td>
<td>Establish in-field grass buffer strips</td>
<td>?</td>
<td>X</td>
<td>No spread zones next to watercourses in the current AP</td>
</tr>
<tr>
<td>10</td>
<td>Loosen compacted soil layers in grassland fields</td>
<td>?</td>
<td>X</td>
<td>Of indirect benefit by improving N uptake by grass</td>
</tr>
<tr>
<td>11</td>
<td>Maintain and enhance soil organic matter levels</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>12</td>
<td>Allow field drainage systems to deteriorate</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>Highly impractical and likely to increase gaseous emissions</td>
</tr>
<tr>
<td>13</td>
<td>Reduce overall stocking rates on livestock farms</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>May be necessary if average manure loading is exceeded</td>
</tr>
<tr>
<td>14</td>
<td>Reduce the length of the grazing day or grazing season</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>Practical constraints?</td>
</tr>
<tr>
<td>15</td>
<td>Reduce field stocking rates when soils are wet</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>16</td>
<td>Move feed and water troughs at regular intervals</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>17</td>
<td>Reduce dietary N and P intakes</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>Potential for decreasing manure N excretion</td>
</tr>
<tr>
<td>18</td>
<td>Adopt phase feeding of livestock</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>Potential for decreasing manure N excretion</td>
</tr>
<tr>
<td>19</td>
<td>Use a fertiliser recommendation system</td>
<td>(\checkmark\checkmark)</td>
<td>(\checkmark)</td>
<td></td>
</tr>
<tr>
<td>20</td>
<td>Integrate fertiliser and manure nutrient supply</td>
<td>(\checkmark\checkmark)</td>
<td>(\checkmark)</td>
<td></td>
</tr>
<tr>
<td>21</td>
<td>Reduce fertiliser application rates below optimum</td>
<td>(\checkmark\checkmark)</td>
<td>X</td>
<td>On average, but be a significant benefit to N loss – but with economic penalty</td>
</tr>
<tr>
<td>22</td>
<td>Do not apply P fertilisers to high P index soils</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>No.</td>
<td>Method</td>
<td>Sig. effect on NO$_3$?</td>
<td>Current AP?</td>
<td>Comments</td>
</tr>
<tr>
<td>-----</td>
<td>------------------------------------------------------------------------</td>
<td>-------------------------</td>
<td>-------------</td>
<td>--------------------------------------------------------------------------</td>
</tr>
<tr>
<td>23</td>
<td>Do not apply fertiliser to high-risk areas</td>
<td>√ √</td>
<td>√</td>
<td></td>
</tr>
<tr>
<td>24</td>
<td>Avoid spreading fertiliser to fields at high-risk times</td>
<td>√ √</td>
<td>√</td>
<td></td>
</tr>
<tr>
<td></td>
<td><strong>Manure management</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>25</td>
<td>Increase the capacity of farm manure (slurry) stores</td>
<td>?</td>
<td>√</td>
<td>Required in circumstances where there is insufficient</td>
</tr>
<tr>
<td>26</td>
<td>Minimise the volume of dirty water produced</td>
<td>?</td>
<td>X</td>
<td>Reduces handling problems</td>
</tr>
<tr>
<td>27</td>
<td>Adopt batch storage of slurry</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>28</td>
<td>Adopt batch storage of solid manure</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>29</td>
<td>Compost solid manure</td>
<td>?</td>
<td>X</td>
<td>Focused on controlling pathogens, but may decrease N availability (and leaching risk)</td>
</tr>
<tr>
<td>30</td>
<td>Change from slurry to a solid manure handling system</td>
<td>?</td>
<td>X</td>
<td>Focused on controlling pathogens, but may decrease N availability (and leaching risk)</td>
</tr>
<tr>
<td>31</td>
<td>Site solid manure heaps away from watercourses/field drains</td>
<td>√ √</td>
<td>X</td>
<td>Good agricultural practice</td>
</tr>
<tr>
<td>32</td>
<td>Site solid manure heaps on concrete and collect the effluent</td>
<td>√ √</td>
<td>X</td>
<td>Good agricultural practice</td>
</tr>
<tr>
<td>33</td>
<td>Do not apply manure to high-risk areas</td>
<td>√ √</td>
<td>√</td>
<td></td>
</tr>
<tr>
<td>34</td>
<td>Do not spread farmyard manure to fields at high-risk times</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>35</td>
<td>Do not spread slurry/poultry manure to fields at high-risk times</td>
<td>√ √</td>
<td>√</td>
<td></td>
</tr>
<tr>
<td>36</td>
<td>Incorporate manure into the soil</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>37</td>
<td>Transport manure to neighbouring farms</td>
<td>?</td>
<td>?</td>
<td>Required where farm N loading is above the limit</td>
</tr>
<tr>
<td>38</td>
<td>Incinerate poultry litter</td>
<td>√ √</td>
<td>?</td>
<td>Reduces farm N loading</td>
</tr>
<tr>
<td></td>
<td><strong>Infrastructure</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>39</td>
<td>Fence off rivers and streams from livestock</td>
<td>?</td>
<td>X</td>
<td>Restricts direct excretal deposition</td>
</tr>
<tr>
<td>40</td>
<td>Construct bridges for livestock crossing rivers and streams</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>41</td>
<td>Re-site gateways away from high-risk areas</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>42</td>
<td>Establish new hedges</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>43</td>
<td>Establish riparian buffer strips</td>
<td>?</td>
<td>X</td>
<td>Uncertain effects for N</td>
</tr>
<tr>
<td>44</td>
<td>Establish and maintain artificial (constructed) wetlands</td>
<td>?</td>
<td>X</td>
<td>Practical constraints for adoption on a large scale</td>
</tr>
</tbody>
</table>
6.5.1 Current AP Measures

Table 6.4 shows that the current AP measures (boxed in the Table) are of a type which can have a significant effect on nitrate leaching and focus on manure and fertiliser management, i.e. trying to control the source of nitrate in the soil.

Section 5 undertook a quantitative assessment of these effects and this is continued under Section 6.1.

The formulation of the measures has a great effect on their impact. For example the ‘high risk period’ (Closed Period) for livestock manure applications as defined within the current NVZ AP applies only to the most risky soils, and is relatively short. Nitrate losses arising from manure applications outside the Closed Period, or on other soils, remain substantial.

6.5.2 Measures with small or uncertain effects on N

Many of these measures (greyed out in Table 6.4) focus on minimising soil damage and/or movement of phosphorus and sediment across the soil surface (i.e. affecting ‘mobilisation’ and ‘transport’ of P/sediment). This loss pathway is generally insignificant for nitrate (e.g. Harris & Catt, 1999). Quantities of surface runoff are small compared to the volume of water moving through the soil, and nitrate concentrations in surface runoff are in general not very different from those in water percolating through the soil. In contrast, concentrations of pollutants such as P, sediment, pathogens and ammonium, which are attenuated by passage through soil, are normally much greater in surface runoff. Therefore prevention of surface runoff will normally have much less impact on nitrate loss than on losses of these pollutants.

The primary aim of manure management measures involved with composting or adopting batch storage is to reduce pathogen numbers (i.e. controlling the ‘source’ of a pollutant), with little benefit to nitrate control.

In terms of cultivation (measures 3 and 4, Table 6.4), physical processes that disrupt the soil structure can influence the mineralisation of soil organic N. Hence, there is apparent scope for manipulating this release (and subsequent leaching) by manipulating time or type of cultivation. Silgram & Shepherd (1999) suggest that this can offer some reduction in nitrate loss in some circumstances. However, there are many practical constraints on time and type of cultivation.

Farms on clay soils are increasingly opting for minimal tillage (or, rather, non-inversion tillage). Minimal tillage is perhaps a misnomer, as the discs and tines used can disrupt the soil almost as much as ploughing. Little if any work comparing N mineralisation and leaching from ploughing and non-inversion tillage has been undertaken, though we would expect losses to be similar. Non-inversion tillage brings with it other practical considerations:

- It is not a suitable technique in a wet autumn – ploughing is better.
- It increases the chance of weed build-up, so that rotational ploughing is often combined with non-inversion tillage.
- It is less appropriate for spring crops.

In summary, there may be scope to reduce nitrate loss to some extent by manipulating timing and type of cultivation. However, other practical considerations usually mean there is, in practice, little scope for this manipulation.

An important exception occurs during establishment of autumn cover crops, where it was found that better results (earlier, more complete plant cover) were generally obtained by minimal cultivation, often allowing weeds and volunteers to play their part ion creating cover, compared with ploughing, which killed natural regrowth, delayed attainment of green cover and increased N mineralisation (Lord et al., 1999).
We have questioned the effectiveness of buffer strips for nitrate control in Table 6.4. Buffer strips can be very effective at trapping sediment from overland flow. Removal of nitrate, which is in solution, is not likely at least during periods of rapid flow (Cuttle et al., 2004). Whether they are of value or not depends on the hydrology of the site; channelling of surface water, preferential subsurface flow, the presence of agricultural drains and direct drainage to groundwater all reduce their effectiveness. A large proportion of the nitrate loss from agricultural land in the UK is via deep seepage (leaching) through soil or drains and will not be controlled by buffer strips.

6.5.3 Potential for other measures?

Manure export from the farm or poultry manure incineration (measures 37 and 38, Table 6.4) may, in effect, already be invoked by the current AP rules as one possible response by a farm that will exceed its annual average manure N loading.

Cover crops, because they are not a part of the current NVZ Action Programme, were not tested specifically within this project, but the review of other data as a part of this project shows them to be of benefit: both in experimental plots and within the pilot NSA scheme (Lord et al., 1999). They clearly have some potential in a revised AP, with the caveat that they are only of benefit in rotations with a significant proportion of spring crops, which typically occur on light to medium, relatively workable soils. They are not effective after late-harvested crops. It is estimated that about 20% of arable land would be appropriate for establishment of an over-winter cover crop.

There are practical issues relating to establishment and destruction of the cover crop and pest and disease control. They also carry a cost.

Reducing N fertiliser applications below the economic optimum has potential for decreasing N leaching losses but is not a part of the current NVZ AP. A quantitative assessment of the likely effect was modelled within this project, as described in Section 5. Clearly, there is potential to decrease nitrate loss, but with a financial penalty both in terms of yield and quality of produce. Formulation of such a measure to minimise disproportionately adverse economic effects would require care.

Many of the measures in Table 6.4 that are not within current AP rules are still in Defra’s (voluntary) Code of Good Agricultural Practice (including those that focus on pollutants other than N, as discussed above).

Many of the other measures listed in Table 6.4 are expensive, practically difficult or of uncertain effect.

Conversion of arable land to low N grassland was extremely effective in the pilot NSA scheme (Lord et al., 1999), but the effects on individual farms of taking land out of production may be extreme unless funded in some way (e.g. through agri-environment schemes).

Similarly, reducing stocking rates will reduce nitrate leaching losses, but can have significant financial implications for a farm business. The framing of such a measure is critical – the benefit of reduction of stocking rates on some farms will be largely offset if the stock are taken up by other farms within the catchment or NVZ.

6.5.4 Other measures not listed in Table 6.4

Table 6.4 is not an exhaustive list of potential measures, though they may be considered the main current approaches. Agricultural researchers in New Zealand take a similar approach, identifying measures that fit into agricultural systems now, but also looking forward to
develop ‘second generation’ measures (S. Ledgard, Pers. Comm.). UK agricultural researchers need to take a similar approach.

Of potential current measures, two extra possibilities are worthy of discussion: fertiliser spreader calibration and nitrification inhibitors.

**Fertiliser spreader calibration** could tackle two sources of inaccuracy when applying fertiliser: (Dilz et al. 1985)

- Incorrect rate of application – applying the correct rate depends on accurately predicting the optimum rate.
- Uneven spreading

Uneven spreading can be a result of spreader design, fertiliser properties or spreader maintenance. Effects on yield (and profitability) of uneven spreading have been estimated in previous projects and the conclusion was that for a fertiliser spreader with a CV below 30%, the effects of spreading on yield were small. Similar conclusions were drawn for manure spreading.

However, effects on nitrate loss are likely to be quite different. This is because of the response of N leaching to fertiliser rate, as described earlier. On areas of the field that would be under fertilised, due to uneven spreading, decreases in N leaching would be small compared with fertilising at the optimum. However, in areas receiving excess fertiliser, leaching would increase compared with fertilising at the optimum. Because the increases in N leaching would be greater than the decreases, the net effect of uneven spreading could be large.

Thus, the mitigation method should be to regularly calibrate fertiliser spreaders. This is a relatively simple, low cost operation that is considered ‘Good Practice’. Ideally, it should be repeated when there is a change in the type of fertiliser being spread, because the quality and density of the material will affect spread pattern. Fertiliser spreader calibration is promoted by the Fertiliser Industry.

In a recent small study of fertiliser calibration as part of the Wagrico project\(^3\), 18 spreaders were tested, with the following results (M. Taylor, Pers. Comm):

- Machines test generally well presented/maintained
- Of 18 tested 60% required some form of adjustment/repair
- 5% were in such poor condition realistically should be replaced
- 10% required mechanical repair to perform accurately
- 22% required replacement of wearing parts to improve spread pattern
- 22% required static calibration for application rates to balance

Of those that required adjustment and that were re-tested after adjustment, the average spread CV decreased from 22% to 3%; a CV of <10% is considered acceptable. Statistics from the British Survey of Fertiliser Practice (2006) suggest that 7% of farms check spread pattern after each change of fertiliser type and 44% check calibration at least once a year. However, 10% of survey respondents checked less than once per year and 16% never checked (Goodlass & Welch, 2007).

**Nitrification inhibitors** work by blocking the conversion of ammonium to nitrate and under suitable conditions can decrease nitrate leaching (Schroder et al., 1993). The chemical compounds are degraded in the soil and their efficacy depends on the rate of this degradation and the duration of their inhibitory action. Inhibitors can be mixed with fertilisers and manure. Logically, the use of inhibitors would be most effective with manure that

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\(^3\) EC LIFE-funded project Water resources in co-operation with agriculture
contains a large ammonium component (i.e. slurries, liquid digested sludge) and when the manure is applied in late summer or early autumn, when the leaching risk is greatest. Their use could potentially shorten the closed period, therefore. However, because the efficacy of the inhibitor is so dependent on environmental conditions and the rate of breakdown, it is unlikely that inhibitors would be accepted as a way of widening the autumn application window for manures in NVZs. They could increase uncertainty as to the crop-availability of manure-derived N, thereby increasing the risk of incorrect fertilisation. There may also be uncertainties regarding the environmental desirability of applying such materials to land, and the risks of them being lost to water.

6.6 Pollution swapping

This NIT18 project did not specifically deal with ‘pollution swapping’, but it is something that needs to be considered when reviewing Action Programme measures. Pollution swapping can take the form of:

- Interchange between different form of N; trading losses of N as nitrate for losses as ammonia (NH₃) or nitrous oxide (N₂O).
- Influence of measures on loss of other potential pollutants to water, most notably phosphorus (P).

6.6.1 Impact on risk of P and other pollutant loss to waters

Phosphorus reaches waters chiefly by surface runoff, or rapid flow through cracking clay soils and drains. Transfers may be as particulate P (chiefly bound to soil and suspended sediment) and as soluble P (for example derived from fertiliser or manure). Factors that reduce the risk of P loss to waters are:

- Prevention of soil erosion
- Avoiding leaving sources of soluble P on the soil surface, at times and in locations where there is a risk, within a few days/weeks, of
  - Surface run-off to streams
  - Drain flow

Other pollutants that have the same risk factors are ammonium and pathogens. Both are present in livestock manures.

Measures that could reduce P loss

The following measures would be expected to reduce the risk of P loss to waters:

- Reduce P inputs derived from manure:
  - Minimise pollution from field heaps of manure
  - Reduce stocking density in grassland systems
  - Reduce numbers of outdoor pigs
  - Limit excretal returns (whether as manure or at grazing)
- Reduce the risk of erosion, surface runoff or direct transfer to waters:
  - Establish cover crops in the autumn
  - Grow crops well
  - Maintain good soil structure
  - Minimise direct loss of organic manures to water bodies
  - Reduce the risk of surface runoff
- Land Use Change
  - Conversion of arable to low-input grass
  - Specification of permitted crops (crops such as potatoes have an increased risk of P loss as well as N loss)
  - Organic farming (potentially, via reduced soluble P inputs as fertiliser lower soil P status, though other factors play a part)
Measures that could increase P loss

One measure carries a risk of increased P emission, depending on how it is implemented:
• Closed period for manure applications.

A closed period for manures in autumn to early winter could
• increase the quantity of manure applied during winter/early spring.
• In arable systems, increase the quantity of manure top-dressed after crop drilling during the winter/early spring (rather than ploughed down in autumn).

Both of these factors could increase the risk of P loss (and loss of ammonium, organic matter and pathogens associated with the manure). Phosphorus loss from freshly applied manures (and fertilisers) is greatest during the first few rainfall events (Withers & Bailey, ##), and depends on whether or not these result in surface runoff and/or rapid flow to drains. Recent experiments on a drained clay soil at Brimstone show that manure applications in winter are most susceptible to loss.

6.6.2 Impact on risk of ammonia emissions to air

The quantity of ammonia volatilised may be reduced by
• Prompt incorporation of manure (or injection)
• Avoidance of applications in summer
• Use of low-emission application techniques.

Manures such as FYM have a low content of ‘readily-available N’ (i.e. ammonium plus uric acid N) and therefore a lower risk of volatilisation per unit of total N applied, or per Livestock Unit, than slurries or poultry manures.

Measures that could reduce ammonia loss

The measures that will reduce NH₃ loss are those that reduce the total quantity of ammonium N applied – either as manure or by direct excretion, or as chemical fertiliser. In addition, any measures that encourage incorporation would be helpful. Measures for nitrate mitigation that could reduce ammonia loss are therefore:
• Reduce ammonium (and uric acid for poultry) inputs as manure:
  • Reduce stocking density in grassland systems
  • Reduce numbers of outdoor pigs
  • Limit excretal returns (whether as manure or at grazing)
  • Dietary N input reduction
• Reduce ammonium inputs as chemical fertiliser:
  • Apply no more fertiliser N than the crop requirement (including full allowance for all sources of N to the crop)
  • Limit fertiliser applications to a rate below that provided in a fertiliser recommendation system

Measures that could increase ammonia loss

One measure carries a risk of increased NH₃ emission, depending on how it is implemented:
• Closed period for manure applications.

A closed period for manures in autumn to early winter could potentially increase the quantity of manure top-dressed to grass during summer and, in arable systems, increase the quantity of manure top-dressed after crop drilling during the winter/early spring (rather than ploughed down in autumn). Both of these factors could increase the risk of ammonia loss.

Ammonia is lost to the atmosphere from manures after spreading. This loss can be minimised by prompt incorporation of the manure into the soil. Experimental data have been
summarised by decay equations for example within the MANNER model (Chambers et al., 1999). These indicate that leaving manures on the soil surface instead of incorporating them may increase ammonia emissions by 5-15% or more of the total N applied as manure.

Ammonia losses are even greater during summer than at other times. These findings are consistent with experimental data on crop N response which has been built into standard fertiliser recommendation systems (MAFF, 2000). These indicate that, due to increased volatilisation of NH₃, there is a reduction in crop-available N from cattle slurry applied on grassland in summer of 15% of total N, compared to the value for early spring or winter applications.

6.6.3 Impact on risk of nitrous oxide emissions to air

Nitrous oxide (a potent greenhouse gas) is produced by microbial action during N transformation processes within the N cycle:
- During denitrification, favoured in wet (waterlogged) and warm soils
- During nitrification, favoured in well aerated and warm soils

Both of these would be considered as ‘direct losses’ of N₂O. Additional emissions can follow from indirect routes:
- Emissions from NH₃ that has been deposited on soils from that atmosphere and then undergoes further transformations (on, average, about 1% of NH₃ emissions according to IPPC guidelines)
- Subsequent denitrification of leached nitrate, considered to be, on average, 2.5% of leached N according to IPPC guidelines

Measures that could reduce nitrous oxide loss

Clearly, minimising the amount of mineral N in the soil will reduce the risk of conversion to N₂O (direct loss) and decreasing nitrate leaching and ammonia volatilisation will reduce indirect losses. Consequently, the AP measures are well aligned with minimising N₂O emissions. Granli & Bockman (1994) identified the key actions for minimising N₂O emissions and also identified Defra’s Codes of Good Agricultural Practice as examples of the measures:
- Keep soil nitrate levels small outside the growing season
- Encourage best use of N from manures to reduce the need for ‘new’ N inputs
- Restricting N availability in the soil by limiting N input to that needed by the crop
- Creating soil conditions generally associated with small emissions
- Reducing unintentional N transfer from agriculture to other ecosystems

Measures that could increase nitrous oxide loss

As described above, any increase in NH₃ emission will indirectly increase N₂O emissions. Thus, if a closed period shifts manure application to spring and summer and applications are predominantly top-dressed, this may have some effects.

However, AP measures are designed to better manage mineral N and most will also benefit nitrous oxide losses.

6.6.4 Summary

It is important to take into consideration the potential for pollution swapping when assessing the effectiveness of existing and future AP measures; reducing nitrate loss is only of benefit if it does not exacerbate losses of other potentially harmful N species or of other contaminants such as P or FIOs.
Whilst the field measurements within the NIT18 project did not measure other loss pathways, a review of the AP measures, based our knowledge of the science of other contaminants has shown that the AP is, on the whole, beneficial towards other losses. This is especially the case for N$_2$O emissions.

There may, however, be concerns for increased NH$_3$ (and indirectly, N$_2$O) if the current AP has caused a move to spring/summer surface applications of manure, from autumn incorporated applications. Similarly, a move from autumn to winter manure applications may increase P losses. Any future changes to the AP measures needs to consider these factors.
CONCLUSIONS

This project has provided:
- Underpinning data on current field-scale nitrate losses from agricultural land under a range of systems that complements river and groundwater data reported by the Environment Agency. Such a micro-catchment monitoring network and associated data is specifically recommended by the EC in their guidance for implementing the Nitrates Directive.
- A method of estimating present and future nitrate loss based on current and predicted farming practice.
- A method by which the effectiveness of individual measures can be assessed both at field scale, and in the catchment context.
- The framework for predicting the time scale over which changes in farming practice will result in changes in nitrate concentrations in ground and surface waters.
- Expert opinion on the implications of the NVZ AP for the risk of eutrophication of rivers and estuaries.
- A method by which the impact of other measures, or externally driven changes in agriculture or climate, can be predicted.

7.1 The effectiveness of current Action Programme measures – field measurements

The measurement data showed that nitrate concentrations in leachate from typical agricultural land within NVZs often exceed 50 mg/l, especially under arable cropping or intensive dairying. Nitrate losses were elevated where manure was used; where fertiliser N inputs were high relative to crop N uptake; and in some intensively stocked grassland systems. Proper adjustment of fertiliser inputs to take account of the crop-available N supplied by manure is effective in reducing the additional losses. However, the data confirm that losses remain greater where manure is used in the rotation than where no manure is applied.

There was some evidence from the field measurements that further reductions in nitrate leaching from slurry (and, by inference, poultry manure) applications could be achieved by extending the Closed Period to cover all soil types, not just sand/shallow soils.

In the surface water micro-catchments (drained clay soils), there was sometimes evidence of increased nitrate concentration in drainage water in the spring (normally, nitrate concentrations were decreasing at this time). This suggests new sources of nitrate were being leached, such as recently applied manure or fertiliser.

Where additional fertiliser N is applied to wheat to boost grain protein, there is an indication that this increases N loss, compared with fertilising for yield only.

Nitrate losses from land were elevated where crop cover was nil or small for a long period prior to the winter – for example, after early harvested peas, and some rotational set-aside. In such situations, the only effective way to reduce nitrate leaching risk is to ensure green cover is established early enough in autumn to take up the high N supply – whether a commercial crop such as oilseed rape, or a purpose-sown cover crop.

7.2 The effectiveness of current Action Programme measures – modelling

The development of a model to scale up and extend field measurements is essential for providing quantitative assessments of the current AP measures and testing the likely impacts of new measures. The choice of two contrasting, real, catchments (arable Meden and grassland Taw) gave the likely range of effects on nitrate leaching of the current AP, and also considered other options for a revised AP: Tables 7.1 and 7.2.
Table 7.1. Modelled results for the Meden, showing N leached per hectare of agricultural land and nitrate concentrations in drainage from agricultural land, along with the average effect of a mitigation measure across the whole catchment and the range of effect in the various sub-catchments.

<table>
<thead>
<tr>
<th></th>
<th>N Leached (kg/ha)</th>
<th>Nitrate Concentration (mg/l)</th>
<th>Average Reduction (%)</th>
<th>Range of Reduction (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>58.4</td>
<td>147</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Current AP measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Do not exceed crop N requirement</td>
<td>54.6</td>
<td>138</td>
<td>6.5</td>
<td>1.7 – 20</td>
</tr>
<tr>
<td>Closed period PLUS do not exceed crop N requirement</td>
<td>49.6</td>
<td>125</td>
<td>15.0</td>
<td>1.9 – 23</td>
</tr>
<tr>
<td><strong>Additional measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10% Fertiliser Reduction</td>
<td>54.1</td>
<td>137</td>
<td>7.4</td>
<td>4.0 – 11.7</td>
</tr>
<tr>
<td>Remove all Manures</td>
<td>43.3</td>
<td>109</td>
<td>25.9</td>
<td>5.8 – 37.4</td>
</tr>
</tbody>
</table>

Table 7.2. Modelled results for the Taw, showing N leached per hectare of agricultural land and nitrate concentrations in drainage from agricultural land, along with the average effect of a mitigation measure across the whole catchment and the range of effect in the various sub-catchments.

<table>
<thead>
<tr>
<th></th>
<th>N Leached (kg/ha)</th>
<th>Nitrate Concentration (mg/l)</th>
<th>Average Reduction (%)</th>
<th>Range of Reduction (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>23.0</td>
<td>29.3</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td><strong>Current AP measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Do not exceed crop N requirement</td>
<td>22.7</td>
<td>28.8</td>
<td>1.5</td>
<td>0.1 – 5.0</td>
</tr>
<tr>
<td><strong>Additional measures:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10% Fertiliser Reduction</td>
<td>22.4</td>
<td>28.5</td>
<td>2.7</td>
<td>0.5 – 5.4</td>
</tr>
<tr>
<td>Remove all Manures</td>
<td>21.0</td>
<td>26.7</td>
<td>9.0</td>
<td>2.7 – 27.5</td>
</tr>
</tbody>
</table>

Both Tables show the range of likely effects within the sub-catchments, depending on the individual circumstances of the sub-catchment (cropping, soil type, livestock numbers and type, etc). The range of likely effect of the AP measures also depends on the farmer practices within the catchments pre-AP (e.g. what proportion of farms made allowance for manure N inputs when planning fertiliser inputs).

These Tables illustrate the main lessons learnt from the NIT18 project:
- **Arable losses often greater than grassland losses** (for the same rainfall) because grassland provides all year cover for N uptake.
- **The relatively large contribution of manures to N losses in many catchments** depending, obviously, on their composition in terms of livestock number and type. The Meden is intensively stocked compared with some arable catchments. Removing all manure reduces nitrate losses by 25% and 9%, respectively, for the Meden and Taw.
• **Good fertiliser practice reduces nitrate loss** – the size of effect varies, but in the two example catchments ranges from 2-7%, on average.
• **The closed period on sandy/shallow soils decreases nitrate loss** – by 15%, on average in the Meden catchment, by about 7% on a more typical catchment with lower stocking densities. This measure was not tested on the Taw because the closed window was not applicable on this soil type.
• **Reducing N fertiliser applications below optimum decreases nitrate loss** - with a greater reduction (c. 7%) on arable land compared with c. 3% on grassland.

### 7.3 Implications for nitrate policy

The analysis of effectiveness of Action Programme measures within this project has clearly demonstrated that the most effective measures are focused on manure and fertiliser management:

- Minimise post harvest soil nitrate (i.e. at risk of winter leaching) by using a fertiliser recommendation system to minimise the risk of over-fertilising the crop.  
- This includes making FULL allowance of manure applications when developing the fertiliser recommendation.  
- Closed window of application for N fertiliser when crop uptake rate is small and leaching risk is high.  
- Closed window of application in the autumn on sandy/shallow soils for manure with a large proportion of readily available N.

Even so, field measurements within the project (from sites within the NVZs) show many instances of high nitrate concentrations. Modelling of the current NVZ AP measures suggests a small effect, when averaged across a catchment or the whole NVZ area:

- A potential reduction in nitrate leaching of c. 7% in a predominantly arable catchment with manure and a significant proportion of sandy/shallow soils (i.e. closed periods for manure application) – half of this benefit is lost if the soil type is such that closed periods do not apply.  
- A potential reduction in nitrate leaching of c. 2% in a predominantly grassland catchment on clay soils, the reduced effect due to no closed periods for manure applications on this soil type.

Therefore, whereas the current AP restricts manure applications on sandy/shallow soils, the assessment from this project and other experiments indicates the need to consider broadening the scope to drained clay soils.

In heavier textured, structured, soils (clays and loams), water generally moves laterally across the surface or through the soil surface layers to cracks, channels (and, ultimately, drains, if installed) – ‘soil water drainage’. Importantly, this movement through e.g. root/worm channels, soil cracks or large pores (‘macropore flow’) can result in rapid water through soils that would at first be considered impermeable. This rapid water flow can occur either when the soil is fully wetted up, but also when the bulk of the soil is dry – then known as ‘bypass flow’. Thus, water (and nitrate) movement through clay soils can occur as:

- Rapid movement through cracks, macropores, etc (bypass flow)  
- Slow movement through the bulk of the soil

The concentration of nitrate in the drainage water (i.e. how much N is leached) will depend on how much contact the water has with sources of nitrate. This results in complex nitrate leaching profiles from drained soils. For example, if most of the soil nitrate is held within the bulk of the soil, water moving rapidly through cracks, so not mixing with the soil, will be low in N. However if, for example, heavy rain falls after a recent fertiliser or manure application, water can pick up N before transferring to cracks.
The importance of this mechanism cannot be overstated. Structured soils can be considered retentive of N when the nitrate is protected in the bulk of the soil and only moves downward with slowly mobile water. However, if rapidly moving water (bypass flow) has access to substantial N (e.g. manure/fertiliser at the soil surface), then losses will increase.

Reviewing current and potential mitigation methods, we can conclude that:

- Good manure and fertiliser management will provide the biggest reductions in nitrate loss (compared with poor practice). These are, to some extent, covered in the current NVZ Action Programme. However, there is increasing evidence from experimental work that drained clay soils should not be considered as ‘retentive’ and that timing restrictions for fresh manure applications need to be placed on these soils.

- There might be other potential candidate measures for inclusion in a revised AP, particularly the use of cover crops. Placing a cap on fertiliser application rates below the level of the economic optimum would similarly reduce nitrate leaching on average (but the effects in individual fields would be variable), though this would carry an economic penalty.

- Many other practices are considered ‘Good Agricultural Practice’ and are included in the current Defra Codes. Many of these relate to maintaining the soil in good condition (avoiding cultivation or stocking when wet, for example) with little direct effect on nitrate loss – though there may be indirect effect from improved crop growth and N use.

- Some other measures could be considered an extreme response to the problem (e.g. arable conversion or reducing stocking numbers) and potentially carry a large cost. Several measures focus on animal management, particularly those relating to feed management with the aim of decreasing N excretion.

Modelling as summarised in Tables 7.1 and 7.2 sometimes shows large effects within the example catchments and sub-catchments of AP measures. However, taking account of land use, soil types and livestock numbers across the whole NVZ area, our assessment suggests that the overall effect of the current NVZ AP is small: somewhere in the order of a 3% reduction in nitrate leaching loss.

### 7.4 Variability and confidence of results

The validity of field-scale estimates can be tested in detail against experimental data. Some examples of such evaluation are given in Appendix III. The response of the model to change in environment or land management can also be tested in this way. Models of biological systems as complex as this are never perfectly accurate, especially when used across a range of sites, because of the inherent variability in factors which are not yet fully understood by science. Input data are also both imprecise and insufficient to fully define the system – for example a full site history is never available, and soil properties are inferred from approximate data such as texture.

Catchment-scale predictions can be evaluated against catchment-scale measurements, at least in the case of surface water catchments with little or no groundwater component, where responses are rapid. Such estimates require real weather data for the years in question and good quality measurement data. Ideally the measurement data should include flow data from the same location, and should not be immediately downstream of a sewage treatment works. In the context of evaluation of suites of multi-pollutant models, a need has been identified by members of the Steering Group for improved data sets specifically for model evaluation/demonstration. The sites would ideally represent a range of ‘typical’ catchments, with multiple pollutants and flow measured to sufficient precision to provide a good estimate of loads, and a good test of model predictions.
Catchment-scale evaluation is more difficult for groundwaters that, in the UK, have response times of decades or more. However, fortunately, the majority of measurements of leaching derive from these conditions, so that we have a good understanding of nitrate losses to the unsaturated zone from agricultural and some other land uses. Investigations to date suggest that for most groundwater catchments in England that are not confined by impermeable layers above, nitrate is conserved. Therefore, a reasonably reliable estimate of total annual inputs can be made.

The accuracy of catchment-scale estimates depends heavily on the accuracy of spatial data and of land management assumptions. Tests of the spatial data have shown that crop areas and livestock numbers can be estimated with good accuracy for catchments of more than about 10-20 km². Somewhat greater areas or additional data may be required if large pig and poultry holdings are involved, because of the large N input from localised sources, whose location is obscured by the census data, and for which the fate of manure is not known.

At the catchment scale, land management, site history and other factors cannot be known with precision, but the larger the catchment, the closer the management practices will approximate to the norm derived from survey data. Survey data do not cover all issues; nor can they disentangle entirely the interplay between, for example, land use, soil type, manure management, and fertiliser input. There is always a need for more data – and in the interim, for careful and informed use of such data as are available to take account of likely patterns of behaviour in agricultural systems.

The single greatest uncertainty, especially when assessing the impacts of relatively modest changes, is in determining exactly how much practice will actually change, and in what way. There is uncertainty both in defining current practice, and in predicting farmer response to regulations.

Finally, there is a need to understand and define ongoing changes in land use and management, which occur in parallel to but largely independently of any Action Programme measures. Analysis of data and predictions such as those of the Business as Usual report (Hodge & Renwick, 2005) is essential if forward predictions of water quality are required, and are important even for proper assessment of the component of change due to the Action Programme alone.

In summary, the changes estimated by the modelling above are unlikely to be accurate to better than +/- 50% for the scale of changes considered within the NVZ AP at present. The expected changes are small relative to the precision of the model system. Relatively better accuracy would be expected for greater changes.

A document exploring some of these issues in more detail is included as Appendix XI.

7.5 Response in groundwaters

Most management changes rapidly affect nitrate loss from the land. Changes are rapidly transmitted through clay soils to surface water catchments. However, it can take many years before effects are evident in groundwater abstractions, or in groundwater-fed rivers.

The approach adopted by BGS to identify broad aquifer types and model their typical response times has been invaluable in demonstrating the range of responses that are possible. Table 7.3 shows that most important aquifers are Chalk and Permo-Triassic Sandstone and, in many instances, times for changes to take effect at abstraction points are long.
Table 7.3. Generalised response times for the main aquifer types in England and rainfall categories.

<table>
<thead>
<tr>
<th>Aquifer type</th>
<th>Area km²</th>
<th>% change after 10 years</th>
<th>Time to 50% (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>&lt;650 mm</td>
<td>&gt;850 mm</td>
</tr>
<tr>
<td>Chalk</td>
<td>19,511</td>
<td>8</td>
<td>10</td>
</tr>
<tr>
<td>Sandstone</td>
<td>15,415</td>
<td>8</td>
<td>10</td>
</tr>
<tr>
<td>Limestone</td>
<td>5,829</td>
<td>28</td>
<td>30</td>
</tr>
<tr>
<td>Crag</td>
<td>3,095</td>
<td>&lt;5</td>
<td>175</td>
</tr>
<tr>
<td>Alluvium</td>
<td>8,289</td>
<td>48</td>
<td>58</td>
</tr>
<tr>
<td>Greensand</td>
<td>5,098</td>
<td>10</td>
<td>15</td>
</tr>
</tbody>
</table>

7.6 Eutrophication

If waters are found to be eutrophic, or in the near future may become eutrophic, and a significant amount of the nitrate present in the waters comes from agricultural sources, then the Member State must designate the land draining into the affected waters as NVZs and put in place Action Programmes or adopt Action Programmes across the total territory. Eutrophication is a complex process involving many factors, including not just nutrients but flow rates and climate.

In rivers, eutrophication is limited by P > Si > N. Since N is rarely limiting, reduction in N loss is unlikely to change river status. If the NVZ AP reduces the risk of surface runoff of P, this could reduce the risk of eutrophication. Estimation of this effect is very difficult because it comprises changes in risk management, for which no prior or post survey data are available.

In estuaries, there may be seasonal changes, with P limiting algal growth in spring, and N in summer. Any changes in measures that increase P inputs in spring are, therefore, likely to be detrimental. The changes in N emissions attributable to the current NVZ AP are unlikely to be large enough to have major impact.

A detailed case study to illustrate the issues for the Taw catchment showed that:
- The Taw Estuary system has the potential to respond relatively quickly to reductions in nutrient inputs. Reduced inputs of nutrients from the catchment would, if sufficient, lead to an improved trophic status within the estuary.
- The NVZ AP measures would not appear to be sufficiently stringent to bring such a change about. Beneficial changes to the trophic status of the estuary are also likely to be impeded if other nutrient inputs are not controlled.

7.7 Knowledge Gaps Identified & Future Research

7.7.1 Clay v permeable soils

Recent research has shown clearly that there are some critical differences in behaviour between permeable soils (typical of groundwater systems) and impermeable soils such as clays (typical of surface waters); and that these differences affect the optimal choice of pollutant mitigation strategy. In particular, loss via rapid flow (through drains or over the surface) is a greater risk on clays, and can result in significant pollution shortly after manure, fertilisers or other soluble materials are applied to wet soils. In contrast, within permeable soils, nitrate moves via the soil profile and takes weeks or months to move out of the root zone by leaching. The effect of manure management strategies is different on permeable than on clay soils. It also affects non-adsorbed pollutants such as nitrate differently from adsorbed pollutants such as ammonium and P. The EC is increasingly concerned to minimise pollution via direct runoff, including losses of pollutants other than nitrate (e.g. phosphorus, pathogens, pesticides). There is a need to understand the effects of
management on diffuse losses of these contaminants from clay soils and the interactions with weather. Only then can sound mitigation strategies be developed. Limited data are now available, but from a very small number of sites. The wide range of sites within this project, on commercial farms, provides a valuable resource for policy support.

7.7.2 Balancing pollution risks

As indicated above, the optimal measures for control of pollutant risk may differ for different pollutants. There are synergies, and conflicts. In addition, there is still relatively little sound, quantitative data showing impacts of the same measures on different pollutant types.

Catchment scale models for N, P, sediment, FIOs and gaseous emissions could usefully be harnessed to work together, for the same AP scenarios, to determine the direction and scale of impacts. Manure management in particular has implications for gaseous emissions/odour, nitrate loss and loss of P, FIOs and ammonia to water which are affected in opposing directions by measures such as closed periods and incorporation of manures. Management of set-aside for ecological benefit can carry risks of increased nitrate loss, and similar issues apply to optimal management of over-winter crop cover.

7.7.3 Improving assessments of effectiveness

The better the understanding of current farm practice and the effects of the AP on this practice, the better the information to feed models to assess the impact of measures. Fertiliser compliance is an area where there are substantial gains to be made but, in practice, it is difficult to assess whether farmers are compliant, and any assessment needs field scale detailed information. Much of this information can now be gathered from Farm Practice Survey and the British Survey of Fertiliser Practice. Also, the BSFP would provide information on impacts (by inside/outside contrasts) but care is needed in this analysis to ensure we are comparing like with like and have enough data.

There are difficulties in assigning impacts to the 'good practice' aspects of the NVZ AP: avoiding manures near to watercourses, or on steep slopes etc. This issue may be more relevant to P than to N, but a good research-based framework needs to be developed for quantifying these effects. Buffer strips, similarly.

In this regard, for catchment scale estimation, it is necessary to build up a surface connectivity index, to predict from existing information the proportion of land which is affected by such ‘surface runoff’ risks and associated measures. The approach should be to build up a mapped estimate of proportion of fields with slopes greater than an agreed value, and their land use and soils; proportion of fields with a stream or ditch along one or more sides, stream edge characteristics (slope of adjacent land, edge management), field sizes.

7.7.4 Assessing Water Framework Directive implementation progress

The challenges faced by the Water Framework Directive of assessing the effectiveness of implementation and demonstrating improvements (or likely improvements) in water quality have much in common with the Nitrates Directive. Given the likely delays in response associated with land management changes and water quality, the approach taken for this NIT18 project could easily be widened to follow other potential contaminants. The surface water sites within NIT18 would be well suited to extending data collection to P and sediment transfers and exploring relationships with N loss. There is also a need to consider other N species, especially NH₄⁺; although the emphasis of this work has been on NO₃⁻, the Nitrates Directive could be interpreted to include other N forms.

Similarly, organic N and P. This was identified in the eutrophication reports by CEH as an issue where data were lacking. We can postulate that there would be more organic N where
manures are applied, and less under arable in general, but we do not have sound data. This source of N also has major implications for greenhouse gases, since eventually it will denitrify (and may encourage denitrification by providing a source of C).
8 REFERENCES


Harrison, R. and Silgram, M. 1998. Final report to MAFF on project NT1508 (cover crops). Includes Appendix: the mineralisation of nitrogen in cover crops – a review


