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Proceedings of the Thirty-ninth Meeting of the Agricultural Research Modellers' Group

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This group, which is concerned with the applications of mathematics to agricultural science, was formed in 1970 and has since met at approximately yearly intervals in London for one-day meetings. The thirty-ninth meeting of the group, chaired by Professor N. Crout of the University of Nottingham, was held in the Kohn Centre at the Royal Society, 6 Carlton House Terrace, London on Friday, 30 March 2007 when the following papers were read.

PLENARY LECTURE

Modelling the sustainability of rural systems: concepts, approaches and tools

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SUMMARY

There is a growing realization that, to understand and manage natural resource systems in a sustainable manner, an integrated approach, taking into account the social, economic and biophysical components of such 'socio-ecological systems' (SESs), is necessary. Agent-based modelling is one approach that allows explicit modelling of human decision-making processes, providing a way of simulating both the social and economic components. When coupled with biophysical process models, they become powerful tools to explore two-way human-environment interactions. However, this approach is not without challenges. Modellers need to work out how to incorporate factors such as incomplete information, multiple goals, trust, loyalty and emotions, how to link agent-based models to existing biophysical models, and how to deal with the inevitable complexity of the models that result. New ways of validating these models also need to be developed as, unlike most biophysical models, there are moral and cost restraints on designing and manipulating experimentally SESs containing humans. There is also a need to ensure the relevance of the models by involving appropriate stakeholders in their development.

INTRODUCTION

Europe is in a period of significant change in relation to the development of future policies for the sustainable development of its landscapes and rural communities. International drivers such as globalization, climate change and European policy (particularly Common Agricultural Policy reform and the Water Framework Directive) are coinciding with national shifts in demographic and social patterns such as residential, leisure and work patterns, quality of life expectations and access to land for recreation. People in rural communities of the future are likely to derive their livelihoods from a variety of activities: farming, forestry, rural industry, services, tourism and recreation, and the provision of environmental qualities for the public good. However, this shift is giving rise to new tensions and conflicts associated with the interaction between production and consumption within the same geographical space. There is considerable debate on how to reconcile these tensions, but this is complicated by the fact that it is not always clear what people in different sectors actually want from the countryside or, even if they do, whether the desired solutions are socially, economically or environmentally possible or acceptable, and whether they are compatible with broader policy decisions. What is needed are concepts and tools incorporating the social, economic and biophysical components of rural systems which can be used to evaluate and test different proposals and policies before they are implemented, and without the time, expense and moral implications of altering the real thing.

As a 'socio-ecological' modeller, it was a great pleasure, therefore, to be asked to give the plenary paper for the 39th Agricultural Research Modellers' Group this year. I was most interested to read the plenary paper given by John Thornley at last year's meeting, particularly in his view that very few models were being developed at the ecosystem level and that they are sorely needed to answer some of the pressing environmental questions of our time (Thornley 2006). I see the opportunity to talk about some of the models we are developing as an opportunity to redress this balance in small measure, but I would go even further than John, and suggest that the next great challenge for modellers of natural resource systems is to incorporate human decision-making and behaviour into such ecosystem process models.

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There are already a number of models linking the human and biophysical components of particular systems which are generally used for integrated assessment, but in many of these models the human dimension is based on economic cost-benefit principles that attempt to optimize use of resources such as capital or labour to maximize a particular output. These approaches have limitations in that they are structured to represent equilibrium when production has stabilized, they presuppose a 'goal' of the system, and do not adequately consider the micro-decisions being made by the various actors within it. Indeed, it has even been argued that the assumptions used in these models are flawed and that their predictions are untrustworthy (e.g. Moss *et al*. 2001).

In this paper, therefore, I would like to discuss ideas and concepts that we have been developing at the Macaulay Institute in Scotland, UK to understand the key processes occurring in rural systems, and new modelling approaches we are using to enhance our understanding of these processes. We are hoping that this understanding can be used to help identify desirable future states of landscapes and rural communities which meet multiple stakeholder needs, and to identify what needs to be done to achieve these desirable states.

CONCEPTUALIZING RURAL SYSTEMS

A useful concept to describe rural systems in this context is that of a 'socio-ecological system' (SES), which takes the view that social, economic and biophysical components considered in isolation can each only provide a partial understanding at best, and that all three aspects must be taken together to obtain a fuller understanding (Berkes *et al*. 2003). In this approach, rural communities and landscapes are 'open systems' operating far from equilibrium (Kay *et al*. 1999), with material, energy and information flowing both into and out of them. It is the way in which their internal social, economic and biophysical components are organized in relation to one another that determines the direction and magnitude of these flows. This implies that humans should be seen as integral parts of such systems, rather than as impartial observers, or as external drivers on ecosystems (but not influenced by them), or as users of the environment (but not influencing it).

What are the dynamics of such SESs? The 'adaptive cycle' concept put forward by Gunderson & Holling (2001) sees such systems cycling through an *exploitation* (r) phase, a *conservation* (K) phase, a *creative destruction* (Ω) phase, followed by a *reorganization* (α) phase. Thus, there is explicit recognition that change is an intrinsic property of SESs, and that they seldom reach a static equilibrium state. Walker *et al*. (2004) have taken the adaptive cycle idea further by conceptualizing SESs as complex systems which self-organize within *basins of attraction*. External perturbations at critical times may, depending on circumstances, transform a system from one basin into a neighbouring one, particularly if it is close to a critical threshold of a particular variable (Walker & Meyers 2004). The concept of system *resilience* is used to describe the amount of effort required to move from one basin into another, while the *adaptability* of the system is the degree to which the components of the system can influence its internal dynamics and hence its resilience. *Sustainability* can then be thought of as the process of maintaining the system in a desirable basin. Resilience may, therefore, be advantageous if the system is already in a desired basin (e.g. conservation of a particular habitat), but disadvantageous if attempts are being made to move it from an undesired basin.

Adaptive cycle patterns have been discerned in several SESs (e.g. Gunderson & Pritchard 2002; Allison & Hobbs 2004). An example of such dynamics in a historical Scottish context is that of the 18–19th Century highland clearances, in which an external perturbation of increased wool prices resulted in a major shift from crofting agriculture to extensive sheep grazing with dramatic consequences on the social, economic and ecological capital (and hence landscapes) of significant areas of Scotland (Richards 2000). This can be thought of as a transformation from the K phase in a basin of attraction representing crofting to a new basin of attraction representing grazing, passing through the Ω phase (e.g. forcible removal of the crofters from the land), α phase (e.g. reorganization of a proportion of the crofting community into other occupations related to sheep production), and r phase (e.g. growth of the new sheep industry) of the adaptive cycle. The persistence of the resulting SES in many of these areas for nearly two centuries would suggest that, under the prevailing socio-economic conditions, it has a high degree of resilience (i.e. its basin of attraction is relatively 'deep').

It is conceivable that current rural policy reform, climate change, and pressure from other drivers such as demographic trends, public demand for recreational landscapes, and trade liberalization, are moving rural systems in Europe into new social, economic and ecological basins. However, it is not yet clear whether this is the case, and if it is, what the nature of the new basins of attraction will be and whether or not they will be desirable, and if so, for whom.

MODELLING SESs

The dominant factor influencing the dynamics of SESs, compared to other ecosystems, is the presence of humans, with their ability to remember and learn from the past, and to perceive both current and future states of their biophysical and social environment. These memories and perceptions, along with the influence of rules, norms and values, are used as a basis to form goals and plans which guide the making of decisions and carrying out of actions. These actions may affect the environment in some way, which, in turn, will influence future perceptions, decisions and actions (non-linear feedbacks and co-evolution) which make SESs complex adaptive systems with emergent behaviour (Gunderson & Holling 2001). There is a need, therefore, to understand the processes by which these decisions are made, and the factors which influence them. Institutions, social networks and social norms are key factors, as these place opportunities for, and constraints on, the number and types of potential decisions available at a given time. Perceptions, attitudes, and values both of land managers and of land users are also important, as these influence the processes by which a particular decision is reached. The underlying biophysical processes of the system must also be taken into account, as they influence the perceptions and decisions being made by the land managers and land users, as well as being influenced by the decisions made and actions taken.

To understand how SESs work, it is first necessary to take each of the components just mentioned and describe their key processes, then to reassemble them to understand how they interact together to contribute to overall system behaviour. Due to the complexity of most SESs, and because it is not possible to manipulate real systems experimentally, modelling needs to play a central role in this process to draw together the different components and allow rigorous testing of hypotheses of how the system can be changed. The question is, however, which modelling approaches best allow the integration of social, economic and biophysical processes?

Agent-based modelling (ABM) is one approach which has aroused the interest of a number of research communities involved in modelling social processes, mainly because it offers a way of replacing differential equations at an aggregate level with decision rules of entities at a lower level (i.e. individuals or groups of individuals). ABM originated from the field of artificial intelligence, and has parallels with Individual Based Modelling (IBM) in ecology (Huston *et al.* 1988). ABMs consist of a number of 'agents' which interact both with each other and with their environment, and can make decisions and change their actions as a result of this interaction (Ferber 1999). Agents may contain their own 'model' of their environment (which may not necessarily be complete or even true) built up from its interactions with it. The behaviour of the whole system depends on the aggregated individual behaviour of each agent. This allows the influence of human decision-making on the environment to be incorporated in a mechanistic and spatially explicit way, also taking into account social interaction, adaptation and multiple scales of decision-making. Agents can interact either through a shared environment and/or directly with each other through markets, social networks and institutions. Higher-order variables (e.g. commodity prices, population dynamics, etc.) are not specified as they are in conventional mathematical programming techniques or econometrics, but, instead, are emergent outcomes. In the land use modelling community, a number of such agent-based land-use models are now beginning to appear (see recent reviews by Parker *et al.* 2002; Bousquet & Le Page 2004; Hare & Deadman 2004), many of which involve the grafting of a multi-agent system representing a number of households onto a cellular automaton 'landscape', with each agent being linked in some way to the cells over which it has influence. Apart from changes in actual land cover, however, these models generally treat the landscape as a relatively static entity, and do not simulate processes such as soil water and nutrient dynamics (e.g. Balmann *et al.* 2002; Deffuant *et al.* 2002; Hoffmann *et al.* 2002; Lynam 2002). Some do include such processes, but somewhat simplistically; for example, Lim *et al.* (2002) uses multiple regression equations for changes in soil characteristics and estimations of crop yields.

However, as the interactions between humans and their environment are two-way, in that actions occurring as a result of human decisions affect processes within the biophysical environment, which in turn may influence further decisions made, it would seem essential, if we are to deepen our understanding of how SESs function, to integrate existing dynamic biophysical simulation models with these emerging ABMs. Some progress is already starting to be made in this direction. The People and Landscape Model (PALM, Matthews 2002, 2006; Matthews & Pilbeam 2005*b*; Matthews *et al.* 2005), for example, is an agent-based model consisting of a number of households located on a landscape made up of a number of heterogeneous land units, each of which contains routines to calculate its water balance and carbon and nitrogen dynamics. Organic matter decomposition is simulated by a version of the soil organic model developed by Parton *et al.* (1988) (CENTURY), while water and nitrogen dynamics are simulated by versions of the generic routines in the Decision Support System for Agrotechnology Transfer (DSSAT) crop models. Various crops are simulated by versions of the same crop models. The soil processes are simulated continuously, and vegetation types (crops, weeds and trees) can come and go in a land unit depending on its management. The emphasis in PALM is on modularity rather than a single model simulating everything and different components (e.g. soil processes) relevant to different problems can be selected from a 'toolbox' and combined to create 'bespoke' models for specific purposes. So far, the model has been used to investigate the survival characteristics of different crop nutrient strategies, including the sale and purchase of excess organic manure between households (Matthews & Pilbeam 2005*a*), and plans are underway to incorporate routines to simulate carbon dynamics in organic soils, and use it to investigate different strategies land managers in Scotland, UK may employ to decouple economic performance of land uses from greenhouse gas emissions.

CHALLENGES

A major advantage that ABM has over other modelling approaches is that it offers the potential to incorporate not only economically rational principles, but also the psychological and sociological factors which influence decision-making, into environmental simulation models. A number of different decision-making models are being developed, but these need to be compared with each other, with decision theory and with observations in the real world, particularly in relation to the implications of their aggregation at the macro-scale to see if particular approaches are more suited to specific situations. A significant challenge also facing agent-based modellers is to include factors such as incomplete information, multiple goals, trust, loyalty and emotions, which are often context dependent, into their models in a realistic way. Agents also need to incorporate mental models of their social and biophysical environment, be able to learn, plan, communicate with each other, evolve rules of behaviour, form social networks and be capable of collective decision-making.

There are also significant challenges in linking agent-based models to biophysical models. Kuhlman (2004) outlined four key challenges when coupling models, based on his experience involving linkages among no less than seven models. Firstly, there may be differences in the understanding of the scenario among members of the interdisciplinary team. Next, there may also be variations in the underlying assumptions among team members, and hence in the sub-models. Thirdly, there can be multiple sources for what are essentially the same data, with the various sub-models not necessarily all using the same source. Finally, there can be overlap in functionality between sub-models, in that two (or more) sub-models, for one reason or another, may happen to contain subcomponents representing the same real world phenomenon, with the risk of telling different stories about the fate of that phenomenon in the coupled whole. The best approach is probably the development of 'fully integrated' models (Antle *et al.* 2001), in which a single model with all the desired components is built from scratch, which, while posing the greatest challenge in terms of required effort, is also the most rigorous as it ensures common understanding of underlying assumptions and theories.

A key issue underlying such combined models, however, is that of complexity, which poses a dilemma for modellers. On one hand, there is the general preference of end users for simple explanations, but on the other, there is no escaping the fact that SESs are complicated systems in that a significant number of components interact with each other. They may also be complex adaptive systems in that they are path dependent with their current and future states depending on their history, and may exhibit non-linear behaviour, self-organized criticality and clustered volatility (Bak 1994). Possibly, the complex nature of such systems requires sufficiently detailed models to be developed in order to capture behaviours that would not be possible with simpler models. However, given that all models are simplifications of reality, the dilemma is what constitutes 'sufficient detail' for such models. One school of thought, referred to as 'greedy reductionism' by Pinker (2002), argues that increasingly detailed models are required that are capable of simulating processes at finer and finer levels. A contrasting point of view is that simpler frameworks, more readily aligned with end-users' modes of action, are required (e.g. Shorter *et al.* 1991). The two approaches may not necessarily be mutually exclusive and the best way forward may be to take a simple framework as the starting point, and incorporate additional details as necessary to describe the processes of interest. A danger of this approach, which needs to be guarded against, is that the resulting model may reflect the prejudices of the user, and only contain the components that he/she thinks are important.

Another challenge for agent-based modellers is to find innovative ways of validating their models. Validation of individual components of an SES model against observed data is possible, although this does not test for any errors introduced through linking them at a higher level. Other approaches of validation include comparison with a historical situation, for example, changes in commodity prices or demographic changes over a particular time period, and comparing the plausibility of simulated results with outcomes expected by 'experts'. However, as the purpose of the model is to *explore* options for effecting change in SESs rather than to *predict* them, it is perhaps more important that the structure of the model and the assumptions incorporated into it are transparent, and therefore well-documented.

Provided these are known, they can provide a focus for debate, and sensitivity analysis can be carried out to determine their relative importance to the overall system.

This last point highlights the growing recognition of the need to improve model relevance by involving stakeholders and potential users of model outputs in their development. In relation to ABMs, Parker *et al.* (2002) distinguish three levels of participation: (a) where stakeholders are involved at all stages of model development, including model conceptualization, building and use, (b) where stakeholders are not involved in model building, but are involved in model running, and (c) where models are presented to policy makers as ready-made software packages with the ability for the users to alter model parameters to test various policy options. ABMs may have particular advantages in a participatory context, as the agents can be made to represent individuals or groups with whom stakeholders can identify. The latter can therefore criticize the models or contribute to their design in ways that make use of their practical knowledge. Several examples describing the use of ABMs as a tool in participatory role-playing games to solve specific problems are given by Bousquet *et al.* (2002). However, participatory modelling should not been seen as a panacea, as it does have its downsides; these include the time and cost of involving stakeholders, possible bias in the stakeholders selected resulting in a poorer representation of the real-world system, and lack of academic credibility of the resulting models. Difficulties in validating such models is a particular issue, although greater 'buy-in' and trust in a model by the participants may be more important in this context than its numerical accuracy (van den Belt 2004).

CONCLUSIONS

While incorporating social processes into ecological process models is not likely to be easy, we believe that it is important and necessary to make the effort to do so. The growing use of ABM in the social sciences opens up new opportunities for this to be done. While the work described in this paper has concentrated so far on land use, we believe that we have to focus increasingly on the whole rural sector, including rural industry and the wider rural population, and to examine its capacity to adapt to drivers such as changes in climate and policy. One theme we are developing is the identification of practical and strategic options for rural communities to move towards low carbon economies by reducing greenhouse gas emissions, increasing carbon storage and switching to alternative energy systems. To do this, we plan to integrate knowledge from the social sciences, economics and ecological sciences to evaluate the impact of various policy options on all aspects of rural systems and investigate the tradeoffs between the individual interests of making a livelihood and the broader societal goals of mitigating and adapting to climate change. Although the work will be grounded in case studies in Scotland, UK, our aim is to develop a broad-based understanding of the realistic options for both adapting to, and mitigating of, climate change, together with distinctive common factors that may be applicable elsewhere.

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ABSTRACTS OF COMMUNICATIONS

Mathematical model for phosphate uptake by mycorrhizal plants. T. ROOSE[1] AND A. SCHNEPF[2].

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Most terrestrial plants form mutualistic relations with fungi and the fungi that interact with plants in this way are termed mycorrhizae. A two part model is reported that estimates the phosphate uptake by a single cylindrical root and associated mycorrhizae. The first part is a model of mycorrhizae growth, based on the ideas of Edelstein (1982) and consists of coupled equations for fungal tip density n and fungal length density ρ:

$$\frac{\partial n}{\partial t} = -\nabla \cdot (n\underline{v}) + f, \qquad \frac{\partial \rho}{\partial t} = n|\underline{v}| - d\rho, \tag{1}$$

where $\underline{v}$ is the velocity of tip movement, d is the rate of hyphae death and f is the function for creation and destruction of fungal tips. If fungal tip branching only occurs via tip splitting then $f = bn$, however, if tip–tip anastomosis and tip–hyphae anastomosis also occur in addition to tip splitting then $f = bn - en^2 - gn\rho$, where b is the rate of tip splitting, e is the rate of tip–tip anastomosis, and g is the rate of tip–hyphae anastomosis. The model was calibrated against the experimental data of Jakobsen *et al.* (1992) and the results for *S. calospora* are shown below in Fig. 1.

The fungal growth model was then incorporated into the single root nutrient uptake model first reported by Tinker & Nye (2000) and further developed

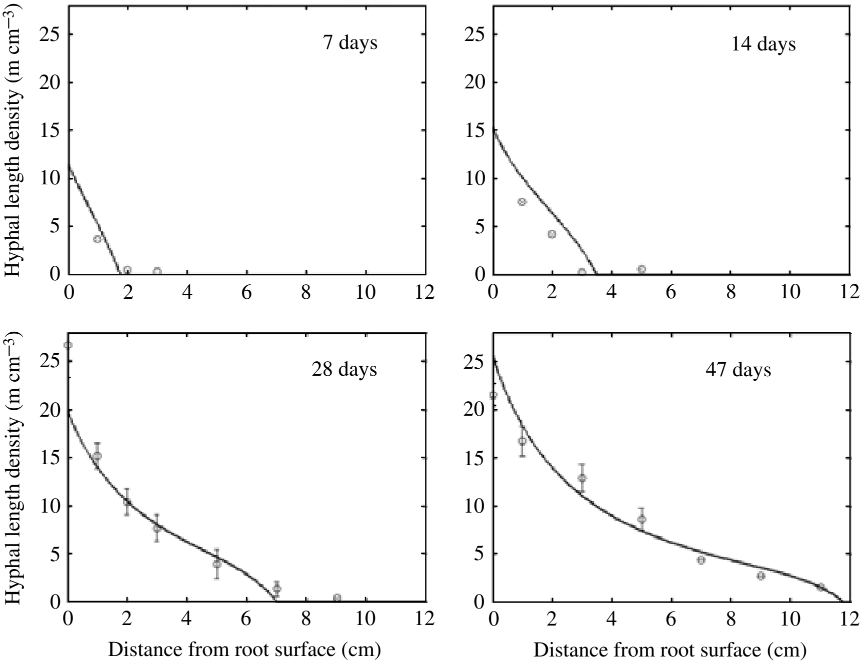

Fig. 1. Results of fitting linear tip branching model to the experimental data for *S. calospora* from Jakobsen *et al.* (1992) at times $t = 7$, 14, 28 and 47 days.

by Roose *et al.* (2001). The detailed derivation of the fungus-root phosphorus (P) uptake model is shown in Schenpf & Roose (2006). The model couples the fungal growth equations (1) with the nutrient dynamics equation:

$$(b+\phi)\frac{\partial c}{\partial t}=\nabla\cdot(Df\nabla c-\underline{v_{H_2O}}c)-2r_{myc}\pi\rho\frac{V_{myc}c}{K_{myc}+c},$$

where c is the concentration of P in soil solution, b is the soil buffer power, ϕ is the soil water content, D is phosphate diffusivity in the soil, f is the impedance factor, $\underline{v_{H_2O}}$ is the fluid flux in the soil, V_{myc} and K_{myc} are mycorrhiza P uptake parameters and r_{myc} is the radius of a hypha.

The model was solved with a combination of analytic and numerical techniques using literature values for mycorrhizae nutrient uptake parameters. Model predictions were strongly dependent on the values for V_{myc} and K_{myc}. In some cases the fungus was able to almost instantaneously deplete all of the P in the soil and therefore the active site for P uptake was a zone within a couple of centimetres near the edge of the fungal colony. This type of response indicates that the host plant relies almost entirely on the fungus to obtain P. Recent experimental studies by Smith *et al.* (2003) support the hypothesis that mycorrhizal plants rely on the fungal symbiont for the majority of their phosphate supply.

Another set of plausible parameter values suggested a slightly different scenario. Whilst the overall P uptake by the plant was still dominated by the mycorrhizae, the active region for P uptake was almost the whole domain of the fungal colony.

These two qualitatively different results highlight the need for more thorough experimental studies that would simultaneously measure the extent of the fungal colony and spatio-temporal amount of phosphate in the soil. The current study also highlights the need for enhanced mechanistic characterization of single hyphae nutrient uptake rates, since these were the parameters that had most uncertainty associated with them and to which the results showed most sensitivity.

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Disentangling the weather, pathogen life cycle and epidemic development chain with an application to yellow rust.

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Published models of plant disease epidemics, incorporating weather, are difficult to compare because each model has its own unique structure. Moreover, these models study how weather changes affect the epidemic growth rate, *r*, and hence the disease severity without assessing the contribution of individual life cycle components. Papastamati & van den Bosch (2007) recently developed a method that does make such a distinction. Their method quantifies the sensitivity of *r* to weather changes as the sum of the effect of a change in a weather variable through the weather variable's effect on all individual life cycle components (i.e. pathogen reproduction, latent period and infectious period). In the current project, this method is extended by developing an elasticity analysis and subsequently linking the model to observed weather patterns which enables a direct comparison between the effects of different weather variables (temperature, surface wetness duration and light quantity) under realistic weather patterns.

Yellow rust, caused by *Puccinia striifromis*, on winter wheat is studied as a key application. The three sites studied represent areas within the UK with contrasting climates. The results show clear differences in elasticities of *r* to weather changes between sites and across seasons. Seasonal differences per individual weather variable are most pronounced at the warmest and driest site whereas seasonal differences between weather variables are most pronounced at the coldest and wettest site. Despite these differences some clear general trends can be observed.

The foremost result is that temperature, and more importantly changes in temperature through their effect on pathogen reproduction, have the largest effect on *r*. Furthermore, the long latent period at low winter temperatures does not appear relevant to the low epidemic development over winter, which is contrary to the general beliefs of many phytopathologists (Zadoks 1961; Daamen *et al.* 1992). The results combined with long-term average yellow rust severity patterns, show that it is winter survival and not summer survival that controls eventual disease severity. Finally, the current UK spraying regime for wheat crops (HGCA 2007) is shown to be timed such that individual sprays are applied at a time when the life cycle component that is most affected by this spray is most sensitive to changes in weather variables.

This project has been funded by the UK Department for Environment, Food and Rural Affairs and the UK Biotechnology and Biological Sciences Research Council.

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Dry drainage: a sustainable management strategy for irrigation areas.

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Many intensively irrigated areas, especially in semi-arid environments, suffer from waterlogging and soil salinity because of the presence of saline groundwater at shallow depth. Conventional wisdom holds that the best solution is to maintain a net flux of salt away from the root zone by deliberate over-irrigation (i.e. leaching) and to control the water table by means of artificial drainage. However, as a result of climatic change, average annual rainfall and surface runoff are predicted to decrease and also to become more erratic in most of the sub-tropics (Watson 2001) so that the amount and quality of water available for leaching will deteriorate.

In some circumstances, it may be possible to devise an alternative strategy based on the concept of 'dry drainage' which aims to achieve the necessary salt balance in irrigated fields by passive transfer to adjacent uncropped areas, which act as evaporative sinks. It may therefore offer twin sustainability benefits: first it is less costly than conventional drainage; second, it obviates the need for disposal of saline effluent into the aquatic environment. Proper design and management of such systems depends on being able to predict the water and salt balances in the irrigated and sink areas. This is now possible with an accurate method of estimating evaporation from bare soil above shallow groundwater (Gowing *et al.* 2006) developed from a series of critical experiments on model soil-water systems (Rose *et al.* 2005). Such evaporation is characterized by liquid transport in the lower part of the soil profile, between the water table and the evaporation front, and vapour transport between the evaporation front and the soil surface, the two fluxes of water acting in series.

An evaluation of the merit of dry drainage requires answers to three key questions: (i) what is the limiting cropping intensity? (ii) What is the limiting water-table depth? (iii) What is the long-term impact of salt accumulation in the sink area? These questions were addressed using a simulation model for a dry-drainage system with various cropping patterns using published soil and climatic data for the Lower Indus Basin, Pakistan, where shallow saline groundwater, intensive irrigation, high evaporative demand and natural dry drainage exist (Konukcu *et al.* 2006). The results showed that dry drainage could satisfy the necessary water and salt balances when the cropped and sink areas were roughly equal and water-table depth was 1·5 m. The system was sustainable because salt concentrations in the sink area increased only slightly over 30 years.

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The Broom's Barn sugar beet growth model and its uses.

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Broom's Barn is the UK national centre for sugar beet research and knowledge transfer. Development of mathematical models has been an aim of sugar beet agronomic research for a long time. The initial Broom's Barn sugar beet growth and yield simulation model has recently been expanded and adapted to a wider range of environments.

The model is process-based and weather-driven so that total crop growth and sugar yield are integrated at a daily time interval. The model is a series of mechanistic and semi-mechanistic equations that simulate the integrated effects of the important environmental variables determining growth and yield of commercial crops. It does not simulate the growth of diseased or nutrient deficient crops. The required inputs are soil available water holding capacity, latitude, sowing date and daily values of temperature, global radiation, rainfall and potential evapotranspiration. With these, the model simulates the effects of temperature on seedling emergence, the growth of foliage cover and the development of the root system down through the soil profile; the effects of diffuse and direct solar radiation, as intercepted by the foliage, on dry matter production; the effect of rainfall, irrigation and the soil water reserve on foliage growth, dry matter production and proportion of sugar to total dry matter yield; the effect of potential evapotranspiration and crop age on intercepted radiation use efficiency; and the effect of plant size on dry matter distribution to sugar yield.

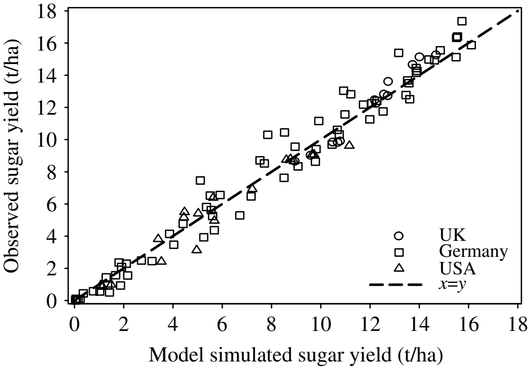

Fig. 1. The relationship of the observed sugar yields with simulated sugar yields in experiments carried out in the UK, Germany and the USA.

The model is not variety-specific, but can simulate the growth and yield for a broad range of cultivars, both in the UK and abroad (Fig. 1). The modelled sugar yields accounted for 0·96 of the variances in the measured sugar yields and the calculated root mean squared error was 0·84. This model has been programmed in a relational database system whereby it can capture and process temporal and spatial data to forecast sugar yield on a national or regional scale. It has now replaced the labour-intensive and costly field sampling method to forecast sugar yield for British Sugar plc (Fig. 2) based on yield from all individual fields. It has also been used to assess the impact on sugar yield of climatic changes in the past and future; monitor real-time crop performance in the growing season; scope the effects of prevailing weather on crop growth and soil water status; profile individual grower's crop performance.

Modelling spatio-temporal starch degradation patterns in apples during fruit growth.

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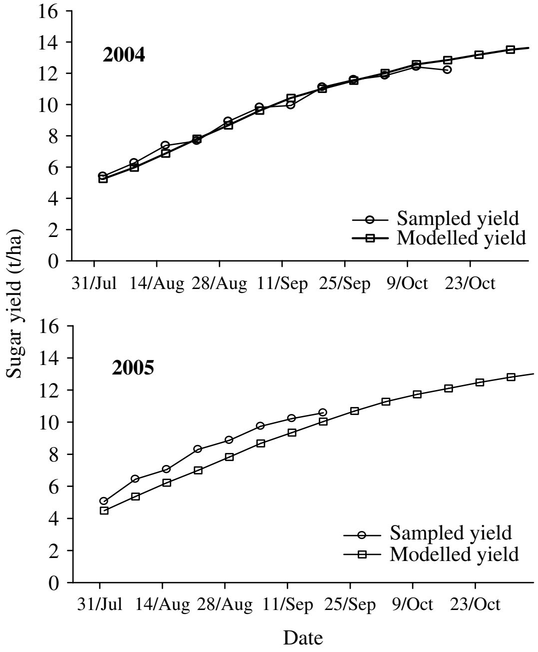

Fig. 2. Comparison of weekly sugar yields derived from 250 field samples with the simulated sugar yields using sowing dates and soil types from all sugar beet fields in 2004 and 2005.

Fruit quality changes are the result of a chain of biochemical and biophysical processes. Until now, most research effort in this area has been directed towards modelling changes in fruit quality and variability at the macroscopic level (Scheerlinck *et al.* 2004). Unfortunately, most of these processes are not fully understood, or are even unknown today, due to the complex underlying mechanisms. Given the economic importance of quality changes, research activities to unravel and model the mechanisms behind quality changes, such as starch degradation, are important.

Fruit is considered as a system which receives inputs from the plant and the environment and produces outputs. The system consists of different organizational levels; each level is a network of interacting elements. The goal is to describe fruit growth, respiration, senescence and quality changes in terms of interacting elements at different spatio-temporal scales. The key challenge is to identify different organizational levels within the fruit.

One such organizational level in fruit growth is the transport of sugars. Sugars either contribute to the metabolism directly or are stored as an energy source. The sugar content of the fruit tissue contributes to water uptake and the accumulation and decay of sugars and the storage of water vary in a spatio-temporal fashion. The internal structure of

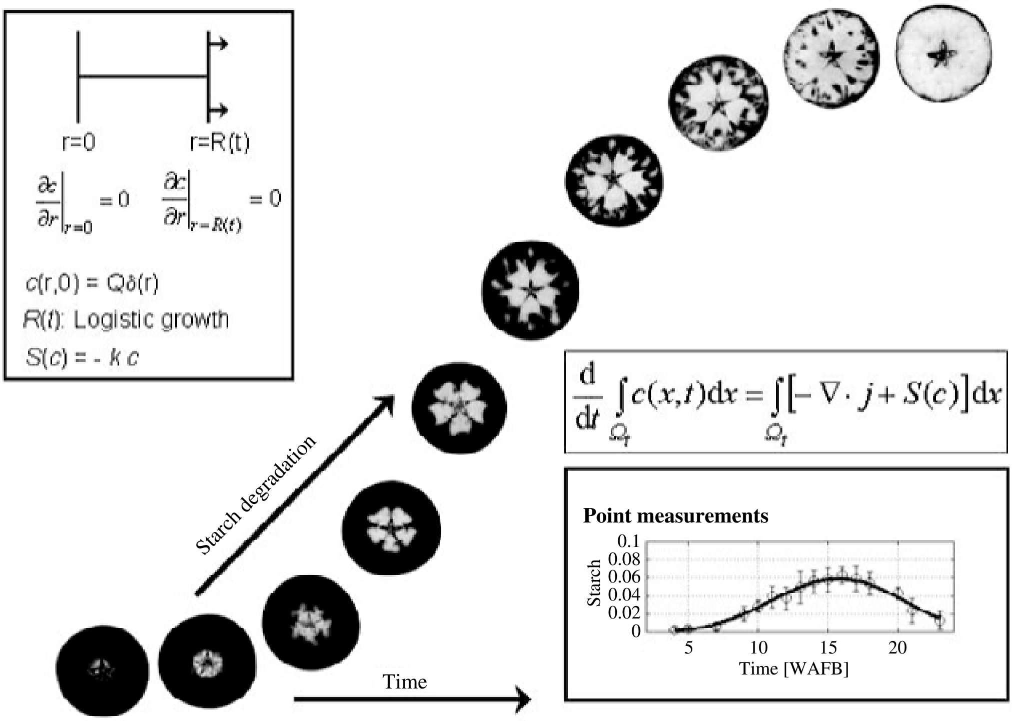

Fig. 1. Starch degradation patterns modelled by means of a conservation law applied to biochemicals moving from the core towards the apple skin. The patterns are shown at different stages during apple maturation, which correspond with the starch index as defined by G. Planton (Eurofru-Ctifl, Paris).

fruit is partly responsible for the spatial variation, but the transport of biochemical components also contributes to the formation of spatial patterns. For example, starch accumulates in the fruit during the growing season and is hydrolysed to sugar in the later stages of maturation. The starch concentration and its degradation rate differ between tissue zones, showing a typical degradation pattern, as shown in Fig. 1, which can be studied and measured using a simple staining method, such as the iodine test (Peirs *et al.* 2004).

Starch degradation begins at the core and expands in a star-shaped fashion to the skin. The hypothesis that the pattern formation is driven by a reaction–diffusion process was drawn from an analysis of starch degradation patterns. Compounds are produced in the seeds and travel through the fruit tissue where they act as signals to enhance the starch degradation process. Since the structure and geometry of fruit change with time, the governing equations need to be solved in a non-fixed computational domain, changing with time and in space (Crampin *et al.* 1999).

Application of a conservation law to the diffusing biochemicals on a growing domain yields a partial differential equation of parabolic type with diffusion, reaction, advection and dilution terms.

The Fund for Scientific Research, Flanders, is kindly acknowledged for financial support. Experiments were conducted at MeBioS-division, Department of Biosystems, Catholic University of Leuven, Leuven, Belgium.

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Development of large area wheat crop model for studying climate change impacts in China. S. LI[1,2], T. R. WHEELER[2] AND A. J. CHALLINOR[1]. [1]*Department of Meteorology, University of Reading, Whiteknights, Reading RG6 6RR, UK,* [2]*Department of Agriculture, University of Reading, Whiteknights, Reading RG6 6AR, UK*

The spatial and temporal scale of climate model output is relatively coarse compared with the usual inputs to impact models, such as dynamic crop models. The general large-area model for annual crops (GLAM) has been developed to simulate crop productivity at the spatial scale of global and regional climate models. The model has been successfully used to simulate groundnut yield over large areas in India (e.g. Challinor *et al.* 2004) and to study crop-atmosphere feedbacks across the tropics (e.g. Osborne *et al.* 2007). The aim of the present work was to develop a large-scale wheat model using the GLAM framework and to evaluate its applicability to wheat production in China under the current climate. Processes such as leaf area development and crop development were modified, and a new wheat parameter set defined. Simulated leaf area index (LAI), crop duration and yield were compared with observations to assess the model performance of GLAM-wheat in China. The observed weather station data was used as climate input of GLAM at both spatial scales: county/city level (70–129 km) and field level (<3 km).

Simulated crop duration at field level agreed well with observations. GLAM-wheat was driven by station weather data, predicted LAI was slightly higher than observations during the early growing season, but the overall pattern of LAI was well simulated thereafter. The yield in these simulations showed a weak correlation ($r = -0{\cdot}134$–$0{\cdot}541$) with observed yield. This is probably because at the field level the yield variability was mainly affected by field managements and diseases, pests and so on. GLAM does not predict the effect of the detailed field management, diseases and pests on yield variability. At county/city level, there were higher correlations ($r = 0{\cdot}549$–$0{\cdot}741$) between observed and simulated yield. Thus model predictions improved when wheat yield was simulated at a larger scale. In one out of five sites at county level, the correlation between observed and simulated yield was significantly different from the correlation at field level at 5% significant level. It was concluded that the GLAM model is suitable to simulate crop yield at large scales (approximately 100 km). Model skill in reproducing spatio-temporal yield patterns is based on the higher yield-climate correlation at large scales.

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Modelling growth and development of bambara groundnut (*Vigna subterranea* (L.) Verdc.) for abiotic stress. A. S. KARUNARATNE, S. N. AZAM-ALI, S. S. MWALE AND N. M. J. CROUT. *Division of Agricultural and Environmental Sciences, School of Biosciences, University of Nottingham, Sutton Bonnington Campus, Loughborough, Leicestershire LE12 5RD, UK*

The leguminous crop bambara groundnut (*Vigna subterranea* (L.) Verdc.) is an indigenous, under-utilized secondary food crop in semi-arid Africa. It produces protein-rich seeds which are eaten unripe or ripe. To explore the potential production of bambara groundnut landraces in various agro-ecological regions and evaluate the possibilities of transferring genotypes among the regions, it is necessary to understand the role of environmental factors. Quantification of the influence of temperature, soil moisture and photoperiod on crop development with a suitable crop model leads to inexpensive and rapid screening methods of landraces for specific environments. The working model (BAMGRO-Stress) is a modified version of the CROPGRO model (Boote *et al.* 1998) and simulates the growth and development for abiotic stress factors such as moisture, temperature and photoperiod. The model should be able to account for differences between landraces in terms of growth, development and yield.

BAMGRO-Stress is a process-oriented model comprising different components that deal specifically with plant development, crop growth, soil water stress, temperature stress and photoperiod. It simulates a crop carbon balance and a crop and soil water balance. Potential aboveground biomass production is predicted from leaf area index (LAI), leaf extinction coefficient and radiation use efficiency. The simulation of growth includes leaf appearance, leaf area expansion, senescence and pod addition. Development is simulated through seven growth stages from sowing to maturity and achievement of each stage is simulated when a predetermined number of thermal units are accumulated. The time step in BAMGRO-Stress is 1 day. Temperature and photoperiod effect is modelled according to Brink *et al.* (2000). Soil water balance is calculated as described in Bannayan (2001).

Parameters of the model for phenological development, leaf appearance, leaf area expansion, radiation interception, biomass accumulation and partitioning and crop water balance are derived from previous data collected at the Tropical Crops Research Unit (TCRU), University of Nottingham, UK. The temperature stress was evaluated growing two contrasting landraces, UniswaRed (Swaziland) and S 19-3 (Namibia) at three temperatures (23 ± 5, 28 ± 5 and 33 ± 5 °C) under irrigation and 12 h day length at TCRU in the summer of 2006. The model for rate of leaf appearance was derived from this experiment and potential leaf area is predicted as a function of leaf number. Subtraction of senescence fraction from potential leaf area provided data for actual leaf area and thereby LAI. The model predicts the leaf number, LAI and dry weights for different temperatures with given temperature stress factors.

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A spatially explicit population model of the effect of spatial separation in grass-clover grazing systems.

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The benefits of using white clover (*Trifolium repens* L.) in pasture grazed by animals have been widely recognized. However, clover is considered inadequate and risky as the main source of nitrogen input, since its abundance in the pasture is patchy, low (typically less than 0·2) and shows great year-to-year variation. This is thought to be due to the costs of nitrogen fixation, competition with grass, the preference for clover by grazing animals, and patchy dung and urine deposition. One solution, suggested by a number of authors, may be to increase heterogeneity within the pasture by spatially separating clover from grass. This method of pasture management, in order to sustain higher clover content in both the sward and diet of grazing animals, would remove inter-specific competition and equalize grazing pressure, allowing clover to grow unimpeded.

An existing spatially explicit grass-clover model (Schwinning & Parsons 1996) was modified, and then used to examine the impact of spatial separation on the content, variability and patchiness of clover in pasture. Simulations show that in the first 10 years spatial separation: (a) increased the clover content by up to 37%, (b) reduced year-to-year variation by over 65%, and (c) increased clover patchiness. Spatially separated pastures were also affected by local and field scale disturbance in the same way as fine mixtures. This study shows the importance that the initial sowing arrangement of plant species may have on the success of clover within a pasture. This is discussed in terms of benefits to nitrogen inputs, herbage dry matter production and animal performance.

Schwinning, S. & Parsons, A. J. (1996). A spatially explicit population model of stoloniferous N-fixing legumes in mixed pasture with grass. *Journal of Ecology* **84**, 815–826.

The implications of farm-scale methane mitigation measures for long-term national methane emissions.

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A number of methane (CH_4) mitigation measures have been suggested and identified, but there is a need to know whether these would be effective over broad spatial scales and under future scenarios. Additionally it is necessary to ascertain whether widespread implementation of these mitigations would have consequences for levels of production and emissions of other pollutants. This research identified a number of potentially effective measures for reducing CH_4 emissions from ruminant livestock farming in England, Wales and Scotland including; increased productivity, increased fertility, improved forage composition, feed additives and vaccination against rumen methanogenesis.

The effectiveness of each of these strategies was quantified within a new modelling framework comprising three linked models. The models included were an animal emissions model (University of Reading), the Sustainable and Integrated Management Systems for Dairy Production ($\text{SIMS}_{\text{DAIRY}}$) and nitrogen field and farm (NGAUGE) models (Institute of Grassland and Environmental Research) and the Atmospheric Emissions for National Environmental Impacts Determination (AENEID) countrywide spatial model (Centre for Ecology and Hydrology). The models required some modifications to enable suitable interfacing and time-step compatibilities. The animal model generated CH_4 emissions for dairy cattle, beef and sheep under a range of intensities, driven by energy, forage quality and animal fertility. Further developments to the existing model allowed herd management decisions affecting replacement rate to be incorporated into the model at a herd level. The $\text{SIMS}_{\text{DAIRY}}$ and NGAUGE models were then used to simulate emissions of CH_4 from manure management as well as emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3^-) leaching, according to soil and weather factors and farm management. In order to take into account the national variation in farm type across dairy, beef and sheep systems, we defined three typologies for dairy farms (extended grazing, conventional intensive and fully-housed intensive) and two typologies each for the beef and sheep farms (upland and lowland) in terms of stocking densities, fertilizer inputs and conception rates. Although emissions of CH_4 were not assumed to be influenced directly by soil or climate, it was necessary to use soil/climate zone data and take these into account when modelling NH_3, N_2O and NO_3^- leaching as a result of CH_4 mitigation methods at the farm and national scale. The emission estimates from the $\text{SIMS}_{\text{DAIRY}}$ and NGAUGE models were then passed on to the AENEID model to assess the impacts of CH_4 mitigation methods against baseline emissions by scaling up by farm typology within soil/climatic regions to the national level.

For dairy cattle, an increase in milk yield per cow (30% in the modelled scenario), coupled with a reduction in dairy cows thereby staying within quota, resulted in the largest reduction in CH_4 emissions at the national level (−24%). The next most effective mitigation strategy was feeding supplemental fat (providing 0·06 lipid in diet dry matter), which delivered an estimated 14% saving in CH_4 emissions. Improving reproductive management as represented by a 30% increase in heat detection rate (HDR) reduced emissions of CH_4 by 7% and a high starch concentrate reduced emissions by 5% from the baseline scenario. A reduction in the milk yield per cow of 30%, coupled with an increase in the number of dairy cows to maintain national milk production, resulted in an increase in CH_4 emissions of almost 15%.

The most effective CH_4 mitigation measure for beef cattle and sheep was vaccination (−10%), while a diet high in starch also appeared effective at reducing emissions from beef cattle at the national level (–5%). Diets high in water soluble carbohydrate appeared to be counter-productive and increased modelled national CH_4 emission estimates slightly.

The effectiveness of increasing milk yield per cow as a measure to decrease CH_4 emissions was matched by similar decreases in emissions of NH_3, N_2O and NO_3^- leaching. While high fat diets for dairy cows appeared to decrease CH_4 emissions by 14%, emissions of NH_3 and nitric oxide were only slightly decreased by applying this mitigation measure, but N_2O emissions and NO_3^- leaching showed a slight increase compared with the base scenario. Small decreases in CH_4 emissions through the introduction of high starch diets or high quality forage were not matched by similar decreases for the nitrogenous compounds under investigation, which showed very marginal decreases following the implementation of these measures.

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Nitrogen intake and excretion from dairy cows as determined by a dynamic model.

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Agriculture is one of the major sources of nitrogen (N) pollution. To increase animal products, cattle, especially dairy cows, are offered increasingly higher amounts of N. However, the efficiency with which N is converted to animal product is low, leading to excess N which is excreted in urine and faeces. From an environmental perspective, losses of N as urine is less desirable due to its greater tendency for leaching and volatilization as ammonia, the major source of which is urea from urine. A data set derived from the present authors' experimental measurements and literature values was used to evaluate an existing model of ruminant digestion and metabolism with regard to its ability to predict N losses in urine and faeces. The model demonstrated that the energy and protein content of the diet affected N utilization within the animal. In line with observations, the model predicted that cows supplemented with maize starch excreted up to 52% less N and exhibited a higher milk protein output. Of particular environmental interest, feeding dairy cows maize-based diets reduced urinary N excretion by almost 30% compared with barley-based concentrates. Based on these results it would appear that feeding maize-based diets has a potential to reduce ammonia emissions by up to 25%. In agreement with literature observations, it was also shown that it is possible to improve N utilization in dairy cows by decreasing dietary protein concentration. Reducing the crude protein content of the diet from 190 to 160 g/kg dry matter reduced urinary N excretion without compromising lactational performance; this could have the potential to decrease ammonia emissions from dairy cows by

21% and nitrous oxide by 15%. Diets with low degradable protein sources also reduced N output in urine with little change in milk production. However, long-term effects of sustained reductions in crude protein intake need further examination in order to avoid unintended health, welfare and production consequences.

The financial support of UK DEFRA project LS3656 is gratefully acknowledged.